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Stand Dynamics of Pinyon-Juniper Woodlands After Hazardous Fuels Reduction Treatments in Arizona☆

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ABSTRACT

Pinyon-juniper ecosystems occur extensively across western North America, and at the landscape scale, variation in structure and composition is influenced by topographic position, soils, disturbance history, and local climate. The persistent pinyon-juniper woodland is a common structural form, and though they are known to be infrequent-fire systems, there is increasing interest in implementation of hazardous fuels reduction treatments in woodlands, especially in the wildland-urban interface. Few studies have quantified stand dynamics following fuels reduction treatments in persistent woodlands or compared treatment outcomes to conditions that develop under natural disturbance and successional processes. In 2004, we established a randomized, replicated study in woodlands of northern Arizona, and monitored stand dynamics and understory responses to determine how stand-level changes differed between common fuels reduction approaches. We compared the resulting structure with a conceptual state-and-transition model. Results showed that, over the 11 yr after treatment, juniper tree densities decreased by 8.4% and 0.9% but increased by 14.0% and 27.3% in Control, Burn, Thin, and Thin + Burn treatments, respectively. Pinyon tree densities decreased by 1.1% and 3.3%, increased by 12.2%, and decreased 7.9% in Control, Burn, Thin, and Thin + Burn treatments, respectively. All treatments showed fuel load reductions throughout the 11-yr study period and minimal rebound of tree recruitment toward pretreatment conditions. Prescribed fire alone (Burn) maintained persistent woodland conditions. Thinning treatments substantially reduced small tree densities and, with the addition of prescribed fire, produced losses of large trees. Thinning with prescribed fire (Thin + Burn) tended to produce conditions qualitatively unlike those described by our state-and-transition model. Evaluation of these commonly used fuels treatments against our state-and-transition model suggested that concerns regarding loss of ecological integrity may be warranted.

Introduction

Pinyon-juniper ecosystems occur extensively across western North America and are found on about 30 M ha in the United States (West, 1999). At a subcontinental scale, species composition, structural characteristics, and historical disturbance regimes are thought to correspond to a seasonal precipitation gradient extending from the Pacific Northwest and western California to southern New Mexico and western Texas (Romme et al., 2009). Much of the distribution of the pinyon-juniper type is found in the Great Basin and Southwest regions, and pinyon-juniper woodlands and savannas comprise over half the forested land in the states of Arizona and New Mexico (Pearson, 1931; Gottfried, 2004; Romme et al., 2009). At the landscape scale, variation in structure and composition is influenced by topographic position, soils, disturbance history, and local climate (Jacobs et al., 2008; Poulos et al., 2009). For example, persistent woodlands occur across the Colorado Plateau on fire-protected sites and where production of fine fuels is limited. In northern Arizona, persistent Utah juniper—Colorado pinyon pine *(Junipeus osteosperma—Pinus edulis)* woodlands are found on upland sites, commonly adjacent to ponderosa pine *(P. ponderosa)* forests. Several studies have concluded that infrequent stand-replacing fire regimes characterized these ecosystems before Anglo-American settlement of the region (Floyd et al., 2000; Huffman et al., 2008; Kennard and Moore, 2013). Long, fire-free periods allow development of uneven-aged stands, with relatively high canopy cover and numbers of "old" trees, and sparse understory plant cover (Romme

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et al., 2003). In contrast, wooded shrublands are thought to be areas of tree expansion and contraction occurring on sites that are favorable to shrub development (Romme et al., 2009). Wooded shrublands are thought to be more open in structure than persistent woodlands, and natural disturbances in this type are severe but not extensive. Mixed-severity fire in wooded shrublands is thought to result in variable tree densities and complex, dynamic structural characteristics. Finally, pinyon-juniper savannas are more open than wooded shrublands and are characterized by widely spaced trees and relatively high grass cover in intervening spaces. Unlike persistent woodlands, structural conditions of pinyon-juniper savannas are thought to be maintained by high-frequency, low-severity surface fire (Margolis, 2014).

High-severity crown fire in dense woodlands and recent tree increases in savannas, as well as expansion of trees into grasslands and shrublands, have prompted ecological restoration and hazardous fuels reduction programs in pinyon-juniper systems (Jacobs and Gatewood, 1999; Huffman et al., 2009; Redmond et al., 2014). Restoration and fuel-reduction treatments are typically designed to address conditions that have resulted from fire exclusion in frequent-fire forests (Agee and Skinner, 2005). Approaches commonly include thinning trees that likely would not have established under a frequent-fire regime and reintroducing surface fire through prescribed burning. Frequent-fire forests following restoration and fuel treatments are intended to be more open with lower canopy cover and have fewer small, young trees; lower surface fuel loading; and increased herbaceous ground cover (Covington et al., 1997; Moore et al., 1999; Schwilk et al., 2009). Aspects of these treatments, however, may be ineffective for reducing fire behavior in infrequent-fire types (e.g., persistent woodland), or may target key structural features and result in negative impacts on ecological function (Brown et al., 2004). For example, thinning for restoration and fuel reduction commonly targets small trees (i.e., "ladder fuels") for removal in order to increase crown base height and lower the potential for transition of surface fire into the canopy (Agee and Skinner, 2005). However, persistent woodlands may exist for centuries without major disturbance, and tree size distributions often follow a negative exponential ("inverse-j") form (Huffman et al., 2013). In these ecosystems, removal of small trees could create novel structural conditions with minimal resemblance to forms developing from natural disturbance (Romme et al., 2003; Huffman et al., 2009). Although there is increasing interest in implementation of hazardous fuels reduction treatments, especially in wildland-urban interface (WUI) areas, more information is needed to better understand long-term woodland responses to treatments and evaluate outcomes against those produced by natural disturbances.

Utilization of strategies that mimic patterns of natural disturbance and management of ecosystems within natural ranges of variation have been suggested as approaches to better conserve species, habitat, and critical ecological functions (Holling and Meffe, 1996; Landres et al., 1999; Lindenmayer et al., 2006). Conceptual state-and-transition models that describe changes in ecological characteristics over time and in response to disturbance can be useful for evaluating management actions (Bestelmeyer et al., 2004; Suding and Hobbs, 2009). Several authors have described state-and-transition models for pinyon-juniper ecosystems (Jameson, 1994; Davenport et al., 1998; Holmgren and Scheffer, 2001; Gori and Bate, 2007). These models predict a range of responses to natural disturbances and expected response pathways depending on type, intensity, and time since disturbance. For example, major fires that occur infrequently (> 100 yr natural fire rotation) in persistent woodlands are commonly severe and result in open patches of early successional species (Fig. 1). Recovery of structural characteristics and complexity after stand-replacing fire may require long periods of time, from several decades to more than a century (Arnold et al., 1964; Erdman, 1970; Huffman et al., 2012). In contrast, in the absence of fire, savanna systems may transition over time toward closed woodlands and cross-environmental thresholds that favor tree cover over success of understory species (Davenport et al., 1998; Jacobs, 2015) (see Fig. 1). Focal attributes distinguishing stable states in these models typically include degree of canopy cover, abundance of "old" trees, relative importance of young trees, and understory composition and development (e.g., Romme et al., 2003, 2009) (see Fig. 1). Nonlinear and nonequilibrium responses are recognized, and woodlands may not recover to initial conditions if physical parameters or feedback processes are altered (Jameson, 1994; Tausch, 1999; Briske et al., 2008). To date, few studies have quantified stand dynamics after fuel reduction treatments in pinyon-juniper woodlands or interpreted outcomes in terms of ecological states and transitions occurring in response to natural disturbances and succession. This information would allow land managers to better evaluate these treatments for their effectiveness in accomplishing both fuel reduction and conservation objectives.

