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Utah juniper and two-needle piñon reduction alters fuel loads

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Abstract. Juniper (*Juniperus* spp.) and piñon (*Pinus* spp.) trees have encroached millions of hectares of sagebrush (*Artemisia* spp.)–bunchgrass communities. Juniper–piñon trees are treated to reduce canopy fuel loads and crown fire potential. We measured the effects of juniper–piñon infilling and fuel-reduction treatments on fuel load characteristics at four locations in Utah. At each location, treatment areas were burned, left untreated, or trees were cut or masticated in a randomised complete-block design. We measured standing and downed fuels by size and type along 30-m transects on 15 subplots (30 × 33 m) per location before and 1–3 years after treatment. Increased tree cover was associated with decreased shrub and herbaceous fuel loads ($P < 0.01$). By 2 years post-treatment, herbaceous fuel loads were greater than pretreatment in all treated areas ($P < 0.01$). Cut and mastication treatments increased surface woody 10- and 100-h fuel loads and wood/bark cover ($P < 0.01$). Masticated-tree depth was a good estimator of fuel loads ($R^2 = 92$). The conversion of canopy fuels to surface fuels reduced fuels that enable crown fire and extreme fire intensity. Cool-season prescribed fire may need to follow mechanical treatments to reduce surface fuel and the potential for wildfire damage to perennial understorey vegetation.

Additional keywords: fire, mulch, resilience, resistance, resource availability, weed invasion.

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Introduction

Desertification with increased woody plants and subsequent reduction of perennial grasses is one of the greatest global changes on rangelands during the last 150 years (Schlesinger *et al.* 1990; Archer *et al.* 2011). The transition from herbaceous to woody plants frequently modifies habitat and trophic structure (Archer *et al.* 2011), lessens primary plant production (Knapp *et al.* 2008) and increases soil erosion (Wainwright *et al.* 2000; Gillette and Pitchford 2004; Breshears *et al.* 2009; Pierson *et al.* 2013). Juniper (*Juniperus* spp.) and piñon (*Pinus* spp.) trees have encroached millions of hectares of sagebrush (*Artemisia* spp.)–bunchgrass communities and reduced understorey plant cover in the western USA, especially on shallow soils (Miller and Wigand 1994; Miller *et al.* 2005). This encroachment is associated with reduced fire frequency resulting from livestock grazing, which reduced fine fuels and herbaceous competition for resources (Miller and Rose 1999). Greater resource availability allowed shrubs to increase in size and improved safe sites for tree establishment (Miller and Rose 1999). The increase in juniper–piñon dominance is also related to a mild and wet climate around the late 1800s and early 1900s that was conducive to tree establishment and growth (Miller and Rose 1999). The shift from sagebrush–bunchgrass dominance to tree dominance has caused concerns of increased soil erosion (Wilcox and Breshears 1995), modified soil fertility, reduced forage production, reduced understorey fuel loads and diversity, altered wildlife habitat and increased fire intensity (Miller and Tausch 2001).

Dense juniper–piñon stands that have limited understorey vegetation are fire resistant because there are insufficient fine surface fuels remaining to carry fire under mild-moderate weather conditions (Miller and Tausch 2001). The high amounts of standing-tree fuel loads usually only burn during dry, windy weather but burn as intense crown fires uncharacteristic for the historic sagebrush–bunchgrass community (Miller and Tausch 2001; Miller *et al.* 2013). These high-intensity wildfires can result in invasive annual plant dominance (Tausch 1999*a*, 1999*b*; Miller and Tausch 2001) by species like cheatgrass (*Bromus tectorum* L.) that grow rapidly following fire with the associated increase in resource availability (McLendon and Redente 1991; Blank *et al.* 1994; Young *et al.* 1999). Invasive annual plant dominance can then lead to short fire-return intervals that prevent re-establishment of the historic sagebrush–bunchgrass community (Young and Evans 1973; Whisenant 1990; Young 1991).

Many residential areas now border rangelands where trees have encroached into the surrounding wildland–urban interface. There is great interest in reducing these fuel loads to reduce extreme wildfire behaviour and risk to life and property (Busse *et al.* 2005) and return the plant community back to the historic sagebrush–bunchgrass community (Smith *et al.* 2006). Unlike tall forests, there is insufficient market demand to make juniper–piñon harvesting profitable for most areas. Prescribed fire is often the least expensive method of reducing fuel loads but dangerously high fuel loads, air quality concerns, unfavourable burning weather and the risk of fire escaping into residential areas make using prescribed fire a challenge (USDA Forest Service 2005). Given these challenges, land managers often focus on mechanical means (USDA Forest Service 2005) to modify fuel beds and potential fire behaviour (Hood and Wu 2006; Kane *et al.* 2006).

Mechanical fuel-reduction treatments can be used to convert standing fuels into compact surface fuels to reduce fire intensity and rate of spread (Bradley *et al.* 2006; Hood and Wu 2006). Common mechanical methods include cutting trees with chainsaws and masticating trees with tractors. Tree mastication consists of a large tractor with rapidly spinning blades or spikes that shred tree canopies and trunks. Major benefits of treatments like tree mastication are the reduction of fuel height, which models have shown to greatly influence fire behaviour (Glitzenstein *et al.* 2006), and the increase in fuel bulk density, which can slow fire spread (Pyne *et al.* 1996). To achieve fuel reduction and post-treatment vegetation objectives at the same time, selecting a fuel-reduction method that not only reduces dangerous canopy fuel loads but also preserves desirable understorey plants would be ideal. Juniper–piñon cutting and mastication are examples of this type of treatment, which allows residual plants to provide habitat and serve as propagule sources following tree control.

Fuel loads are often analysed by fuel type and size to improve estimates of potential fire behaviour and to evaluate fuel-reduction effects on remaining vegetation. This is important because smaller fuels dry quickly and can support a faster rate of fire spread than larger fuels, whereas larger fuels greatly influence burning duration, fire severity and soil heating (Pyne *et al.* 1996). The longer burning times of larger fuels can be especially damaging to the growing points of remaining plants. Dead fuels are commonly categorised by the time-lag fuel moisture (TLFM) classes of 1, 10, 100 and 1000 h using the diameter ranges of 0–0.64, 0.64–2.54, 2.54–7.62, >7.62 cm respectively. This classification of fuels is based on the time required for fuels to lose approximately two-thirds of their initial moisture content (Bradshaw *et al.* 1983) because fuel moisture along with fuel size greatly influence fire behaviour.

