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SPECIAL FEATURE:
SAGEBRUSH STEPPE TREATMENT EVALUATION PROJECT

Treatment longevity and changes in surface fuel loads after pinyon–juniper mastication

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Abstract. In the Intermountain West, land managers masticate pinyon pine (*Pinus* spp.) and juniper (*Juniperus* spp.) trees that have encroached sagebrush steppe communities to reduce canopy fuels, alter potential fire behavior, and promote growth of understory grasses, forbs, and shrubs. At three study sites in Utah, 45 sampling plots spanning a range of tree cover from 5% to 50% were masticated. We measured surface fuel load components three times over a 10-yr period. We also measured tree cover, density, and height as indicators of treatment longevity. Changes in these variables were analyzed across the range of pre-treatment tree cover using linear mixed effects modeling. We detected decreases in 1-h down woody debris by 5–6 yr post-treatment, and from 5–6 to 10 yr post-treatment, but did not detect changes in 10-h or 100 + 1000-h down woody debris. By 10 yr post-treatment, there was very little duff and tree litter left for all pre-treatment tree cover values. Herbaceous fuels (all standing live and dead biomass) increased through 10 yr post-treatment. At 10 yr post-treatment, pinyon–juniper cover ranged 0–2.6%, and the majority of trees were <1 m in height. Given that 1-h fuels were the only class of down woody debris that decreased, it may be beneficial to masticate woody fuels to the finest size possible. Decreases in 1-h down woody debris and duff + litter fuels over time may have important implications for fire behavior and effects, but increases in herbaceous and shrub fuel loads should also be taken into account. At 10 yr post-treatment, understory grasses and shrubs were not being outcompeted by trees, and average pinyon–juniper canopy cover was <1%. Therefore, tree regeneration was not sufficient to support a crown fire. In areas where sage-grouse are a management concern, we recommend monitoring tree regeneration at mastication treatments at 10–15 yr post-treatment.

Key words: decomposition; down woody debris; pinyon pine; sagebrush; Special Feature: Sagebrush Steppe Treatment Evaluation Project.

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Introduction

Degradation of rangelands is a global issue and often results in decreased plant cover and a shift from herbaceous to woody vegetation (Geist and Lambin 2004, D'Odorico et al. 2013). In the past 160 yr in the Intermountain West, USA, sagebrush (*Artemisia* spp.) steppe communities have experienced substantial declines in quality and quantity of habitat for sagebrush-obligate species (Cottam and Stewart 1940, Miller and Rose 1999, Miller and Eddleman 2000). One important factor in the decline of these communities is the expansion and infilling of pinyon–juniper (*Pinus* spp. and *Juniperus* spp.) woodlands (Miller and Tausch 2001). Before Euro-Americans settled the Intermountain West, frequent wildfires limited persistent pinyon–juniper woodlands to rocky outcrops and rimrock—places that lacked the understory vegetation often needed to carry fire (Burkhardt and Tisdale 1976, Miller and Tausch 2001, Waichler et al. 2001, Miller et al. 2008, Miller and Heyerdahl 2008). Due to changes in land management, such as fire suppression, livestock grazing that reduced fine fuels, and a reduction in Native American fire use, fires have become less frequent in the elevation ranges that pinyon–juniper woodlands are able to occupy (Cottam and Stewart 1940, Burkhardt and Tisdale 1976, Miller and Rose 1999, Gruell 1999, Miller et al. 2008). Without wildfires that kill pinyon pine and juniper trees, these woodlands have greatly increased in density and area (Miller et al. 2008). Pinyon–juniper woodland expansion has also been facilitated by increases in atmospheric CO_2 (Polley et al. 1996) and an unusually wet climate during the late 1800s and early 1900s that aided pinyon pine and juniper regeneration (Miller and Tausch 2001). Thus, pinyon–juniper woodlands have expanded into or infilled more than 18 million ha in Intermountain West since Euro-American settlement (Miller et al. 2008).

As sagebrush-bunchgrass communities transition to dense pinyon–juniper woodlands in the absence of periodic fire, there are many changes to wildlife habitat, ecosystem functions, and fuel loads. During this transition, shrubs, grasses, and forbs decrease due to competition with trees for water (Roundy et al. 2014*a*, Ray et al. 2019) and nutrients (Bates et al. 2000, Rau et al. 2011, Young et al. 2014). These changes in vegetation reduce forage for ungulates such as cattle (*Bos taurus*; Miller et al. 2005) and mule deer (*Odocoileus hemionus*; Rosenstock et al. 1989) and reduce suitable habitat for sagebrush-obligate species such as sage-grouse (*Centrocercus urophasianus*; Baruch-Mordo et al. 2013, Bates et al. 2017) and pygmy rabbits (*Brachylagus idahoensis*; Larrucea and Brussard 2008). Due to reduced density of understory plants that aid in water infiltration, pinyon–juniper woodlands often experience increased runoff and soil erosion (Reid et al. 1999, Roundy et al. 2014*a*, Pierson et al. 2015). As pinyon–juniper woodlands mature, the fuel structure of the system changes from one dominated by fine, surface fuels (e.g., herbaceous and shrub fuels), to a system dominated by fuels not commonly found in sagebrush steppe communities: tree litter and duff, and canopy fuels that include coarse woody fuels (Miller and Tausch 2001, Sabin 2008, Tausch 2009, Miller et al. 2013, Young et al. 2015). In older pinyon–juniper woodlands, risk of high-intensity crown fires increases as canopy fuel load and continuity increases (Brown and Davis 1973, Pyne et al. 1996, Miller et al. 2013, Strand et al. 2013, Keane 2015). High-intensity crown fires are not only difficult for wildland firefighters to control, but may also lead to undesirable ecological outcomes, such as water-repellant soils (Zvirzdin et al. 2017) and an invasive, annual grass-dominated state that is difficult and costly to restore and often decreases fire return intervals (Miller et al. 2013, Chambers et al. 2014).

One treatment that land managers use to reduce pinyon pine and juniper trees where they have expanded into sagebrush-bunchgrass communities is mechanical mastication. During this treatment, whole trees are shredded to finer-sized down woody debris (i.e., mulch), thereby converting canopy fuels to surface fuels (Fig. 1). In addition to reducing canopy fuels, mastication treatments release understory plants from competition with trees and reduce the risk of high-severity crown fires. However, the increase in masticated down woody debris on the soil surface can lead to longer smoldering times and greater soil heating during subsequent fires (Busse et al. 2005, Sikkink et al. 2017), especially in areas where masticated debris overlays tree litter and duff (Sikkink et al. 2017). Quantifying fuel loads after mastication of pinyon–juniper woodlands is important because the quantity of fuel and its distribution among different fuel classes can alter fire behavior, severity, and effects (Pyne et al. 1996, Strand et al. 2013, Weiner et al. 2016). In addition, different-sized woody fuels often decompose at different rates (Harmon et al. 1986, Fasth et al. 2011, Battaglia et al. 2015, Ostrogović et al. 2015, Reed 2016, Coop et al. 2017), but decomposition rates may vary with soil moisture and temperature patterns (Harmon et al. 1986, Berbeco et al. 2012, Ostrogović et al. 2015). Many studies have described the changes in shrub and herbaceous cover after pinyon–juniper mastication treatments (Ross et al. 2012, Redmond et al. 2014, Roundy et al. 2014*b*, Bybee et al. 2016, Coop et al. 2017, Fornwalt et al. 2017), and others have documented a corresponding expansion of sagebrush-obligate birds into masticated sites (Knick et al. 2014).