In this study, we remeasured hazardous fuel reduction treatments in persistent pinyon-juniper woodlands on the Kaibab National Forest in northern Arizona. Thinning and prescribed burning were applied, both respectively and in combination, in 2005 − 2006 as part of a randomized, replicated, experimental design. Overstory structure, tree regeneration, surface and canopy fuels, and understory vegetation responses were monitored for 11 yr in order to meet the following objectives: 1) determine how stand-level dynamics differ between common fuels reduction approaches; 2) compare ecological conditions resulting from treatments with a conceptual model of states and transitions driven by natural disturbances in pinyon-juniper ecosystems; and 3) use state-and-transition comparisons to assess treatment effectiveness for long-term fuel management and ecosystem conservation?

Methods

Study Site

We conducted our study at a 760-ha site in the Tusayan Ranger District of the Kaibab National Forest in northern Arizona (Fig. 2). The site was identified by the US Forest Service for hazardous fuel reduction treatments to help safeguard Grand Canyon National Park and the town of Tusayan, Arizona from wildfire. Elevation at the site ranges from 2 005 to 2 073 m, and vegetation is transitional with representation of Rocky Mountain and Madrean montane conifer forests, primarily composed of ponderosa pine *(Pinus ponderosa)* overstories occupying canyon bottoms and Great Basin conifer woodlands, dominated by Colorado pinyon *(Pinus edulis)* and Utah Juniper *(Juniperus osteosperma)*, occurring on the uplands (Brown, 1994; Huffman et al., 2008). Gambel oak *(Quercus gambelii)* is found scattered in low densities throughout the site. Our study was focused on responses of the pinyon-juniper woodlands occurring on the upland microsites. Soils in these communities are Typic Haplustalfs with gravelly, loam texture, derived from limestone parent material (USDA Forest Service, 1991). Precipitation at the study site is typical of northern Arizona and falls mainly in summer as punctuated monsoonal rains generated from convection storms and moist air that moves into the region from the Gulf of California and Gulf of Mexico and in winter as snow from Pacific frontal systems. Common shrubs occurring at the site include Stansbury's cliffrose (*Purshia mexicana* var. *stansburyana*), Apache plume *(Fallugia paradoxa)*, big sagebrush and black sagebrush (*Artemisia tridentata* and *A. nova*), rubber rabbitbrush *(Ericameria nauseosa)*, and broom snakeweed *(Gutierrezia sarothrae)*. Numerous forbs are found in these communities with winged and redroot buckwheat (*Eriogonum alatum* and *E. racemosum*), fineleaf hymenopappus *(Hymenopappus filifolius)*, toadflax penstemon and thickleaf penstemon (*Penstemon linarioides* and *P. pachyphyllus*), and mid bladderpod *(Physaria intermedia)* being among the more common species. Common graminoids (grass and grasslike species) include native blue grama *(Bouteloua gracilis)*, Junegrass *(Koeleria macrantha)*, muttongrass *(Poa fendleriana)*, and White Mountain sedge *(Carex geophila)*. The site has not been grazed by domestic livestock since the late 1990s; however, native ungulates

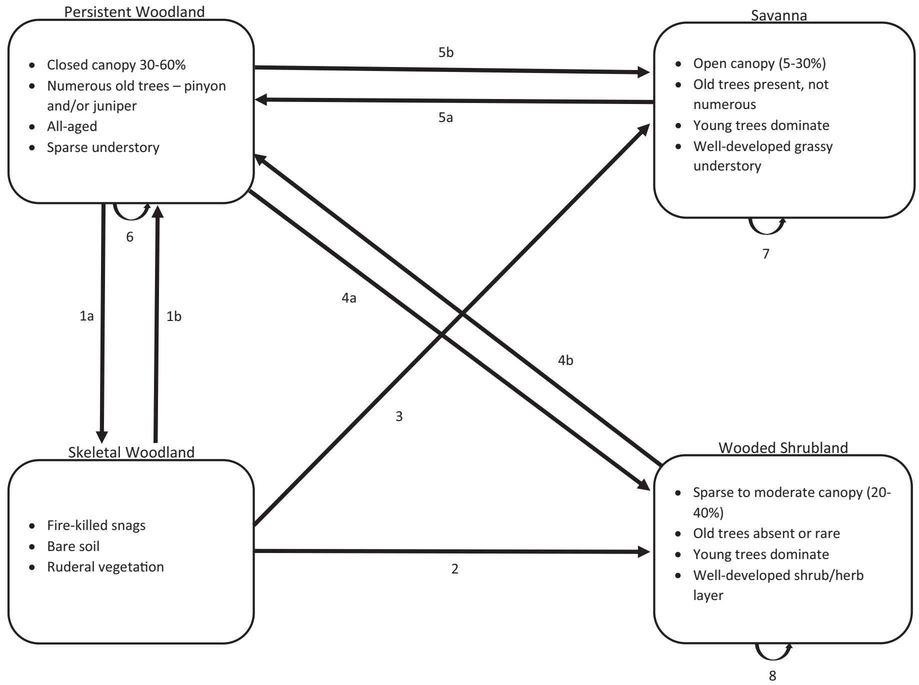

Figure 1. Conceptual model of states and transitions for southwestern pinyon juniper ecosystems. Boxes indicate stable structural states, and *arrows* indicate transitions between states resulting from natural disturbance or succession. *1a*, Stand-replacing fire occurring at natural fire rotations of up to 600 yr. *1b*. Succession occurring over 100 − 300 yr; *2*. Succession occurring over 10 − 100 yr; *3*. Succession occurring over 10 − 100 yr; *4a*. Moderate severity fire and/or periodic severe drought occurring over 10 − 100 yr; *4b*. Succession occurring over 100 − 200 yr; *5a*. Succession occurring over 100 − 200 yr; *5b*. Moderate-high severity fire occurring at natural fire rotations of > 100 yr; *6*. Low-severity fire occurring at smaller spatial scales at return intervals < 50 yr; *7*. Low- moderate severity fire occurring at return intervals < 50 yr; *8*. Low-severity fire occurring at smaller spatial scales at return intervals of < 50 yr.

(deer and elk) do use the site and intensive grazing is known to have occurred historically throughout the region (Jacobs and Gatewood, 1999; Landis and Bailey, 2005; Huffman et al., 2008).

Historical fire rotation at the site was 340 yr (Huffman et al., 2008). Infrequent, stand-replacing fires resulted in a fine-grained mosaic of stand ages across the landscape. Dendrochronological evidence suggests that no large, stand-replacing fire has occurred for several centuries, and stands with pinyon pine trees ≥ 200 yr in age occurred on > 80% of the study site area before fuel reduction treatments.

Experimental Design

In 2004, we established a randomized complete block design to examine effects of two hazardous fuel reduction approaches—tree thinning and prescribed fire—on forest structure, tree mortality, tree recruitment and regeneration, and understory plant community characteristics. Six blocks were selected on the basis of topographic characteristics and location within the study area (see Fig. 2). We used a geographic information system (GIS) to delineate blocks and identify locations (points) on which experimental treatment units and field sample plots would be centered. Eight points (two replicates of each of four treatments—see below) were randomly selected from a rectangular grid (200-m) overlaid on each experimental block polygon in the GIS. Selected points were then randomly assigned to one of four treatments with two examples of each treatment in each block ($n = 12$). Treatments assigned were as follows: 1) untreated control (Control); 2) prescribed fire (Burn); 3) tree thinning (Thin); and 4) thinning followed by prescribed fire (Thin + Burn) (see Fig. 2). Thinning prescriptions (both Thin and Thin + Burn) were modeled after actual fuels reduction treatments planned for the site by the U.S. Forest Service. The prescriptions called for removing juniper trees up to 30.5 cm in diameter at root collar (drc) and pinyon pine trees up to 22.9 cm drc. Ponderosa pine trees up to 22.9 cm in diameter at breast height (1.37 m aboveground; dbh) were removed, whereas all Gambel oak were retained and not cut. On our study units, spacing protocols were not used, although on the larger operational treatment at the site, smaller trees (i.e., below prescription limits) were left in situations where removing them would create undesirably large (e.g., > ca 0.01 ha) openings (H. McRae, US Forest Service, personal communication).