Our goal was to quantify the effects of juniper–piñon infilling and fuel-reduction treatments on fuel load characteristics in sagebrush–bunchgrass ecosystems. We analysed fuel loads before and after burn, cut and mastication treatments and compared them with untreated control areas by fuel type and TLFM classes where appropriate, and quantified fuel depth and bulk density. Our objectives included: (1) determine the effects of juniper–piñon infilling represented by pretreatment tree cover on fuel loads; (2) determine the effects of fuel-reduction treatments on fuel loads over time; (3) determine if masticated-tree bulk density changes with tree infilling or masticated-tree depth; and (4) determine if masticated-tree depth accurately estimates total masticated-tree fuel loads. This is important because if depth measurements are found to accurately estimate fuel loads, that would save land managers time in the field compared with collecting and weighing fuels.

Materials and methods

Study locations

We measured fuel loads at Greenville (38°12′N, 112°48′W), Onaqui (40°13′N, 112°28′W), Scipio (39°17′N, 112°4′W) and Stansbury (40°35′N, 112°39′W) in western Utah. Each location had: loamy skeletal soils with carbonatic, mesic Typic Calcixerepts at Greenville; carbonatic, mesic, shallow Petrocalcic Palexerolls at Onaqui; mixed, superactive, mesic, shallow Calcic Petrocalcids at Scipio; and mixed, active, frigid Pachic Haploxerolls at Stansbury (Rau *et al.* 2011). Elevation ranges from 1700 to 1900 m across these four locations. Precipitation occurs mainly in the form of snow in winter and rain in spring and autumn leaving summers mostly dry. Each location had a gradient of tree cover that ranged from dominant sagebrush (*Artemisia* spp.)–bunchgrass vegetation nearly devoid of trees considered Phase I (*sensu* Miller *et al.* 2005) to tree-dominated areas with little remaining sagebrush–bunchgrass vegetation in Phase III. Utah juniper (*Juniperus osteosperma* [Torr.] Little) was the dominant tree at Onaqui and Stansbury and was co-dominate with two-needle piñon (*Pinus edulis* Engelm.) at Greenville and Scipio. Common shrub and grass species at these four locations included Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle and Young), rabbitbrush (*Chrysothamnus viscidiflorus* [Hook.] Nutt.), black sagebrush (*Artemisia nova* A. Nelson), mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* [Rydb.] Beetle), antelope bitterbrush (*Purshia tridentata* [Pursh] DC.), bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] A. Löve), Sandberg bluegrass (*Poa secunda* J. Presl), needle-and-thread (*Hesperostipa comata* [Trin. and Rupr.] Barkworth) and Indian ricegrass (*Achnatherum hymenoides* [Roem. and Schult.] Barkworth). Scipio and Stansbury also had patches of cheatgrass (*Bromus tectorum* L.) dominance. Our research was part of the Sagebrush Steppe Treatment Evaluation Project (SageSTEP) comparing the effectiveness of sagebrush-steppe restoration treatments (http://www.sagestep.org). More detailed descriptions of these locations are in McIver *et al.* (2010) and Young (2012).

Treatment implementation

The US Bureau of Land Management, US Forest Service and contractors burned, cut, masticated and left untreated 6–50-ha plots at Onaqui in the autumn of 2006 and Greenville, Scipio and Stansbury in the autumn of 2007. Onaqui was burned 30 September 2006 with 5–16 km h^{-1} winds, 12–19% relative humidity and 22–26°C dry temperatures. Greenville was burned 16 October 2007 with 0–24 km h^{-1} winds, 22–32% relative humidity and 14–19°C dry temperatures. Scipio was burned 16 October 2007 with 2–14 km h^{-1} winds, 28–41% relative humidity and 14–18°C dry temperatures. Stansbury was burned 20 September 2007 with 5–32 km h^{-1} winds, 13–18% relative humidity and 19–27°C dry temperatures. Greenville, Onaqui and Scipio required follow-up burns with fire fighters burning individual trees and shrubs to achieve at least 90% burn. This was completed within days of initial prescribed burns. Weather conditions and fuel loads at Stansbury supported a self-sustained backing fire. Cut treatments were applied by hand-crews with chainsaws to trees >0.5-m tall. Tractors masticated trees >0.5-m tall with Fecon® Bull Hog® attachments (Fecon, Lebanon, OH, USA).

Study design

Our randomised complete block experimental design included four research locations. Each location (block) contained a randomly located burn, cut and mastication treatment plot and untreated control plot. Research locations represented a randomly selected subpopulation of a larger population of potential research locations in the Great Basin. Treatment plots (6–50-ha) within locations had similar soils and pretreatment vegetation. Treatments were applied once at each of the four locations to allow us to determine regional responses across the eastern Great Basin. Woodland encroachment is a regional problem and knowing regional responses will assist government agencies that manage millions of hectares. Each treatment plot covered the range of tree encroachment from Phase I through Phase III (*sensu* Miller *et al.* 2005) to account for the effect of tree encroachment. We randomly dispersed 14–16 subplots (30 × 33 m) across the gradient of tree encroachment encompassed in each treatment plot to develop the tree cover covariate for mixed-model analysis of covariance. An example of quantifying the covariate with subplots can be found in section 7.8 of Littell *et al.* (2006). We used this tree cover covariate developed across subplots within treatment plots to adjust treatment plot comparisons for fuel loads so that comparisons were made at matching levels of tree cover. This allowed us to compare treatment effects across all locations at 5% increments of tree cover, which covered the range of tree encroachment, so that we could identify potential differences in fuel load response to treatment at low, medium and high levels of tree cover. If fuel load response differed by the level of tree cover, this would signify potential interaction between treatment and tree cover. Treatment effects were only compared across multiple locations (blocks) and not at individual locations or among individual subplots to prevent pseudoreplication. Vegetation, climate, soil nutrients and the soil environment were measured pretreatment and 1–3 years post-treatment in all treatment plots.