Few studies, however, have described post-treatment changes in masticated sites in terms of fuel loads (Young et al. 2015, Coop et al. 2017), especially on a decadal timeframe in pinyon–juniper woodlands (Coop et al. 2017). A few studies have quantified changes in masticated, down woody debris fuel loads over time in pinyon–juniper woodlands (Shakespear 2014, Battaglia et al. 2015, Coop et al. 2017), but several of these studies took place outside of the Great Basin in Colorado (Battaglia et al. 2015, Coop et al. 2017). These studies detected decreases in fine woody fuels over 5–10 yr post-treatment; yet only one of these studies has been published (Coop et al. 2017), and its inferences may be limited because the finest size classes of woody debris were analyzed together. The quantity of masticated woody debris left onsite is of ecological importance because thick layers of masticated debris can (1) alter seedling establishment of the site (Young et al. 2013*a*), (2) alter nutrient cycling processes (Rhoades et al. 2012, Young et al. 2013*a*), (3) increase soil moisture (Young et al. 2013*b*), and (4) smolder for long periods of time and result in severe fire effects if burned in wildfires (Busse et al. 2005, Kreye et al. 2014).

Research that examines changes in fuel loads after pinyon–juniper mastication is important

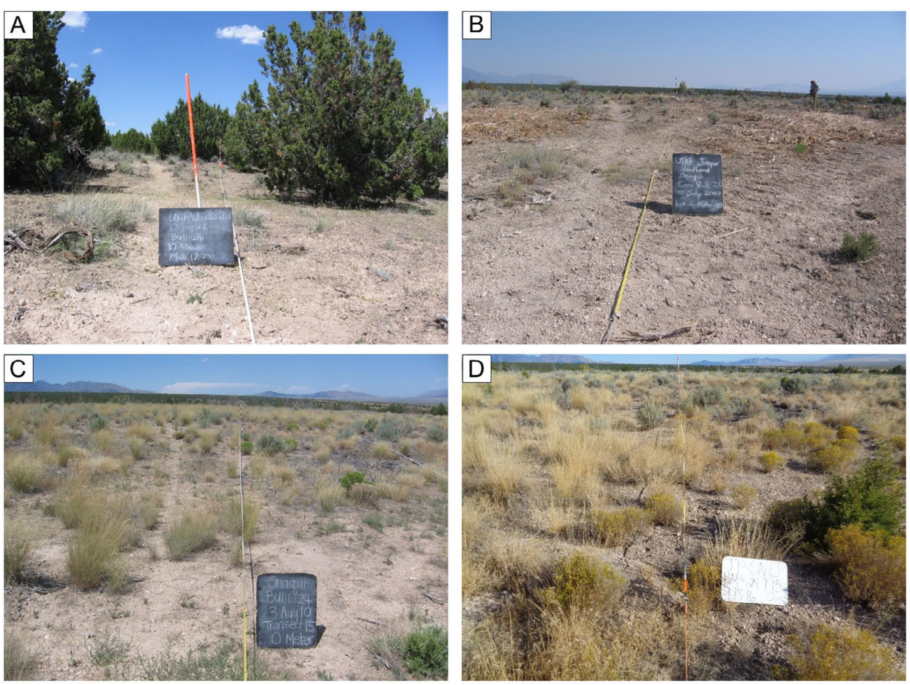

Fig. 1. Photoseries of increases in herbaceous fuels and decreases in bare ground: (A) pre-treatment, (B) 1 yr post-treatment, (C) 6 yr post-treatment, and (D) 10 yr post-treatment. This sampling plot is located at the Onaqui study site.

because there are few studies of masticated pinyon–juniper woodlands that: extend out to 10 yr post-treatment, account for variability in masticated fuel loads along a gradient of pre-treatment tree cover, or analyze other surface fuel loading components in addition to down woody debris (e.g., herbaceous, shrub, tree litter, and duff fuels). The primary objectives of this study are to analyze changes in (1) components of surface fuel loads (tree litter and duff, down woody debris, herbaceous, and shrub fuels), and (2) indicators of treatment longevity (pinyon–juniper cover and density) across 10 yr after mastication of pinyon–juniper woodlands. The intent of analyzing surface fuel loading components is to gain a better understanding of how quickly down woody debris, tree litter, and duff decompose, and how quickly herbaceous and shrub fuel loads increase following pinyon–juniper mastication. Land managers are also interested in how long it takes for trees to re-invade a site, and therefore how frequently these sites need to be treated.

This work is an extension of Young et al. (2015) and Shakespear (2014). In Young et al. (2015), pre-treatment fuel loads were compared to fuel loads at 1, 2, and 3 yr post-treatment. Therefore, this analysis will not include information on pre-treatment fuel loads or short-term changes in fuel loads.

Methods and Materials

Study locations and treatment implementation

Data were collected at three study sites situated along a north to south gradient in western Utah—Onaqui, Scipio, and Greenville Bench (see McIver and Brunson 2014 for a map). These study sites and data are part of the Sagebrush Steppe Treatment Evaluation Project (SageSTEP; McIver and Brunson 2014). An additional mastication treatment at the Stansbury SageSTEP site was not included in this analysis because the site burned in the Big Pole wildfire in 2009 (2 yr post-treatment). Elevation of the sampled plots ranged from 1674 to 1761 m. Soils were classified as follows: Loamy-skeletal, carbonatic, mesic, shallow Petrocalcic Palexerolls at Onaqui; Loamy-skeletal, mixed superactive, mesic, shallow, Calcic Petrocalcids at Scipio; and Loamy-skeletal, carbonatic, mesic Typic Calcixerepts at Greenville Bench (Rau et al. 2011). The three sites were located in the 305–356 mm (12–14 in.) precipitation zone (Bourne and Bunting 2011). Daily precipitation was measured using a tipping bucket at each site as described by Roundy et al. (2014*a*). The October–June precipitation was generally at or below the 30-yr average (1988–2018) for the course of the study, except for the water year (1 October–30 September) of 2010–2011, which was >75 mm above average at each site (Fig. 2; PRISM Climate Group 2004 data were substituted for missing values).

The study sites were comprised of Wyoming big sagebrush (*Artemisia tridentata* ssp.

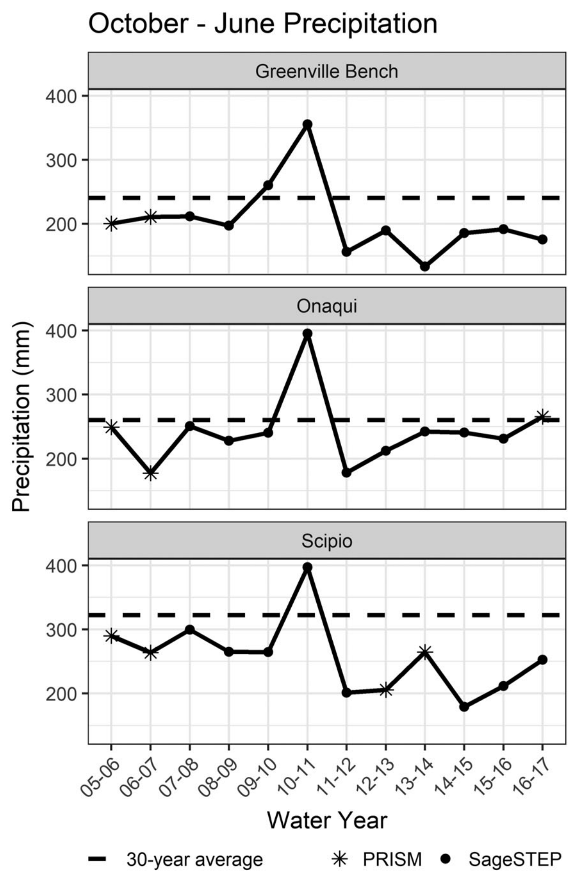

Fig. 2. October–June precipitation recorded at the three study sites across the course of the study. Data from PRISM Climate Group were used to estimate October–June precipitation for years with missing data (i.e., years before precipitation gauges were installed or years in which the gauges malfunctioned), and a 30-yr average.

wyomingensis)-bunchgrass communities encroached by Utah juniper (*Juniperus osteosperma)* and Colorado pinyon pine (*Pinus edulis*). Utah juniper is the dominant tree species at Onaqui and Scipio and is co-dominant with Colorado pinyon pine at Greenville Bench. The dominant bunchgrasses were bluebunch wheatgrass (*Pseudoroegneria spicata*) at Onaqui and Scipio, and needle-and-thread (*Hesperostipa comata*) at Greenville Bench. Wyoming big sagebrush was the dominant shrub at all these sites; the resprouting shrubs yellow rabbitbrush (*Chrysothamnus viscidiflorus)* and rubber rabbitbrush (*Ericameria nauseosa*) occurred infrequently. Prior to treatment, cover of the introduced annual cheatgrass (*Bromus tectorum*) ranged between 0% and 31% on the sampling plot level. The sites were fenced to exclude livestock for the duration of the study.