Field Sampling

In 2004, selected points described earlier were located in the field using hand-held global positioning systems (GPS). Experimental treatment units, 100 × 100 m (1 ha) in size, were traversed around each of the 48 points and demarcated using tree paint and flagging tape. Centered within each treatment unit, we established one circular sample plot, 0.04 ha in size. Sample plot centers were permanently monumented and georeferenced for relocation and long-term monitoring.

Before fuel reduction treatment implementation, we conducted measurements on field plots to quantify overstory structural characteristics, surface and canopy fuel loading, tree regeneration density and species composition, and understory vegetation cover and species richness. On each plot, standing trees (> 1.37 m in height) were numbered and fitted with aluminum tags nailed at 40 cm aboveground, and condition (live or dead) and species were recorded. Diameter (drc for juniper and pinyon pine; dbh for ponderosa pine and Gambel oak) and total height were recorded for all standing trees, and height-to-crown base was recorded for live trees. In addition, average crown radius was estimated for live juniper trees. Dead and down trees were located, and drc or dbh was estimated as determined by species. On a smaller (100-m^2) nested subplot, we tallied tree regeneration (< 1.37 m height) by species, condition (live or dead), and height class. Height classes were as follows: 1) < 40 cm; 2) 40–80 cm; and 3) > 80–137 cm in total height. Dead woody

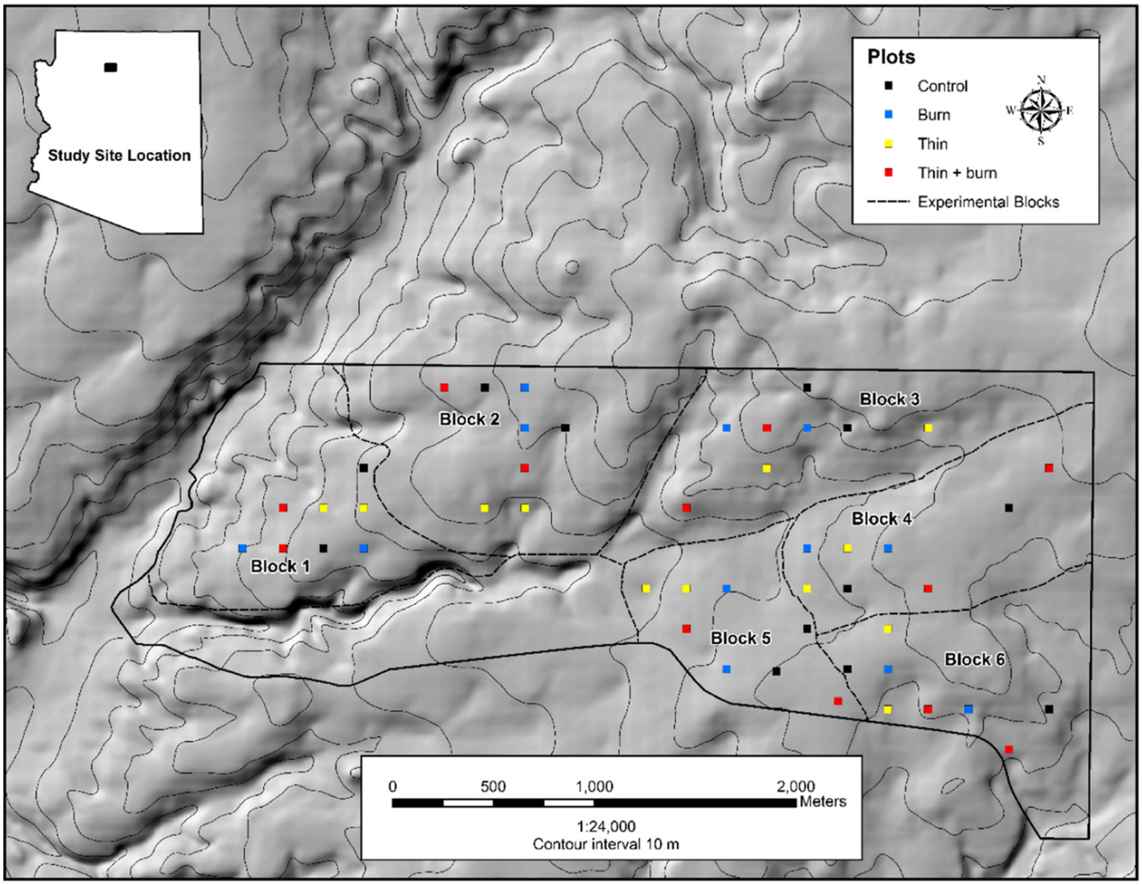

Figure 2. Pinyon-juniper study site (750 ha) on the Tusayan Ranger District of the Kaibab National Forest in northern Arizona. Shown are experimental blocks and plot locations ($n = 12$) within 1-ha units for untreated (Control), prescribed fire only (Burn), tree thinning only (Thin), and combined thinning and fire (Thin + Burn) treatments.

surface fuels were sampled on a randomly oriented planar transect (15 m) following methods described in Brown (1974). On these transects, we tallied intersecting woody fuels by moisture release timelag class and measured depth of the forest floor detrital layer at points every 1.5 m. Forest floor depth measurements were taken separately for litter (Oi) and duff (Oe + Oa) O-horizon soil layers. Fuels timelag classes followed those described in Brown (1974).

Understory community characteristics were sampled along a 50-m line transect centered on each plot with the 25-m point located at plot center. Transects were systematically oriented parallel with plot slope. Species occurrence was recorded within a 10 × 50 m belt arrayed along the sampling transect at each plot. Plant nomenclature was verified using local keys (e.g., Kearney and Peebles, 1964; Springer et al., 2009), and nativity, duration, and life form were determined using the US Department of Agriculture Plants Database (https://plants.usda.gov/). Species and substrate cover (%) were estimated in 10 quadrats (50 × 200 cm) per plot located at 5-m intervals and on alternating sides of each transect (but see below). Substrate types identified were bare soil, litter (fine forest floor debris), rock, and wood (> 7.6 cm width). In addition, tree canopy cover was estimated along each transect by using a densitometer and taking readings at 16 equally spaced points. Plot measurements were taken in 2004 before treatment implementation ("pretreatment"), as well as in 2007 (1 yr post treatment) and 2017 (11 yr post treatment). In addition to the above years, understory vegetation measurements were also collected in 2011 (five yr post-treatment). Due to time constraints, 5 of the 10 understory sampling quadrats on each plot were systematically selected for remeasurement in 2017. Plots were measured between late May and early June in all sampling years.

