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Long-term effects of chaining treatments on vegetation structure in piñon–juniper woodlands of the Colorado Plateau



Miranda D. Redmond^{a,*}, Neil S. Cobb^b, Mark E. Miller^c, Nichole N. Barger^a

^a Ecology and Evolutionary Biology Department, University of Colorado, Boulder, CO 80309, United States

^b Merriam-Powell Center for Environmental Research, Northern Arizona University, P.O. Box 6077, Flagstaff, AZ 86011, United States

^c Southeast Utah Group, National Park Service, 2282 S. West Resource Blvd., Moab, UT 84532, United States

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ABSTRACT

Over the last half-century a range of methods have been utilized to reduce trees and shrubs in order to reduce wildfire risk and promote herbaceous vegetation to support livestock and wildlife. We examined the long-term (20–40 year) effects of past tree-reduction treatments on vegetation and ground cover in piñon–juniper woodlands, which is the third most extensive vegetation type in the continental United States. Tree-reduction treatments were conducted between 1963 and 1988 in Grand Staircase-Escalante National Monument, Utah by the US Bureau of Land Management and involved chaining followed by seeding to remove trees and often shrubs. Treatments were effective at increasing perennial grass cover and reducing tree cover over multiple decades. The increase in perennial grass cover was predominantly due to a nonnative species that was seeded, *Agropyron cristatum* (crested wheatgrass). Surface fuel loads were nearly twice as high in treated areas, likely changing fire behavior and increasing habitat complexity. Treated areas had higher bare mineral soil cover and lower biocrust cover, which may influence soil erosional processes. Interestingly, treated areas had significantly less *Pinus edulis* (piñon pine) recruitment compared to untreated areas, while there was no change in *Juniperus osteosperma* (Utah juniper) recruitment. These results indicate that treated areas may become more *J. osteosperma* dominated in the future due to increased establishment of *J. osteosperma* compared to *P. edulis*. Our results show that while treatments were effective at reducing tree cover and increasing herbaceous cover, there were long-term (40 year) treatment effects on vegetation composition and ground cover that need to be taken under consideration when developing future management strategies.

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1. Introduction

Across the western US, there is an increasing need to effectively manage ecosystems to both mitigate hazardous wildfires and maintain, and in some cases restore, the structure, function, diversity and dynamics of forest and rangeland ecosystems. Of particular importance to the management of public lands in the western US are ecosystems dominated by various species of piñon (e.g., *Pinus edulis* and *P. monophylla*) and juniper (e.g., *Juniperus monosperma*, *J. osteosperma*, and *J. occidentalis*), which collectively represent the third most extensive vegetation type in the continental US and are one of the predominant vegetation types administered by federal land-management agencies in the US (Romme et al., 2009).

During the past half-century, piñon–juniper (P–J) ecosystems have been a major focus for land-management activities due to their great spatial extent, the multiple ecosystem services they provide, and historic changes in the structure and extent of P–J

populations. Across the western US, P–J populations have established in adjacent grassland and shrubsteppe vegetation, and existing woodlands have experienced increased tree recruitment and stand densities over the last century (Barger et al., 2009; Miller and Rose, 1999; Miller et al., 2008; Tausch et al., 1981). In thickening woodlands and where trees have established in adjacent grasslands and shrubsteppe, competition from trees has contributed to declines in forage production for livestock and diminished habitat quality for some wildlife species (Bates et al., 2005; Clary and Jameson, 1981; Noson et al., 2006). In some settings, increasing tree dominance and decreasing herbaceous cover have contributed to increases in runoff and soil erosion (Wilcox, 1994), with implications for long-term ecosystem sustainability.

Thus, since the 1950s a variety of tree-reduction methods have been used across large tracts of public lands in the western US in attempts to restore herbaceous cover and production. These early treatment methods, which predominantly involved green chaining and seeding, were effective at increasing herbaceous cover (Aro, 1971; Tausch and Tueller, 1977) and, in certain cases, reducing runoff and soil erosion in the short-term (Farmer et al., 1999;

* Corresponding author. Tel.: +1 (415) 300 6901; fax: +1 (303) 492 8699.

E-mail address: MirandaRedmond@gmail.com (M.D. Redmond).

Gifford, 1973). However, the effectiveness of these treatments at maintaining low tree densities, high herbaceous cover, and reducing soil erosion in the long-term are largely unknown (but see Pier-son et al., 2007 and Skousen et al., 1989).

Despite this lack of knowledge, concern over threats posed by wildland fire over the last decade has led to sharp increases in the number and extent of new tree-reduction projects, some of which use the same chaining techniques common in the 1960s. Many of these projects have been conducted in conjunction with the US National Fire Plan, which aims to reduce the risk of catastrophic wildland fire while restoring ecological functions to forests and woodlands across the US. Whereas past management treatments primarily focused on forage production for livestock, managers are now tasked with treating for multiple objectives, including fire prevention and maintenance of ecosystem attributes such as soil stability and fertility, hydrologic processes, and ecosystem resistance to invasion by exotic plants. Thus, there is a clear need for understanding the longer-term effects of tree-reduction on vegetation structure and soil properties in these ecosystems.