Mastication treatments were implemented at Onaqui in 2006 and at Scipio and Greenville Bench in 2007 and ranged 10–20 ha in size. Tractors equipped with Fecon Bullhog masticators (horizontal shaft) were used to shred pinyon–juniper trees >0.5 m in height. At each site, 15 randomly placed, rectangular sampling plots (30 × 33 m) were established in the treated area, for a total of 45 sampling plots. Six transects measuring 30 m in length were placed parallel to each other within the plot. Treatments were implemented in locations such that sampling plots would cover a range of pre-treatment tree cover: 7–34% at Greenville Bench, 5–36% at Onaqui, and 9–50% at Scipio.

Field measurements

Masticated down woody debris (DWD) were collected within 0.25 × 0.25 m quadrats placed every other meter along two 30-m transects (30 quadrats per sampling plot). Down woody debris were defined as dead, detached woody material within 2 m of the soil surface (Keane 2015). Down woody debris were collected at Scipio and Greenville Bench at 1, 5, and 10 yr post-treatment, and at Onaqui at 1, 6, and 10 yr post-treatment. The difference between sites in years since treatment for data collection of DWD was due to logistical constraints with field crews. In successive sampling periods, fuels were collected at different positions along the transects to avoid destructively sampling the same area twice. Fuels that were partially outside of the quadrat were cut to the length inside in the quadrat. Masticated DWD were weighed by time-lag fuel moisture class (1-, 10-, 100-, 1000-h) after being dried at 60°C for at least 96 h (Young et al. 2015). Time lag, as defined by Fosberg (1970), is the time it takes for a piece of wood (of a specific diameter) to lose 63% of the difference between its initial moisture content (after a precipitation event) and its equilibrium moisture content when in an environment of 27°C and 20% relative humidity. Given this definition, DWD were classified by time-lag fuel moisture classes based on their diameters. The 1-h fuels have a diameter of 0–0.64 cm, 10-h DWD have diameters of 0.64–2.54 cm, 100-h DWD have a diameter of 2.54–7.62 cm, and 1000-h DWD have a diameter >7.62 cm (Fosberg and Deeming 1971, Keane 2015).

Tree litter and duff were collected from 0.25 × 0.25 m quadrats at 1 and 10 yr post-treatment. Within each sampling plot, duff and litter were collected in six quadrats placed at one-third the distance from the bole of the tree to the edge of the masticated debris pile. A quadrat was placed under the four trees closest to the corners and two trees closest to the center of the sampling plot. Only trees with masticated debris piles >2 m in diameter were sampled for litter and duff because small, young trees have not had sufficient time to develop litter mounds large enough for sampling. Collected samples were dried at 50°C for 48 h. Tree litter refers to debris (e.g., leaves) from trees that have fallen to the ground and are easily recognizable because they have not yet decomposed (Robichaud and Miller 1999). Duff is the layer of decomposing organic material between the litter layer and mineral soil (Robichaud and Miller 1999, Keane 2015). Tree litter and duff fuels were collected together due to the difficulty of distinguishing between juniper litter and duff.

Herbaceous fuels (a combination of standing live, standing dead, and interspace herbaceous litter) were collected in 0.50 × 0.50 m quadrats placed every other meter along one 30-m transect for a total of 15 quadrats per sampling plot. Herbaceous fuels were sampled at 1, 6, and 10 yr post-treatment. These fuels were weighed after being dried at 50°C for 48 h.

Shrub volume measurements were collected for shrubs taller than 15 cm within five nested-circular frames with a radius of 1, 2, or 3 m so that at least 10 shrubs of each common species were measured per sampling plot (Bonham 1989, Young et al. 2015). The five nested-circular frames were evenly spaced along one transect. Shrub volume measurements were collected at 1, 6, and 10 yr post-treatment. Site and species-specific allometric equations developed by Bourne and Bunting (2011) were used to estimate shrub fuel loads at each time interval. R^2 values for the allometric equations are available in Bourne and Bunting (2011) and ranged from 0.62 to 0.97.

Bare ground cover (%) was measured using the line-point intercept method with data recorded every 0.5 m along five 30-m long transects for a total of 300 points per sampling plot. A point was considered bare ground if the only contact point was mineral soil (i.e., masticated debris did not count as bare ground).

Tree cover was collected pre-treatment and at 10 yr post-treatment, and tree density was collected at 1, 6, and 10 yr post-treatment. Tree cover was estimated after measuring the longest canopy diameter and the perpendicular diameter of each tree >0.5 m in height, within each 30 × 33 m plot. Using these canopy diameter measurements, a total canopy area was estimated and divided by the area of the sampling plot. Tree density was measured using different methods depending on the size class of the tree. Within each 30 × 33 m plot, every tree >0.5 m in height was counted. Trees between 0.05 and 0.5 m in height were counted in three 30 × 2 m belt transects. Trees under 0.05 m in height were measured in the same 0.5 × 0.5 m quadrats used for sampling herbaceous biomass, but there were not enough trees under 0.05 m in height to statistically analyze.

Data analysis

We modeled fuel loads using linear mixed effects modeling in the statistical program R (R Development Core Team 2017) with the lme4 package (Bates et al. 2015). A separate model was created for each of the following surface fuel loading components: 1-h DWD, 10-h DWD, 100-h + 1000-h DWD, Duff + Litter, Herbaceous, and Shrub. The 1000-h DWD were combined with 100-h DWD because there were not enough 1000-h DWD left after the mastication treatment to analyze these fuels separately. Herbaceous fuel loads were analyzed as the sum of live standing herbaceous fuel, dead standing herbaceous fuel, and interspace litter. Tree density was also analyzed using a linear mixed effects model. Pre-treatment tree cover, years since treatment, and the interaction between the two were used as fixed effects for all models. Young et al. (2015) demonstrated pre-treatment tree cover is a reasonable predictor of post-treatment fuel loads and can be used as a covariate to explain variability in sampled fuel loads. Years since treatment was treated as a factor in each model, because the effect of each year since treatment was not incremental. Site and sampling plot were included in the models as random effects, with sampling plot nested within site. Response variables were square-root transformed for all models to better meet assumptions of homoscedasticity as assessed using residual plots. Differences in fuel loads by years since treatment were analyzed using linear contrasts at the following pre-treatment tree cover values: 10%, 20%, and 40%; these values can be interpreted as low, medium, and high tree covers for pinyon–juniper woodlands in Utah. Linear contrasts were not performed for pre-treatment tree cover values >40% due to a lack of data; there were only two sampling plots with pre-treatment tree cover >40%. Linear contrasts and Wald tests were conducted using the trtools package (Johnson 2019), and marginal and conditional R^2 (Nakagawa et al. 2017) were estimated using the MuMIn package (Barton 2018). Marginal R^2 estimates the variance explained by the fixed effects of the model, and conditional R^2 estimates the variance explained by both the fixed and random effects of the model (Nakagawa et al. 2017). A conservative critical value of $\alpha = 0.01$ was used to determine significance of linear contrasts to reduce familywise Type I error rates. Tree height and cover were only measured at pre-treatment and 10 yr post-treatment, so these variables were not analyzed statistically. A summary table of means and standard deviations based on the raw data is also provided in Table 1.