Field measurements

Standing-tree fuel loads were quantified pretreatment from tree height, canopy base height, widest canopy diameter (D1) and canopy diameter perpendicular (D2) to the widest diameter using allometric equations that produced fuel load estimates by TLFM class (Tausch 2009). Pretreatment tree cover (A) was estimated using $A = \pi (D1 \times D2)/4$. We estimated standing-tree fuel loads and tree cover for trees >0.5-m tall. We applied TLFM class terminology to standing woody plants as well as to downed fuels to make it evident that the plant material included in a particular TLFM class of masticated-tree fuel refers to the same type and size of plant material in standing trees. Surface woody fuel loads except masticated fuels were measured pretreatment and 3 years post-treatment using the planar-intersect method modified from Brown (1974) and Brown *et al.* (1982). We measured 10- and 100-h TLFM classes along three 30-m transects and the 1000-h TLFM class along five 30-m transects per subplot because the 1000-h TLFM class fuels were rare. These TLFM class values were adjusted according to the length of transects measured. The scarcity of the 1-h class of surface woody fuels in this semidesert system and amount of time required to search the 11 km of transects for twigs did not warrant its measurement with the planar-intersect method but these fuels were accounted for in line-intercept measurements of cover. Each fuel piece was assigned to a TLFM class by measuring the diameter perpendicular to fuel length where the transect intersected the fuel piece. Equations from Brown (1974) and Brown *et al.* (1982) converted TLFM class piece counts to fuel loads per unit surface area. All sizes of masticated-tree fuel loads were measured 1 year after treatment using 15 quadrats (0.25 × 0.25 m) along each of two 30-m transects per subplot with values averaged to the subplot level for analysis. We measured masticated fuel depth in each quadrat with a ruler to the nearest 0.5 cm. Masticated fuel depth was usually measured near the centre of these small quadrats and depth measurements were visually adjusted to account for surface undulation but when necessary additional depth measurements were taken. Masticated fuels were cut along the perimeter of each quadrat with pruning loppers, collected by hand, oven-dried for a minimum of 96 h at 60°C, and weighed by 1-, 10-, 100- and 1000-h TLFM classes. Shrub fuel loads were estimated for shrubs taller than 15 cm at five sampling points within a radius of 1, 2 or 3 m depending on shrub density so that at least 10 shrubs were measured per subplot using this nested circular-frame method (Bonham 1989). We measured shrub fuel loads along one 30-m transect per subplot on pre- and post-treatment years except at Onaqui pretreatment. Shrub fuel loads were estimated from allometric regression equations using the shrub canopy measurements of height, widest diameter and diameter perpendicular to the widest diameter using site-specific equations. We developed these regression equations using shrubs located outside the subplots by first measuring the canopy dimensions of 19–21 shrubs per major species per location across the range of shrub sizes. Second, we destructively sampled these shrubs, cut them into TLFM classes, oven-dried them at 50°C for 48 h, and weighed them by TLFM class. Third, the canopy dimensions of these shrubs were used to estimate shrub fuel loads through regression analysis. Last, we used these regression equations and shrub dimensions measured inside subplots to estimate subplot fuel loads without destructive sampling. Tree mound fuel loads measured pretreatment and 1 year post-treatment were collected in 0.25 × 0.25-m quadrats (Bonham 1989) from six trees per subplot unless fewer than six trees were found in a subplot. Tree mound fuel was collected near the base of the tree, one-third of the canopy radius from the base of the tree and, if canopy diameter was >4 m, a third sample was collected at two-thirds of the canopy radius. Sampled tree-mound wet weight was measured in the field and adjusted to dry weight using subsamples oven-dried at 50°C for 48 h. Herbaceous fuel loads were measured at pre- and 1–3 years post-treatment and collected in fifteen 0.5 × 0.5-m quadrats (Bonham 1989) along one 30-m transect per subplot. Herbaceous fuels included dead herbaceous material on the ground and live and dead standing herbaceous material clipped 1 cm above ground. Collected samples were oven-dried at 50°C for 48 h. Wood/bark cover was measured pre- and 1–3 years post-treatment using the line-intercept method (Bonham 1989) with pins dropped at 0.5-m increments along five 30-m transects that included fuels up to 2 m aboveground.

Data analysis

We analysed fuel load and cover response variables using mixed-model analysis of covariance and Proc Glimmix (SAS v9.3, SAS Institute, Cary, NC, USA). Analysis of covariance data requirements were met for total masticated fuel loads without data transformation and for the other response variables after square-root transformation based on evaluation of residuals plots. The 1000-h surface woody and masticated-tree fuel loads and burned-shrub fuels were mostly zero so these categories were excluded from analysis to meet the normally distributed residuals requirement. Onaqui treatments were applied 1 year earlier than at other locations in a stagger-start design (Loughin 2006). We accounted for this time difference by evaluating the number of years since treatment instead of calendar years. The stagger-start design and analysis of all locations together extends the scope of our inferences beyond one specific site and the weather patterns of one specific year. The fixed effects included masticated fuel depth, TLFM classes, treatment type, tree cover and years since treatment. Pretreatment tree cover was the covariate for most response variables whereas masticated-tree fuel depth was the covariate for total masticated fuel loads. We analysed fixed effects using F-tests from maximum likelihood estimations. Random effects included locations and subplots (30 × 33 m) where appropriate. Subplots were nested in treatment and location to account for potential spatial correlation. Years since treatment was included as a repeated measure where appropriate to account for potential temporal correlation. We compared fixed effects along the covariate at 5% increments using linear contrasts and least-squares means. We adjusted for false positives from multiple comparisons by using a critical α level of 0.01.

Results

Temperature and precipitation

Temperatures during our study were similar to long-term averages. Greenville, Onaqui and Scipio had annual-average air temperatures of 9–10°C, minimum temperatures of 0–2°C and maximum temperatures of 17–19°C. Annual average temperatures during 1970–2007 included minimum air temperatures of 0–3°C and maximum air temperatures of 16–17°C (PRISM Climate Group 2008). Greenville annual precipitation was 193 mm in 2009 and 387 mm in 2010, and the 1997–2007 annual-average precipitation was 334 mm (PRISM Climate Group 2008). Onaqui annual precipitation was 259 mm in 2008, 287 mm in 2009 and 370 mm in 2010, and the 1997–2007 annual-average precipitation was 311 mm (PRISM Climate Group 2008). Scipio annual precipitation was 280 mm in 2009 and 443 mm in 2010, and the 1997–2007 annual-average precipitation was 349 mm (PRISM Climate Group 2008). Stansbury precipitation and temperature data are not available during our study because a wildfire damaged data loggers in August 2009, but the 1997–2007 annual-average precipitation was 389 mm (PRISM Climate Group 2008).

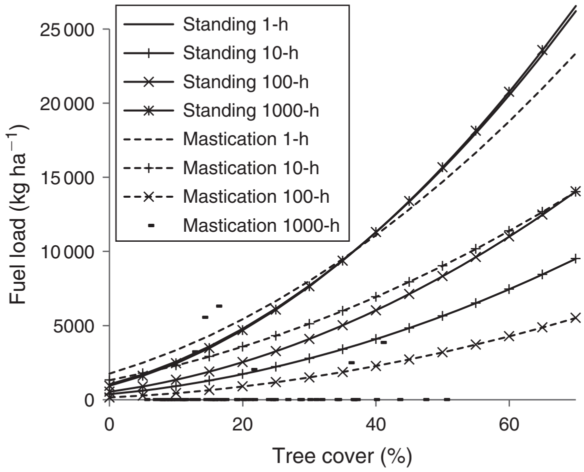

Fig. 1. Tree fuel loads for pretreatment standing trees and 1 year post-treatment masticated trees by time-lag fuel moisture (TLFM) class. Masticated tree 1000-h TLFM was not statistically analysed because only 6 of 60 measurements were > 0.

