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Spatially explicit measurements of forest structure and fire behavior following restoration treatments in dry forests

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ABSTRACT

Restoration treatments in dry forests of the western US often attempt silvicultural practices to restore the historical characteristics of forest structure and fire behavior. However, it is suggested that a reliance on non-spatial metrics of forest stand structure, along with the use of wildland fire behavior models that lack the ability to handle complex structures, may lead to uncharacteristically homogeneous rather than heterogeneous forest structures following restoration. In our study, we used spatially explicit forest inventory data and a physics based fire behavior model to investigate the effects of restoration driven, variable retention harvests on structural complexity, both of horizontal and vertical dimensions, and potential fire behavior. Structural complexity was assessed at stand and patch scales using a combination of point pattern analyses, a patch detection algorithm, and nearest-neighbor and tree patch indices of height variation. The potential fire behavior before and after treatment was simulated across a range of open wind speeds using a 3-D physics based fire behavior model, the Wildland-urban interface Fire Dynamics Simulator (WFDS). Our results show that treatments resulted in an aggregated spatial pattern of trees consisting of a matrix of individual trees, clumps and openings similar to descriptions of historical dry forests. Treatments had inconsistent effects on vertical complexity across sites likely due to differences in treatment of ladder fuels; lack of reference conditions hinder evaluation of this structural aspect. Simulation modeling using WFDS suggest that treatments moderated fire rate of spread, fireline intensity and canopy consumption across all wind speeds tested and shifted potential fire behavior towards historical ranges. Our findings suggest that current restoration-based variable retention harvests can simultaneously fulfill objectives of altering structural complexity and of reducing fire behavior, though we recommend further research on desired ranges of vertical complexity to inform treatment design.

1. Introduction

Land management practices such as livestock grazing, fire suppression, and timber-oriented management following Euro-American settlement have contributed to varying degrees of altered forest structure across the western U.S. Changes in forest structure are particularly evident in dry forest types that historically had frequent low to moderate severity fire regimes (Brown et al., 2015; Franklin et al., 2013; Fulé et al., 2009; Naficy et al., 2010; Veblen et al., 2000). As a result, modern forests are often thought to have increased surface and canopy fuel loads and less complex forest structural patterns (Tuten et al., 2015). These changes are believed to result in an increased potential for extensive uncharacteristic wildfires exceeding historical ranges of fire behavior and severity with impaired ecosystem function and lower long term forest resiliency (Fulé et al., 1997; Hessburg et al., 2005; Savage and Mast, 2005). Concerns over the potential impacts of changes in forest structure have resulted in an increased emphasis on utilizing forest treatments to modify forest structure such that both contemporary forest structure and potential fire behavior are similar to historical patterns (Allen et al., 2002; Fulé et al., 1997; Moore et al., 1999; Schultz et al., 2012; Underhill et al., 2014; Youngblood et al., 2004).

These treatments seek to restore the historic patterns of forest structure and potential fire behavior to contemporary forests often using variable retention harvesting as a silvicultural practice. Several restoration-focused variable retention harvest implementation

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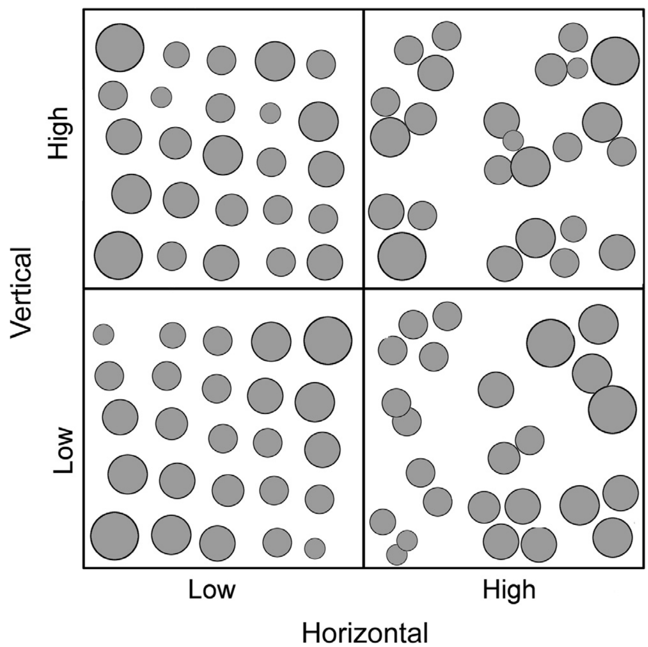

Fig. 1. Overhead view of the same rearranged trees (filled dots), scaled by height, across two levels of horizontal and vertical complexity.

strategies have been developed (Churchill et al., 2013a; Fulé et al., 2001; Graham et al., 2006; Moore et al., 1999; Reynolds et al., 2013; Youtz et al., 2008). Common to all of these is treatments using variable retention harvesting for the purpose of restoration specifically differ from other silvicultural strategies such as traditional fuel hazard reduction treatments in that they seek to create complex forest structures analogous to the natural range of variation in forest structure by modifying two interrelated components of forest structure: the type, number and size of individual elements (e.g. trees) that make up the forest and their structural complexity (Churchill et al., 2013b).

Structural complexity refers to the degree of spatial heterogeneity of forest structure (Zenner, 2004). Structural complexity can be characterized across both horizontal and vertical dimensions and at multiple scales (Fig. 1). Given that forest management typically operates at the spatial scale of a stand (O'Hara and Nagel, 2013) we focused this work on describing complexity at the stand and tree patch (i.e. within-stand) scales (Fig. 1). At the stand scale, horizontal complexity is often described by classifying the general spatial pattern exhibited by trees (e.g. Harrod et al., 1999) while patch scale measures describe the horizontal mosaic of different sized patches of trees, generally ranging from 2 to 20 trees, individual trees, and openings (Larson and Churchill, 2012). Vertical complexity at stand scales focus on the degree to which differently sized trees spatially intermingle over a stand (Franklin and Van Pelt, 2004), while at a finer scale, variability in tree heights within patches of trees is often measured (Cooper, 1960; Mast and Veblen, 1999; White, 1985).

