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Effects of fuels reductions on plant communities and soils in a Piñon-juniper woodland

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ABSTRACT

Over the past decade, a variety of fuels reduction strategies have been implemented across western US forests to lower the risk of high severity fires. In two separate studies, we evaluated the short-term effects (<two growing seasons) of hand thinning (lop & scatter, pile burn) and mechanical mastication on understory plant communities and soil resources in an upland Piñon-juniper woodland. All treated sites were compared to a nearby untreated control site. After one growing season, understory plant cover was 4–5.5 times greater in hand-thinned treatments (lop & scatter pile burn), while understory cover in mastication treatments was 15 times greater following two growing seasons, compared to untreated controls. *Bromus tectorum*, an invasive annual grass, was present in all treated sites and absent from controls. Soil aggregate stability, an indicator of overall soil quality, was lower in the pile burn and mastication sites. Nitrogen fixation potential was low across all sites, but lowest in two treated sites (lop & scatter and mastication). This study suggests that different fuels reduction techniques generally have positive effects on total understory plant cover, but treatments that involve burning of slash materials may have more negative effects on site stability than alternative treatment options.

1. Introduction

In 2003 legislation was introduced under the Healthy Forest Restoration Act (HFRA) with the intent to reduce fire risk through a combination of controlled burns and fuels reduction treatments across a range of US forest types. Along with reducing risk of high severity fires, the legislation includes the explicit goal of restoring forest ecosystems by decreasing fuel loads in a manner that maintains ecological integrity and promotes a return to historic fire cycles (HFRA, 2003). Within the context of maintaining ecological integrity, however, the effects of fuels-reduction treatments on plant communities and soil processes are not well documented (Boerner et al., 2009), especially in the semi-arid woodland environments of the western U.S. (Brockway et al., 2002; Owen et al., 2009).

Piñon-juniper woodlands cover ca. 136 million hectares of arid and semi-arid land in the western U.S. and are among the most extensive vegetation types in the United States (Mitchell and Roberts, 1999). In the past 150 years, Piñon-juniper landscapes have undergone significant expansion and increases in stand density (Blackburn and Tueller, 1970; Miller and Tausch, 2002; Mitchell and Roberts, 1999). There is evidence that following Piñon-juniper woodland expansion into grassland and shrubland ecosystems, understory cover declines, especially forage species for livestock and wildlife (Pierson et al., 2008). Studies have shown that declines in understory plant cover may lead to increased susceptibility to soil erosion (Davenport et al., 1998; Pierson et al., 2007). Thus, to maintain site integrity (HFRA, 2003), land managers are tasked not just with reducing fuel loads, but also the restoration of understory plant communities.

Upland Piñon-juniper woodlands (~1930–2330 m.a.s.l) occurring on shallow soils tend to have lower productivity and vascular plant cover (Bowker et al., 2006). In these ecosytems, interspaces between vascular plants are often heavily colonized by biological soil crusts (BSCs), which may represent up to 70% of the biotic groundcover (Belnap, 1990). The communities of lichens, fungi, mosses, and cyanobacteria that colonize the top few millimeters of soil surfaces effectively bind soil particles (Belnap and Lange, 2002), decreasing wind and water erosion (Belnap and Gillette, 1998; Evans and Johansen, 1999).

In addition to preventing nutrient loss from erosion, nitrogen (N) fixation by diazotrophic cyanobacteria is an important N input

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pathway (Belnap, 2002). BSCs play an important role in soil nutrient dynamics of these nutrient poor systems (Evans and Ehleringer, 1993; Evans and Johansen, 1999). However, BSCs are highly susceptible to soil surface disturbance (Barger et al., 2006; Belnap and Lange, 2002; Mack and Thompson, 1982) and the suite of fuels-reduction treatments applied to Piñon-juniper woodlands may have direct, negative effects on BSC communities caused by the use of prescribed fire (Owen et al., 2009) and soil surface disturbance related to foot-traffic and heavy machinery (Belnap and Gillette, 1998).

In conducting this study, our goal was to explore the balance between ecological benefits and the potential ecological risks of several commonly implemented fuels reduction treatments in an upland Piñon-juniper woodland. Possible benefits from fuels reduction include increased understory plant cover (Owen et al., 2009; Brockway et al., 2002), increased plant diversity (Brockway et al., 2002), decreased soil erosion (Hastings et al., 2003), and increased soil moisture (Owen et al., 2009; Miller and Seastedt, 2009). Risks may include changes in plant community composition (Miller and Seastedt, 2009; Wolk and Rocca, 2009), increases in invasive species specifically *Bromus tectorum* (Owen et al., 2009; Wolk and Rocca, 2009), and increased soil erosion (Barger, unpublished data). Different treatment types may differ in their effects on the balance between risks and benefits associated with fuels reduction. For example, Owen et al. (2009) found that sites where mulch was burned in piles had lower soil aggregate stability, lower understory plant cover, lower soil moisture, and higher soil temperature than mastication sites where mulch was spread across the landscape.

Following a suite of fuels reduction treatments conducted by the Bureau of Land Management (BLM) in an upland Piñon-juniper woodland that was similar in climate, parent material, elevation, and vegetation communities in southeastern Utah, we examined changes in key ecological attributes within treated sites as compared to nearby untreated control sites. Downward trends in plant and forage cover, and increases in soil erosion over the past several decades suggested that this site was becoming highly degraded, a characteristic of other upland Piñon-juniper woodlands in the region (http://www.blm.gov/ut/st/en/fo/monticello/fire/fuels_management/Shay_Mesa_Restoration.html). In two separate studies (hand-thinning study and mastication study) we examined the short-term (one and two growing seasons) plant community and soil responses to treatments. We hypothesized that ecological risk such as exotic species introduction and soil surface disturbance is greatest within the first two growing seasons after treatment, and we focused our efforts on the shorter-term ecological responses in these time periods. Specifically, we addressed the following questions: 1) Do fuels reduction treatments result in increases in understory plant cover and if so, do these cover increases include colonization by exotic species? 2) Is there evidence that fuels reduction treatments and the associated soil surface disturbance results in declines in soil stability? 3) Do fuels reduction treatments result in differences in soil nutrient status particularly in relation to BSC cover and nitrogen fixation potential?