Results

Down woody debris

We detected decreases in fuel loads of 1-h DWD from 1 to 5–6 yr post-treatment at 10%,

Table 1. Means ± standard deviations of fuel loads (Mg/ha), bare ground cover (%), and tree density (stems/ha) for sampling plots that had ranges of pre-treatment tree cover from 5% to 15%, 15–25%, and 25–50%.

Response variable	Years post-treatment	Pre-treatment tree cover range (%) 5–15	15–25	25–50
1-h DWD fuel load	1	3.39 ± 2.16	7.04 ± 4.46	10.87 ± 4.49
	5–6	1.67 ± 1.59	3.68 ± 2.87	5.38 ± 1.68
	10	0.89 ± 0.81	2.23 ± 1.44	3.12 ± 2.06
10-h DWD fuel load	1	1.93 ± 1.11	4.44 ± 1.7	6.62 ± 2.22
	5–6	2.17 ± 1.16	3.68 ± 1.89	4.46 ± 2.15
	10	2.57 ± 2.28	3.98 ± 2.23	5.46 ± 2.96
100 + 1000-h DWD fuel load	1	1.37 ± 2.13	1.59 ± 2.84	4.01 ± 2.95
	5–6	0.56 ± 0.62	1.24 ± 1.61	2.58 ± 3.27
	10	0.94 ± 1.05	1.9 ± 3.07	3.7 ± 3.6
Tree litter + duff	1	5.27 ± 2.72	10.59 ± 3.03	15.96 ± 6.82
	10	0.34 ± 0.59	0.33 ± 0.43	0.53 ± 1.02
Herbaceous fuel load	1	0.72 ± 0.28	0.37 ± 0.2	0.3 ± 0.2
	6	0.65 ± 0.29	0.6 ± 0.39	0.7 ± 0.4
	10	1.02 ± 0.35	1.43 ± 0.64	1.2 ± 0.42
Shrub fuel load	1	1.84 ± 1.62	0.86 ± 0.7	0.29 ± 0.51
	6	2.16 ± 1.6	1.69 ± 1.21	0.39 ± 0.33
	10	2.66 ± 1.95	1.68 ± 1.31	0.76 ± 0.58
Total fuel load	1	14.53 ± 5.38	24.43 ± 7.74	32.38 ± 11.17
	10	8.41 ± 4.83	12.02 ± 7.12	13.23 ± 7.07
Bare ground cover	1	27.57 ± 11.37	30.51 ± 8.84	28.42 ± 7.91
	6	22.68 ± 6.09	21.29 ± 7.31	17.93 ± 9.4
	10	22.98 ± 6.98	20.13 ± 5.58	17.04 ± 5.36
Tree density	1	91.75 ± 85.11	81.6 ± 91.23	77.9 ± 111.68
	6	202.9 ± 176.86	170.33 ± 172.32	161.1 ± 187.67
	10	219.7 ± 161.64	193.6 ± 190.26	159.2 ± 207.85

Note: Means and standard deviations provided are based on raw data.

20%, and 40% pre-treatment tree cover ($P < 0.01$; Figs. 3 and 4, Tables 1, 2, 3). We also detected decreases from 5–6 to 10 yr post-treatment at 20% and 40% pre-treatment tree cover. The model of 1-h DWD had a marginal $R^2 = 0.51$, and conditional $R^2 = 0.66$ (Table 2). In terms of raw data, sampling plots with pre-treatment tree cover ranging 15–25% decreased from a mean and standard deviation of 7.04 ± 0.43 Mg/ha at 1 yr post-treatment to 2.23 ± 0.43 Mg/ha at 10 yr post-treatment. From 1 to 5–6 yr post-treatment, the mean (±SE) decrease in 1-h DWD load was 34.9% ± 8.8%, and from 1 to 10 yr post-treatment was 62.6 ± 6.5%. We failed to detect changes in fuel loads for the 10-h and 100 + 1000-h classes of DWD (Fig. 4, Table 1).

Tree litter + duff

We detected decreases in tree litter + duff from 1 to 10 yr post-treatment at 10, 20, and 40% pre-treatment tree cover (Fig. 5; Table 3). The estimated marginal and conditional R^2 values for the model were 0.89 and 0.93, respectively (Table 2), demonstrating that the fixed effects of the model explained 89% of the variability in the data. By 10 yr post-treatment, there were very low litter and duff loads at all levels of pre-treatment tree cover. Means and standard deviations for fuel loads in sampling plots with 5–15% pre-treatment tree cover were 0.34 ± 0.59 Mg/ha, 15–25% pre-treatment tree cover were 0.33 ± 0.43 Mg/ha, and 25–50% pre-treatment tree cover were 0.53 ± 1.02 Mg/ha (Table 1). From 1 to 10 yr post-treatment, the mean (±SE) decrease in duff + litter load was 95.3 ± 1.1% (relative to the litter and duff load at 1 yr post-treatment). Although we did not quantify tree litter or duff separately, there was so little tree litter + duff left at 10 yr post-treatment that both tree litter and duff likely decreased in fuel load.

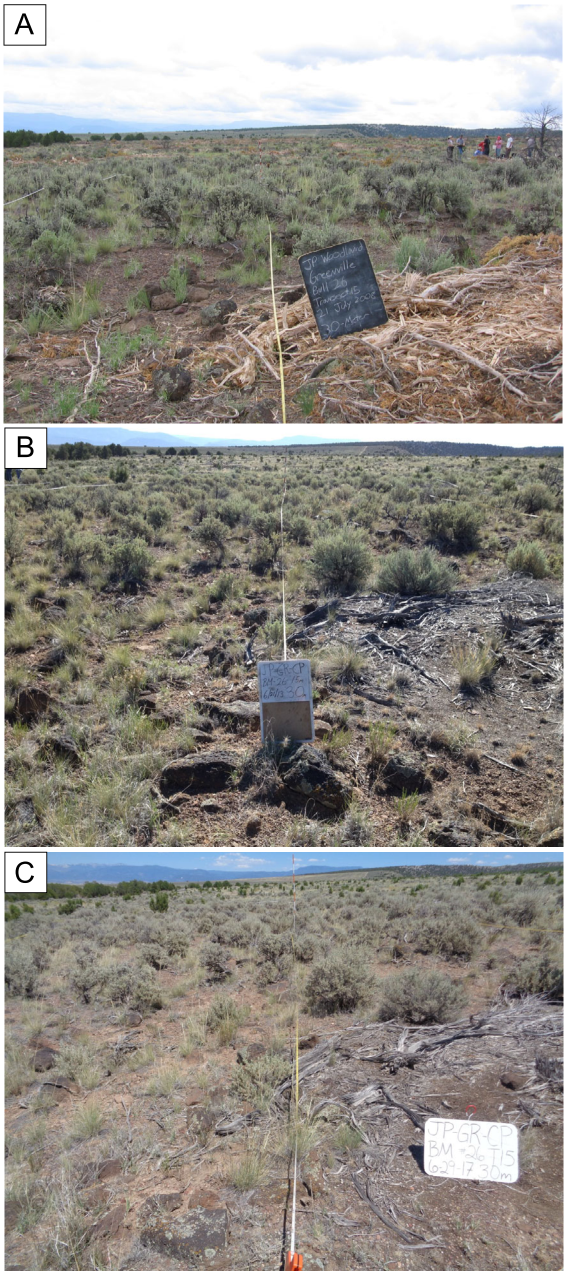

Fig. 3. Photoseries of decomposition of fine-sized down woody debris at 1 yr post-treatment (A), 5 yr post-treatment (B), and 10 yr post-treatment (C). This sampling plot is located at the Greenville Bench study site.