In this study, we examined how past chaining treatments influenced plant communities and soil surface characteristics at 17 paired (treated vs. untreated) sites that were chained and seeded between 1963 and 1988. We also examined whether vegetation and soil surface responses changed as time since chaining treatment increased (from 18 to 43 years). We predicted that these past chaining treatments would lead to increased herbaceous cover and decreased tree cover. Additionally, we hypothesized that as time since chaining treatment increased there would be an increase in tree cover leading to a decline in herbaceous cover. We focused our study in Grand Staircase-Escalante National Monument located in southern Utah on the Colorado Plateau, where numerous chaining treatments have occurred (<http://www.mpcer.nau.edu/pj/pjwood/>).

2. Materials and methods

2.1. Study area and treatment methods

From May thru August 2006, we sampled 17 paired (treated vs. untreated) sites located within P–J woodlands in Grand Staircase-Escalante National Monument, Utah that had been treated

between 1963 and 1988 by the Bureau of Land Management (BLM) (see Table 1 and Fig. 1). For our treated sites, we focused on tree-reduction treatments involving chaining and seeding (Table 1), as those were the most common treatment methods used by the BLM and were applied to over 169,000 ha of land across the Colorado Plateau since the 1940s (<http://www.mpcer.nau.edu/pj/pjwood/>). The chaining treatment method involves two tractors pulling heavy chains (18–40 kg/link) in a “U” or “J” shaped pattern to pull over and uproot trees and often shrubs (BLM, 2008). Smooth chains were used at all sites except for site 126 where Ely chaining was used. An Ely chain had short pieces of hardened railroad rails welded perpendicular to each link to increase soil disturbance and uproot more trees and shrubs. Both smooth and Ely chaining disturb soils, and in all of the treatments examined in this study, plant debris was left in the treatment to reduce erosion (BLM, 2008). All treatments involved seeding, however, the seeding methods and species mixes seeded into the area varied (see Table 1 for details). Seeding was done either using aerial or hand broadcasting, where seeds are left on the soil surface, or by drilling or using a dribbler, where equipment is used to bury seeds (BLM, 2008). While all treatments involved chaining and seeding, the combination of treatment methods varied (Table 1). For example, some sites were double chained and then seeded (denoted as chain/chain/seed in Table 1) while other sites were chained once and then plowed and seeded (denoted as chain/plow/seed in Table 1). All paired untreated sites were adjacent to the treatment area and of similar slope ($\pm 9^\circ$), aspect ($\pm 75^\circ$), elevation (± 75 m) and of the same soil map unit (Natural Resource Conservation Service, 2006).

Sites were located in either persistent P–J woodlands or wooded shrublands (Romme et al., 2009), with vegetation consisting of overstory *P.edulis* Engel. (twoneedle piñon) and *J. osteosperma* (Torr.) Little (Utah juniper). All paired sites had at least one tree with a basal trunk diameter greater than 22 cm, suggesting that trees had established prior to the 20th century at our study sites (Despain, 1989; Barger et al., 2009). The dominant shrub in the area is big sagebrush (*Artemisia tridentata* Nutt. and *A.tridentata* Nutt. ssp. *tridentata*), which ranged from 0 to 35% cover in the untreated sites. Other shrub species common at many of the sites included broom snakeweed (*Gutierrezia sarothrae* (Pursh) Britt. and Rusby), Mormon tea (*Ephedra viridis* Coville), and antelope bitterbrush (*Purshia tridentata* (Pursh) DC.).

Table 1

Treatment year, treatment method, species seeded, amount seeded (in parenthesis next to species seeded in kg ha^{-1}), method of seeding for all treated sites, and mean slope, aspect, and elevation for each paired site at Grand Staircase-Escalante National Monument, Utah. The order in which treatment combinations were applied is denoted by slashes with methods in chronological order (i.e. Chain/Plow/Seed indicates the site was chained, then plowed, then seeded).