2. Methods

2.1. Study area

Our study sites were located on Shay Mesa in southeastern Utah, USA (2200 m.a.s.l; 57°58′N, 109°34′). Mean annual precipitation at these sites is highly variable ranging from 167 to 586 mm with a mean of 386 mm over the last century (1902–2010 average, Monticello, UT, 2079 m.a.s.l., Western Regional Climate Center, WRCC, 425805). Annual mean minimum and maximum temperatures at the sites are 0.5 °C and 15 °C respectively (WRCC). Soils were classified as a Bond-Rizno fine sandy loam complex (Lammars, 1991). These soils are shallow (10–50 cm), with high sand content (65–74%, Table 1) and are characterized as upland shallow loam (Piñon-juniper) ecological sites (NRCS, 2004). Plant species present across all sites were Colorado Piñon (*Pinus edulis*), Utah juniper (*Juniperus osteosperma*), Broom snakeweed (*Gutierrezia sarothrae*), rock goldenrod (*Petradoria pumila*), blue gramma (*Bouteloua gracilis*), Barkworth squirreltail (*Elymus elymoides*), rubber rabbitbrush (*Ericameria nauseosa*), and big sagebrush (*Artemisia tridentate*) (Barger, unpublished data).

In the 1960's the Shay Mesa study sites were chained, a thinning procedure that involves uprooting trees by dragging a thick chain between two vehicles, and were subsequently seeded with forage species. The study site was grazed by cattle until 2005, but since then has been free of livestock (Paul Plemons, BLM, personal communication).

2.2. Site selection

Approximately 1000 ha of Shay Mesa was treated for fuels reduction and habitat restoration from 2007 through 2009. In two separate studies we examined the short-term (one and two growing seasons) plant community and soil responses to treatments. In the first study, we assessed the effects of two different hand-thinning treatments: *pile burn*; where trees were manually cut with slash placed in discrete piles that were later burned, and *lop & scatter*; where trees were manually removed and debris scattered across the site. The pile burn and lop & scatter treatment areas were roughly 1 km^2 and approximately 1 km distant from each other. For each of these hand-thinned sites, field crews hand-cut trees with chainsaws. In the lop & scatter site, which was left unseeded, trees were left whole or cut into smaller pieces and then distributed evenly across the landscape by field crews 10 months prior to sampling (September, 2008, Fig. 1). In the pile burn site trees were cut into smaller sections and placed in 2 × 2 m paraboloid shaped piles that were spaced approximately 2 m apart (August, 2008). Piles were allowed to dry for 6 months and then burned (February, 2009, Fig. 1). Based on the geometry and spacing of the piles, these pile burn treatments burn approximately 30–40% of the land surface of the site. Approximately one month after the prescribed burn, pile burn sites were then harrowed and seeded by all terrain vehicles to imbed the seeds in soil.

In a second study, we established sites in areas that had been mechanically masticated one (GS1) and two (GS2) growing season prior to our study. Mastication was done with a Tigercat feller buncher with an attached Fecon brush cutter. The two mastication sites were aerially seeded before treatment in May 2009 (GS1) and December 2007 (GS2, Fig. 1). The GS1 and GS2 treatment areas were approximately 0.75 km^2 and approximately 300 m apart.

In all sites overstory tree cover was reduced from an estimated greater than 30% cover to less than 3% cover. We compared all

Table 1
Soil texture and bulk density. Values are means (±1 SE, $n = 20$). Lowercase letters indicate significant differences between the GS1 and GS2 sites.

	Hand-thinned study		Control	Mastication study	
	Pile burn	Lop & scatter		GS1	GS2
% Sand	74.4 (1.8)	72.3 (2.2)	70.6 (1.3)ab	76.3 (1.5)a	65.4 (2.6)b
% Silt	18.8 (1.5)	19.5 (1.6)	22.0 (0.9)ab	17.6 (1.3)a	26.43 (2.4)b
% Clay	6.80 (0.6)	8.23 (0.1)	7.41 (0.6)	6.14 (0.4)	8.14 (0.7)
Bulk density (g/cm^3)	1.22 (0.0)	1.26 (0.0)	1.20 (0.3)	1.23 (0.4)	1.18 (0.4)