Herbaceous fuels

We detected increases in herbaceous fuel loads from 1 to 6 yr post-treatment at 20% and 40% pre-treatment tree cover, and from 6 to 10 yr at 10%, 20%, and 40% pre-treatment tree cover (Fig. 1, Table 3). The marginal and conditional R^2 values were both 0.45. By 10 yr post-treatment, mean herbaceous fuel loads were >1 Mg/ha across the range of pre-treatment tree cover (Table 1). From 1 to 6 yr post-treatment, the mean (±SE) increase in herbaceous fuel load was 180 ± 87.0%, and from 1 to 10 yr post-treatment was 413.4 ± 110.4%. The increases in herbaceous fuels were due to a combination of increases in cheatgrass and native grasses and forbs.

Shrub fuels

We detected increases in shrub fuel loads from 1 to 6 yr post-treatment at 20% pre-treatment tree cover, and increases at 10%, 20%, and 40% pre-treatment tree cover from 1 to 10 yr post-treatment (Fig. 1, Table 3). We failed to detect differences in shrub fuel loads between 6 and 10 yr post-treatment. The estimated marginal and conditional R^2 values were 0.34 and 0.77 (Table 2). The large difference between the conditional and marginal R^2 values demonstrates that the random effects—site and sampling plot—accounted for a substantial portion of variation explained by the model. Based on the raw data, mean shrub fuel loads in sampling plots between 5–15% and 15–25% increased almost twice as much as mean shrub fuel loads between 25% and 50% tree cover from 1 to 10 yr post-treatment (Table 1). From 1 to 5–6 yr post-treatment, the mean (±SE) increase in shrub fuel load was 134.4 ± 34.7%, and from 1 to 10 yr post-treatment was 232 ± 61.4%.

Total fuel load

We detected decreases in total fuel loads from 1 to 10 yr post-treatment at 10%, 20%, and 40% pre-treatment tree cover (Tables 1 and 3, Fig. 6). The marginal and conditional R^2 values were 0.63 and 0.75 (Table 2). Based on the raw data, the mean (±SD) total fuel load for sampling plots with 5–15% pre-tree cover decreased from 14.53 ± 5.38 to 8.41 ± 4.83 Mg/ha, 15–25% pre-treatment tree cover decreased from 24.43 ± 7.74 to 12.02 ± 7.12 Mg/ha, and 25–50% pre-treatment tree cover decreased from 32.38 ± 11.17 to 13.23 ± 7.07 Mg/ha (Table 1) from 1 to 10 yr post-treatment. At 1 yr post-treatment, tree litter + duff and 1-h down woody debris comprised the majority of the mean total fuel load

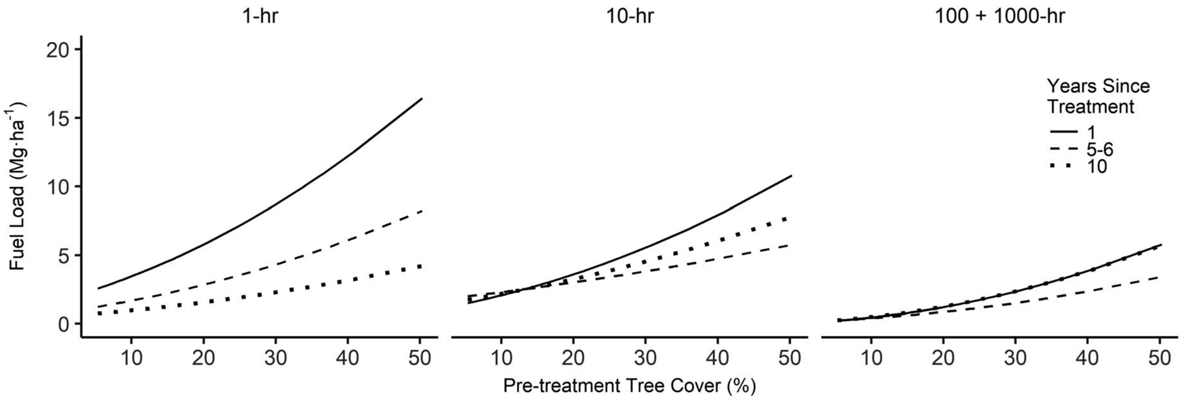

Fig. 4. Model-based estimates of the median of down woody debris fuel loads (Mg/ha) by pre-treatment tree cover (%), year since treatment, and time-lag fuel moisture classes: 1-h down woody debris (left), 10-h down woody debris (center), and 100 + 1000-h down woody debris (right). No significant differences were detected in 10-h or 100 + 1000-h fuel loads between years sampled.

(Fig. 7). These fuels decomposed such that at 10 yr post-treatment, the mean (±SE) total fuel load decreased 48.6 ± 3.9%, even though there were significant increases in herbaceous and shrub fuels (Fig. 7).

Bare ground cover

We detected decreases in bare ground cover (%) at 20% and 40% pre-treatment tree cover from 1 to 6 yr post-treatment, but we failed to detect significant changes in bare ground cover between 6 and 10 yr (Table 3, Fig 6). Bare ground cover varied substantially by site and sampling plot, which is demonstrated by the large difference between the marginal and conditional R^2 values of 0.20 and 0.39 (Table 2).

Tree density, cover, and height

Tree density increased between 1 and 6 yr post-treatment at 10% and 20% pre-treatment tree cover (Table 3, Figure 8). Tree density varied substantially among sampling plots and sites (Figs. 2D and 8), demonstrated by the difference between the marginal and conditional R^2 values of 0.10 and 0.72 (Table 2). At 10 yr post-treatment, trees were recorded in 107 of 135 sampling plots. In sampling plots with trees, the tree density was composed of 72 ± 39% trees between 0.05 and 0.5 m in height. In sampling plots where there were trees >0.5 m in height, the mean tree height and standard deviation of trees >0.5 m in height were 0.9 ± 0.2 m. At 10 yr post-treatment, mean tree cover and standard deviation were 0.6 ± 0.7%, with a range of 0–2.6%. All of the sampling plots with greater than or equal to 1% tree cover occurred at the Greenville Bench site.

Discussion

Changes in surface fuel loads

Several studies have shown that pinyon–juniper litter decomposes relatively quickly, but most of these studies are short-term (Murphy et al. 1998, Bates et al. 2007, Vanderbilt et al. 2008). Bates et al. (2007) found a 27% mean mass loss of juniper litter 2 yr after a juniper cutting treatment. Murphy et al. (1998) also found that after 2 yr, juniper and pinyon pine litter lost 25–35% of its mass in the elevation ranges that pinyon–juniper woodlands occur. Our analysis shows that by 10 yr after mastication, there was little tree litter or duff left on site (4.5 ± 7.5% of the tree litter and duff fuel loads measured at 1 yr post-treatment were left at 10 yr post-treatment).

We also detected significant decreases in the finest size fuel class of DWD (1-h), but did not detect changes in coarser fuels. Several studies have shown that finer-sized fuels (intact or masticated) decompose at a higher rate than coarser fuels (Mattson et al. 1987, Harmon et al. 1995, Hyvönen et al. 2000, Lyons and McCarthy 2010, Berbeco et al. 2012, Battaglia et al. 2015, Ostrogović et al. 2015, Reed 2016, Coop et al. 2017), but

Table 2. Summary of output from Wald tests on linear mixed effects models.