Paired site	Slope	Aspect	Elev. (m)	Year treated	Treatment method	Species seeded	Seeding method
137	2°	S	1960	1963	Chain/Seed	<i>A. cristatum</i> (7.8)	Drill
139	2°	E	2027	1964	Chain/Windrow/Seed	<i>A. cristatum</i> (6.7)	Drill
127	1°	W	1840	1965	Plow/Chain/Seed	<i>A. cristatum</i> (6.7)	Aerial Broadcast
129	1°	E	1960	1965	Chain/Chain/Seed	<i>A. cristatum</i> (6.7)	Aerial Broadcast
130	4°	S	1973	1965	Chain/Chain/Seed	<i>A. cristatum</i> (6.7)	Aerial Broadcast
133	3°	SE	1604	1965	Chain/Chain/Seed	<i>A. cristatum</i> (6.7)	Aerial Broadcast & Drill
135	5°	SE	2148	1965	Chain/Seed	<i>A. cristatum</i> (6.7)	Aerial Broadcast
110	2°	SE	1886	1966	Chain/Seed/Chain	<i>A. cristatum</i> (6.7)	Aerial Broadcast
131	4°	S	2002	1966	Chain/Chain/Seed	<i>A. cristatum</i> (6.7)	Aerial Broadcast
132	5°	E	2042	1968	Chain/Plow/Seed	<i>Elymus junceus</i> (5.6)	Aerial Broadcast
134	3°	SE	1762	1969	Chain/Chain/Seed	<i>A. cristatum</i> (5.6), <i>Atriplex canescens</i> (0.3), <i>Medicago</i> sp. (5.6), <i>Purshia tridentata</i> (0.3)	Aerial Broadcast
113	5°	SE	1950	1971	Chain/Seed/Seed	<i>A. intermedium</i> (5.8)	Aerial Broadcast
150	5°	SW	1892	1981	Chain/Chain/Seed	<i>A. cristatum</i> (4.5), <i>A. trichophorum</i> (2.2), <i>E. junceus</i> (3.4), <i>Melilotus officinalis</i> (1.1), <i>P. tridentata</i> (0.3)	Aerial Broadcast
126	1°	S	2028	1982	Chain/Plow/Seed	Other Herbs (7.1)	Aerial or Hand Broadcast
128	6°	SW	2192	1982	Chain/Seed	<i>A. cristatum</i> (9.0), <i>E. junceus</i> (6.7), <i>A. intermedium</i> (2.2), <i>M. officinalis</i> (2.2), <i>A. canescens</i> (0.3)	Aerial Broadcast & Dribbler
107	2°	E	2036	1983	Chain/Seed	<i>A. cristatum</i> (4.5), <i>A. trichophorum</i> (3.4), <i>M. officinalis</i> (0.9), <i>Onobrychis</i> sp. (0.9)	Aerial Broadcast
123	1°	E	1859	1988	Chain/Seed	<i>A. cristatum</i> , <i>E. junceus</i>	Drill

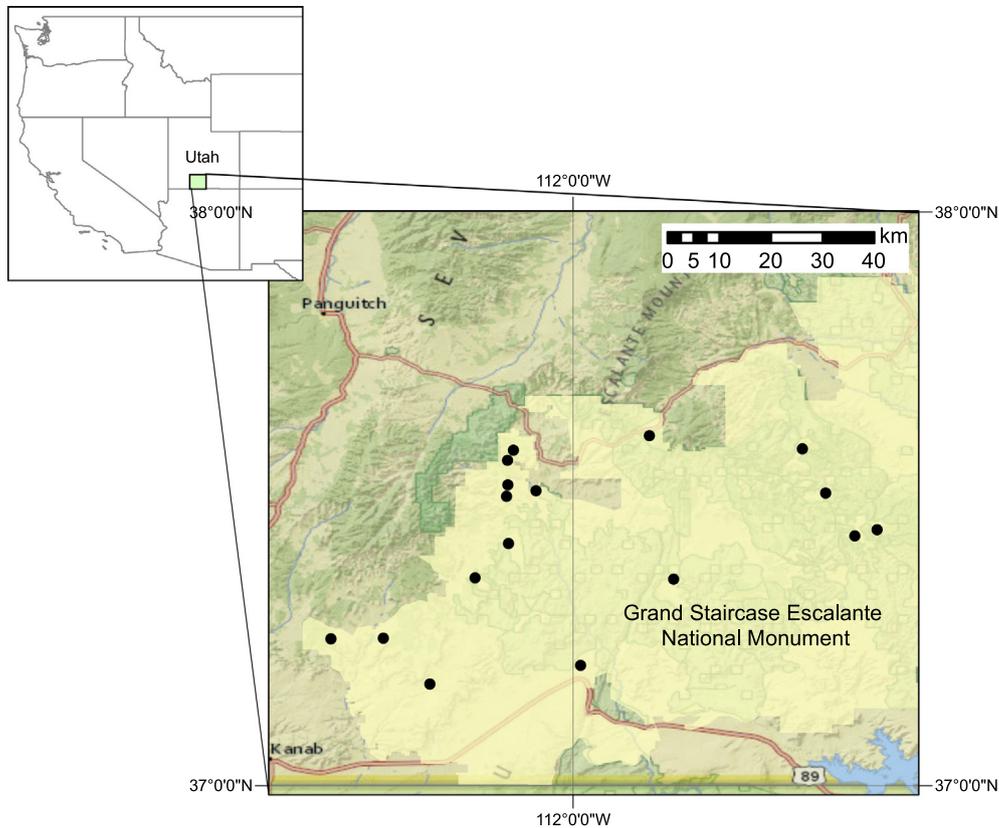


Fig. 1. Map of the 17 paired sites (black circles) scattered throughout Grand Staircase-Escalante National Monument in southern Utah, USA.