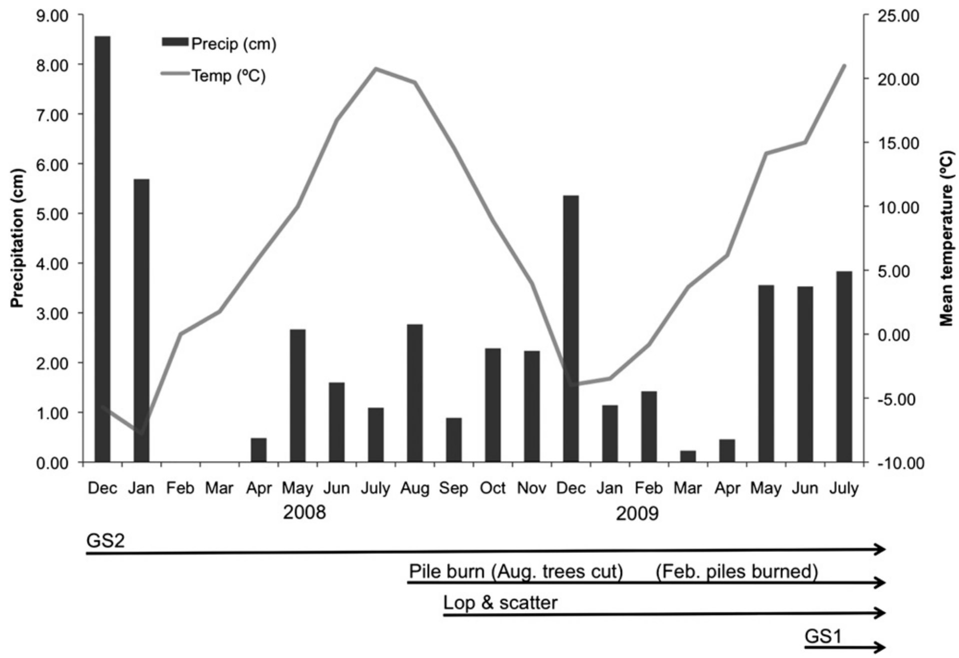

Fig. 1. Timing of treatments and local climate. Total precipitation by month and average monthly temperatures from December 2007 to July 2009. Timing of treatments is shown with arrows indicating the month during which thinning treatments were conducted. Note: pile burn was conducted in two phases, which are separately indicated on the same arrow.

treated sites to an untreated control site (0.4 km^2) located within 3 km of the treatment sites, that occurred on the same soil type, and which had similar grazing histories to the treated sites. A total of five separate sites were sampled following treatment by the BLM. We replicated within each site by establishing 10, 35 m transects ($n = 10$). Transect locations were randomly generated using Hawths Analysis Tool for ArcGIS (ESRI, Redlands, CA). All transects were located on slopes with a grade of 8% or less. The random placement of transects incorporated both burned and unburned parts of the landscape in the pile burn sites. The seed stock applied in the mastication and pile burn sites was a mixture of native and non-native species composed of 61% perennial grasses, 24% perennial forbs, and 15% shrubs.

2.3. Cover measurements

We used a line-point intercept method as described in Herrick et al. (2005) to estimate percent cover of vascular plants, ground cover, and soil surface characteristics. Due to difficulty in identifying seedlings to the species level in the field, plants were categorized into six functional groups: perennial forb, perennial grass, annual forb, annual grass, shrub, and tree. There were two perennial forb species, however, that were easily identifiable and widely distributed throughout the site. Thus we identified *G. sarothrae* and *P. pumila* (Laycock, 1967; Ralphs and Banks, 2009) to the species level (*Torr.*). Of particular interest in our study were the changes in *B. tectorum* cover with treatment. *B. tectorum* is an invasive annual grass that has been reported to increase in abundance following disturbances such as fire (Keeley, 2006). We included identification of *B. tectorum* to the species level since increases in cover may have future negative impacts in these ecosystems and *B. tectorum* is easily identified. Soil surface characteristics were categorized as bare soil, rock, lichen, moss, light or dark BSCs. We distinguished between light and dark crusts because the coloration of BSC communities can indicate different species composition and function (Yeager et al., 2004). Dark crusts tend have higher concentrations of the nitrogen-fixing cyanobacteria, *Nostoc commune* and *Scyntoma myochrus* which accounts for the higher nitrogen fixation rates observed in dark crusts (Barger et al., 2006). We recorded understory cover every 0.5 m, for a total of 70 data points per 35 m transect (700 per site).

Plant canopies can protect soil from erosion caused by raindrop impacts and wind, thus the relative proportion of open ground between plant canopies (canopy gap) can be an important indicator of susceptibility to wind and water erosion (Derner and Whitman, 2009; NRCS, 2010). Moreover, in large canopy gaps (>100 cm), wind velocity is higher near the soil surface, which can increase the risk to wind erosion (NRCS, 2010). We organized canopy gap sizes into four size classes: 20–50 cm, 51–100 cm, 101–200 cm, and >200 cm. We followed the gap-intercept protocol from Herrick et al. (2005) to measure canopy gap using the same transects. Gap distances were measured to the nearest 1 cm with canopy defined as >50% plant matter in a 3 cm window and a minimum gap size of 20 cm (Herrick et al., 2005).

Coarse woody debris embedded at the soil surface provides additional obstructive capacity to wind and water erosion. We measured interspaces between embedded woody debris (hereafter wood gap). Wood gaps were also organized into size classes as with plant canopies. Debris was defined as pieces of wood greater than 3 cm in diameter that would leave an indentation in the soil surface or significantly disturb the soil surface when removed (Herrick et al., 2005). Minimum gap size between woody debris was set at 20 cm.

2.4. Soil sampling and analysis

Treatment effects on soil surface stability were determined by field-based soil aggregate stability tests (Herrick et al., 2005). Along each transect, nine surface aggregate stability samples were collected at 4 m intervals. Values for aggregate stability range from 1 (no aggregate stability or formation of aggregates) to 6 (strong aggregation and highly resistant to disintegration).

In order to evaluate whether treatments resulted in soil compaction we measured soil bulk density across all treatments. To obtain soil bulk density measurements, two soil cores were collected along each transect with a PVC cylinder of known volume to a depth of 5 cm. Soils were dried at 105 °C and then weighed. Bulk density was obtained by the calculation: Dry soil g/110 cm^3.