Several previous studies have shown that restoration treatments can be effective for meeting non-spatial forest structure restoration objectives and reducing potential fire behavior (Fulé et al., 2012; Harrod et al., 1999; Hudak et al., 2011). However, there is concern that the lack of spatially explicit reference conditions across the geographic range of dry-forest types lead to a reliance on non-spatial measures of forest structure such as trees per acre or basal area which do not account for the juxtaposition of trees or of their attributes. This could result in uniform implementation of treatments and lead to homogeneous stands with low structural complexity (Churchill et al., 2013b; Larson and Churchill, 2012; North et al., 2009; Underhill et al., 2014). In such cases, the resulting forest structure may not mimic desired conditions which are considered to consist of a spatially aggregated, or sometimes random, pattern of overstory trees within a matrix of canopy openings intermixed with individual trees, and various sized tree patches (Abella et al., 2007; Brown et al., 2015; Harrod et al., 1999; Larson and Churchill, 2012; Reynolds et al., 2013; Sanchez-Meador et al., 2011; Youngblood et al., 2004). These forests historically also demonstrated some degree of vertical complexity, as tree sizes de-coupled from ages owing to variable growing conditions (Mast and Veblen, 1999). Unfortunately, as Larson and Churchill (2012) pointed out in a review of spatial patterns in dry forests, the effects of restoration treatments on spatial aspects of forest structure remain largely undocumented. Some studies have been conducted in the Sierras (B. Collins, unpublished), northern Rocky Mountains (Larson et al., 2012), and the Southwest (Abella et al., 2006), though these have had limited geographic scopes and had not focused on effects of treatments on vertical complexity.

Wildland fires have been shown to have considerable fine scale spatial variability in their rate of spread and intensity because of heterogeneity in wildland fuels (Anderson, 1982; Cheney and Gould, 1995; Fernandes et al., 2000), as well as complex interactions between forest structure and the environment. Furthermore, local variations in fire behavior due to fine scale structural complexity in forest structure may result in non-uniform fire effects leading to a range of post-fire conditions (Thaxton and Platt, 2006). Much of the previous research evaluating effects of restoration treatments on fire behavior utilize non-spatial fire behavior models. While such modeling approaches can be useful in fire management planning, the lack of consideration of fuel heterogeneity or the interactions between the pattern of fuel, the wind, and the fire may limit applications of non-spatial fire behavior models (Hoffman et al., 2012; Linn et al., 2013; Parsons et al., 2011). More recently, several studies have used physics based fire behavior models such as the Wildland Urban Interface Fire Dynamics Simulator (WFDS; Mell et al., 2007, 2009) or FIRETEC (Linn et al., 2002). Results from these recent studies have suggested both non-spatial and spatial aspects of forest structure play an important role in determining fire behavior (Hoffman et al., 2012, 2015; Linn et al., 2013; Pimont et al., 2011). Despite the increasing evidence of the importance of structural complexity on fire behavior, no studies to date have directly measured and accounted for the spatial structures resulting from restoration treatments on potential fire behavior.

The overall goal of this study was to explore how aspects of structural complexity and potential fire behavior were changing, on balance, as a result of variable retention harvesting across a range of restoration treatments. We were interested in finding were treatments were having consistent effects and where treatments may provide mixed results. To meet this goal, we applied spatial statistics and spatially explicit fire modeling to field derived, stem mapped data from restoration treatments across the Colorado Plateau and southern Rocky Mountains. Three questions were investigated: First, how did non-spatial metrics of forest structure and fuels change?; Second, how did treatments alter vertical and horizontal structural complexity at stand and patch scales?; And third, how was potential fire behavior altered following treatment?

2. Methods

2.1. Study areas

We sampled seven sites that had been treated within the past 10 years across the Colorado Plateau and southern Rocky

Table 1
Restoration treatments and their two-letter site codes sampled in this study.

Treatment name	Site code	Treatment year	Location	Managing agency
PA5	HB	2012	Boulder Cty, CO	Boulder County Parks & Open Spaces
Lookout Canyon	LC	2003	Coconino Cty, AZ	Kaibab Nat'l Forest
Messenger Gulch II	MG	2012	Park Cty, CO	Pike Nat'l Forest
Phantom Creek I	PC	2010	Teller Cty, CO	Pike Nat'l Forest
Dowdy Lake	DL	2012	Larimer Cty, CO	Roosevelt Nat'l Forest
Unc Mesa	UM	2010	Montrose Cty, CO	Uncompahgre Nat'l Forest
Bluewater	BW	2010	Cibola Cty, CO	Cibola Nat'l Forest

Mountains (Table 1). All treatments occurred in forest stands which were considered to be overstocked and reduced ecosystem resilience compared to historical conditions. For these reasons treatments were implemented with the expressed goals of restoring forest structural patterns and potential fire behavior akin to historic ranges. A majority of sites were implemented under Collaborate Forest Landscape Restoration Projects (e.g. Underhill et al., 2014) and in addition, these sites' treatments all were considered demonstrations of ecologically-based forest restoration by their respective management agencies.

These restoration treatments were implemented using variable retention harvesting and were managed for multiple age classes. Specifications across treatments varied in terms of target ranges for basal area and trees per acre though all sought to increase the proportion of ponderosa pine (*Pinus ponderosa* Lawson). Aside from these specifications, common silvicultural prescriptive elements such as diameter caps, proportions of size classes to retain or remove were not specified. Rather, the emphasis was on reducing stand stocking in a manner congruent with historic patterns of structural complexity.

The guidelines emphasized a creation of openings and retention of distinct tree patches and residual individual trees. Prescriptions showed a preference for leaving as individuals, the largest trees or whose crown morphology suggested open-grown conditions in the past; leaving patches of 2–15 trees with preference for patches that display interlocking crowns and those with uneven sizes of trees; and opening sizes ranging from 1 to 4 tree lengths in diameter. Some prescriptions prohibited cutting within patches, while others gave leeway to cutting perceived ladder fuels. All guidelines gave focus towards patch scale structural complexity, mostly related to the horizontal dimensions, and half specifically mentioned aggregation as desired at the stand-scale. Guidelines did not focus on basing leave or cut tree decision making based on existing remnant material (i.e. pre-settlement trees, stumps, and logs) nor did any guideline reference site-level historic spatial reference conditions as aids. In all sites, cut-to-length or tree-length mechanized harvesting was done with unmerchantable crown material lopped and scattered. More information on the implementation strategy used for the majority of our sites has been described in Reynolds et al. (2013) and Underhill et al. (2014).

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2.2. Field sampling

In 2012 and 2013, from June through early August, a single 200 m × 200 m plot was surveyed, established and sampled within each treated site. Four hectare plots were used, as this is the maximal scale suggested to capture a distribution of patch scale elements (Larson and Churchill, 2012). Plots at each restoration treatment site were selected in consultation with local forest managers based on the criteria that plots must be relatively flat (i.e. less than 5% average slope), not contain complex topography and areas with over 20% slope within the plot and must have site conditions and stand structure typical of dry forest stands targeted for restoration. Any additional sources of heterogeneity (e.g. roads, streams, ridges, management unit edges, property boundaries) were avoided.

Each plot was divided into a grid with sixty-four 25 m by 25 m cells. Within each cell, all trees at least 1.37 m tall were mapped by measuring the azimuth, to the nearest 0.1° and distance, to the nearest 0.1 m, from grid intersections to each tree. We recorded species, tree height, diameter at stump height (DSH), diameter at breast height (DBH), crown radius, and crown base height. Crown radii were calculated as the average of two distances from tree bole to crown edge measured at random directions. We visually compacted lower limbs to determine crown base height (Toney and Reeves, 2009).