Mean annual temperature and precipitation from 1960 to 2010 in our study area was 11.8 °C and 251 mm, respectively (<http://prism.oregonstate.edu>). Annual precipitation in 2006 was slightly below average (240 mm), but within one standard deviation of the 50 year mean (<http://prism.oregonstate.edu>). Mean annual temperature in 2006 was similar to the 50 year mean (11.7 °C) (<http://prism.oregonstate.edu>).

2.2. Field methods

For all paired sites, we used GIS to randomly locate three points within each treated and untreated area. At each point we established three 10 × 10 m subplots, each located 75 m apart, for a total of 9 subplots per site.

To quantify tree density and cover, we recorded the species status (live or dead), height (live trees only), two perpendicular canopy widths (live trees only), and the basal trunk diameter (BTD) for all trees (adults, ≥5 cm BTD; saplings, ≥2.5 cm and <5 cm BTD; and seedlings, BTD <2.5 cm) rooted within each subplot. To quantify shrub density, we counted the number of each shrub species located within each subplot. Following, two observers did an ocular estimate of percent cover for each shrub species within each 10 × 10 m subplot, which were then averaged. To quantify herbaceous and soil cover, we randomly placed one 1 m² quadrat within each quadrant of the 10 × 10 m subplot and estimated percent cover bare ground, rock, litter, and biocrusts located within each 1 m² quadrat as well as percent cover for each plant species.

To quantify surface fuel loads, we randomly established one 10 m transect going through the center of each subplot and recorded the small and large diameter of each downed, dead woody material (twigs, stems, branches, bolewood) from trees and shrubs that intersected the transect (see methods in Brown, 1974). For

fuels greater than 7.6 cm we classified wood as sound (≤50% decay) or rotten (>50% decay).

2.3. Data analysis

To understand the effects of chaining and seeding on vegetation, biocrusts, and surface fuel load we performed a paired (treated vs. untreated) Student's *t*-test across the 17 sites. In cases where the response variable was not normally distributed, even after multiple transformations were attempted, we performed a paired Wilcoxon Signed Rank test. We omitted sites 110, 113, and 123 from all surface fuel load analyses due to missing data.

We evaluated understory diversity, which included all shrubs, graminoids, and forbs, using two indices: species richness (N_0), calculated as the mean number of species per 10 × 10 m subplot and species heterogeneity (N_2), calculated as the reciprocal of Simpson's Index ($1/\sum p_i^2$), where p_i represents the proportional representation of each species in a sample, calculated on the basis of cover (Hill, 1973). Species heterogeneity shares the same units as species richness and incorporates both species richness and species evenness (Peet, 1974).

To understand the long-term effects of chaining on tree recruitment dynamics, we examined the number of *P. edulis* and *J. osteosperma* trees that likely recruited after each treatment among the sites that were treated between 1963 and 1971. Based on our size-age relationship for *P. edulis* ($\text{age}_{\text{treeX}} = 2.45 * \text{BTD}_{\text{treeX}} + 26.8$; Appendix Table A.1 and Fig. A.1) and *J. osteosperma* ($\text{age}_{\text{treeX}} = 21.1 * \text{canopy}_{\text{treeX}} + 26.6$; Appendix Table A.1 and Fig. A.1), we considered trees with a BTD ≤5 cm for *P. edulis* and trees with a canopy ≤0.64 m² for *J. osteosperma* to have recruited after treatment (i.e. ≤40 years old). Using this size cut-off, we examined the differences in the numbers of trees that recruited in the last 40 years (i.e. recruited post-treatment) in the treated and un-

treated sites by conducting paired (treated vs. untreated) Student's *t*-test using data across the 12 sites that were treated between 1963 and 1971. Additionally, we did paired (treated vs. untreated) Student's *t*-tests to examine if *P. edulis* dominance (calculated as: $\text{density}_{P.edulis} / (\text{density}_{P.edulis} + \text{density}_{J.osteosperma})$) differed between the treated and untreated sites among the trees that recruited post-treatment as well as among the trees that recruited prior to treatment. Lastly, to examine changes in recruitment patterns across all of our 17 paired sites, we examined differences in seedling (BTD <2.5 cm) and sapling (BTD <5 cm and ≥ 2.5 cm) densities among the treated and untreated sites by doing paired (treated vs. untreated) Student's *t*-tests. In cases where the response variable was not normally distributed, even after multiple transformations were attempted, we performed a paired Wilcoxon Signed Rank test.