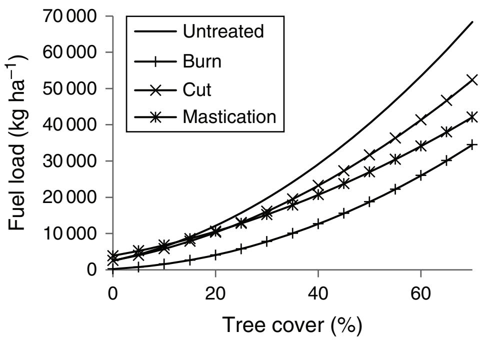

Fig. 2. Tree mound fuel loads for untreated and 1 year post-treatment plots.

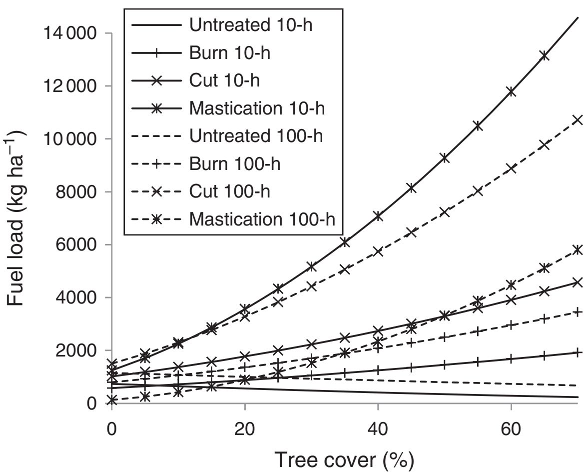

Fig. 3. Surface wood fuel loads for untreated and 1 year post-treatment plots by time-lag fuel moisture (TLFM) class.

Tree cover effect

Tree cover, representing the level of tree infilling, affected all fuel loads ($P < 0.001$; Figs 1–6; Table 1). Tree fuels generally increased with tree cover whereas non-tree fuels (shrubs,

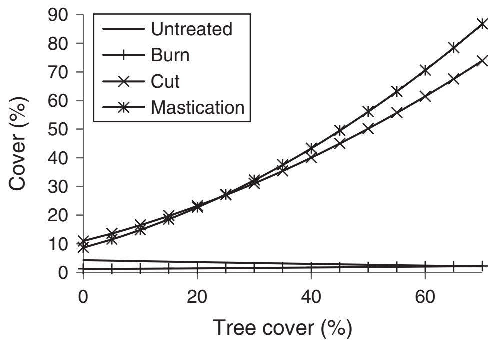

Fig. 4. Wood/bark cover fuel loads for untreated and 1 year post-treatment plots.

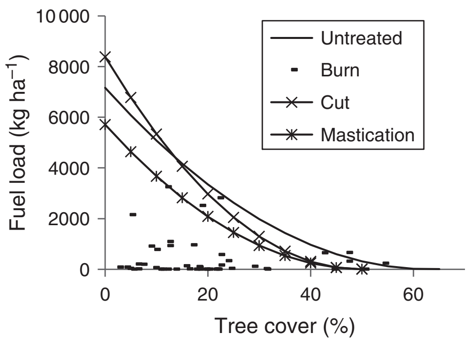

Fig. 5. Standing shrub fuel loads for untreated and 3 years post-treatment plots. So few shrubs remained post-burn that they could not be statistically analysed.

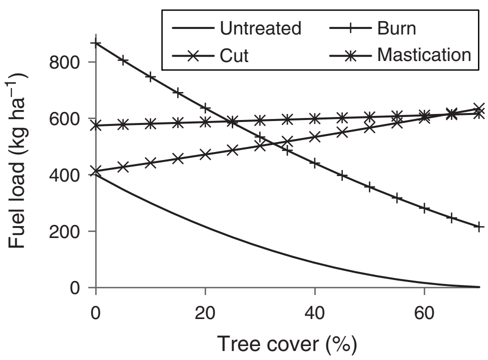

Fig. 6. Herbaceous fuel loads for untreated and 3 years post-treatment plots.

herbaceous) usually decreased. Tree-mound fuel loads increased with tree cover in all areas pre- and post-treatment ($P < 0.01$; Fig. 2). After treatment, surface-woody 10- and 100-h fuel loads increased with tree cover in burn and cut treatments ($P < 0.01$; Fig. 3). Masticated-tree 1-, 10- and 100-h fuel loads increased with tree cover ($P < 0.01$; Fig. 1). Wood/bark cover increased with tree cover in cut and mastication treatments ($P < 0.01$; Fig. 4). Post-treatment differences in surface-woody fuels and wood/bark cover were not due to pre-existing conditions because pretreatment surface-woody 10- and 100-h fuel loads and wood/bark cover did not change with tree cover ($P > 0.01$; Figs 3, 4). Unlike fuel types derived from trees, shrub fuel loads consistently decreased with increased tree cover during pre- and post-treatment years in cut, mastication and untreated areas ($P < 0.01$; Fig. 5). Herbaceous fuel loads decreased with increased tree cover before treatment and 1 year post-treatment but by 3 years post-treatment cut and mastication herbaceous fuels no longer decreased with increased tree cover ($P < 0.01$; Fig. 6).

Tree reduction and years since treatment effects

All tree reduction treatments increased surface-woody 10- and 100-h fuel loads whereas only cut and mastication increased wood/bark cover ($P < 0.01$; Fig. 3; Table 2). Burn was the only treatment to decrease tree mound fuel loads and wood/bark cover ($P < 0.01$; Figs 2, 4; Table 2). Cut and mastication had little effect on shrub fuel loads whereas burned shrubs could not be statistically analysed because of insufficient fuel remaining with 42% of burned subplots lacking shrubs ($P < 0.01$; Fig. 5; Table 2). A basic average value could be calculated for the burned shrub fuels, which was 242 kg ha^{-1} (s.d. 529) with values that ranged from 0 to 3254 kg ha^{-1}. Years since treatment was also important because all treatments increased herbaceous fuel loads but not until 2 years post-treatment ($P < 0.01$; Fig. 6; Table 2).

Mastication converted large branches and tree stems into masticated 10- and 100-h fuel loads ($P < 0.01$; Fig. 1; Table 2). Mastication reduced tree stem fuel loads so much that only 6 of the 60 masticated subplots had 1000-h fuels (Fig. 1). Masticated-tree 1000-h fuels averaged 391 kg ha^{-1} (s.d. 1281) and ranged from 0 to 6299 kg ha^{-1} 1 year post-treatment.