To reconstruct the initial forest structure, all stumps were mapped and had their species and DSH recorded. No activities, such as prescribed fire or harvesting equipment, removed stumps. In cases where equipment damaged stumps, we visually estimated the surface area of stumps using remnant material to estimate DSH. We used linear regressions for each species to estimate each cut tree's DBH, crown radius and crown base height based on DSH (Ziegler, 2014).

Fine surface fuels, consisting of downed, dead wood, aboveground herbaceous mass, and litter within each treated plot. We estimated fuel load of woody fuels less than 2.5 cm in diameter using methods from Brown (1974) from two 2 m planar intercept transects laid out in random directions from each of the 64 cell centers. We measured litter depth and collected litter within a 0.10 m^2 quadrats placed randomly in each cell following methods outlined by Ottmar and Andreu (2007). In those random locations, we also collected herbaceous fuel within a 1 m^2 quadrat after measuring the average height. The collected biomass was dried in an oven at 100 °C until mass equilibration was reached, generally for 24 h, to determine total surface fuel loading.

We repeated the surface fuels sampling method in an untreated plot adjacent to each treated plot. The design of these untreated plots mirrored those of the treated site. Untreated plots were selected in consultation with local forest managers with the requirements of similar biophysical conditions, species composition and stand structure to the treated plots prior to treatment.

2.3. Analysis of non-spatial stand structure and structural complexity

We tested effects of treatment on non-spatial stand structure variables with a mixed-effects (PROC GLIMMIX; SAS Institute Inc, Cary, NC, USA) model with status (before and after treatment) as a fixed effect and site as a random effect. Pairwise differences between treatment for basal area, trees per hectare, quadratic mean diameter, canopy base height, and total fine surface fuel load were determined using least squares means with a Tukey-Kramer adjustment. For all significance tests we used an α of 0.05.

Table 2
Framework for assessing structural complexity across scales and dimensions.

Scale	Method	What was examined
Horizontal dimension		
Stand	Pair correlation functions	Spatial pattern of plots and change in pattern following treatment
Patch	Spatial patch detection	Areal cover of individual trees, patches and openings; distribution of trees among patch sizes
Vertical dimension		
Stand	Height differentiation index	Spatial mingling of differently sized trees across each plot
Patch	Patch scale coefficient of variation of tree heights	Variability of tree heights among patches

We examined changes in both vertical and horizontal structural complexity at both the stand and patch scales using a combination of methods to evaluate changes due to treatment (Table 2). To assess effects on horizontal complexity at the stand scale we estimated the empirical pair correlation function before treatment, $g_{pre}(r)$, and after treatment, $g_{post}(r)$. Specifically, $g_{post}(r)$ was calculated from the locations of residual trees and $g_{pre}(r)$ was measured from both the locations of residual trees and trees removed during treatment. This spatial statistic is used to assess the degree of aggregation, in this study the attraction or repulsion of trees to other trees. Because the spatial statistic is normalized by the expectation of complete spatial randomness (CSR), a value of 1 indicates CSR while a value less than and greater than 1 indicate uniform and aggregated patterns, respectively (Wiegand and Moloney, 2004). To test for departures from CSR for both the $g_{pre}(\mathrm{r})$ and $g_{post}(\mathrm{r})$ we used a goodness of fit test developed by Loosmore and Ford (2006). This approaches compares the observed pattern to a null hypothesized spatial process model. In this study we generated a spatial processes model of CSR, by independently and identically redistributions of the points using a Poisson process 999 times. We tested for goodness of fit of the observed pattern to the simulated spatial process over a range of distances, r, from 0 m to 10 m. This analysis was performed in Programita (Wiegand, 2004) using Ripley's (1988) method of edge correction.

To test whether restoration produced more uniform or aggregated patterns than was present prior to treatment we examined the difference between the pre- and post-spatial pattern, $g_{post}(r) - g_{pre}(r)$, to a random labeling null model (Wiegand and Moloney, 2012). We utilized a random labeling null model given that the post-treatment pattern is conditioned on the pre-treatment pattern (Goreaud and Pelissier, 2012). The null hypotheses is therefor that the $g_{post}(r) - g_{pre}(r)$ will have a value of zero (Aakala et al., 2012; Goreaud and Pelissier, 2012). Indicating that the post-treatment pattern is effectively a random thinning of the pre-treatment pattern and therefore preserves the pre-treatment pattern. Alternatively a positive departure from zero indicate a shift towards greater aggregation and a negative departures indicate a shift towards uniformity. To test for departures from our hypotheses, $g_{pre}(\mathrm{r}) - g_{post}(\mathrm{r}) = 0$, we again utilized the goodness of fit test developed by Loosmore and Ford (2006). However for this test the hypothesized spatial process model was generated using the random labeling hypothesis, where 999 simulations were produced by random relabeling of residual trees and cut trees. We tested for goodness of fit of the observed pattern to the simulated spatial process over a range of distances, r, from 0 m to 10 m. This analysis was performed in Programita (Wiegand, 2004) using Ripley's (1988) method of edge correction.

Patch scale spatial patterns were characterized using a patch detection algorithm to identify the elements of individual trees, tree patches and openings (Clyatt et al., 2016; Plotkin et al., 2002; Sanchez-Meador et al., 2011). We determined individual patches by identifying all trees with overlapping crowns, based on measured crown radii, and then joining the contagious overlapping sets of trees to form each unique patch. We then determined the proportion of trees in the following patch size classes: small (2–4 trees), medium (5–10 trees), large (11–20 trees), and patches of continuous cover (21+ trees) (Churchill et al., 2013a; Sanchez-Meador et al., 2011). Similar to several previous studies we elected not to incorporate an edge correction factor (Clyatt et al., 2016; Plotkin et al., 2002; Churchill et al., 2013b; Larson et al., 2012). Several previous studies have shown that there is no advantage in using edge corrections when only a description of the pattern is reported as done here, however the use of edge correction factors is an important consideration when testing for departure from a null hypothesis (Wiegand and Moloney, 2012; Churchill et al., 2013b; Plotkin et al., 2002; Yamada and Rogerson, 2003). Restoration treatment effects were assessed by descriptively evaluating changes in the proportion of trees in each of the size classes.

We measured vertical complexity at the stand scale using the height differentiation index (TH) (Kint et al., 2000) and by estimating the coefficient of variation of tree heights at the patch scale. The height differentiation index estimates the dissimilarities between each tree and its three nearest neighbors for all trees across the stand such that values range from 0 to 1 with higher values reflecting greater dissimilarities (Gadow and Hui, 2002; Kint et al., 2000; Saunders and Wagner, 2008). At the patch scale, we calculated the coefficient of variation of tree heights (CV_{HT}) for each individual patch. We then compared the medians of TH and CV_{HT} of before to after treatment, by site, with a Wilcoxon rank sum test using R v2.15.2 (R Development Core Team, 2010).