We also performed linear regressions to examine how year since treatment influenced the percent difference in herbaceous cover, shrub cover, tree density and basal area, ground cover, and fuel load (1–1000 h fuels). All analyses were performed using the statistical software R (R Development Core Team, 2011), with $\alpha = 0.05$.

3. Results and discussion

3.1. Changes in herbaceous cover

Past chaining treatment methods were effective at increasing understory cover, even 40 years post-treatment. Total herbaceous cover was over four times as high (8.1% as opposed to 1.7%) in sites that had been treated as compared to untreated sites (paired *t*-test, $P = 0.001$; Fig. 2). Interestingly, while there was a trend of higher herbaceous cover among sites that had been treated more recently (i.e. 1980s), there was no strong relationship between year of treatment and difference in herbaceous cover among paired treated and untreated sites ($R^2 = 0.16$, $P = 0.07$; Table 2). This lack of a strong relationship may be due to our paired sampling design (i.e. rather than comparing pre-treatment and post-treatment data at each treated site, we compared post-treatment data with data from an adjacent untreated site). Additionally, there was high variability in vegetation cover among our untreated sites (Table 2), which may also reduce our ability to detect how vegetation may change over time following treatment.

When comparing differences in cover among herbaceous functional groups, there was no significant difference between treated

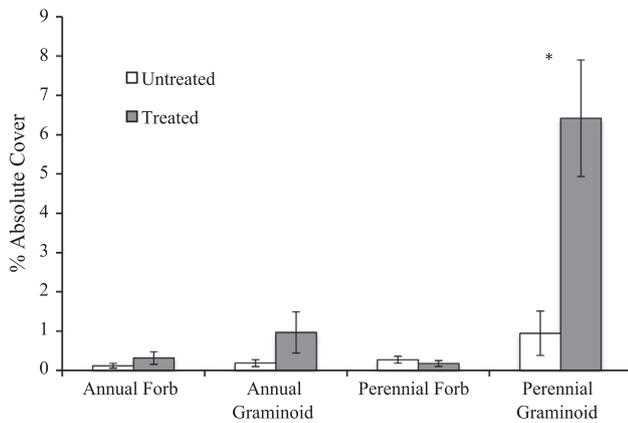


Fig. 2. Percent absolute cover of annual forbs, annual graminoids, perennial forbs, and perennial graminoids in treated (chained and seeded) and untreated sites at Grand Staircase-Escalante National Monument, Utah. Data are means ± 1 SE and an asterisk denotes significant differences between treated and untreated sites, with $\alpha = 0.05$.

Table 2 Herbaceous cover (%), *Artemisia tridentata* cover (%), other shrub sp. cover (%), tree cover (%), tree density (trees per ha), tree basal area (m^2 per ha), and 1–1000 h surface fuel loads (metric t ha^{-1}) in each untreated (U) and treated (T) paired site at Grand Staircase-Escalante National Monument, Utah. Bottom row includes mean ± 1 SE.

Paired site	Year	Herbaceous cover		A. tridentata cover		Other Shrub sp. cover		Tree cover		Tree density		Tree basal area		Surface fuel loads	
		U	T	U	T	U	T	U	T	U	T	U	T	U	T
137	1963	0	1	0	17	7	18	3	18	522	178	31	2	11	35
139	1964	2	2	1	25	9	5	5	14	389	244	16	3	9	5
127	1965	10	7	10	27	23	14	7	11	189	211	11	10	1	14
129	1965	5	1	36	10	3	10	12	14	367	356	11	7	14	18
130	1965	0	6	6	36	6	7	0	20	411	33	19	0	16	12
133	1965	1	1	2	8	15	14	0	4	78	22	12	1	9	41
135	1965	1	9	0	13	8	13	4	21	478	144	27	4	12	19
110	1966	1	12	20	19	0	3	11	14	178	256	11	9	-	-
131	1966	0	1	0	15	13	9	8	20	356	289	19	8	36	76
132	1968	1	12	0	5	14	17	4	13	300	178	27	4	16	13
134	1969	0	1	0	27	21	16	0	12	256	44	39	1	41	81
113	1971	1	23	8	26	16	0	0	30	311	0	27	0	-	-
150	1981	0	0	0	13	9	9	4	15	456	256	31	4	13	57
126	1982	1	17	4	37	7	8	0	24	500	22	18	0	16	2
128	1982	0	20	3	2	19	15	6	13	511	367	21	4	28	51
107	1983	1	15	11	21	5	9	2	20	444	89	14	1	1	46
123	1988	2	8	18	5	10	11	0	10	33	0	3	0	-	-
Mean (± 1 SE)		2 (1) ^a	8 (2) ^b	7(2) ^a	18 (3) ^b	11 (2) ^a	10 (1) ^a	4 (1) ^b	16 (1) ^a	340 (30) ^a	158 (30) ^b	20 (2) ^a	4 (1) ^b	16 (3) ^a	33 (7) ^b