Across treatment differences

All pretreatment fuel loads were similar among treatments for most of the tree cover range, indicating that differences among treated and untreated areas were not due to pre-existing conditions ($P > 0.03$; Table 3). Mechanical treatments increased 10-h and 100-h fuel loads and wood/bark cover whereas burning only increased 100-h fuel loads and reduced wood/bark cover compared with untreated areas ($P < 0.01$; Figs 3, 4; Table 3). Cut treatments had two to seven times greater surface-woody 100-h fuel loads than burn and mastication treatments ($P < 0.01$; Fig. 3; Table 3). Only burn treatments reduced tree mound fuel loads ($P < 0.01$; Fig. 2; Table 3). Starting 2 years post-treatment in all treated areas, herbaceous fuel loads were greater than untreated areas ($P < 0.01$) but not different from each other ($P > 0.01$; Fig. 6; Table 3).

Treatment effects on fuel load distribution among time-lag fuel moisture classes

Treatments altered the distribution of fuel loads among TLFM. Before treatment, standing-tree 1-h fuel loads and tree stems (1000-h) were similar in quantity ($P > 0.1$; Fig. 1; Table 4) with two to three times more fuel than 10- and 100-h fuels ($P < 0.01$). Standing tree 100-h fuels had twice the fuel of 10-h fuels ($P < 0.01$; Fig. 1; Table 4). However, after treatment masticated-tree 10-h fuels were three to seven times greater than 100-h fuels ($P < 0.01$; Fig. 1; Table 4) because mastication converted larger TLFM fuels into smaller fuels. Masticated-tree 1-h fuels were

Table 1. Mixed-model analysis of covariance and Type III *F*-tests from restricted maximum likelihood estimation
Den DF, denominator degrees of freedom; Num DF, numerator degrees of freedom; TLFM, time-lag fuel moisture

Fuels analysed	Effect	Num DF	Den DF	*F*-value	*P*-value
Standing tree	Treatment	3	184	0.7	0.5782
$R^2 = 92$	TLFM	3	57	38.3	<0.0001
	Tree cover	1	236	3227.9	<0.0001
	Treatment × TLFM	9	57	0.3	0.9761
	Treatment × tree cover	3	243	1.2	0.3001
	Tree cover × TLFM	3	687	367.0	<0.0001
	Tree cover × treatment × TLFM	9	686	2.0	0.0344
Mastication	Tree cover	1	62	183.0	<0.0001
$R^2 = 82$	Year[A]	1	73	1.1	0.3087
	TLFM	3	37	9.4	<0.0001
	Year × TLFM	2	37	7.4	0.0020
	Tree cover × year	1	63	1.4	0.2497
	Tree cover × TLFM	3	282	18.0	<0.0001
	Tree cover × year × TLFM	2	282	3.2	0.0425
Wood	Treatment	3	23	5.7	0.0046
$R^2 = 78$	Year	3	156	2.9	0.0375
	TLFM	1	21	8.2	0.0092
	Tree cover	1	232	79.9	<0.0001
	Year × treatment	7	158	1.2	0.3149
	Treatment × TLFM	3	21	5.4	0.0066
	Tree cover × treatment	3	238	27.5	<0.0001
	Year × TLFM	3	133	2.0	0.1233
	Tree cover × year	3	510	29.8	<0.0001
	Tree cover × TLFM	1	268	2.5	0.1179
	Year × treatment × TLFM	7	131	2.4	0.0226
	Tree cover × year × TLFM	3	811	0.2	0.8888
	Tree cover × year × treatment	7	615	5.8	<0.0001
	Tree cover × treatment × TLFM	3	242	2.2	0.0906
	Tree cover × year × treatment × TLFM	7	804	1.9	0.0683
Shrub	Treatment	2	33	2.2	0.1287
$R^2 = 92$	Year	3	92	1.2	0.3253
	Tree cover	1	174	150.1	<0.0001
	Treatment × year	6	95	1.9	0.0952
	Tree cover × treatment	2	128	0.7	0.5137
	Tree cover × year	3	276	0.9	0.4699
	Tree cover × treatment × year	6	312	0.3	0.9492
Tree mound	Treatment	3	24	1.7	0.1856
$R^2 = 65$	Year	1	28	4.4	0.0461
	Tree cover	1	232	895.1	<0.0001
	Treatment × year	3	27	8.3	0.0004
	Tree cover × treatment	3	232	1.8	0.1430
	Tree cover × year	1	235	11.6	0.0008
	Tree cover × treatment × year	3	234	2.1	0.0966
Herbaceous	Treatment	3	17	1.7	0.1977
$R^2 = 84$	Year	3	70	4.5	0.0061
	Tree cover	1	238	126.8	<0.0001
	Treatment × year	9	70	3.0	0.0043
	Tree cover × treatment	3	240	8.2	<0.0001
	Tree cover × year	3	531	15.2	<0.0001
	Tree cover × treatment × year	9	588	4.9	<0.0001
Wood/bark cover	Treatment	3	20	24.5	<0.0001
$R^2 = 95$	Tree cover	1	266	208.7	<0.0001
	Year	3	106	8.3	<0.0001
	Treatment × tree cover	3	267	69.7	<0.0001
	Treatment × year	9	106	10.2	<0.0001
	Tree cover × year	3	526	30.3	<0.0001
	Treatment × year × tree cover	9	578	10.1	<0.0001
Masticated depth-to-load $R^2 = 92$	Masticated-tree depth	1	58	622.8	<0.0001

[A]Years since treatment including pretreatment where applicable.

Table 2. Tree cover (%) where post-treatment fuel loads (kg ha^{-1}) and wood/bark cover (%) differ from pretreatment and where post-treatment years differed among years since treatment as determined from linear-contrasts ($P < 0.01$)

–, no differences among years since treatment; na, no measurement of fuels

Treatment	Fuel type	Pretreatment –1	Pretreatment –2	Pretreatment –3	Post-treatment 1–2	Post-treatment 1–3	Post-treatment 2–3
Untreated	Surface wood 10 h	–	–	–	–	–	–
	Surface wood 100 h	–	–	–	–	–	–
	Shrub	–	–	–	–	–	–
	Tree mound	–	na	na	na	na	na
	Herbaceous	–	–	–	–	–	–
	Wood/bark cover	–	10–20	–	–	–	–
Burn	Surface wood 10 h	40–70	–	–	–	–	–
	Surface wood 100 h	30–70	45–70	30–70	–	–	–
	Shrub	na	na	na	na	na	na
	Tree mound	5–70	na	na	na	na	na
	Herbaceous	–	15–70	5–70	5–70	5–70	5–20
	Wood/bark cover	10–45	5–70	10–50	–	–	–
Cut	Surface wood 10 h	10–70	20–70	20–70	–	–	–
	Surface wood 100 h	10–70	10–70	15–70	–	–	–
	Shrub	–	–	–	–	–	–
	Tree mound	35–70	na	na	na	na	na
	Herbaceous	–	25–70	15–70	25–70	15–70	–
	Wood/bark cover	5–70	5–70	5–70	–	10–35	15–20
Mastication	Surface wood 10 h	5–70	na	na	na	na	na
	Surface wood 100 h	35–70	na	na	na	na	na
	Shrub	10–40	–	–	–	5–25	–
	Tree mound	45–70	na	na	na	na	na
	Herbaceous	–	25–70	15–70	25–70	15–70	–
	Wood/bark cover	5–70	5–70	5–70	–	25–35	–
	Mastication[A] 1 h	–	na	na	na	na	na
	Mastication 10 h	5–45	na	na	na	na	na
	Mastication 100 h	10–70	na	na	na	na	na

[A]Pretreatment mastication fuels were standing tree fuels and post-treatment mastication fuels were surface fuels.

greater than 1000-h fuels but insufficient 1000-h fuels remained after treatment in the burn, mastication and untreated areas for their statistical analysis. The 1000-h fuels averaged 885 kg ha^{-1} (s.d. 1 779) with individual values ranging from 0 to 20 047 kg ha^{-1}.