2.4. Fire modeling background

WFDS is a fire behavior model developed by the National Institute of Standards and Technology and the US Forest Service Pacific Northwest Research Station. WFDS employs computational fluid dynamics methods to solve the spatial and temporal evolution of fire using a three dimensional numerical grid. This approach allows for representation of fuels and prediction of fire behavior through time in three dimensions. A more detailed description of the physical and mathematical formulations in WFDS can be found in Mell et al. (2007, 2009). WFDS is an extension of FDS, the Fire Dynamics Simulator (McGrattan et al., 2016a). Verification and evaluation studies of FDS and WFDS are described in Hoffman et al. (2015), McGrattan et al. (2016a, 2016b), Mell et al. (2007, 2009) and Mueller et al. (2015).

2.5. Fire behavior simulations

Potential fire behavior for each site pre- and post-treatment was simulated using WFDS under 4 different open wind speeds. All WFDS simulation used a simulation domain measuring 1000 m × 400 m × 100 m in the x, y and z dimensions respectively, discretized as 1 m cubic cells in the x and y dimensions with a vertically stretched discretization of cells in the z dimension that ranged from 0.5 m at $z = 0$ m to 2.0 m at $z = 100$ m (Fig. 2). The overall domain was divided into five parts: a wind entry field at x = 0, an exiting wind field along x = 1000 m, a wind field development zone from x = 0 to x = 700, an area of interest from x = 700 to x = 900, y = 100 to y = 300, and the outflow zone from x = 900 to x = 1000. The entering wind flow was simulated following an atmospheric power law function with four different 10-m wind speeds, 2.2 m s^{-1}, 4.0 m s^{-1}, 9.0 m s^{-1}, and 13.4 m s^{-1}. The exiting wind flow boundary was simulated with an open condition, while the two lateral boundaries ($y = 0$ m and $y = 400$ m) and the ceiling ($z = 100$ m) were simulated as free-slip, no-flux walls. We assumed neutral atmospheric conditions.

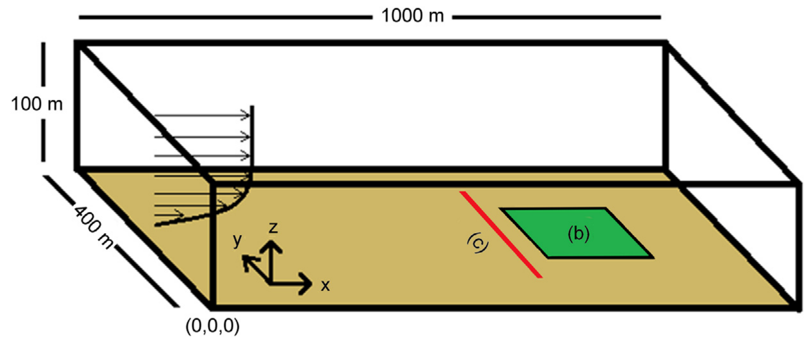

Fig. 2. Simulation set-up for fire behavior modeling with (a) wind inflow, (b) the area of interest, and (c) the fireline origin.

We input the measured stem-maps for each site, before and after treatment, in the area of interest. To ensure that the wind field approaching our zone of interest was well developed and representative of typical forest flow we populated the rest of the domain with analogous forests created from rotations and reflections of each site's stem map.

Individual tree crowns were simulated as right circular cones with properties of a surface area/volume ratio of 4000 m^{-1} (Brown, 1970; Mell et al., 2009; Scott and Burgan, 2005), bulk density of 1.2 kg m^{-3}, drag constant of 0.125 and foliage moisture set to 100%. Surface fuels were modeled as a homogeneous fuel layer defined by each site's mean fine fuel load and mean fuelbed depth, a surface area/volume ratio of 5710 m^{-1} (Brown, 1970) and fuel moisture of 5%. Coarse woody fuels were not accounted for as WFDS assumes that all fuels are thermally thin and that thermally thick fuels do not significantly contribute to fire propagation (Anderson, 1969; Mell et al., 2009). The heat of combustion of all fuels was set to 17,700 kJ kg^{-1} following Mell et al. (2009). To initiate a fire in the simulations we ignited a 300 m long fire at x = 650.

2.6. WFDS output and analysis

We estimated the mean fire rate of spread, fireline intensity, and canopy consumption within the area of interest for each site, before and after treatment, for each wind scenario to assess fire behavior. The fire rate of spread and fireline intensity were estimated as the mean from ten separate 1 m by 200 m belt transects spaced 20 m apart in the *y* dimension across the area of interest. Within each belt transect the fire rate of spread was estimated by calculating the distance traveled in the x-dimension every second and then averaging across all time steps and transects. Similarly, we estimated fireline intensity by summing the total heat release rate on each transect every second and averaging across all time steps and transects. Canopy consumption was calculated as a ratio of the post-fire canopy mass over the initial canopy mass within the area of interest.

We used a mixed-effects (PROC GLIMMIX; SAS Institute Inc, Cary, NC, USA) three-way analysis of variance with status and 10-m wind speed (m/s) as fixed effects and site as a random effect to examine the treatments' effects. Pairwise differences between statuses and wind speed levels of the three fire behavior parameters were determined using least squares means with a Tukey-Kramer adjustment.

3. Results

3.1. Effect on stand structure and structural complexity

Treatments significantly reduced trees per hectare (p = 0.002) and basal area (p = 0.001) (Table 3). These tree attributes were respectively 2.3 and 1.8 greater before than after treatment. We did not detect significant changes in quadratic mean diameter (p = 0.524), canopy base height (p = 0.232) or surface fuel loading (p = 0.247) following treatment. Six of the seven sites were dominated by ponderosa pine (*Pinus ponderosa* Lawson) and one site (PC) was more mixed, dominated by Douglas-fir (*Pseudotsuga menziesii* var. *glauca* [Mirb.] Franco). Composition did not change dramatically following treatment, though a preference for retaining ponderosa pine and quaking aspen is evident.

Table 3
Effects of restoration treatments on non-spatial stand structure and species composition, with significant changes in stand structure denoted in bold.