Soil nutrient status was evaluated by examining soil organic carbon (SOC) and total soil nitrogen. Three soil cores were collected along each transect at 9 m intervals and sampled to a depth of 10 cm after removing surface litter. These 10 cm cores were then split into three depths: 0–2 cm, 2–5 cm, 5–10 cm. Replicate samples were composited to yield one soil sample of each depth per transect ($n = 10$). Each soil sample was dried at 60 °C, sieved to 2 mm, and ground in a mortar and pestle. Samples were analyzed for total carbon and nitrogen on an elemental combustion analyzer (Costech 4010 CHN, Valencia, CA). We used a modified pressure calcimeter method to determine soil inorganic C content (Sherrod et al., 2002). SOC was calculated by subtracting soil inorganic C from total soil C.

Soil chlorophyll α concentration may be used as an indicator of N fixation potential of biological BSC communities and the development level of the crust; higher soil chlorophyll α content is related to higher N fixation potential and a more well developed BSC (Belnap et al., 1993; Yeager et al., 2004). Samples were collected to a depth of 2 cm at nine points along each transect. We extracted chlorophyll α from the soil sample with buffered 100% methanol (Castle et al., 2011). Extracts were analyzed on a spectrophotometer (Beckman DU-64).

2.5. Acetylene reduction assay

To provide an estimate of treatment effects on soil nitrogen fixation, acetylene-reduction assays (ARA) were conducted in the laboratory. Schollhorn and Burris (1967) found that acetylene (C_2H_2) competitively inhibits fixation of N_2 and is reduced to ethylene (C_2H_4) by nitrogenase enzymes. Because there is uncertainty about the conversion rates between moles of C_2H_2 fixed to moles of N_2, the acetylene-reduction assays were used to estimate relative differences between sites as opposed to absolute amount of nitrogen fixed (Evans and Johansen, 1999).

Soils were moistened in the field to insure collection of an intact soil surface. Three soil cores were collected at 9 m intervals on each transect using PVC cylinders (2.4 cm radius × 6.4 cm height). Thirty soil samples were collected in each of the treatment sites and the control for a total of 150 samples ($n = 30$). Soil cores were then brought into the lab and allowed to air-dry. The cores were then placed in 500 mL airtight jars with rubber stoppers. Four mL of deionized water (equivalent to an average rain event for the study site of 2.8 mm of rain) was added to each core, the core was allowed to equilibrate for two hours. The addition of water wetted only the top 1 cm of soil in each core. A 10% C_2H_2 atmosphere was created in each chamber by first removing a volume of headspace and replacing that same volume with C_2H_2 gas. Following a 4-hr incubation period, a syringe was inserted into the jar and pumped several times to ensure a well-mixed sample. Twenty-four mL of headspace was sampled from each jar and then removed and stored in an evacuated, airtight glass vial prior to analysis. Samples were analyzed for ethylene on a Hewlett–Packard 5890 series II gas chromatograph (Palo Alto, CA).

2.6. Statistical analyses

The mastication study (control, GS1, and GS2) and the hand-thinned treatment study (control, pile burn, and lop & scatter) were analyzed separately. Due to unequal variance across sites, even after log transforming data we were unable to use analysis of variance (ANOVA) in our data analysis. As a result, percent understory cover and soil surface cover were analyzed with the non-parametric Brown-Forsythe robust equality of means followed by Games-Howell post-hoc tests. Since annual grasses and annual forbs were not present in the control sites, these cover types were analyzed using independent samples *t*-tests with unequal variances between treated sites. Percent canopy gap and wood gap, bulk density, soil C and N, and chlorophyll α concentration were all analyzed with one-way ANOVA with Tukey post-hoc tests for each sub-category of gap size and total gap cover. Gap data were arcsin-transformed to provide normality of data and equal variances between sites. Median aggregate stability test scores are reported on an ordinal scale; these data were analyzed with the non-parametric Kruskal–Wallis test with Kolmogorov–Smirnov pair-wise tests. ARA data were analyzed with a Kruskal–Wallis test and Mann–Whitney pair-wise comparisons. SPSS Statistics 17 (IBM, North Castle, NY) was used for all data analyses. Significance levels were evaluated at levels of $P < 0.05$.

3. Results

3.1. Plant response

Following hand thinning, understory plant cover was 4–5.5 fold higher in the pile burn and lop & scatter respectively relative to the untreated control ($P < 0.001$, Fig. 2A). This pattern was primarily