Masticated-tree depth and bulk density

Masticated-tree debris depth accounted for 92% of the variation in total masticated-tree fuel loads (Fig. 7; Table 1). One centimetre of masticated-tree depth represented 9086 kg ha^{-1} of masticated fuel ($P < 0.001$). However, masticated-tree bulk density was not associated with masticated-tree depth or tree cover as covariates ($P > 0.01$). Masticated-tree bulk density had a mean of 105 kg m^{-3} (s.d. 40) and ranged from 52 to 260 kg m^{-3}.

Discussion

Tree cover effect

Increased tree cover was associated with increased standing-tree fuel loads as expected but was not associated with surface-woody fuel loads before treatment. The lack of increased pretreatment surface wood and wood/bark cover with increased tree cover was likely due to the young age of these woodlands. Juniper and piñon trees can live for several hundred years and these woodlands have mostly established since the late 1800s (Miller and Tausch 2001). The scarcity of pretreatment large surface fuels in our study supports results found across other juniper–piñon woodlands (Despain and Mosley 1990; Miller and Rose 1995, 1999). After treatment, surface-woody fuels, masticated-tree fuels, wood/bark cover and tree mounds increased with former tree cover. Even after the burn treatments, surface-woody fuel loads increased with former tree cover probably because partially burned limbs fell to the ground replacing consumed, pretreatment surface-woody fuels. The difficulty of keeping prescribed fires burning in mild to moderate weather conditions on our sites illustrates the resistance of juniper–piñon woodlands to burning after the trees have greatly reduced understorey fine fuels (Miller and Rose 1999; Miller and Tausch 2001). In contrast, under extreme weather conditions crown fires in juniper–piñon can burn intensely (Miller and Tausch 2001) and endanger life and property, especially at the wildland–urban interface. These intense crown fires can reduce forage production, alter wildlife habitat and increase the risk of erosion and invasive annual plant species dominance (Wilcox and Breshears 1995; Tausch 1999*a*, 1999*b*; Miller and Tausch 2001). Invasive annual plants mature and dry earlier in summer than perennial grasses and can result in shorter fire-return

Table 3. Tree cover (%) where fuel loads (kg ha^{-1}) and wood/bark cover (%) differed among treatment types as determined from linear-contrasts ($P < 0.01$)

–, no differences among treatment types; na, no measurement of fuels

Fuel type	Years since treatment	Untreated–Burn	Untreated–Cut	Untreated–Mastication	Burn–Cut	Burn–Mastication	Cut–Mastication
Standing tree 1 h	Pretreatment	–	–	–	–	–	–
Standing tree 10 h	Pretreatment	–	–	–	–	–	–
Standing tree 100 h	Pretreatment	–	–	–	–	–	–
Standing tree 1000 h	Pretreatment	–	–	–	–	–	65–70[A]
Surface wood 10 h	Pretreatment	–	–	–	–	–	–
	1	–	20–70	10–70	20–70	5–70	15–70
	2	–	25–70	na	30–40	na	na
	3	–	25–70	na	–	na	na
Surface wood 100 h	Pretreatment	–	–	–	–	–	–
	1	45–70	15–70	5–15, 40–70	10–70	5	5–70
	2	60–70	10–70	na	5–70	na	na
	3	30–70	15–70	na	15–60	na	na
Shrub	Pretreatment	na	–	–	na	na	–
	1	na	–	5–35	na	na	–
	2	na	–	–	na	na	–
	3	na	–	–	na	na	–
Tree mound	Pretreatment	–	–	–	–	–	–
	1	5–70	–	–	5–65	5–40	–
Herbaceous	Pretreatment	–	–	–	–	–	–
	1	–	–	–	–	5	–
	2	20–70	25–70	10–70	–	–	–
	3	5–70	25–70	15–70	5	–	–
Wood/bark cover	Pretreatment	–	–	–	–	–	–
	1	5–20	5–70	5–70	5–70	5–70	–
	2	5–20	5–70	5–70	5–70	5–70	–
	3	5–30	5–70	5–70	5–70	5–70	–

[A]Range of tree cover (%) where standing tree 1000-h fuel loads differed between cut and mastication treatments ($P < 0.01$).

intervals that limit the resilience and return of the historic sagebrush–bunchgrass community (Miller and Tausch 2001; D'Antonio and Chambers 2006; Tausch and Hood 2007).

The decrease in pretreatment shrub and herbaceous fuel loads with increased tree cover supports results from earlier studies (Miller and Tausch 2001). Prior to Eurasian settlement, the historic fire return interval was sufficient to restrict juniper–piñon encroachment of sagebrush–bunchgrass communities (Miller and Tausch 2001; Miller *et al.* 2005). However, the fine fuels that historically helped carry fire were reduced by over grazing in the late 1800s and early 1900s, resulting in reduced fire frequency (Miller and Tausch 2001). Additionally, milder winters and greater precipitation during 1850–1916 favoured juniper growth (Miller and Tausch 2001). Under these conditions, trees proved superior competitors for resources over the historic plant community (Miller and Tausch 2001). Several characteristics have enabled these trees to acquire resources and reduce the sagebrush–bunchgrass community. Juniper trees start transpiring early in spring and reduce the amount of soil water remaining for understorey plant species (Angell and Miller 1994). Shallow juniper roots compete with grass roots for resources (Emerson 1932). Juniper roots hydraulically move soil water deeper into the soil profile away from shallow-rooted species (Leffler *et al.* 2002). Juniper canopies intercept precipitation and thereby reduce the amount of water reaching the ground (Evans 1988). Their canopies and mounds result in hydrophobic soil layers that reduce the availability of water to subcanopy plants (Madsen *et al.* 2011). Tree canopies and mounds shade part of the surrounding area and thereby reduce soil temperatures and growth rates (Facelli and Pickett 1991; Lebron *et al.* 2007; Matsushima and Chang 2007; Lin 2010). Additionally, juniper trees redistribute soil nutrients from surrounding areas to their subcanopy mounds through roots and litter-fall (fallen foliage, twigs and berries; Klopatek 1987; McDaniel and Graham 1992; Davenport *et al.* 1996).