Site name	Status	Species (% of TPH)	**TPH**	**Live BA** (m^2/ha)	QMD (cm)	CBH (M)	SFL (kg/m^2)
HB	Pre	98 PP, 2 RM	418	19.1	24.1	3.5	1.0
	Post	98 PP, 2 RM	295	14.5	25.0	3.5	0.8
LC	Pre	100 PP	487	25.6	25.9	3.7	0.4
	Post	100 PP	203	19.7	35.2	6.5	0.3
MG	Pre	95 PP, 5 DF	685	22.7	20.5	3.1	0.4
	Post	97 PP, 3 DF	269	10.6	22.4	2.6	0.6
PC	Pre	20 PP, 54 DF, 7 AS, 19 SP	934	24.8	18.4	2.1	1.3
	Post	32 PP, 41 DF, 17 AS, 10 SP	352	9.5	18.5	2.5	1.3
DL	Pre	95 PP, 4 DF, 1 AS	516	14.1	18.7	2.1	1.2
	Post	97 PP, 3 DF	181	9.0	19.0	2.9	0.6
UM	Pre	87 PP, 6 AS, 7 SP, 1 GO	504	30	27.5	4.6	1.2
	Post	87 PP, 7 AS, 6 SP, 1 GO	314	17.6	26.7	4.9	1.1
BW	Pre	81 PP, 8 RM, 11 PI	327	19.3	27.4	4.6	0.2
	Post	88 PP, 3 RM, 9 PI	63	7.5	38.9	4.7	0.2

Pre, pre-treatment; post, post-treatment; TPH, trees per hectare; BA, basal area; QMD, quadratic mean diameter; CBH, canopy base height; SFL, surface fuel loading; Species include RM (*Juniperus scopulorum* Sarg), SP (*Picea engelmanii* Parry ex Engelm. and/or *Picea pungens* Engelm.), PI (*Pinus edulis* Englm), PP (*Pinus ponderosa* Lawson), AS (*Populus tremuloides* Michx), DF (*Pseudotsuga menziesii* var. *glauca* [Mirb.] Franco), and GO (*Quercus gambelii* Nutt.), and may not add to 100% due to rounding.

On initial visual inspection, all of the treatments appeared to have produced an aggregated pattern of trees composed of a mosaic of patches and openings, though some patterns of implementation are nuanced (Fig. 3). At sites HB and UM, most of the locations of stumps are near other stump locations, having carved out patches which do not appear to be cut from within; cuttings at MG and PC occur throughout areas of high tree density, resulting in residual mixes of smaller patches and single trees. Tree removal at BW and LC occurred in areas of highest density, reducing spatial gradients of tree density at the stand scale; conversely removals at DL are concentrated, creating a second large opening and a greater tree density gradient.

Analysis of horizontal forest structural complexity at the stand scale revealed that aggregation, across the 0–20 m range tested, was the dominant spatial pattern before treatment, $g_{pre}(r)$, with the exception of site HB which exhibited a uniform spatial pattern (Table 4). Following treatment, all sites were classified as having an aggregated spatial pattern. However, comparisons of the post-treatment to pre-treatment pair correlations function ($g_{post}(r) - g_{pre}(r)$) suggest that treatments did not necessarily result in increased levels of aggregation (Table 4). The post-treatment spatial patterns on four sites (HB, MG, PC, and DL) had increased levels of aggregation, while two sites (LC, UM), decreased levels of aggregation. One site (BW) showed no change in aggregation following treatment (Table 4).

All treatments resulted in a greater proportion of openings and a reduced proportion of patch cover (Table 5). The effect on the areal coverage of individual trees was mixed across the sites with both increased and decreased coverages (Table 5). However, the percent of trees classified as individuals increased on all sites. The proportion of trees within continuous cover patches (>20 trees) and large patches (11–20 trees) were reduced on all sites,

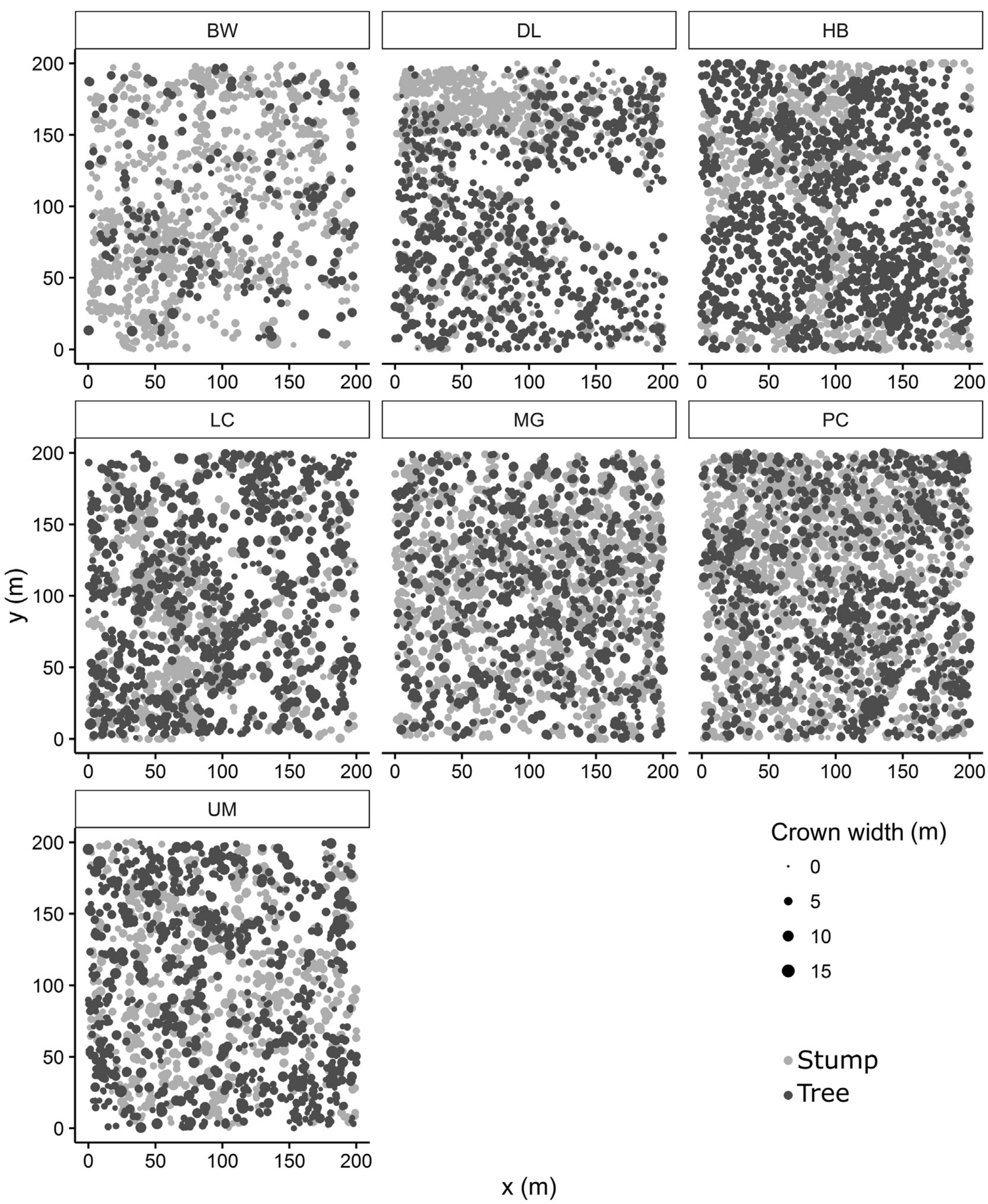

Fig. 3. Stem-maps of sampled sites before and after restoration treatments with crown width not to scale.