Treatment Implementation

Treatment of the experimental units was implemented by the US Forest Service in 2004 − 2006. Trees were hand-felled using chainsaws, and thinning debris was lopped to ~16 cm in length and scattered. Tree boles were not removed from the site, and dispersed fuelwood gathering by the public was allowed but resulted in minimal wood removal from our sample plots. Prescribed fire was implemented on experimental units in fall of 2006 using drip torches and hand ignition. Discontinuous surface fuel cover required crews to target fuel "jackpots" (i.e., heavier aggregations) rather than use more common broadcast burning techniques. Relative humidity was 15 − 20% and wind speeds (6 m height) were up to 16 km h^{-1} on the days of burning. More details concerning treatment implementation and prescribed burn conditions can be found in Huffman et al. (2009 and 2013)

Analysis

Huffman et al. (2009) reported first-yr post-treatment effects of thinning and prescribed fire at our site; therefore, we limited our interest in treatment effects on overstory structure and fuels to 2017 responses. We used 2-way analysis of variance (ANOVA) for blocked designs and analyzed tree density (trees ha^{-1}), stand basal area (BA; m^2 ha^{-1}), individual tree basal area growth rate (cm^2 yr^{-1}), tree canopy cover (%), crown fuel load (kg m^{-2}), canopy bulk density (kg m^{-3}), sum of 1-100-hr timelag surface fuels classes (Mg ha^{-1}), and litter cover Tree diameter (drc) was included as a covariate ($P < 0.05$) in models for testing treatment effects on tree growth. Canopy cover and litter cover were arcsine-transformed before analysis to stabilize variance. Crown fuel loading and canopy bulk density were calculated

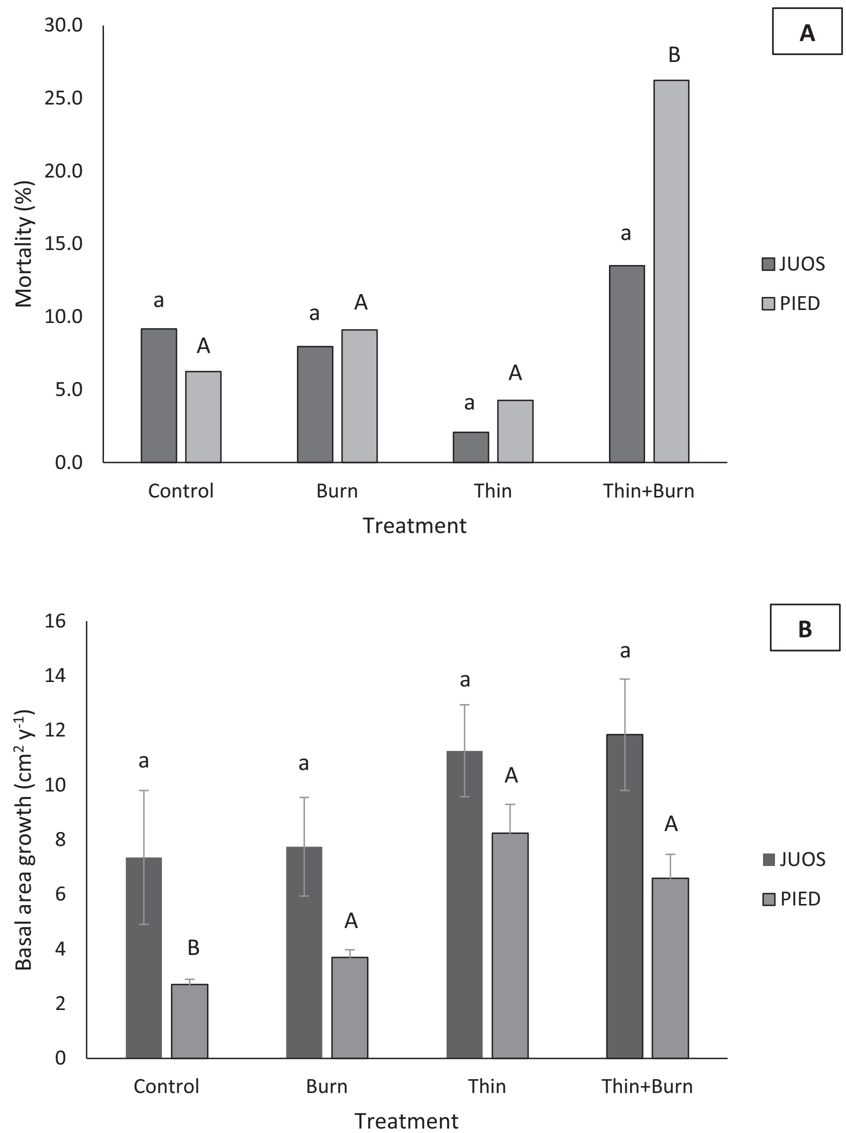

Figure 3. Tree mortality (%) by species and treatment (**A**) and individual tree growth (mean ± standard error) (**B**) from 2007 to 2017 for overstory juniper (JUOS) and pinyon pine (PIED) trees within four experimental fuels reduction treatments at the Tusayan study site. Different lowercase letters denote significant differences ($P < 0.05$) between treatments for juniper, and different uppercase letters denote significant differences for pinyon pine.

using allometric equations that related tree diameter and crown area to foliage biomass (Miller et al., 1981; Grier et al., 1992; see also Huffman et al., 2009). Before ANOVA, we used Shapiro-Wilk to test for data normality and Levine's test to assess for equal variance. When data showed non-normal distributions, we used either the square-root or natural log transformation to better meet this assumption. We used Welch's test to analyze for treatment effects when variances were unequal. When significant ($P < 0.05$) treatment effects were found, we used Tukey's honestly significant difference (HSD) post-hoc tests for pairwise comparisons (Kuehl, 1994). An *alpha* level of 0.05 was used for these ANOVA tests to determine statistical significance. To test for differences in post-treatment (2007 − 2017) pinyon and juniper tree mortality, we used z-tests for independent proportions (i.e., two-by-two contingency tables). We used a Bonferonni correction and $alpha = 0.0125$ to determine statistical significance for these multiple pairwise comparisons.

To test for treatment effects on regeneration over time, we used ANOVA for repeated measures and analyzed mean density within each seedling size class, as well as all size classes combined (see earlier section "Field Sampling"). We also used ANOVA for repeated measures to test for treatment effects on understory plant cover and species richness. Total cover data were transformed using the square root to meet ANOVA assumptions for normality. When time × treatment interactions were found, we performed 1-way ANOVA tests within treatment years. For these tests, Tukey's HSD post-hoc tests were used for pairwise comparisons of treatment group means. An *alpha* level of 0.05 was used for all ANOVA tests to determine statistical significance.

To examine effects of fuel treatments on ecological states and transitions, we first constructed a conceptual state and transition model (see Fig. 1) based on descriptions found in the published literature (Arnold et al., 1964; Romme et al., 2003; Landis and Bailey, 2005; Gori and Bate, 2007; Romme et al., 2009). We next explored post-treatment structural states using plot values and *k*-means cluster analysis (Kaufman and Rousseeuw, 1990). *K*-means is an iterative fitting process that can allow partitioning of multivariate data into groups using a set of attributes that is assumed to be important in determining a plot's membership or allocation to a particular state. On the basis of our conceptual model, we selected the following attributes for clustering: 1) number of large trees (> 35 cm drc); 2) number of small trees (≤ 35 cm drc); 3) canopy cover; 4) graminoid cover; and 5) shrub cover. These variables emerged in the literature as being important for describing pinyon-juniper structural states. Attribute values on each plot were calculated from 2017 field data and then standardized (*z*-score) before cluster analysis. We chose the range of k to be 3 − 16 clusters or groups, based on the number of ecological states in our conceptual model ($N = 3$, with exception of "skeletal woodland"—see Fig. 1) and number of sample plots ($N = 48$). Group number maximizing the cubic clustering criteria (CCC) was chosen as the optimal solution. Using the final set of groups (i.e., ecological states), we identified proportions of plots within treatments that were associated with each ecological state. We then used these proportions to calculate a weighted pooled mean score for each treatment group. Pooled mean scores for each attribute were used to qualitatively evaluate structural characteristics for each treatment against our conceptual model.

Results

Overstory Dynamics

Over the post-treatment period 2007 − 2017, mean juniper tree mortality was 2.1 − 13.5% and pinyon tree mortality was 4.3 − 26.2% (Fig. 3A). Although juniper mortality tended to be low in the Thin treatment, we found no statistically significant differences among treatments. Pinyon mortality in Thin + Burn was significantly greater than that of the other treatments (see Fig. 3A). Mortality and ingrowth (i.e., trees recruited into the overstory (≥ 1.37 m height) from the seedling class (< 1.37 m height) jointly affected net change in tree density. For juniper, mean tree density decreased by 8.4% and 0.9% in Control and Burn treatments, respectively, between 2007 and 2017. In contrast, juniper tree density increased by 14.0% and 27.3% on average for Thin and Thin + Burn treatments, respectively. We found a similar pattern for pinyon pine tree density changes. In Control and Burn treatments, pinyon tree density decreased 1.1% and 3.3%, respectively. In the Thin treatment, pinyon tree density increased by 12.2%. In contrast to juniper, pinyon tree density in the Thin + Burn treatment decreased 7.9% from 2007 to 2017.