Shrub fuel loads 3 years post-treatment and herbaceous fuel loads 1 year post-treatment continued to decrease in all treated areas where tree cover had increased. This decrease was likely due to the sagebrush–bunchgrass community having been displaced by decades of juniper–piñon dominance. These results illustrate that the pretreatment plant community and its response to juniper control treatments largely determine post-treatment plant community composition (Miller and Tausch 2001). By 2 and 3 years after cut and mastication treatments, herbaceous fuel loads in areas of dense juniper encroachment had returned to unencroached levels. This likely resulted from the increased resource availability (Young *et al.* 2013*a*; Roundy *et al.* 2014*b*) that commonly follows removal of the main resource users (D'Antonio *et al.* 2009), especially where tree cover was highest. Prediction of which plant species will dominate long term

Table 4. Tree cover (%) where fuel loads (kg ha^{-1}) differed among time-lag fuel moisture (TLFM) classes as determined from linear-contrasts ($P < 0.01$)

–, no differences among TLFM classes; na, no measurement of fuels

Treatment	Years since treatment	TLFM class	Standing tree	Surface wood[A]
Untreated	Pretreatment	1–10 h	5–70	na
	Pretreatment	1–100 h	5–70	na
	Pretreatment	1–1000 h	–	na
	Pretreatment	10–100 h	15–70	–
	Pretreatment	10–1000 h	5–70	na
	Pretreatment	100–1000 h	5–70	na
	1	10–100 h	na	–
	2	10–100 h	na	–
	3	10–100 h	na	–
Burn	Pretreatment	1–10 h	5–70	na
	Pretreatment	1–100 h	5–70	na
	Pretreatment	1–1000 h	–	na
	Pretreatment	10–100 h	15–70	–
	Pretreatment	10–1000 h	5–70	na
	Pretreatment	100–1000 h	5–70	na
	1	10–100 h	na	–
	2	10–100 h	na	–
	3	10–100 h	na	30–70
Cut	Pretreatment	1–10 h	5–70	na
	Pretreatment	1–100 h	5–70	na
	Pretreatment	1–1000 h	–	na
	Pretreatment	10–100 h	15–70	–
	Pretreatment	10–1000 h	5–70	na
	Pretreatment	100–1000 h	5–70	na
	1	10–100 h	na	15–70
	2	10–100 h	na	5–70
	3	10–100 h	na	5–70
Mastication	Pretreatment	1–10 h	5–70	na
	Pretreatment	1–100 h	5–70	na
	Pretreatment	1–1000 h	–	na
	Pretreatment	10–100 h	15–70	–
	Pretreatment	10–1000 h	5–70	na
	Pretreatment	100–1000 h	5–70	na
	1	1–10 h	na	20–70
	1	1–100 h	na	5–70
	1	10–100 h	na	5–70

[A]Includes 1-h fuels with leaf scales and fruit.

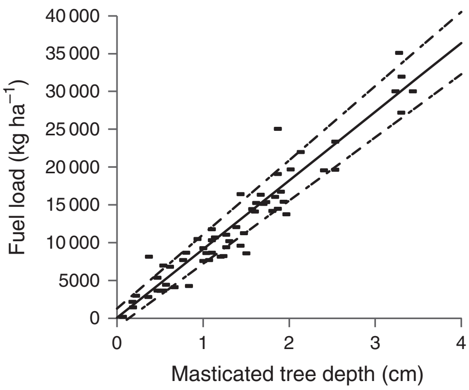

Fig. 7. Masticated-tree depth-to-fuel-load relationship 1 year post-treatment. Dashed lines are confidence intervals (95%).

following juniper–piñon reduction remains to be seen and will require long-term studies like the ongoing SageSTEP (McIver *et al.* 2010). For sites where weedy species dominate the pretreatment understorey, ecosystem resistance to invasive species is considered to be low (Chambers *et al.* 2014*a*; Roundy *et al.* 2014*a*) because the weedy species will likely dominate post-treatment and maintain a degraded ecological state (Miller and Tausch 2001). Because cheatgrass is more adapted to warmer sites (Chambers *et al.* 2007), its dominance is more likely after fuel control treatments on sites with a mesic soil temperature regime (Chambers *et al.* 2014*b*; Miller *et al.* 2014). Ecosystem resilience (recovery after disturbance) and resistance to weed dominance is most likely for sites with higher perennial grass cover (Chambers *et al.* 2014*b*; Roundy *et al.* 2014*a*). Perennial grasses reduce growth of invasive annual grasses by limiting resources available to them (Chambers *et al.* 2007), and help reduce interspace run-off and erosion after tree reduction (Williams *et al.* 2014).

Treatment and years since treatment effects

Burn treatments as well as cut and mastication treatments increased herbaceous fuel loads by 2 and 3 years post-treatment compared with pretreatment levels. We expect that this increase in herbaceous production resulted from increased soil water and N availability after tree reduction. Burning or cutting of juniper–piñon trees has increased the time of available soil water in spring by 15–26 days and mastication of trees has quadrupled inorganic soil N in Phase III (*sensu* Miller *et al.* 2005) dense woodland encroachment of sagebrush–bunchgrass communities (Young *et al.* 2013*a*, 2013*b*, 2014; Roundy *et al.* 2014*b*). Burn treatments increased surface-woody 100-h fuel loads, especially where tree cover was greatest. This result was due to fire intensity not being sufficient to fully consume standing trees and allowing partially burned branches and stems to collect on the ground, as shown in previous research (Bradley *et al.* 2006). The burn treatment also increased surface-woody 10-h fuel loads 1 year post-treatment but not 2–3 years post-treatment, probably because physical weathering broke down the remaining charred 10-h fuels into smaller TLFM classes. These small increases in burnt surface-woody fuel loads are not expected to greatly increase potential fire behaviour because of their low quantity and the charred wood has already lost some of its volatile material that supports flaming combustion (Pyne *et al.* 1996). However, more research is needed to predict fire behaviour in these systems and how other factors like weather and site characteristics influence fire behaviour and severity in these types of fuel load conditions. Burn treatments decreased shrub and tree mound fuel loads largely because fire crews had to follow initial ignition with individual ignition of shrubs and trees where fine surface fuels were insufficient to carry fire in the mild-moderate weather conditions. It is common for fire crews to employ multiple techniques, including modifying fuels before treatment (e.g. Bates *et al.* 2011), to achieve desired outcomes, especially where fine surface fuels are insufficient to carry prescribed fire. The amount and type of methods used to burn an area have the potential to cause high variability in fire behaviour and burn severity among prescribed burns but taking extra measures to complete a burn and meet the burn prescription where surface fuels are minimal is common.