Table 4
Pre-treatment ($g_{pre}(r)$), post-treatment ($g_{post}(r)$), patterns from a distance, r, of 0 m to 10 m, either uniform (unif.) or aggregated (agg.) and their differences with p-values describing deviation from a random pattern and of no difference in patterns ($g_{post}(r) - g_{pre}(r)$).

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	More unif.	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	More unif.	0.026
BW	Agg.	0.001	Agg.	0.001	None	0.138

Table 5
Patch scale elements of stem-mapped sites pre- and post-restoration treatment.

Site	HB		LC		MG		PC		DL		UM		BW	
Status	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post	Pre	Post
Areal cover (%)														
Individual 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
Percent of trees belonging to individual trees and patch sizes														
Individual tree	21	27	13	25	10	28	13	36	16	51	11	14	27	58
Small (2–4 trees)	35	38	19	36	24	37	20	35	19	24	18	28	43	42
Medium (5–9 trees)	26	25	17	30	27	25	17	16	18	12	17	27	19	0
Large (10–19 trees)	15	11	20	10	20	6	20	8	19	12	20	15	11	0
Continuous cover (20+ trees)	3	0	32	0	19	4	31	5	28	0	35	16	0	0

whereas the proportion of trees within the smallest sized patches (2–4 trees) increased or remained the same. Treatments had mixed effects on the proportion of trees located in medium sized patches (5–10 trees).

Effects on stand scale vertical complexity were mixed across sites (Fig. 4a). The median height differentiation decreased on three sites (BW, $W = 206{,}020$, $p < 0.001$; DL, $W = 951{,}840$, $p < 0.001$; and LC, $W = 350{,}590$, $p < 0.001$), showed no change on two sites (PC, $W = 892{,}830$, $p = 0.436$; and UM, $W = 385{,}150$, $p = 0.272$) and increased on two sites (HB, $W = 329{,}240$, $p = 0.005$ and MG, $W = 522{,}710$, $p < 0.001$). Although treatments showed no consistent effect on the median stand scale vertical complexity the interquartile range of the height differentiation index increased on all sites following treatment.

Effects on patch scale vertical complexity were also inconsistent across sites (Fig. 4b). Patch scale vertical complexity increased on one site (MG, $W = 33{,}889$, $p = 0.001$), decreased on three sites (BW, $W = 7432$, $p = 0.004$; DL, $W = 21{,}836$, $p < 0.001$; and LC, $W = 25{,}088$, $p < 0.001$) and showed no change on the other three sites (UM, $W = 19{,}113$, $p = 0.330$; HB, $W = 38{,}026$, $p = 0.155$; and PC, W = 40,163, $p = 0.611$). Unlike our measure of stand scale of vertical complexity, changes in the interquartile range were not

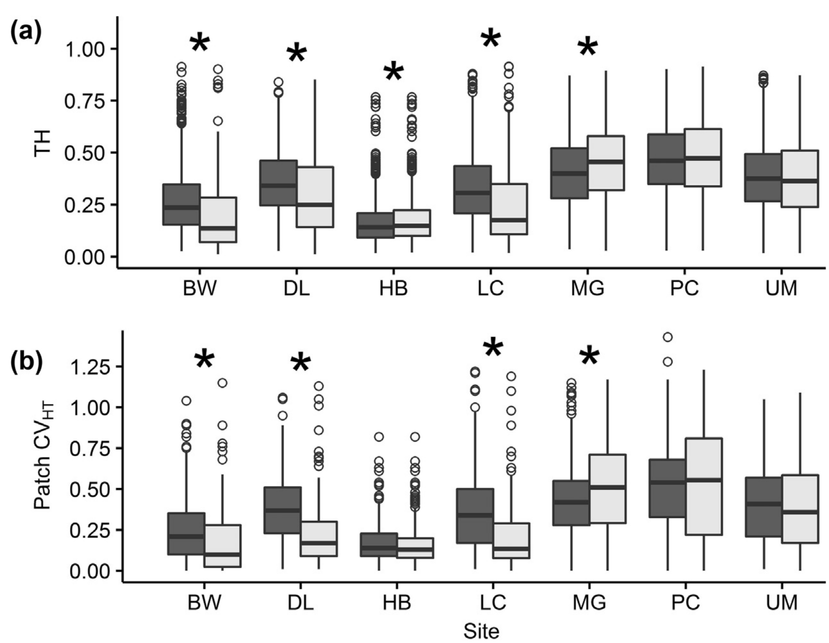

Fig. 4. Tukey boxplots of vertical complexity at stand (a; height differentiation index (TH)) and patch (b; coefficients of variation of patches' tree heights) scales among each site before (dark grey) and after (light grey) restoration treatments with asterisks noting significant differences in median complexity due to treatment.