Different letters denote significant differences between treated and untreated sites, with $\alpha = 0.05$.

and untreated sites in percent cover of annual forbs, annual graminoids, or perennial forbs (all paired Wilcoxon tests, all $P > 0.12$; Fig. 2). However, perennial graminoids had significantly higher cover in treated sites (6.4% cover) than untreated sites (0.3% cover) (paired Wilcoxon test, $P = 0.006$; Fig. 2). Previous research examining the impacts of chaining immediately (1–2 years) following chaining found increases in annual and perennial forbs (Skousen et al., 1989; Tausch and Tueller, 1977). However, 8–10 years following chaining these studies report increases in perennial graminoids and shrubs while forbs return to pre-treatment levels (Skousen et al., 1989; Tausch and Tueller, 1977), results which concur with our findings.

The high cover of perennial graminoids in the previously treated sites is predominately due to the increase in *Agropyron cristatum* (L.) Gaertn. (crested wheatgrass), the most commonly seeded species (Table 1). This species also accounts for the large increase in nonnative species cover, which was over ten times higher in the treated sites (5.6% cover on average) as compared to the untreated sites (0.5% cover on average) (paired Student's *T* test, $P = 0.006$). Among native species cover, there was an insignificant trend of higher native species cover in the treated sites (2.0% cover on average) as compared to the untreated sites (0.9% cover on average) (paired Wilcoxon test, $P = 0.09$).

3.2. Seeded species

A. cristatum was the most commonly seeded species and was seeded in at least 14 of the 17 treated sites (Table 1). This species had significantly higher cover in the treated sites (4.4% cover) as compared to the untreated sites (0.4% cover) (paired Wilcoxon test, $P = 0.006$) and was the dominant species at the majority of the treated sites (10 of 17 sites). The only other species seeded that was present at the sites was *Elymus junceus* Fisch. (Russian wildrye), which was present in 2 of the 3 seeded sites. Sites 123 and 128, which were seeded with *E. junceus* and *A. cristatum* (see Table 1), had low *E. junceus* cover (0% and 0.2% cover, respectively) in the treated sites, similar to the paired untreated sites (both 0% cover). In the other *E. junceus* seeded site (site 132), which was seeded with only *E. junceus*, there was high *E. junceus* cover in the treated site (10% cover) unlike the paired untreated site (0% cover). While both *A. cristatum* and *E. junceus* are adapted for heavy grazing, *E. junceus* is more drought tolerant than *A. cristatum* and is also able to tolerate more alkaline levels than *A. cristatum* (Monsen et al., 2004). In our study area, however, it appears that *A. cristatum* may be more competitive than *E. junceus*. When seeded without *A. cristatum*, *E. junceus* was effective at becoming established and remained a dominant herbaceous species for 40 years post-treatment.

Our results show that the commonly seeded *A. cristatum* is effective at becoming established following seeding, which is consistent with other tree-reduction studies (Ott et al., 2003; Skousen et al., 1989), and remains a dominant herbaceous species for 40 years post-treatment. *A. cristatum* is one of the most frequently planted nonnative grasses in western North America due to its high productivity, ease of establishment, grazing resistance, and ability to survive droughts (Lesica and DeLuca, 1996; Smoliak and Dormaar, 1985). The dominance and persistence of *A. cristatum* following these chaining treatments suggest that *A. cristatum* may be outcompeting other native species that would otherwise become more dominant (Henderson and Naeth, 2005; Walker, 1997; Wilson and Gerry, 1995).

3.3. *Bromus tectorum*

Bromus tectorum L. (cheatgrass), a highly invasive annual grass, was detected in 8 of the 17 paired sites. Within those 8 paired sites,

B. tectorum had low cover ($\leq 3\%$), with no significant difference in cover between treated (mean \pm SE: $0.2 \pm 0.1\%$) and untreated ($1.0 \pm 0.5\%$) sites (paired Student's *t*-test, $P = 0.09$). These results suggest that the chain and seed method did not affect *B. tectorum* abundance 40 years post-treatment. Similarly, while Skousen et al. (1989) found large increases in *B. tectorum* cover immediately (2 years) following chaining and seeding in P–J woodlands in central Utah, there was little increase in *B. tectorum* in the long-term (10–20 years) (Skousen et al., 1989). The lack of a long-term increase in *B. tectorum* cover following chaining and seeding in our study and others may be due to the establishment and growth of perennial grasses, which may have resulted in competitive exclusion of *B. tectorum* (Chambers et al., 2007; Ott et al., 2003; Thompson et al., 2006). Further, *A. cristatum* may be particularly effective at outcompeting *B. tectorum* compared to other native perennial grasses (Cox and Anderson, 2004; Leffler et al., 2011). Taken together, these results suggest that seeding with perennial graminoids, especially *A. cristatum*, may reduce the risk of *B. tectorum* invasion. Another potential explanation for the low cover of *B. tectorum* in our treated sites may be that *B. tectorum* has low invasibility in P–J woodlands in our study area (Crall et al., 2006), which is supported by the low cover of *B. tectorum* in our untreated sites. Thus, areas with larger populations of *B. tectorum* may potentially see long-term increases in *B. tectorum* following chaining and seeding.