The most apparent effect of cut and mastication treatments is the conversion of standing trees to surface fuels. This conversion should reduce overall fire intensity by eliminating tree canopy fuels (Pollet and Omi 2002) and the opportunity for crown fire but the additional surface fuels could increase fireline intensity. Prior to woodland encroachment in the sagebrush steppe, high-intensity fires were infrequent but killed most of the shrubs and encroaching trees (Baker and Shinneman 2004; Romme *et al.* 2009). These pre-woodland encroachment fires produced mosaics of burned and unburned vegetation (Keane *et al.* 2008) ideal for sage-grouse habitat (Crawford *et al.* 2004). In areas of dense juniper infilling, surface fuels are inadequate to carry a surface fire that will kill mature trees (Bates *et al.* 2011; Romme *et al.* 2009). After woodland encroachment the mean fire return interval is largely determined by the probability of ignition and fire season length, with weather conditions greatly influencing fire spread dynamics and tree mortality (Romme *et al.* 2009). To maintain prescribed fire under desirable weather conditions, cutting up to 25% of juniper trees is necessary to increase surface fuels after dense juniper infilling has greatly reduced surface fine fuels (Bates *et al.* 2011). Published reports of detailed fire behaviour in cut juniper woodlands are rare but flame lengths of 2–11 m and burn durations of 5–55 min after cutting 25–50% of juniper trees have been recorded during October burns (Bates *et al.* 2011).

Increases in surface fuel loads, wood/bark cover, fuel continuity and the conversion of large fuels into small masticated fuels are expected to increase fire size, severity, residence time, rate of spread and homogenise post-fire vegetation (Pyne *et al.* 1996; Crawford *et al.* 2004; Bradley *et al.* 2006; Keane *et al.* 2008; Busse *et al.* 2010). Severe large-extent fires result in degradation of sage-grouse habitat until mature sagebrush re-establishes (Crawford *et al.* 2004). These changes also slow the return of sufficient sagebrush for sage-grouse cover due to lack of propagules and short distance seed dispersal by sagebrush (Crawford *et al.* 2004). Over time, surface fuels from fuel-reduction treatments are expected to decompose, compact and pose less of a fire hazard but the long-term rate of change in fire behaviour is unknown (Bradley *et al.* 2006; Shakespear 2014).

Standard fire behaviour models are not available for masticated-tree fuel loads. Some studies have tried to apply standard models to masticated fuel beds but the results are commonly an over-prediction of the rate of fire spread and under-prediction of flame length (Knapp *et al.* 2011). Because fire behaviour research is lacking in masticated juniper trees, we refer to the limited fire behaviour work done with masticated understories in mixed conifer forests of the Sierra Nevada Mountains (e.g. Stephens and Moghaddas 2005; Knapp *et al.* 2011). Land managers want to reduce potential fire behaviour in both juniper–piñon-encroached sagebrush steppe and forests after years of fuel build up, and they often do this by mechanically converting standing fuels into surface fuels. Methods in forested systems often include thinning by removal of mid-size trees that have commercial value before mastication of small trees or shrubs. In forest ecosystems with commercial timber, managers also want to preserve selected tall timbers that have major economic and ecological value. However, in juniper-encroached sagebrush steppe (not historic juniper woodlands), trees were not part of the historic plant community and not a major priority for protection from fire. In these systems, retaining perennial grasses, forbs and shrubs is critical to ecosystem resilience and restoration of historic plant communities. In ponderosa pine (*Pinus ponderosa* Laws) forests, mastication of small trees moderates fire behaviour by reducing ladder fuels and the potential for crown fire but increases surface fuel loads and continuity (Stephens and Moghaddas 2005). These surface fuels are expected to increase the period of flaming combustion and fire severity (Stephens and Moghaddas 2005). Average rates of fire spread in the masticated understorey of mixed-conifer stands have been reported between 33 and 222 m h^{-1} with average flame lengths between 0.7 and 1.1 m (Kobziar *et al.* 2009; Knapp *et al.* 2011). With increased surface fuels soil heating may increase, raising concerns about the negative effects of fire severity on desirable residual plants. In controlled experiments, burning of masticated fuel beds can heat near-surface soil temperatures to the point of plant mortality (Busse *et al.* 2005). If maximum juniper mortality is the goal, burning during the warm season is most effective, but may also damage desirable understorey species (Bates *et al.* 2011). To minimise the risk of damaging desired understorey species and the risk of runaway prescribed fires, burning of mechanically treated juniper trees is sometimes conducted during winter or early spring (Bates *et al.* 2011) because high soil moisture conditions minimise heat-related mortality (Busse *et al.* 2010). Following mechanical fuel control and prescribed fire, surface fuels and potential flame lengths are reduced. This should make future fires easier to control compared with crown fires (Agee and Skinner 2005).

Masticated-tree fuel bed characteristics

Masticated-tree fuel depth accurately estimated masticated-tree fuel loads, which supports results from earlier research (Hood and Wu 2006; Kane *et al.* 2009; Battaglia *et al.* 2010). These results suggest that measurements of masticated-tree fuel depth can rapidly assess masticated-tree fuel loads on sites similar to this study without the need to collect and process large samples of masticated-tree material. Before this rapid assessment method can be applied to new areas, regression equations relating fuel depth to fuel loads should be developed for the target area and plant community. Fuel depth and bulk density are both important determinants of fire behaviour (Pyne *et al.* 1996). These factors are part of the Rothermel fire-spread model and an increase in these factors increases soil heating and fire severity (Pyne *et al.* 1996). If masticated-tree fuel loads settle over time as we visually observed, then the expected increase in bulk density should also slow the rate of fire spread over time (Pyne *et al.* 1996) compared with initial post-mastication conditions.

Conclusions

Our study provides practical information related to commonly used fuel-reduction treatments to aid fuel management planning and fire behaviour estimations in juniper–piñon encroached sagebrush–bunchgrass communities. Tree encroachment into sagebrush–bunchgrass ecosystems greatly increases canopy fuel loads and can severely reduce understorey vegetation. These tree-dominated ecosystems are often resistant to fire because of insufficient surface fuel but can burn intensely with crown fires under extreme weather conditions (Miller *et al.* 2013). All treatments reduced tree canopy fuels and the potential for crown fire but only prescribed fire reduced fuel loads. Increased surface fuel loads from mechanical juniper control treatments can be reduced by cool-wet season, spot burning to reduce future fire severity unless fuel removal is practical. The use of periodic prescribed fire could extend the time of treatment effectiveness and lessen the future need to mechanically retreat juniper–piñon encroachment. As efficient use of time and resources is always a management concern, there is potential for using masticated-tree fuel depth measurements to rapidly assess masticated-tree fuel loads after the initial development of regression equations that relate fuel depths to fuel loads.

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