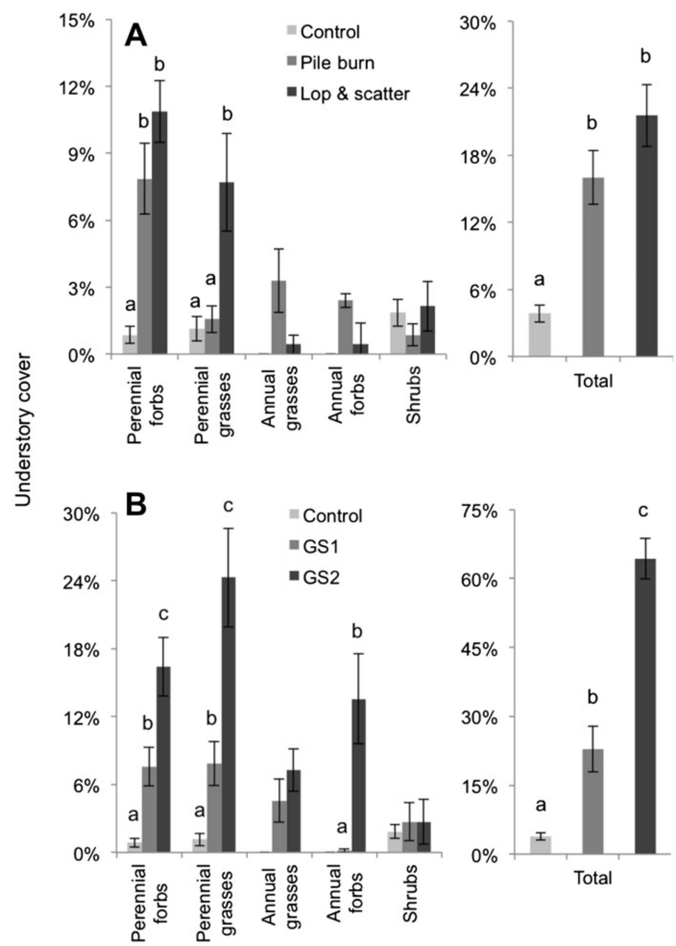

Fig. 2. Understory cover in treatments and control. Functional group and total understory cover for the hand-thinned study (A) and the mastication study (B). Values are means (±1 SE, $n = 10$). Different lowercase letters indicate significant differences in the mean across treatment types with $P < 0.05$.

due to greater perennial forb cover in the pile burn, and the combination of greater perennial forb and grass cover in lop & scatter compared to the control ($P < 0.01$, Fig. 2A). Greater perennial forb cover in the pile burn was driven by higher cover of two native sub-shrub species, *G. sarothrae* and *P. pumila* which, together, composed more than 78% and 70% of total perennial forb cover in the pile burn and lop & scatter sites respectively. *B. tectorum*, an invasive annual grass and the only annual grass observed in any sites, was not present in the control but comprised more than 18% of the total understory cover in the pile burn (3% of total area cover). *B. tectorum* was also present in the lop & scatter but constituted less than 2% of the total understory cover (0.4% of total area cover, Fig. 2A). Annual forbs were also not present in the control but comprised 13% and 2% of the understory cover in the pile burn and lop & scatter respectively (Fig. 2A).

Total understory cover in sites that have been masticated in the previous two growing seasons was 5–16 folder higher than controls ($P < 0.001$, Fig. 2B). GS2 had significantly higher perennial forb, perennial grass, and annual forb cover, followed by the GS1 site and then the control ($P < 0.05$, Fig. 2B). Similar to the hand thinning treatments, *G. sarothrae* and *P. pumila* made up more than 84% and 70% of total perennial forb cover in GS1 and GS2 respectively. *B. tectorum* was the only annual grass present in the mastication treatment, which comprised 20% and 11% of the understory cover in GS1 and GS2 respectively.

3.2. Soil stability indicators

Canopy gap was higher in the pile burn (87%) as compared to the lop & scatter (76%) and the control (67%, $P < 0.01$, Fig. 3A). A larger proportion of canopy gaps were in the largest gap size class (>200 cm) in the control compared to the lop & scatter and pile burn ($P < 0.05$, Fig. 3A). Wood gap was significantly lower ($P < 0.01$) in both the lop & scatter (67%) and the pile burn (68%) relative to the control (83%) (Fig. 3B). Furthermore, in the control, more than 55% of the wood gaps were in the largest size class (>200 cm) as compared to 23% in the pile burn and 32% in the lop & scatter ($P < 0.01$, Fig. 3B). Despite differences in canopy and wood gap, there was no difference in bare soil cover for both hand-thinned treatments (Table 2).

In the mastication study, the total canopy gap was greater in GS1 (82%) as compared to the control (67%) and GS2 (57%, $P < 0.05$, Fig. 3C). Although there were no significant differences in total canopy gap between the control and GS2, the majority of canopy gaps in GS2 were in smaller size classes ($P < 0.01$, Fig. 2C). The addition of mulch in both of the mastication sites is presumably the reason for significantly reduced wood gap ($P < 0.001$) for both the GS1 site (46%) and the GS2 site (33%) compared to the control (83%, Fig. 3D). Additionally, treated plots had most of the wood gaps in the smaller size classes ($P < 0.01$, Fig. 2D). Total bare soil was significantly lower ($P < 0.01$) in both GS1 (38%) and GS2 (26%) as compared to the control (54%) (Table 2).

Soil aggregate stability was significantly lower in the pile burn than in the control ($P < 0.01$), but not significantly different from the lop & scatter (Table 2). In the mastication study, aggregate stability was significantly lower in GS2 than the control ($P < 0.05$) but not significantly different from GS1 (Table 2). Changes in aggregate stability were not explained by changes in BSC cover. There were no significant differences in BSC cover between the lop & scatter, pile burn, & control or in the GS1, GS2, & control.