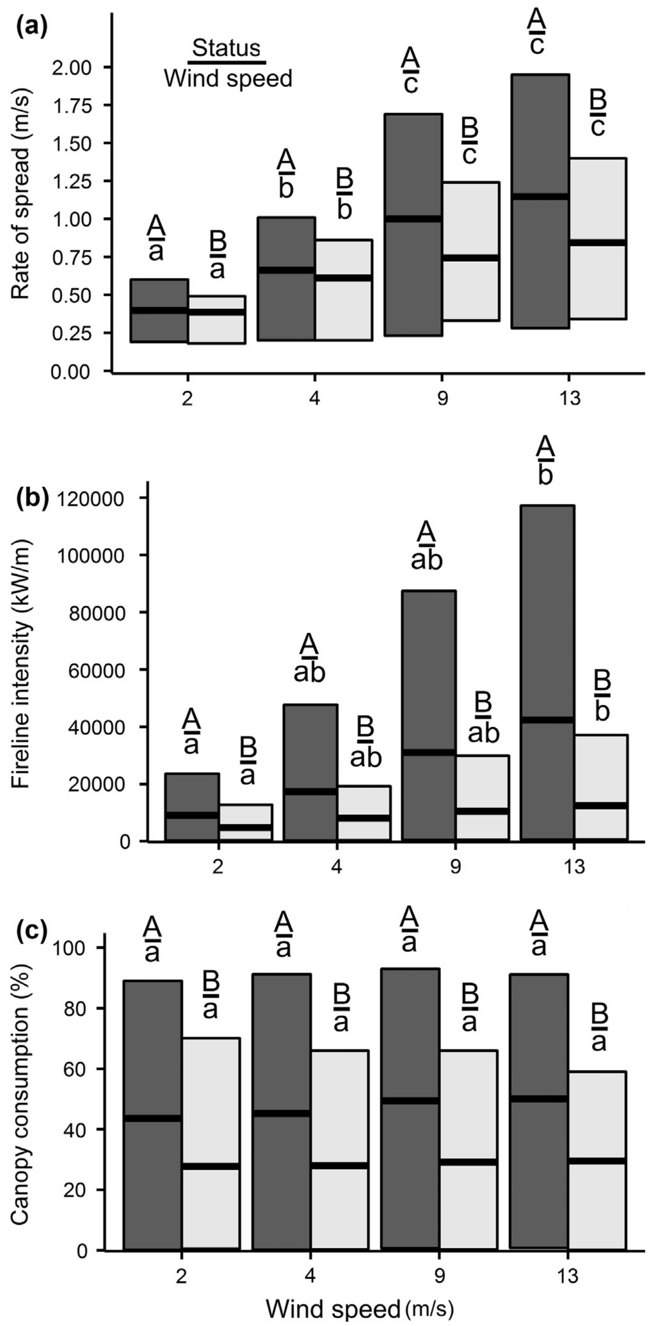

Fig. 5. Measures of modeled fire behavior, rate of spread (a), fireline intensity (b), and canopy consumption (c), before (dark grey) and after (light grey) restoration treatment with means, by wind speed, the crossbars and letters denote significant pairwise grouping difference by status, pre- or post-treatment (uppercase) and by wind speed level (lowercase).

consistent across sites, though we observed that as median patch scale vertical complexity increased so too did the interquartile range.

3.2. Effects on simulated fire behavior

Treatments reduced all measures of fire behavior across all wind speeds tested. Fire rate of spread was lower after treatment for all wind speeds ($p = 0.007$), with an average reduction of 24%. Fire rates of spread significantly increased with higher levels of wind speed ($p = 0.001$), though with no difference between 9 m/s and 13 m/s win speed (Fig. 5a). Similarly, treatments reduced fireline intensity across all wind speeds ($p = 0.009$), by an average of 177% (Fig. 5b), though pairwise comparisons reveal wind speed was less of a discriminatory factor. In addition, our simulations suggest that the proportion of canopy fuel consumed was reduced by 64% ($p < 0.001$) following treatment (Fig. 5c), however canopy consumption ($p = 0.670$) did not significantly change at any wind speed levels.

4. Discussion

In this study, we evaluated the effect of forest restoration treatments in dry forest types of the Colorado Plateau and southern Rocky Mountains on non-spatial stand structure, structural complexity and fire behavior. Specifically, we stem-mapped multiple sites where restoration treatments were implemented and then (1) examined changes in traditional, non-spatial measures of stand structure following treatment, (2) used spatially explicit statistics to examine changes in structural complexity in the vertical and horizontal dimensions at stand and patch scales, and then (3) used the stem-maps to model fire behavior over a range of wind scenarios using WFDS, a spatially explicit physics based fire model.

We specifically sought out to evaluate the primary objectives of these treatments, which are the primary emphasis of most restoration treatments (North et al., 2007). Other factors may influence the success of restoring the functions of natural processes. For example, it has been advocated that restoration treatments should account for soil pedon heterogeneity (Abella et al., 2013) and wildlife utilization among patches, large trees, and openings (Allen et al., 2002), minimize ground disturbances (Allen et al., 2002), and ensure adequate understory response (North et al., 2009), among others. Although we did not assess these factors further research that investigates a broader suite of forest restoration goals is needed.

Furthermore, our results evaluated a single entry. As suggested by Reynolds et al. (2013) the process of forest restoration may require multiple entries and reintroduction of fire to meet broad goals, yet little information currently exists on the long-term temporal dynamics of forest structure and processes following restoration treatments.

4.1. Treatment effects on non-spatial measures of stand structure

We observed that treatments effects on non-spatial stand structure were typical of fuel hazard reduction treatments in some regards and atypical in others. Tree densities, basal areas and canopy fuel loads were all decreased consistent with expectations of thinning or harvesting and in fuel hazard reduction in particular, as found by others (Agee and Skinner, 2005; Churchill et al., 2013a; Covington et al., 1997; Fulé et al., 2012; Hudak et al., 2011; Stephens et al., 2009). However, we did not find an increase in CBH contradicting the results reported by Fulé et al. (2012) and violating one of the principle objectives in fuel hazard reduction treatments outlined by Agee and Skinner (2005). The lack of effect on CBH across our sites may reflect an emphasis on retaining or cutting trees based on their relative positions rather than emphasizing preferential selection of smaller diameter trees as is commonly suggested for fuels hazard reduction (Agee and Skinner, 2005; Churchill et al., 2013b). In addition, we found that surface fuel loadings were not different following treatment across our sites which contradicts results from several previous studies that have found increased surface fuel loads following mechanical cutting operations (Fulé et al., 2012; Reinhardt et al., 2008; Schwilk et al., 2009). These differences are possibly due to either various harvesting systems utilized and their relative impacts on activity fuels (Stephens et al., 2009) or from delayed sampling. Further work is required to fully understand how various harvesting systems in restoration treatments impact surface fuel loads and the implications of these impacts on potential fire behavior. Overall, our findings suggest that restoration treatments using variable retention of tree patches and single trees may produce slightly different non-spatial stand structures than typical fuel hazard reduction treatments may produce.

Looking back, the approach of stem-mapping and making measurements after treatments was efficient at capturing tree locations. That we found numerous sapling stumps less than 2.54 cm in diameter gives us confidence in capturing the stand structure before treatment. Inevitably, evidence of some cut saplings may have been missed which likely would have led to a more aggregated pattern pre-treatment, possibly most vertical height differentiation and, if these saplings served as ladder fuels, greater pre-treatment fire behavior. Trading space for time to measure pre-treatment surface fuels may not have captured treatment effects as well as on-site measurements pre-treatment as surface fuels can be highly spatially variable (Vakili et al., 2016).

4.2. Effects on structural complexity

Our evaluation of horizontal and vertical structural complexity, how trees were distributed across space and the juxtaposition of their heights, showed that restoration treatments could reproduce structural complexity qualitatively similar to historic, dry forests even without use of site-specific reference conditions. All post-treatment stands were best described as having an aggregated spatial pattern of trees consisting of a matrix of individual trees, clumps and openings, similar to descriptions of historical dry forests across the western U.S. (Abella and Denton, 2009; Clyatt et al., 2016; Harrod et al., 1999; Larson and Churchill, 2012) and specifically within the regions we sampled (Boyden et al., 2005; Brown et al., 2015; Tuten et al., 2015).