Post-treatment (2007 − 2017) basal area growth rates of individual trees were 7.4 − 11.8 $cm^2\ yr^{-1}$ and 2.7 − 8.2 $cm^2\ yr^{-1}$ for juniper and pinyon pine, respectively (see Fig. 3B). Mean growth rates of juniper trees tended to be greater in Thin and Thin + Burn treatments than Control and Burn, but treatments were not statistically different. Pinyon pine trees showed a pattern similar to that of juniper, and mean growth rates in all treatments were significantly greater than that of the Control treatment. Mortality, ingrowth, and tree BA growth rates jointly affected net change in overall stand-level BA ($m^2\ ha^{-1}$). In the Control treatment, juniper BA decreased by 6.2% between 2007 and 2017. In Burn and Thin treatments, juniper BA increased by 7% and 8.8%, respectively. Juniper BA decreased in Thin + Burn treatment by 5% from 2007 to 2017. Pinyon pine BA showed an increase of 8.4% in the Control treatment, no change in Burn, an increase of 3.5% in the Thin treatment, and a decrease of 19.1% in Thin + Burn. Combined species BA in 2017 was significantly greater in Control and Burn compared with Thin and Thin + Burn treatments (Table 1).

In 2017, 11 yr after treatment implementation, mean tree density (pinyon and juniper combined) was significantly lower in the Thin and Thin + Burn treatments compared with the Control and Burn treatments (see Table 1). Tree density in the Thin + Burn treatment was about 24% that of the Control and 32% that of the Burn treatment. Effects of thinning from below (i.e., small size classes) remained evident in 2017 (Fig. 4A and B). Thinning shifted diameter class distribution from negative exponential (all-sized) forms in 2004 (see Fig. 4A) to unimodal-shaped distributions, which were still apparent in 2017 (see Fig. 4B). Before treatment (2004), the smallest-diameter class (≤ 15 cm drc) was largely dominated by pinyon pine and had more trees than the next larger class (15.1 − 35.0 cm) by 2.2- to 2.5-fold. In 2017, the smallest-size class in thinned treatments showed more even numbers of pinyon and juniper trees and about half the abundance observed in the next-larger class (see Fig. 4B). In contrast, although total tree number was reduced in the Burn treatment, the distribution of diameters for this treatment in 2017 maintained a negative exponential form (see Fig. 4B).

Canopy and Surface Fuels

In 2017, means for canopy cover, crown fuel load, and canopy bulk density showed a consistent pattern of decrease with increasing treatment intensity, as follows: Control > Burn > Thin > Thin + Burn (see Table 1). Mean canopy cover was significantly greater in the Control compared with Thin and Thin + Burn (see Table 1). Canopy cover in the Control was similar to Burn, and Burn was similar to the Thin treatment. Crown fuel load was significantly higher in the Control and Burn treatments compared with Thin and Thin + Burn. Crown fuel load in Thin + Burn was 38% that of the Control, and means in Thin and Burn treatments were 57% and 83% of the Control mean, respectively (see Table 1). Canopy bulk density was significantly higher ($P < 0.05$) in the Control and Burn treatments compared with Thin + Burn (see Table 1). Dead woody surface fuels tended to increase with intensity of treatment. However, we found no statistical differences between treatments in mean loading of combined 1- to 100-hr timelag classes (see Table 1). In contrast, litter cover systematically decreased with treatment intensity (see Table 1). In 2017, litter cover was significantly higher in the Control treatment compared with Thin and Thin + Burn treatments, which in turn were not statistically different.

Tree Regeneration and Understory Vegetation

Mean total density of tree seedlings was not affected by treatment, nor were densities of juniper or pinyon when analyzed by species and size class (Fig. 5). Further, we found no effect of treatment on density

Table 1
Means (and standard errors) in 2004 (pretreatment) and 2017 (11 yr after treatment) for tree density, stand basal area (BA), canopy cover, crown fuel load (CFL), canopy bulk density (CBD), dead woody surface fuel loading (sum of 1- to 100-hr timelag classes), and litter cover (%) in four hazardous fuels reduction treatments in pinyon-juniper woodlands in northern Arizona. Different uppercase letters denote significantly different ($P < 0.05$) means. No statistical differences were found among treatments in 2004.

	Density (trees ha^{-1})	BA ($m^2\ ha^{-1}$)	Canopy cover (%)	CFL (kg m^{-2})	CBD (kg m^{-3})	1-100-hr (Mg ha^{-1})	Litter (%)
2004							
Control	960.4 (92.4)	35.4 (2.9)	40.1 (4.2)	1.126 (0.082)	0.119 (0.010)	3.70 (1.22)	63.7 (5.1)
Burn	910.4 (72.7)	39.0 (3.7)	40.1 (5.0)	1.191 (0.101)	0.122 (0.012)	2.81 (1.13)	60.3 (4.6)
Thin	820.0 (70.8)	31.0 (3.0)	28.6 (4.4)	1.402 (0.255)	0.157 (0.033)	3.16 (0.88)	51.7 (5.6)
Thin + Burn	1008.3 (109.5)	39.2 (4.5)	29.2 (3.2)	1.157 (0.105)	0.103 (0.016)	2.96 (0.62)	53.5 (4.2)
2017							
Control	900.0 (79.3) A	37.1 (1.4) A	46.3 (4.1) A	0.899 (0.069) A	0.092 (0.006) A	4.29 (1.08) A	66.1 (6.1) A
Burn	677.1 (61.6) B	34.0 (3.2) A	40.6 (6.1) AB	0.745 (0.078) A	0.071 (0.010) AB	4.49 (2.47) A	57.1 (4.8) AB
Thin	225.0 (22.0) C	23.1 (2.5) B	27.1 (3.6) BC	0.460 (0.551) B	0.049 (0.005) BC	5.24 (1.28) A	45.5 (6.0) B
Thin + Burn	214.6 (40.6) C	19.9 (3.8) B	17.7 (4.9) C	0.341 (0.068) B	0.037 (0.007) C	6.63 (2.13) A	39.5 (3.1) B

BA, basal area; CFL, crown fuel load; CBD, canopy bulk density.