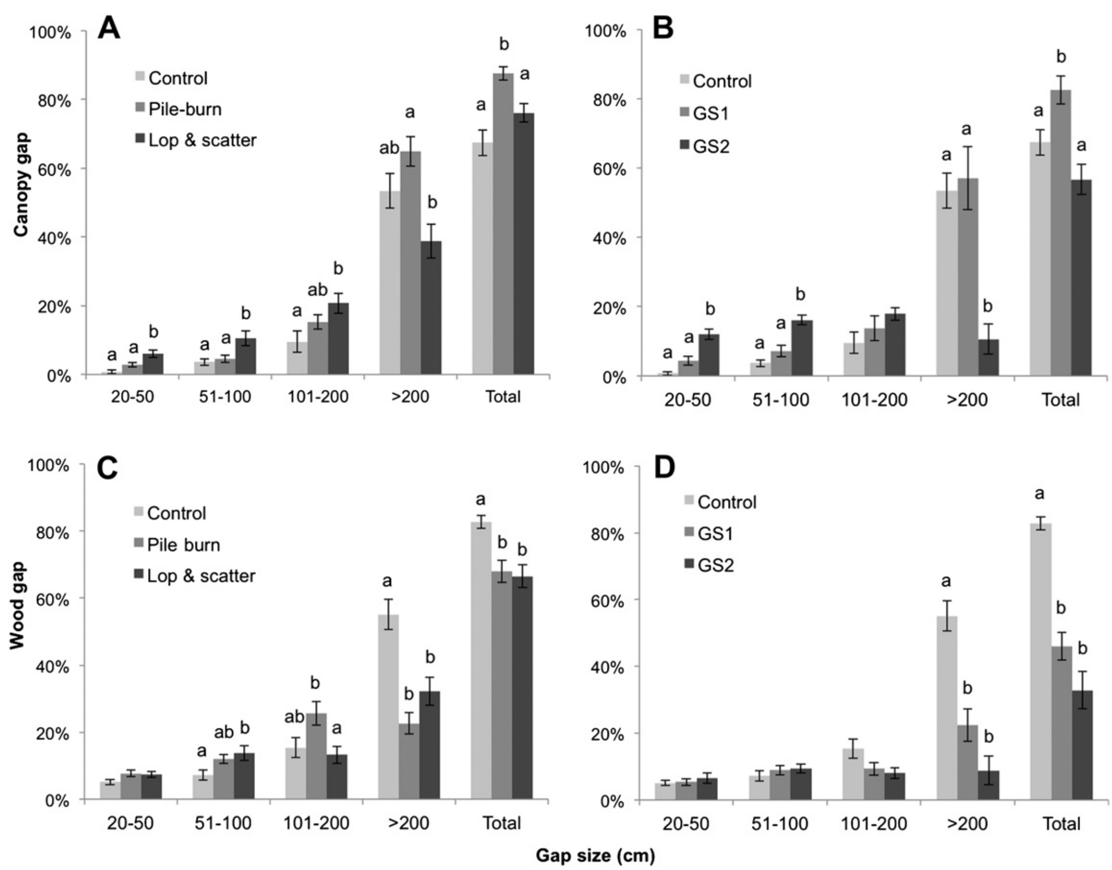

Fig. 3. Canopy and wood gap. Percent gap cover for canopy gap and wood gap for the hand-thinned study (A, B) and the mastication study (C, D). Gaps are organized into four size classes based on gap size with total gap cover shown on the far right. Values are means (±1 SE, $n = 10$). Lowercase letters indicate significant differences among sites with $P < 0.05$.

Table 2
Soil physical and chemical properties. Data are presented as means (±1 SE, $n = 10$), and median (±median absolute deviation) aggregate stability test scores for the hand-thinned study and the mastication study. An aggregate stability score of 0 indicates no aggregation as in sand, while a score of 6 indicates strongly bound aggregates such as found in mature BSCs. Bare soil is defined as soil with no plant, litter, woody debris, or BSC cover. Nitrogen fixation is shown as the total ethylene produced per soil core ($n = 30$). Different underlined uppercase letters indicate significant differences among sites for the hand-thinned study, whereas italicized lowercase letters indicate significant differences across sites for the mastication study with $P < 0.05$. BSC cover for the hand-thinned study had significance for ANOVA but no significant differences in pair-wise comparisons.

	Horizon (cm)	Hand-thinned study		Control	Mastication study	
		Pile burn	Lop & scatter		GS1	GS2
% Soil organic carbon	0–2	2.13 (0.28)	1.72 (0.22)	1.80 (0.24) *a*	1.32 (0.20) *a*	3.27 (0.51) *b*
	2–5	1.62 (0.41)	1.88 (0.25)	1.87 (0.27) *ab*	1.10 (0.23) *a*	2.96 (0.40) *b*
	5–10	2.16 (0.031)	2.16 (0.32)	2.58 (0.41)	1.30 (0.34)	2.69 (0.44)
% Total nitrogen	0–2	B 0.11 (0.01)	A 0.08 (0.01)	AB 0.09 (0.01) *a*	0.07 (0.01) *a*	0.15 (0.02) *b*
	2–5	0.08 (0.01)	0.08 (0.01)	0.09 (0.01) *a*	0.06 (0.00) *a*	0.15 (0.02) *a*
	5–10	0.10 (0.01)	0.09 (0.01)	0.09 (0.00) *a*	0.06 (0.01) *b*	0.13 (0.02) *a*
Median aggregate stability class	–	B 2 (1.72)	AB 3 (1.72)	A 4 (1.69) *a*	3 (1.86) *ab*	2 (1.76) *b*
% Bare soil	–	52 (3.06)	50 (3.16)	54 (6.06) *a*	38 (3.42) *b*	26 (1.30) *c*
% BSC cover	–	37.86 (3.20)	23.67 (3.89)	39.15 (5.91)	38.29 (2.07)	28.71 (5.50)
Chlorophyll *a* (μg/g soil)	0–2	1.42 (0.59)	1.10 (0.29)	1.62 (0.33)	1.07 (0.20)	1.28 (0.28)
Nitrogen fixation (nmol/ethylene/hr/cm^2)	–	AB 0.57 (0.40)	B 0.03 (0.02)	A 1.62 (0.75) *a*	0.00 (0.00) *b*	1.19 (0.51) *ab*

3.3. *Soil nutrient status*

Percent total soil nitrogen was significantly higher in the pile burn (0.11%) relative to the lop & scatter (0.08%) and control (0.09%, $P < 0.05$), but only in the 0–2 cm horizon (Table 2). There were no differences in soil C or C:N ratios across hand thinning treatments.

In the mastication treatment GS2 had nearly twice the amount of total N in surface soils (0–5 cm, $P < 0.001$) compared to both GS1 and the control (Table 2). GS1 had significantly less N in the 5–10 cm horizon ($P < 0.05$) than both GS2 and the control. In the 0–2 cm horizon, total SOC was higher in GS2 than GS1 and the control ($P < 0.05$, Table 2). GS2 had higher total SOC in the 2–5 cm horizon compared to GS1 but not the control ($P < 0.001$). There were no differences in soil C:N ratios across sites. Chlorophyll α concentration, as an indicator of total site nitrogen fixation potential, was not significantly different among any sites (Table 2).