We found that restoration treatments did not always increase the amount of aggregation compared to the initial spatial patterns and in some cases reduced the level of aggregation. Although these results seem to contradict the overall objectives to restore or maintain structural complexity they should not be interpreted as a failure of restoration treatments. Several previous studies have suggested that contemporary spatial patterns may be more or less aggregated than the historical conditions at a given location (Larson et al., 2012; North et al., 2007; Tuten et al., 2015; White and Vankat, 1993) and therefore treatments need not increase the level of aggregation in a given stand but rather seek to produce a range of conditions that mimic the historical range of variability in spatial patterns across a landscape.

Furthermore, these treatments resulted in a range of distributions of individual trees and patches that are within the reported historical range of variability for similar ecosystems (Churchill et al., 2013b; Clyatt et al., 2016; Sanchez-Meador et al., 2011). Given the lack of spatially explicit targets within our study area, it is difficult to determine how closely these treatments mimic historical site-specific conditions. Again, as suggested by Larson and Churchill (2012), the exact replication of specific reference patterns may not be as important as ensuring that a broader characteristic aggregated pattern is present to maintain intact pattern-process linkages.

We found no regular effect of restoration treatments on vertical structure and that a wide diversity of vertical complexity existed on these sites before and after harvesting. Given that silvicultural prescriptions for our sites did not directly address manipulation of vertical structure or desired size class distributions, the lack of a consistent effect of treatments was not surprising.

What specific range of vertical complexity is desirable for restoring vertical complexity is unknown. It is generally agreed upon that some level of stand-scale vertical complexity in restoring natural patterns of structure is valued (Franklin and Van Pelt, 2004; Graham et al., 2006; North et al., 2009; Reynolds et al., 2013; Youtz et al., 2008), though these studies' mixed recommendations regarding patch vertical complexity as well as differing interpretations of historic patch complexity (e.g. Cooper, 1961; White, 1985) lead to contrasting viewpoints and competing operational guidelines for forest restoration within patches (Tuten et al., 2015). One point of agreement is that smaller trees were likely more infrequent than today (Brown and Wu, 2005; Fulé et al., 1997; Kilgore, 1981) due to greater susceptibility of small trees to fire (Battaglia et al., 2009). This would have likely reduced vertical complexity by reducing potential juxtapositions of small and large trees. In this regard, managing for high levels of vertical complexity may not be appropriate and preferential removal of ladder fuels, while retaining a mixture of moderate- to larger-sized trees (Agee and Skinner, 2005; Churchill et al., 2013b), may produce a more moderate degree of vertical complexity analogous to historic forests. We recommend future reconstructions of the natural range of variation in dry forests explore the spatial distribution of variously sized trees to further inform ecological restoration projects.

Last, we note that despite inconsistent alterations to median vertical complexity, there was always more fine-grain variability in vertical complexity at the stand scale, and sometimes at the patch scale following treatments. While managers are not creating more complex structures at the average location, they are creating more variability in complexity to a degree. This is an attribute that has been observed to distinguish uneven-aged from even-aged stands, which have less variability in vertical complexity (Motz et al., 2010). Given the importance of vertical complexity for a variety of ecological processes such as wildlife habitat and stand dynamics (Franklin and Van Pelt, 2004; MacArthur and MacArthur, 1961; Murdoch et al., 1972; Peck et al., 2014), a conservative approach of creating a wider range of conditions may be a hedge against undesirable disturbances and change agents. Further research that quantifies the potential tradeoffs of lower versus higher vertical complexity among various perspectives, objectives and ecological processes could help advance restoration treatment design.

4.3. Effects on simulated fire behavior

Prior to this study, few studies have explicitly accounted for structural complexity during assessments of restoration treatments and questions remain regarding the potential impact of various forest structures on potential fire behavior (Fulé et al., 2009). Yet, previous studies have suggested that structural complexity can influence potential fire behavior by altering local thresholds for crown fire ignition and spread, and the within canopy wind speed and turbulence (Hoffman et al., 2012; Pimont et al., 2011).

Our results indicate that the restoration treatments resulted in a decreased fire rate of spread, fireline intensity and canopy fuel consumption compared to pre-thinned forests across all wind scenarios tested. Although we were not able to directly compare our fire behavior predictions to historical data in this area, simulated canopy consumption suggest that following treatment these forests exhibit low to moderate levels of fire severity (20–35% canopy consumption) which are in line with several estimates of historic fire severity in these systems (Fule et al., 2014; Sherriff et al., 2014). The simulated rates of spread from these simulations and other physics-based fire behavior models have shown reasonable agreement with published fire behavior data (Hoffman et al., 2015) and are within the range (<1000 kW m^{-1} for surface fires and between 30,000 and 150,000 kW m^{-1} in crown fires) of fireline intensities that have been reported in the literature (Alexander and Cruz, 2016; Kiil and Grigel, 1969; Stocks et al., 2004). These results provide additional support to a growing body of literature that has suggested treatments do result in reduced fire behavior (Fulé et al., 2012).

In brief, our fire behavior simulations indicate that thinning or harvesting for restoration purposes need not result in an evenly spaced pattern of residual trees, common in traditional fuels reduction treatments (Larson and Churchill, 2012), to achieve significant reduction in potential fire behavior. However, further research that assesses the potential tradeoffs associated with various horizontal and vertical spatial patterns is needed to better inform the design of restoration treatments.

4.4. Conclusion

Hessburg et al. (1999) suggested that restoration treatments should result in a variety of target stand structures that reflect the diversity of reference spatial patterns across a landscape. Ideally, the variety in stand structures created through restoration would also result in a distribution of potential fire behavior and severity that reflect the historical range of variability. Despite a lack of documented reference spatial patterns in our study area, our results suggest that the range of structural complexities created by these treatments are emulating the range of stand structural reference conditions reported for dry conifer forests of the western U.S. Furthermore, our fire behavior simulations demonstrate that restoration treatments reduce potential fire behavior.

As restoration treatments continue to be implemented across dry forests, our findings can be used to generate further null hypotheses on how exact treatment implementations might alter structure and what those changes might mean for processes such fire behavior or other forest resources. While we specifically sought out to evaluate the primary objectives of these treatments it is important to recognize that a variety of goals may be important within forest restoration (North et al., 2007). For example, it has been advocated that restoration treatments should account for soil pedon heterogeneity (Abella et al., 2013), wildlife utilization among patches, large trees, and openings (Allen et al., 2002), minimize ground disturbances (Allen et al., 2002), and ensure adequate understory response (North et al., 2009), among others. Given the large number of potential objectives that need to be simultaneously considered in the design of restoration treatments further research that quantifies the potential tradeoffs among various objectives and ecological processes could help advance restoration treatment design.

Ultimately, our results indicate that current restoration treatments implementing variable retention harvesting in the Colorado Plateau and southern Rocky Mountains are at least qualitatively meeting non-spatial and spatial stand structure objectives while simultaneously reducing potential fire behavior.

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