Response variable	Fixed effect	Estimate	SE	Lower 99% CI	Upper 99% CI	*T*	*P*
1-h DWD	Intercept (YST 1)	1.317	0.191	0.827	1.808	6.9	**<0.0001**
	Pre Tree Cover	0.054	0.009	0.031	0.077	6.1	**<0.0001**
	YST 5–6	−0.408	0.225	−0.988	0.172	−1.8	0.0700
	YST 10	−0.592	0.224	−1.168	−0.016	−2.6	**0.0081**
	Pre Tree Cover × YST 5–6	−0.015	0.011	−0.043	0.012	−1.5	0.1400
	Pre Tree Cover × YST 10	−0.028	0.011	−0.055	−0.001	−2.7	**0.0076**
10-h DWD	Intercept (YST 1)	0.985	0.205	0.457	1.513	4.8	**<0.0001**
	Pre Tree Cover	0.046	0.008	0.025	0.066	5.7	**<0.0001**
	YST 5–6	0.315	0.229	−0.275	0.905	1.37	0.1700
	YST 10	0.167	0.228	−0.419	0.754	0.73	0.4600
	Pre Tree Cover × YST 5–6	−0.024	0.011	−0.051	0.004	−2.22	0.0260
	Pre Tree Cover × YST 10	−0.013	0.011	−0.041	0.015	−1.22	0.2200
100 + 1000-h DWD	Intercept (YST 1)	0.247	0.373	−0.713	1.208	0.663	0.5100
	Pre Tree Cover	0.043	0.009	0.019	0.067	4.578	**<0.0001**
	YST 5–6	0.074	0.258	−0.590	0.738	0.288	0.7700
	YST 10	0.034	0.256	−0.625	0.694	0.134	0.8900
	Pre Tree Cover × YST 5–6	−0.012	0.012	−0.043	0.019	−1.038	0.3000
	Pre Tree Cover × YST 10	−0.001	0.012	−0.032	0.030	−0.076	0.9400
Tree litter + duff	Intercept (YST 1)	1.470	0.206	0.940	2.000	7.1	**<0.0001**
	Pre Tree Cover	0.078	0.006	0.064	0.093	13.7	**<0.0001**
	YST 10	−1.149	0.168	−1.583	−0.715	−6.8	**<0.0001**
	Pre Tree Cover × YST 10	−0.072	0.008	−0.092	−0.052	−9.1	**<0.0001**
Herbaceous	Intercept (YST 1)	0.896	0.063	0.734	1.058	14.24	**<0.0001**
	Pre Tree Cover	−0.012	0.003	−0.019	−0.004	−3.97	**<0.0001**
	YST 6	−0.129	0.089	−0.359	0.101	−1.44	0.1500
	YST 10	0.088	0.089	−0.142	0.317	0.98	0.3300
	Pre Tree Cover × YST 6	0.012	0.004	0.001	0.023	2.91	**0.0036**
	Pre Tree Cover × YST 10	0.016	0.004	0.006	0.027	3.93	**<0.0001**
Shrub	Intercept (YST 1)	1.560	0.153	1.166	1.954	10.2	**<0.0001**
	Pre Tree Cover	−0.035	0.007	−0.053	−0.016	−4.84	**<0.0001**
	YST 6	0.114	0.129	−0.219	0.447	0.88	0.3800
	YST 10	0.242	0.127	−0.084	0.568	1.91	0.0560
	Pre Tree Cover × YST 6	0.004	0.006	−0.011	0.020	0.73	0.4600
	Pre Tree Cover × YST 10	0.005	0.006	−0.011	0.020	0.8	0.4200
Total fuel load	Intercept (YST 1)	2.8474	0.324	2.0127	3.682	8.79	**<0.0001**
	Pre Tree Cover	0.0979	0.0108	0.0702	0.1256	9.1	**<0.0001**
	YST 10	−0.4905	0.3089	−1.2862	0.3052	−1.59	0.1120
	Pre Tree Cover × YST 10	−0.0554	0.0145	−0.0927	−0.0181	−3.82	**0.0001**
Bare ground cover	Intercept (YST 1)	4.889	0.304	4.105	5.673	16.07	**<0.0001**
	Pre Tree Cover	0.020	0.011	−0.008	0.049	1.85	0.0640
	YST 6	−0.165	0.311	−0.965	0.635	−0.53	0.6000
	YST 10	−0.037	0.311	−0.836	0.763	−0.12	0.9100
	Pre Tree Cover × YST 6	−0.031	0.015	−0.069	0.006	−2.14	0.0330
	Pre Tree Cover × YST 10	−0.040	0.015	−0.078	-0.003	−2.76	**0.0057**
Tree density	Intercept (YST 1)	4.916	3.630	−4.433	14.270	1.35	0.1756
	Pre Tree Cover	0.098	0.074	−0.093	0.290	1.32	0.1853
	YST 6	5.109	1.843	0.362	9.860	2.77	**0.0056**
	YST 10	6.998	1.843	2.250	11.740	3.8	**0.0002**
	Pre Tree Cover × YST 6	−0.038	0.086	−0.261	0.180	−0.45	0.6562
	Pre Tree Cover × YST 10	−0.069	0.086	−0.291	0.150	−0.8	0.4253

Notes: Estimates, standard errors, and confidence intervals are on the square-root transformed and cannot be back-transformed. R^2_m and R^2_c are the marginal and conditional R^2. “YST” represents year since treatment. *P* values for significant results appear in boldface ($P < 0.01$).The R^2_m and R^2_c values, respectively, for each response variable are as follows: 1-h DWD, 0.51, 0.66; 10-h DWD, 0.28, 0.42; 100 + 1000-h DWD, 0.20, 0.60; Tree litter + duff, 0.89, 0.93; Herbaceous, 0.45, 0.45; Shrub, 0.34, 0.77; Total fuel load, 0.64, 0.73; Bare ground cover, 0.20, 0.39; and Tree density, 0.10, 0.72.

few have demonstrated that this pattern of decomposition in fine masticated fuels occurs on a timescale relevant to land managers in arid and semi-arid regions of the Intermountain West (Shakespear 2014, Coop et al. 2017). Reed (2016) found that 1-h masticated down woody debris decreased significantly over 8–9 yr post-treatment in northern California and southern Oregon; 1-h fuels lost 69% of their mass over 8–9 yr post-treatment. The 69% mass loss over 8–9 yr post-treatment is slightly greater than the 65% mass loss over 10 yr that we documented. Battaglia et al. (2015) documented a mass loss of ~50% for pine mulch chips placed in a pinyon–juniper woodland in Colorado. Reed (2016) showed that 10-h masticated fuels decompose significantly, but at a slower rate than 1-h fuels on the same time scale. We did not detect changes in 10-h fuel loads by 10 yr post-treatment in our study area. Other locations may experience different decomposition rates than observed in our study due to many factors including climate, substrate quality (species of wood or litter), carbon to nitrogen and carbon to phosphorous ratios, microbial and fungal communities, soil nutrient availability, and solar photodegradation (Harmon et al. 1986, Murphy et al. 1998, Bates et al. 2007, Gallo et al. 2009).

The substantial decreases in tree litter + duff and 1-h DWD documented in this study have important implications for wildfires that occur within a couple years vs. 5–10 yr after pinyon–juniper mastication. Both tree litter + duff and masticated debris tend to smolder for long periods of time, resulting in extensive soil heating, increased fire severity, bunchgrass mortality, and a potential increase in exotic species (Stephan et al. 2010, Strand et al. 2013, Kreye et al. 2014,

Table 3. Summary of linear contrast estimates; significant contrasts are bolded ($P < 0.01$).