3.4. Changes in shrub cover

Treated sites had 60% higher shrub cover than untreated sites (paired Student's *t*-test, $P = 0.009$; Table 2), which was due to the large increase in *A. tridentata*. In particular, *A. tridentata*, the dominant shrub, was over twice as high in chained and seeded sites (Table 2; paired Student's *t*-test, $P = 0.002$). While there was an increase in shrub cover overall, when *A. tridentata* was excluded from the analysis, there was no difference in shrub cover between treated and untreated sites (Table 2; paired Student's *t*-test, $P = 0.7$). These results highlight how certain shrub species benefit from chaining treatments, while other shrub species are unaffected, which is consistent with observations from other tree-reduction studies in P–J woodlands (Rippel et al., 1983).

3.5. Changes in understory diversity

While previous research found a decline in understory diversity immediately following chaining (O'Meara et al., 1981), our results suggest there were no long-term effects of chaining treatments on understory plant diversity: both in terms of species richness (paired Student's *t*-test, $P = 0.4$) and species heterogeneity (paired Student's *t*-test, $P = 0.7$). Species richness was much higher in both treated (mean \pm SE: 11.0 ± 1.0) and untreated sites (10.4 ± 0.8) than species heterogeneity (3.0 ± 0.3 in treated sites and 2.9 ± 0.3 in untreated sites), highlighting how a select number of species, primarily *A. cristatum* (treated sites only) and *A. tridentata* (both treated and untreated sites), dominated understory cover.

3.6. Changes in soil surface characteristics

The percent bare mineral soil cover was significantly higher in the treated sites as compared to the untreated sites (paired Student's *t*-test, $P = 0.01$; Fig. 3), while litter, biocrust, and rock cover was significantly lower in treated sites as compared to untreated sites (all paired Student's *t*-tests, all $P < 0.02$; Fig. 3). Higher bare mineral soil cover may increase wind and water erosion (Wilcox, 1994; Davenport et al., 1998). Additionally, biocrusts can be important sources for fixed nitrogen in these semi-arid ecosystems (Belnap, 1996), and also for preventing soil and nutrient loss through

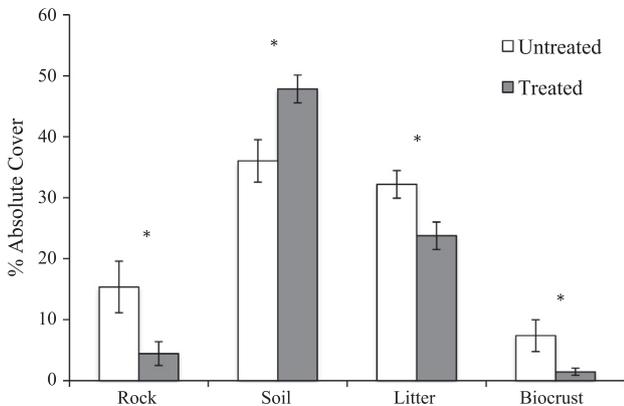


Fig. 3. Percent absolute cover of rock, bare mineral soil, litter, and biocrust cover in treated (chained and seeded) and untreated sites at Grand Staircase-Escalante National Monument, Utah. Data are means \pm 1 SE and an asterisk denotes significant differences between treated and untreated sites, with $\alpha = 0.05$.

wind and water erosion (Barger et al., 2006; Belnap and Gillette, 1998). While both higher bare mineral soil cover and lower biocrust cover may increase erosion, the higher herbaceous cover in the treated areas (Table 2) has likely reduced soil and water loss, because vegetation patches can serve as a sink for both water and sediment (Wilcox et al., 2003; Ludwig et al., 2005). Therefore, the potential negative effects of increased bare mineral soil cover and decreased biocrust cover on soil erosion may be offset by the increase in herbaceous vegetation. Indeed, previous research found a decrease in soil erosion one to six years following chaining in a P–J woodland in northern Utah, which was associated with an increase in understory vegetation (Farmer et al., 1999). The change in vegetation cover may also offset the potential losses of fixed nitrogen due to reduced biocrust cover.