3.4. *Nitrogen fixation*

All sites showed low nitrogen fixation potential (<2 nmol ethylene/hr/cm^2), but there were significant differences in ethylene production. Soils from the control site had higher ethylene production rates than the lop & scatter but not the pile burn ($P < 0.05$, Table 2). Ethylene production rates were lower in the GS1 site as compared to the control but not GS2 ($P < 0.05$, Table 2)

4. Discussion

4.1. *Plant responses*

One of the primary restoration goals in implementing fuels reduction treatments across public lands is to enhance herbaceous understory plant production. We observed that fuels reduction treatments significantly increased understory cover one year following treatments relative to untreated sites, suggesting that in fact this management goal is being met. Based on ecological sites descriptions, potential understory cover in these upland Piñon-juniper ecosystems is approximately 55 percent (NRCS, 2004). Two growing seasons after treatment in the mastication sites understory cover was 64 percent; values that clearly exceed the restoration target for plant cover increases after treatment.

Although plant cover was higher across all treated sites representing more historical conditions in total understory cover, the short-term species level responses may not promote desired native species or those species that were historically present at these sites. In treated sites, the positive understory plant cover response was driven by two sub-shrub species, *G. sarothrae* and *P. pumila*, and the exotic annual grass *B. tectorum* all of which have been targeted for removal in other rangeland restoration projects because of their capacity to outcompete other species in rangelands and their low palatability to livestock and game (Knapp, 1996; Laycock, 1967; Ralphs and Banks, 2009). These three plants comprised nearly 50% understory after one growing season in mastication plots, 40% in lop & scatter, and 30% in the second growing season mastication and pile burn.

Disturbance during fuels reduction efforts may partially explain the changes in plant species composition. *G. sarothrae* is known to be a species that increases on heavily grazed ranges, following fire, and after chaining due to soil disturbance and removal of competitors (Arnold et al., 1964; Brockway et al., 2002; Ralphs and Banks, 2009). Previous studies have observed increases in *B. tectorum* following both pile burn and mastication treatments in Piñon-juniper woodlands (Owen et al., 2009) and thinning in ponderosa pine (Wolk and Rocca, 2009). Similar to Owen et al. (2009) we observed dense rings of *B. tectorum* surrounding burned areas in the pile burn. *B. tectorum* quickly establishes in a post-fire environment, outcompeting native plant species (Evangelista et al., 2004; Hassan and West, 1986). Wolk and Rocca (2009) found that sites where mulch was added had lower *B. tectorum* cover than sites where cut trees were removed from the site, which supports findings from Wicks (1997) that mulch additions reduce *B. tectorum* germination. Although our data suggest that the post-treatment environment promotes colonization of *B. tectorum* (this species was not present in the untreated control), we did not find evidence of continuing increases in cover through the short time frame of our study.

Both *G. sarothrae* and *B. tectorum* can establish and become dominant in heavily utilized rangelands and alter forage productivity (Evans et al., 2001; Ralphs and Sanders, 2002) and *B. tectorum* may alter fire-return intervals, and nutrient cycling, (Evans et al., 2001). Though the initial cover response from these plants may reduce erosion immediately following treatment, in the long term, *G. sarothrae* may increase soil erosion by promoting an understory with large interspaces relative to the other native plant species they replace (Wood and Mosley, 2010). Similarly, *P. pumila* is a poor soil stabilizer because it has a taproot with few lateral roots and, consequently, may have little effect on decreasing erosion despite significant increases in cover (Laycock, 1967). *B. tectorum* may effectively stabilize topsoil immediately following treatments and on longer time scales (Knapp, 1996). The 10-fold difference in cover for these three plant species between all treated sites and the untreated controls suggests that treatment promotes species that are considered to be undesirable range species (Stubbendieck et al., 1997). These changes in plant species composition, however, may be temporary until other species respond to treatment. For example, there was some evidence from our study that perennial grass cover does increase over time. Perennial grass cover in mastication sites two growing seasons after treatment was triple (24%) that of sites that had been treated one year previous (8%). Although this study lends insight into short-term vegetation responses to fuels reduction treatments, studies that evaluate decadal and multi-decadal treatment responses will lead to a better understanding of vegetation responses to fuels reduction treatments over longer time scales.

4.2. Soil responses

Land managers have mandated goals of reducing fire risk, increasing understory cover but also of maintaining soil and site stability. Soil aggregate stability can be an overall indicator for rangeland health (Herrick et al., 2001) and is correlated with susceptibility to erosion (Blackburn and Pierson, 1994). In this study, soil aggregate stability was generally low for all sites including controls and was in the range of what would be expected from an early-successional BSC community (Barger et al., 2006). All of these sites (including controls) were previously treated for rangeland improvement in the 1960's and have a long history of livestock grazing, factors which may regulate the cover and composition of the BSC cover and soil stability over long time scales (Evans and Johansen, 1999).

The control had the highest median aggregate stability class among all sites with a median score of 4, which corresponds to 10–25% of soil in stable aggregates (Herrick et al., 2001). All treated sites had median classes of 3 or less, which are more typical of unstable sites with less than 10% of soil in stable aggregates (Herrick et al., 2001). Effects specific to pile burning may explain this drop in aggregate stability. The center of pile burn piles can reach temperatures of more than 300 °C (Esquilin et al., 2007), a temperature at which soil-binding polysaccharides and other stabilizing organics are combusted (Badía and Marti, 2003), likely leading to destabilization of the soil surface. Pile burning also resulted in increased canopy gap particularly in large gaps, which may contribute to increased erosion potential (Derner and Whitman, 2009).