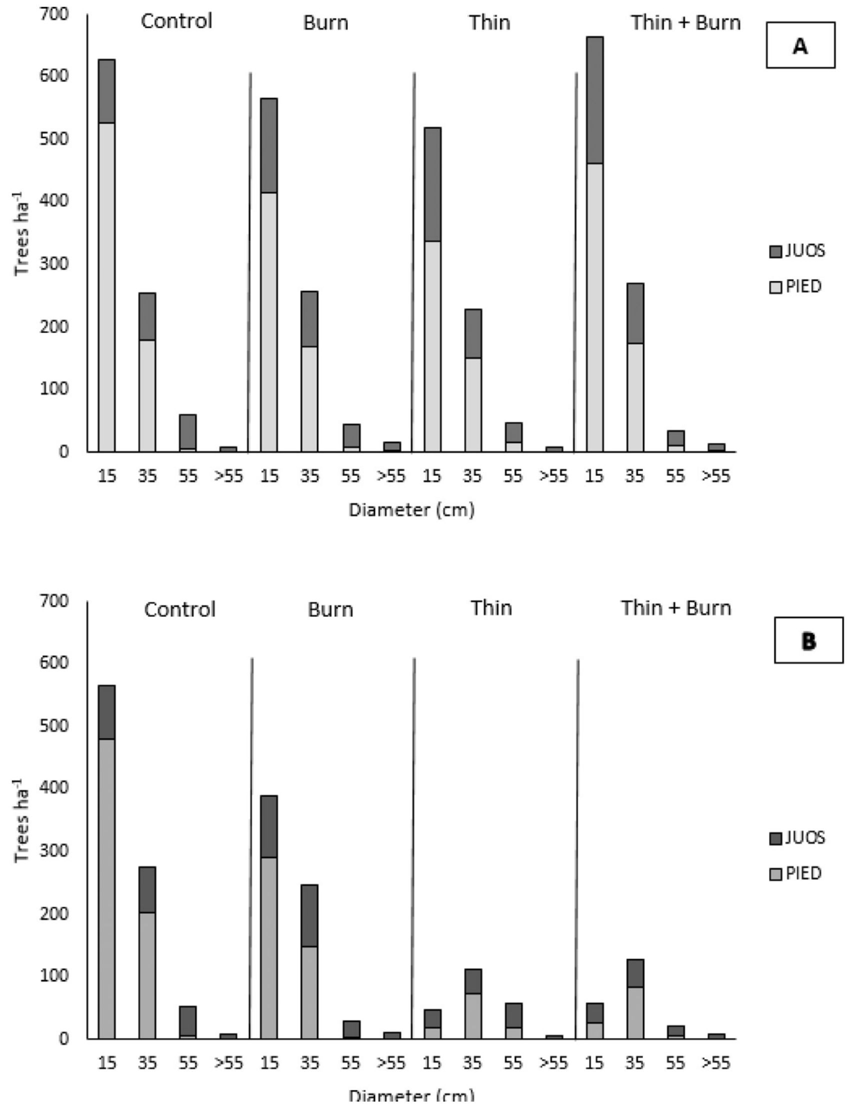

Figure 4. Diameter distributions for juniper (JUOS) and pinyon pine (PIED) trees in 2004 before treatment (**A**) and in 2017 (**B**) 11 yr after treatments were implemented, for four fuel-reduction treatments at the Tusayan study site.

of seedling within size classes for either species. In general, seedling number decreased from 2004 (pretreatment) to 2007 (1-yr post treatment) across all treatments, including the Control. With the exception of juniper seedlings in Thin and Thin + Burn, densities further decreased from 2007 to 2011 in all species × treatment combinations (see Fig. 5). From 2011 to 2017, total seedling densities tended to increase. Size class distributions were dominated by seedlings < 40 cm in height, regardless of the measurement year, species, or treatment (see Fig. 5). This was especially pronounced for pinyon pine.

Understory plant cover showed a significant treatment effect, as well as a significant treatment × year interaction (Fig. 6). When measurement years were analyzed separately, we found significant differences in total plant cover among treatments for each year. Cover in the Control tended to remain constant over the study period, whereas that in Burn decreased immediately after treatment implementation and then increased from 2011 to 2017 (see Fig. 6). Cover in the Thin treatment increased steadily over time, whereas that in Thin + Burn showed an initial decrease and then a higher rate of increase than other treatments for the remainder of the study period. By 2017, the Control showed significantly lower plant cover than Thin and Thin + Burn treatments but was not different than Burn. The Thin + Burn treatment showed the highest total plant cover but was not significantly different than that of Thin (see Fig. 6). In all treatments, plant composition was dominated by the graminoid functional group, followed by shrubs, then forbs (see Fig. 6). Graminoid cover in Thin + Burn tended to be higher than cover of other functional groups in other treatments but was < 6%. Similarly, shrub cover in Thin + Burn was < 3% (see Fig. 6). Non-native species cover across all treatments averaged < 1% in 2017. Cheatgrass *(Bromus tectorum)* was found on < 4% of Control, Burn, and Thin plots but occurred on 23% of Thin + Burn plots. We found no significant effect of treatment on species richness. In 2004, mean number of species found per 500 m^2 transect was 26.9 − 28.6. Number of species tended to increase over time in all treatments, and by 2017 richness averaged 31.3 − 38.0 species per 500 m^2. Species composition was dominated by forbs, followed by graminoids, then shrubs.

Cluster Analysis

Twelve groups (i.e., ecological states) were identified by *k*-means cluster analysis (CCC = −0.3777) using standardized plot values for five key structural attributes. Treatments generally segregated into unique groups, and these were evaluated against state descriptions in our conceptual model. For example, Control plots were associated with 5 of the 12 groups and were characterized by relatively high numbers of small trees, high tree canopy cover, and low graminoid cover (Fig. 7). Pooled mean scores for large trees and shrub cover in these