Response variable	Years compared	Pre-treatment tree cover (%) 10	20	40
1-h DWD fuel load	1:5–6	**−0.56**	**−0.72**	**−1.03**
	5–6:10	−0.31	**−0.44**	**−0.69**
	1:10	**−0.87**	**−1.15**	**−1.71**
10-h DWD fuel load	1:5–6	0.08	−0.16	−0.64
	5–6:10	−0.04	0.07	0.28
	1:10	0.04	−0.09	−0.35
100 + 1000-h DWD fuel load	1:5–6	−0.05	−0.18	−0.43
	5–6:10	0.08	0.19	0.00
	1:10	0.03	0.02	0.42
Tree litter + duff	1:10	**−1.90**	**−2.59**	**−4.00**
Herbaceous fuel load	1:6	−0.01	**0.12**	**0.36**
	6:10	**0.26**	**0.30**	**0.38**
	1:10	**0.25**	**0.42**	**0.74**
Shrub fuel load	1:6	0.16	**0.20**	0.29
	6:10	0.13	0.14	0.14
	1:10	**0.29**	**0.34**	**0.43**
Total fuel load	1:10	**−1.04**	**−1.60**	**−2.71**
Bare ground cover	1:6	−0.48	**−0.79**	**−1.41**
	6:10	0.04	−0.05	−0.24
	1:10	−0.44	**−0.84**	**−1.65**
Tree density	1:6	**4.72**	**4.34**	3.57
	6:10	1.58	1.28	0.672
	1:10	**6.31**	**5.62**	4.24

Note: Estimates are on the square-root transformed scale.

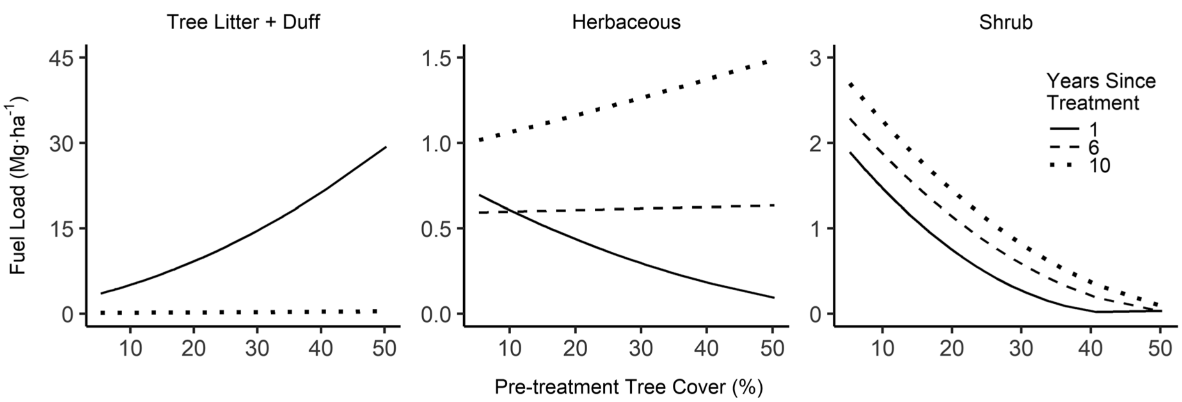

Fig. 5. Model-based estimates of the median fuel loads (Mg/ha) of tree litter + duff (left), herbaceous (center), and shrub (right) across a gradient of pre-treatment tree cover, and at 1, 6, and 10 yr post-treatment. Note: tree litter + duff fuel loads were not collected (nor estimated) at 6 yr post-treatment.

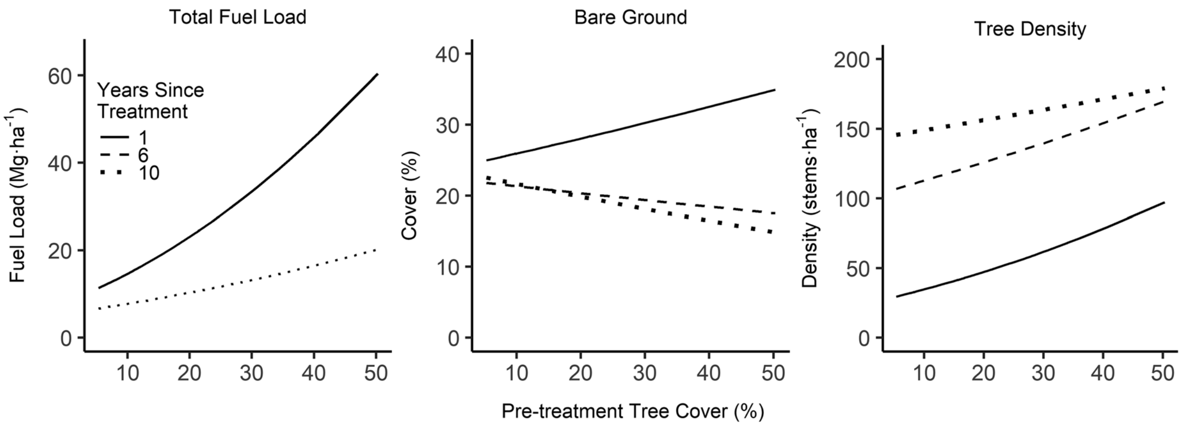

Fig. 6. Model-based estimates of median total fuel load (Mg/ha), bare ground cover (%; top), and tree density (stems/ha; bottom) across a gradient of pre-treatment tree cover, and at 1, 6, and 10 yr post-treatment. Note: Total fuel load was only estimated at 1 and 10 yr post-treatment because tree litter + duff fuel loads were not collected 6 yr post-treatment.

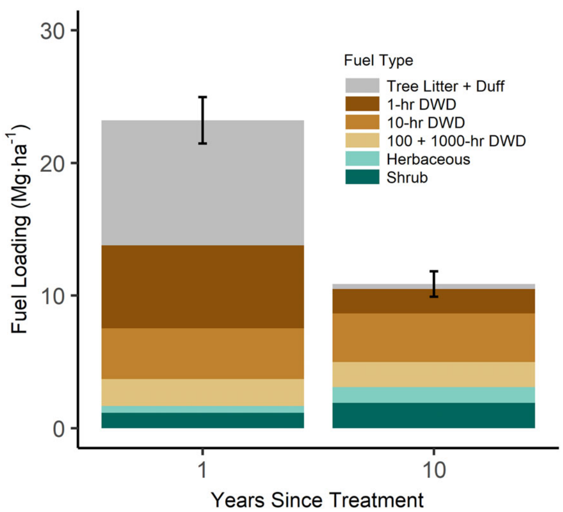

Fig. 7. Total fuel load (Mg/ha; mean ± SE) by fuel type at 1 and 10 yr post-treatment.

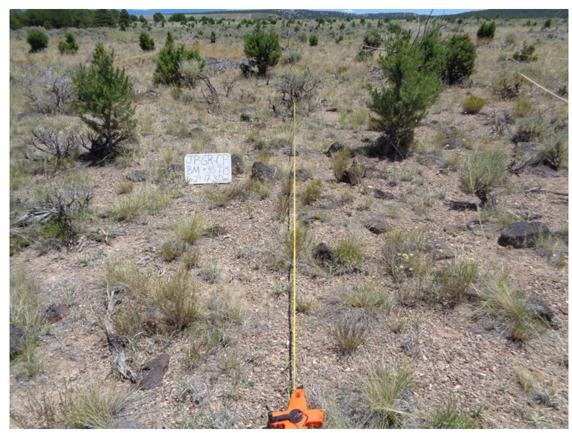

Fig. 8. High tree density at a Greenville Bench sampling plot at 10 yr post-treatment.