Two growing seasons following mastication treatment, soil aggregate stability was lower than controls, which could be explained by the delayed degradation of photosynthetic BSC organisms through time due to limited light availability beneath mulch piles (Belnap and Lange, 2002). Despite lower soil aggregate stability two growing seasons after mastication, the addition of mulch and consequent decrease in wood gap for both mastication sites will likely mitigate erosion potential. Slash additions have been observed to prevent significant soil erosion from direct rain impacts and overland flow (Hastings et al., 2003; Stoddard et al., 2008) Additionally, after two growing seasons, it appears that

Table 3
Review of plant and soil responses to treatment. Observed trends in studies examining soil and plant responses to treatment. Symbols indicate the direction of response such that "−" reflects an observed decrease, "+" an increase, and "0" indicates no response. When possible, the magnitude of response is given in parenthesis. Values are the mean difference between treatment and control sites responses. Instances where no data were collected are indicated by an n/a.

Trees targeted for removal	Time since treatment	Treatment	Plant responses		Soil responses			Notes	Study
			Understory cover	Non-native species	Soil exposure	Aggregate soil stability	Observed soil erosion		
J. occidentalis	10 years	Lop	+ (17.60%)	0	n/a	n/a	− (~85 fold decrease)	No effort was made to spread debris across landscape.	Pierson et al., 2007
J. monosperma and *P. edulis*	1–3 years	Mechanical mastication	+ (62%)	n/a	− (12% less bare soil)	n/a	0		Brockway et al., 2002
J. monosperma and *P. edulis*	3–5 years	Lop & slash	+	n/a	n/a	n/a	− (~100 fold decrease)	~10 fold increase in ground cover (slash, forbs, grasses, and litter)	Hastings et al., 2003
J. osteosperma and *P. edulis*	5 years	Pile burn	+ (~20%)	+ (~24%)	n/a	n/a	n/a	Plant cover does not include interspaces and does include tree cover.	Haskins and Gehring, 2004
J. osteosperma and *P. edulis*	2.5–3.5 years	Pile burn	− (22%)	+ (75%)	+	−	n/a	Measurements were taken exclusively within burn piles.	Owen et al., 2009
		Mechanical mastication	+ (~17%)	0	−	0	n/a	n/a	
J. osteosperma and *P. edulis*	10 months	Lop & scatter	+ (18%)	0	0, − (canopy gap, wood gap)	0	n/a	n/a	This study
	11 months	Pile burn	+ (12%)	+	+,−	−	n/a	n/a	
	18 months	Mechanical Mastication	+ (60%)	+	−,−	−	n/a	n/a	

understory plant cover increases to the point where the majority of canopy gaps are in smaller gap size classes, which are less vulnerable to soil erosion (Derner and Whitman, 2009).

We found few changes to nutrient status in the hand-thinned study except for a small increase of soil N on the surface soil layer in the pile burn plots, five months after burning. In the mastication study there were notable increases in total soil organic carbon and total nitrogen, but only after two growing seasons post-treatment. This finding indicates that large inputs of mulch and litter, as a result of treatments, may be incorporated into the soil over longer time scales as others have suggested (Stubbs and Pyke, 2005).

Because BSCs play an important role in the nutrient budget of aridland systems (Barger et al., 2006, 2005; Belnap, 1990, 2002), it is important to understand the effects of treatment on BSC community function, particularly nitrogen fixation. Soil disturbance from treatment could have caused the low observed nitrogenase activity observed in the one growing season mastication plot and the lop & scatter plot. However, this change in BSC function does not correspond to changes in soil aggregate stability and BSC cover, which may be a relict from previous disturbance regimes. Overall, nitrogenase activity across all sites was seven fold lower than rates previously reported for late successional biological soil crust communities (Belnap, 2002), which most likely is a result of historical disturbance from chaining and cattle grazing. Both of these activities can dramatically decrease BSC cover (Belnap and Gillette, 1998) and favor early succesional BSC communities that have low nitrogen-fixing bacterial biomass (Barger et al., 2006). On sites with less historical disturbance, changes in BSC cover and function could be more dramatic following treatment.

5. Management implications

Land managers often have multiple goals for forest ecosystem management. The perceived benefits of forest thinning activities are the reduced risk of catastrophic fires and restoration of understory plant communities to historical conditions. However, these benefits must be weighed against the potential risks of decreasing site stability and increasing invasive species cover. Although assessing the full balance between all the possible risks and benefits associated with fuels-reduction treatments is difficult, data from this study and others suggests that removal of slash by pile burning in semi-arid environments can have more negative impacts than other alternatives (Table 3).

With few exceptions (Owen et al., 2009), studies examining the ecological effects of fuels reduction overwhelming suggest that these treatments tend to result in increased understory cover. However, pile burn treatments consistently had the highest increases of non-native cover (Haskins and Gehring, 2004; Owen et al., 2009, Table 3). Additionally, strong evidence suggests additions of slash to the landscape during mechanical mastication or hand thinning can significantly decrease observed soil erosion (Hastings et al., 2003; Pierson et al., 2007, Table 3). While no such data exists for pile burn studies, significant decreases in aggregate stability and increases in canopy gap, suggest these landscapes are highly susceptible to erosion post-treatment for at least 3.5 years (Owen et al., 2009, Table 3). Additional studies over longer time scales that include measurements of observed soil erosion and different possible responses to seeding of specific species are needed to draw more robust conclusions, but current understanding of plant and soil responses following treatment suggest alternatives to the pile burn method.

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