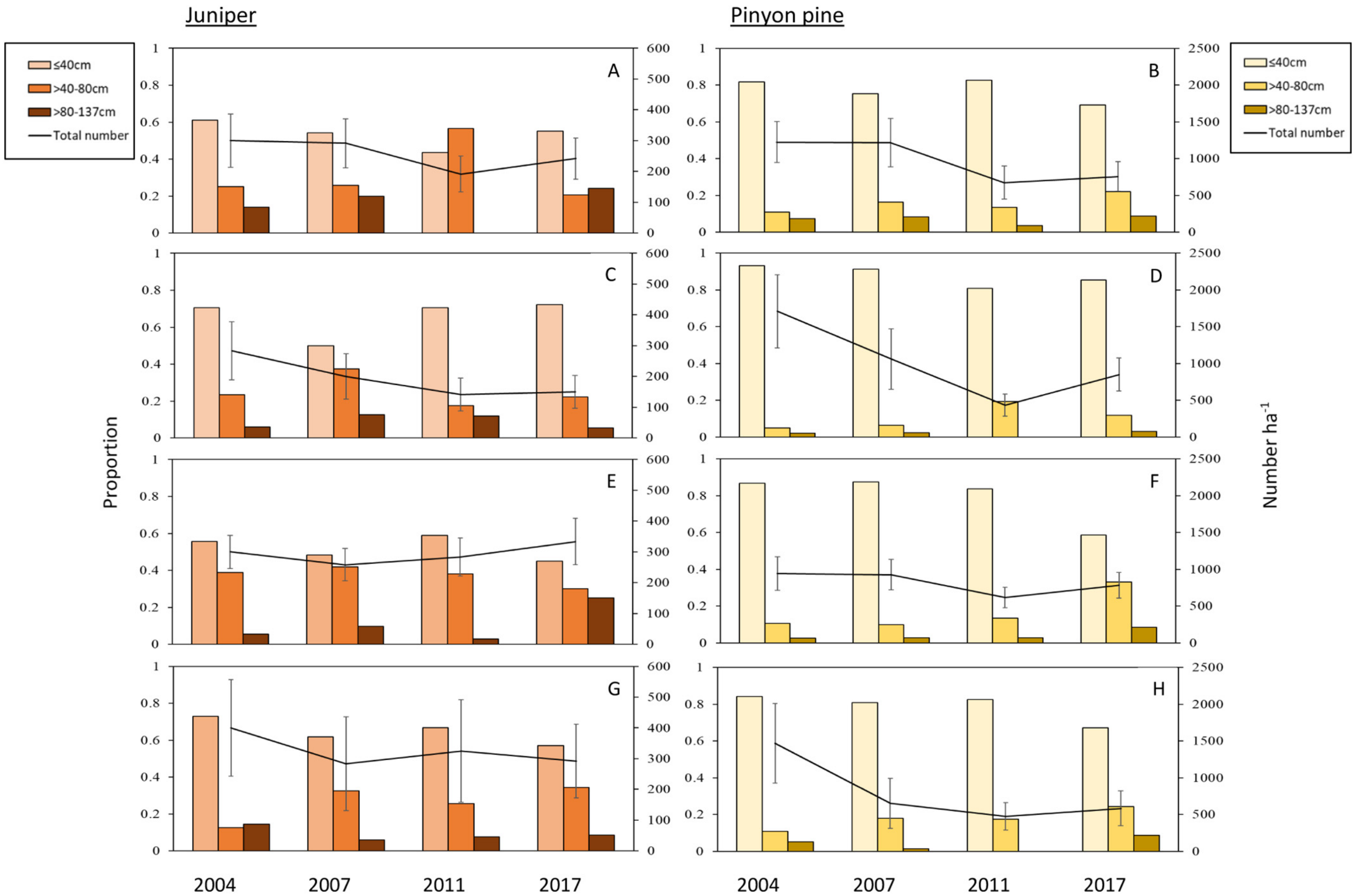

Figure 5. Proportion (bars and left y-axis) of total density (line and right y-axis) of tree seedlings within three height classes in 2004 (pretreatment) and three post-treatment measurement yr (2007, 2011, and 2017). Shown are juniper (orange bars, left) and pinyon pine (yellow bars, right) in Control (**A, B**), Burn (**C, D**), Thin (**E, F**), and Thin + Burn (**G, H**) treatments at the Tusayan study site.

groups were near zero (−0.2 – 0.2). These groups showed attribute qualities similar to those described for the persistent woodland state in our conceptual model (see Fig. 1). Burn plots showed greater variability than the Control treatment and were associated with 7 of the 12 groups. Pooled mean scores showed that Burn plots were similar in structure to Control plots, but with higher relative numbers of large trees and higher relative shrub cover (see Fig. 7). Plots in the Thin treatment were associated with 4 of the 12 groups and were characterized mainly by low numbers of small trees, low canopy cover, and high graminoid cover. Open canopy and high graminoid cover were qualities that fit descriptions of the pinyon-juniper savanna state (see Fig. 1). Thin + Burn plots were associated with low numbers of large and small trees, low canopy cover, high graminoid cover, and high shrub cover (see Fig. 7). Conditions in Thin + Burn plots did not correspond well with any of the state descriptions in our model.

Discussion

Fuel Treatment Effects on Stand Dynamics

Hazardous fuels reduction and restoration treatments that include tree thinning and prescribed fire are designed to address ecological changes that have come about due to anthropogenic exclusion of fire in historically frequent-fire ecosystems (Graham et al., 2004; McIver et al., 2009). Although frequent-fire systems have been identified as high priority for fuel reduction and restoration (e.g., Brown et al., 2004), such treatments are commonly implemented in persistent pinyon-juniper woodlands, particularly within wildland-urban interface areas (Schoennagel and Nelson, 2011), where protection of human communities from wildfire may outweigh conservation concerns. However, few studies have analyzed longer-term overstory dynamics after treatments in these woodlands.

At our site in northern Arizona, stand dynamics over the 11-yr study period differed among treatments for tree mortality, individual tree growth, ingrowth, and stand-level changes in density. Untreated Controls generally maintained stand structural characteristics that existed before implementation of the study, although with a minor shift toward increasing pinyon pine importance. Decreases in overstory density from 2007 to 2017 in the Control treatment indicated that climate, in concert with treatment, affected outcomes. Prescribed fire alone (Burn) produced minor effects on stand structure, and low rates of subsequent mortality resulted in only small reductions in canopy fuels over the study period. In the Thin treatment, tree density and stand-level BA increased for both juniper and pinyon pine, indicating maintenance of overstory species composition and patterns that could lead to eventual recovery of pretreatment conditions. Consistent patterns of tree recruitment and BA increases for both juniper and pinyon on Thin plots suggested that structural conditions and fuels in this treatment may rebound more rapidly toward pretreatment conditions than those in Thin + Burn. In the Thin + Burn treatment, gains in juniper tree density with losses of stand-level juniper BA suggested that mortality was focused on larger trees, and small trees were recruited (i.e., ingrowth) from understory seedlings banks, which offset mortality. Pinyon pine declined in tree density and BA, suggesting that individual tree growth and recruitment did not offset the high level of mortality, which in turn disproportionately affected larger trees. These shifts in stand composition and structure are similar to those observed after severe outbreaks of pinyon ips beetle *(Ips confusus)* (Negrón and Wilson, 2003).

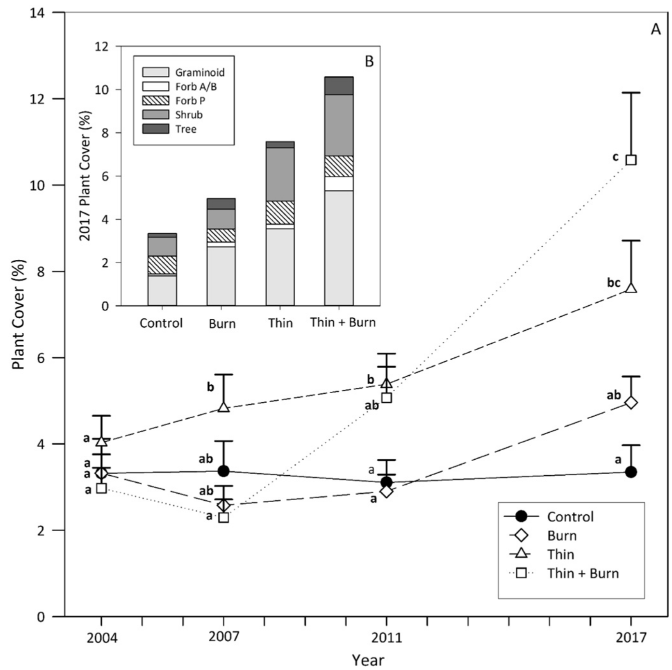

Figure 6. Mean understory plant cover (%) in 2004 (pretreatment) and three post-treatment measurement yr (2007, 2011, and 2017) (± standard error) for four fuel-reduction treatments in southwestern pinyon-juniper woodlands. Different lowercase letters denote significantly different ($P < 0.05$) means within measurement years. Also shown (inset) is cover by functional group in 2017.

Structural changes initiated by thinning and burning treatments persisted for 11 yr. Persistence of treatment effects illustrated typical slow recovery to disturbance and management activities in these ecosystems (Erdman, 1970). Further, some fuels characteristics showed lower values in 2017 than those reported immediately after treatment. For example, mean crown fuel loads continued to decline over the study period (see Table 1) and were lower in 2017 compared with those reported in Huffman et al. (2009). Decline in crown fuel load reflected continuing mortality of larger overstory trees in all treatments, but particularly in Thin + Burn. Delayed mortality and reductions in fuel loading continuing for several years after treatment have been reported in other systems (Stephens et al., 2012; Roccaforte et al., 2018). We did not attempt to identify factors related to secondary tree mortality, but interacting effects of residual damage from fire, bark beetle infestation, and drought were likely important. For example, Negrón and Wilson (2003) found that pinyon ips beetle infestation increased with pinyon

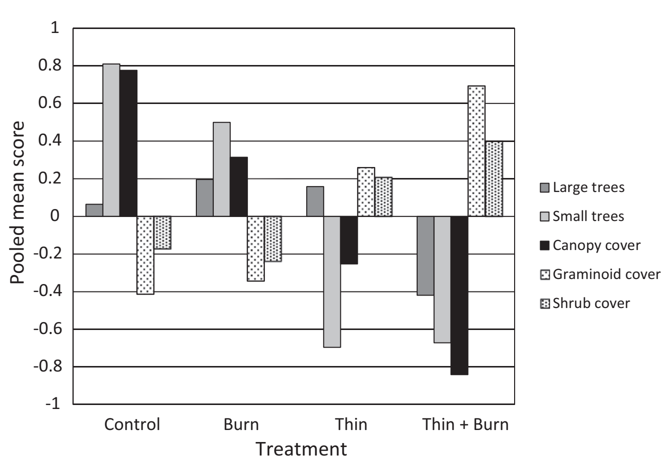

Figure 7. Mean standardized scores from cluster analysis for five structural attributes in 2017 for four fuel reduction treatments at the Tusayan study site.

stand density and individual tree size. Breece et al. (2008) reported that prescribed fire increased bark beetle attacks, and tree morality was positively related with crown damage and bark beetle attack in southwestern ponderosa pine forests. Morillas et al. (2017) found reduced soil water content and tree sap flow rates after experimental reduction in pinyon density during a droughty period (2011 − 2012) in pinyon-juniper woodlands of New Mexico.