Weiner et al. 2016, Sikkink et al. 2017). Sikkink et al. (2017) demonstrated that smoldering duration of masticated fuels was more than twice as long when the masticated fuels were burned over duff rather than sandy soil. Greater fuel loads of masticated debris can increase soil heating (Busse et al. 2005) and increase fireline intensity (Kreye et al. 2014). These aspects of potential fire behavior and effects would likely be reduced by 10 yr after mastication treatments, due to reduced fuel loads of tree litter, duff, and 1-h down woody debris (via decomposition). Kreye et al. (2016), however, found that masticated debris older than 10 yr may smolder 50% longer, but burn with lower fire intensity and flame height than younger masticated debris. The decreases in tree litter + duff and 1-h DWD were much greater in magnitude than the increases in herbaceous and shrub fuel loads, and therefore, total fuel loads decreased about 42–59% from 1 to 10 yr post-treatment depending on pre-treatment tree cover (Table 1). In this study, trees were masticated using horizontal shaft masticators, which are more effective at reducing a high proportion of coarse fuels to finer-sized mulches than vertical shaft masticators (Jain et al. 2018). If decomposition of masticated fuels is a primary management goal, it would be beneficial to use horizontal shaft masticators and contract experienced operators; operator skill can have a substantial impact on masticated fuel size (Jain et al. 2018).

Decreases in bare ground cover and increases in herbaceous and shrub fuel loads indicate an increase in fuel continuity over the course of our study. Bare ground cover decreased significantly from 1 to 6 yr post-treatment at 20% and 40% pre-treatment tree cover, but no significant change was detected at 10% pre-treatment tree cover. This trend could be expected because on sites with low tree cover, there are only minor changes in understory cover after treatment; whereas sites with a high tree cover experience greater increases in understory herbaceous and shrub cover after treatment (Miller et al. 2005). At 10 yr after pinyon–juniper reduction treatments, land managers should expect high herbaceous fuel continuity and, therefore, the potential for increased risk of fire ignition and rate of spread (Keane 2015). Some areas treated with mastication in the Intermountain West may have lower herbaceous fuel loads than those analyzed in our study due to differences in ecological site and/or herbaceous biomass removal via grazing.

Although shrub fuels increased at 10%, 20%, and 40% pre-treatment tree cover by 10 yr post-treatment, there was still a substantial effect of pre-treatment tree cover on shrub fuel loads. Sampling plots treated at high pre-treatment tree cover had substantially lower shrub fuel loads at 10 yr post-treatment than sampling plots treated at lower pre-treatment tree cover (Tables 1 and 3). A similar trend of slower recovery of shrubs (especially sagebrush) after treating dense pinyon–juniper woodlands (e.g., Phase III as defined by Miller et al. 2005) was demonstrated in Bates et al. (2017). Shrub biomass and fuel loads likely increased in response to an increase in soil water and nutrient availability after removing trees (Roundy et al. 2014*a*, Ray et al. 2019). In our study, the majority of the increase in shrub fuel loads was due to sagebrush, but there were also small increases in other shrubs such as yellow rabbitbrush and rubber rabbitbrush. Increased sagebrush biomass and cover plays an important role in wildlife habitat and ecosystem functions, but increases in shrub fuels can also play important roles in fire behavior and effects. In extreme weather conditions, sites with high shrub canopy continuity and fuel loads can carry fire even in areas where herbaceous fuel loads and continuity are very low (Launchbaugh et al. 2008). In addition, fire intensity is typically greater under sagebrush and can result in higher bunchgrass mortality under sagebrush than in interspaces (Boyd et al. 2015, Hulet et al. 2015).

Treatment longevity

Although our linear mixed effects models detected significant increases in tree density from 1 to 6 yr post-treatment at 10% and 20% pre-treatment tree cover, these results should be interpreted conservatively. Since we could not statistically analyze trees <0.05 m tall, it is difficult to determine the magnitude of increase in tree density depicted in our model that was due to new recruitment from seed germination, or to trees <0.05 m in height growing into taller trees by 10 yr post-treatment. Our model depicted a high variability in tree density and cover due to site and sampling plot, with the Greenville Bench site having many sampling plots with high tree density and cover compared to the Onaqui and Scipio sites. This result may represent a greater number of resprouting trees or small trees that were not killed in the treatment at Greenville Bench because it was the only site that had a very rocky soil surface which made mastication more difficult. Other studies, however, have documented mean increases in tree density of about 5–10 stems$\cdot$ha$^{-1}\cdot$yr^{-1} following mechanical reduction of pinyon–juniper woodlands (Bristow et al. 2014, Bates et al. 2017). By 15 yr post-treatment, Bates et al. (2017) found that western juniper density in a cut treatment reached pre-treatment levels and that three-fourths of these trees were recruited after the treatment.

Treatment longevity is a frequently used term that is context-specific and difficult to define, especially when land managers are implementing treatments to address multiple objectives. If defined in terms of risk of crown fire, tree density did not increase sufficiently to support a crown fire at 10 yr post-treatment. Bates et al. (2017), however, suggested a treatment longevity of 25–30 yr for western juniper cutting treatments on

Steens Mountain, Oregon, based on the goal of maintaining dominance of understory perennial bunchgrasses and shrubs. In many areas of the Intermountain West, however, many mastication treatments are implemented to improve sage-grouse habitat. If treatment longevity is defined in terms of sage-grouse potential use of the site, treatment longevity may be much shorter. Baruch-Mordo et al. (2013) suggest that tree cover of 4% can influence sage-grouse to abandon lek sites, and Coates et al. (2017) suggest treating encroaching pinyon pine and juniper at tree cover values as low as 1.5% to improve sage-grouse survival. Knick et al. (2014) found that sagebrush-obligate birds recolonized mechanical cutting and mastication treatments at Onaqui within 5 yr post-treatment, but did not find that sagebrush-obligate birds recolonized other sites within the study. They suggested that almost complete removal of trees and the presence of adjacent, extensive sagebrush habitat are crucial in order for sagebrush-obligate birds to recolonize a tree-invaded sagebrush community.

In our study, tree density and tree cover were highly dependent on site and sampling plot. Based on the Coates et al. (2017) interpretation, the Greenville Bench site in our study should be re-treated at 10–15 yr post-treatment because more than one-third of the sampling plots had tree cover values ranging 1.5–2.6%. There were not any sampling plots at the Onaqui or Scipio sites that had >0.7% tree cover at 10 yr post-treatment. Once trees are established, however, tree cover can increase quickly. Bates et al. (2017) documented mean tree cover of <1% by 12 yr after a cutting treatment, but 3.8% cover by 25 yr post-treatment.

Management implications

After mastication of pinyon–juniper woodlands, there are complex changes in surface fuel loads due to some components decreasing (tree litter + duff and 1-h DWD), other components increasing (herbaceous, shrub, and small trees). Land managers should account for changes in all components of surface fuel loads when analyzing potential fire behavior and effects after mastication treatments and expect high spatial variability in fuel loads among and within sites. Areas that were treated at high pre-treatment tree cover will likely be at greater risk of ignition and rate of fire spread as herbaceous fuels increase. These effects may be coupled with a decrease in potential lethal soil heating as tree litter + duff and 1-h down woody debris fuels decompose. If reducing down woody debris fuel loads via decomposition is a primary management goal, land managers should seek skilled operators who utilize horizontal shaft masticators to produce a high percentage of 1-h fuels. Increases in tree cover and density are highly site dependent, and depending on management goals, treatment longevity may be defined differently. In areas where sage-grouse productivity is a management priority, we recommend monitoring mastication treatments 10–15 yr post-treatment to assess the need for follow-up treatment. This recommendation may be conservative, but there are many benefits to reducing pinyon–juniper trees when tree cover is still low, and trees are not yet dominating the ecological processes occurring on site.

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