Many restoration and fuels reduction studies in pinyon-juniper ecosystems have focused primarily on understory vegetation, soils, and hydrologic responses to treatments. For example, Ross et al. (2012) reported that understory plant cover was 4- and 15-fold greater in thinned and masticated sites, respectively, compared with an untreated control 1 − 2 yr after pinyon-juniper woodland treatment in southeastern Utah. Mastication (shredding, chipping, or mulching thinning debris) may improve understory development by moderating extremes in soil temperature and moisture (Owen et al., 2009). However, mechanical disturbance on mastication sites and heat transfer to soils due to smoldering combustion may increase establishment of nonnative plant cover (Kreye et al., 2014; Coop et al., 2017). In our study, thinned trees were lopped into small lengths and this debris was then scattered. Like mastication, lop-and-scatter slash treatments have been shown to improve environmental conditions for herbaceous understory plant establishment (Jacobs and Gatewood, 1999; Stoddard et al., 2008). Jacobs (2015) described positive understory cover responses and reduced runoff and sediment production, 3 − 5 yr after thinning and lop-and-scatter in northern New Mexico. Ashcroft et al. (2017) found that thinning intensity in northern New Mexico woodlands affected runoff, with areas of low stand BA showing less runoff than those with higher BA and untreated controls. Our results suggested that slash treatment (broadcast burn versus no burning) was less important in affecting understory cover response than changes to tree density and overstory canopy cover (see Fig. 6). Although Thin + Burn resulted in significant increases in abundance and tended to be higher than other treatments, total cover averaged < 10% in 2017. Limited understory responses to thinning have been reported in other pinyon-juniper woodlands (e.g., Redmond et al., 2014; Huffman et al., 2017), and more research is necessary to better understand community-level responses across the broad range of environmental conditions and site histories associated with pinyon-juniper ecosystems.

Comparisons with Conceptual State and Transition Model

Some ecologists have suggested that implementation of fuel reduction activities designed to address anthropogenic changes in frequent-fire forests may be ecologically inappropriate and damaging to infrequent-fire types (Romme et al., 2003; Floyd et al., 2004; Shinneman and Baker, 2009; Schoennagel and Nelson, 2011). One approach for testing these concerns is to evaluate treatment outcomes against models of ecosystem responses to natural disturbances. High tree density and canopy cover, all-aged stands, and low understory abundance in untreated units at our study site fit descriptions of pinyon-juniper forest (Romme et al., 2003) and persistent woodlands (Romme et al., 2009). In the US Southwest, natural fire rotations in these systems have been estimated to be up to 600 yr and possibly longer on some (Floyd et al., 2008; Kennard and Moore, 2013). Major fire events are typically severe and driven by wind and droughty conditions. High-severity fire causes rapid transition to a “skeletal forest” condition, characterized by standing dead trees and dominated by ruderal plants (see Fig. 1). Various models of succession after severe fire have been described, but most authors agree that return to a closed woodland structure may require ≥ 2 centuries. For example, Arnold et al. (1964) described successional transition to two possible intermediate states after stand-replacing fire—one dominated by shrubs, the other dominated by perennial grasses—and reoccurrence of fire-causing transition from the shrub-dominated state to the grass-dominated state. Similarly, Gori and Bate (2007) depicted severe fire moderating transitions among six states characterized by differences in canopy cover and understory structure, which in turn were influenced by fire-free period length. Other studies have described considerable changes in woodland structure due to drought- and bark beetle – related tree mortality, mainly of large/old pinyon pine (Negrón and Wilson, 2003; Breshears et al., 2005; Mueller et al., 2005; Floyd et al., 2015). In contrast to stand-replacing fire in which most or all trees are killed, these disturbances may create more open structures that enable more abundant understory development and where large, old trees are less important. In our conceptual state and transition model (see Fig. 1), we attempted to capture these dynamics while also simplifying structural variations to evaluate responses to treatments. To date, no quantitative models are available that provide expected ranges of values for structural attributes in various states. However, variability among sites and communities would likely reduce the utility of quantitative models for evaluating general response patterns and improving management decisions (Bestelmeyer et al., 2004). Thus, with the exception of value ranges for canopy cover, our evaluation of treatment effects against natural disturbance processes and structural conditions was qualitative.

We identified treatment outcomes that both paralleled and contrasted with descriptions of structural states in our model. Results from cluster analysis indicated similarity in structural states between Control and Burn treatments in 2017. Although the Burn treatment produced only minor decreases in tree density and fuels (crown fuel load was 83% of Control), low-severity prescribed fire also conserved structural integrity of these persistent woodlands by maintaining occurrence of large trees, as well as an all-sized diameter distribution and higher canopy cover. Further research could be done to determine whether woodland structural characteristics would be sustained while more intensive fuels reduction objectives were met incrementally with repeated fire entries. The Thin treatment also showed greater than average occurrence of large trees but low densities of small trees, as well as low canopy cover and high graminoid and shrub cover. These characteristics corresponded closer to expectations for the savanna structural state than for persistent woodland or wooded shrubland. However, thinning sharply reduced dominance of small trees and understory development was limited, and these characteristics contrast with those described for pinyon-juniper savannas. Further monitoring is necessary to determine the effects of individual tree growth and tree recruitment on future structural characteristics in the Thin treatment. Structural characteristics in the Thin + Burn treatment did not correspond well to expectations for any of the four states in our conceptual model. Cluster analysis for Thin + Burn showed lower than average large trees, small trees, and canopy cover and higher than average graminoid cover and shrub cover. These results reflect both small tree reductions from thinning and larger tree mortality in responses to prescribed fire. Continued mortality of large trees, in addition to recruitment of young junipers in this treatment, may lead to conditions more similar to early successional stages after severe fire (Huffman et al., 2012).

Implications

Results of our study showed that common treatments for restoration and hazardous fuels reduction are effective for reducing tree density and canopy fuel loads in persistent pinyon-juniper woodlands of the Southwest. Woody surface fuels did not appear to be affected by thinning and prescribed fire, although this component of the fuels complex may have low importance to crown fire spread and propagation in persistent woodlands (Floyd et al., 2015). Thinning and prescribed fire treatments continued to show canopy fuel loading reductions through the 11-yr study period and minimal rebound in terms of tree recruitment toward pretreatment conditions. Thus, treatments may have a long-term effect on woodland structure. Although priority for design of treatments is presently given to protection of human communities in the WUI, urban expansion into wildlands is increasing (Radeloff et al., 2018) and more attention should be given to ecological conservation when planning for fuels reduction in infrequent-fire systems in these areas. Evaluation of commonly used treatments against our state and transition model suggested that concerns regarding loss of ecological integrity may be warranted. Although the Burn treatment was least effective in reducing fuel loads, it was more effective than Thin and Thin + Burn in maintaining conditions expected for persistent woodlands. Thinning treatments were designed to substantially reduce small tree densities, and this approach appeared to conflict with expectations for ecological states produced by natural disturbances and successional processes. Thinning followed by prescribed fire showed additional losses of large trees and led to conditions not described by our conceptual model. Further research is needed to test alternative approaches for reducing fuel loading in these systems while maintaining ecological integrity by more closely adhering to reference states resulting from natural disturbance.

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