The changes in ground cover in the treated areas may be due to the physical effects of chaining or the post-treatment effects of livestock grazing. Livestock grazing has been an important economic activity in our study area (Grand Staircase-Escalante National Monument) since the late 1800s and is still common today. Grazing is known to reduce biocrusts in piñon–juniper woodlands (Beymer and Klopatek, 1992). Additionally, in sagebrush-dominated ecosystems within Grand Staircase-Escalante National Monument, persistent heavy grazing has significantly impacted soil hydrologic conditions (Miller, 2008). Our treated sites may have been grazed more often than the untreated sites due to their higher herbaceous cover (Fig. 2). Thus, post-treatment management (i.e. grazing) may have played a role in the differences in bare mineral soil and biocrust cover between the treated and untreated sites.

There was no relationship between year of treatment and difference in soil, litter, rock, or biocrust cover among paired treated and untreated sites (linear regressions, all $R^2 < 0.0$ and $P > 0.35$). These results suggest that ground cover does not vary among sites that were chained 20 and 40 years ago. However, the lack of a relationship may be partly due to a relatively small sample size (17) given the heterogeneity of the sampling area and the paired study design.

3.7. Changes in tree cover, density, and recruitment

Chaining was effective at reducing tree abundance for 40 years following treatment: tree basal area was five fold higher in the untreated sites as compared to the treated sites (paired Student's t -test, $P < 0.001$; Fig. 4A) and tree density was twice as high in the

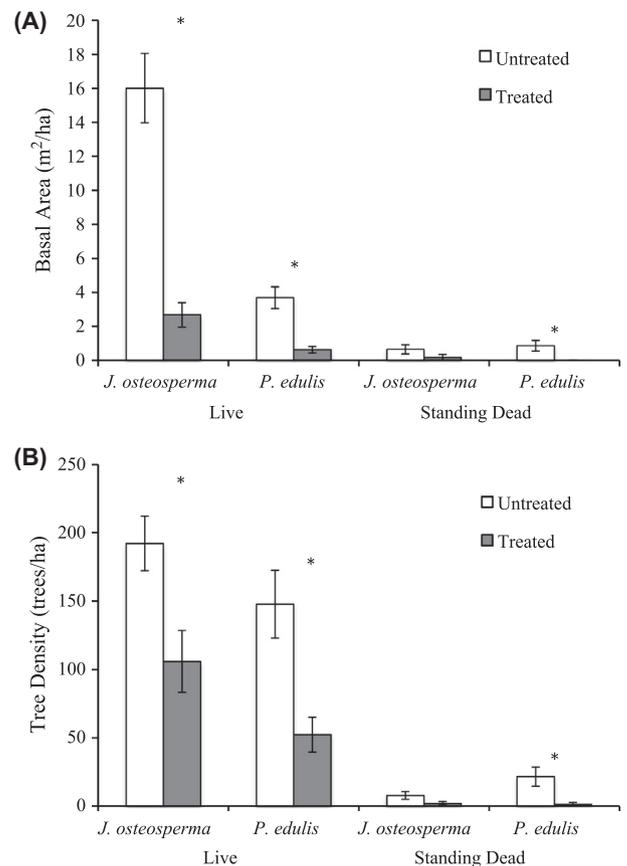


Fig. 4. Differences between treated (chained and seeded) and untreated sites in *J. osteosperma* and *P. edulis* basal area (A.) and tree density (B.) among live (left side) and standing dead (right side) adult trees (BTD ≥ 5 cm) at Grand Staircase-Escalante National Monument, Utah. Data are means \pm 1 SE and an asterisk denotes significant differences between treated and untreated sites, with $\alpha = 0.05$.

untreated sites as compared to the treated sites (paired Student's t -test, $P < 0.004$; Fig. 4B). These results suggest that chaining can be effective at decreasing tree density and basal area for 40 years, counter to what previous research has suggested (Skousen et al., 1989; Tausch and Tueller, 1977).

The low levels of adult tree basal area and density in the chained areas is likely due to the slow regeneration rate of *P. edulis* and *J. osteosperma*. Based on our linear regression of tree size and tree age from two of our paired-sites (Fig. A.1), 40 year old *P. edulis* trees were only 5 cm in basal trunk diameter (BTD) (Fig. A.1). Similarly, 40 year old *J. osteosperma* trees had a small canopy (0.64 m²) and were only ~6 cm in BTD (Fig. A.1 and Table A.1). Thus, the majority of adult trees in the treated area were trees that had established prior to treatment and were not killed during treatment. This is further highlighted by the low canopy cover (<0.5%) of trees that recruited after treatment as compared to trees that recruited prior to treatment (16%) in the treated areas (Fig. 5a). These results illustrate the slow regeneration rate of these woodlands and how the number of trees surviving treatment strongly determines the rate of recovery. Thus, the long-term effectiveness of chaining may be largely due to the number of juvenile trees that survived the treatment (Skousen et al., 1989; Fig. 5a), which may explain the difference in rates of recovery between our study and others (Skousen et al., 1989; Tausch and Tueller, 1977). Given the low cover of newly recruited trees over the past 40 years in our study area (Fig. 5a), our results suggest that if all trees were killed during treatment there may be little tree cover (<1%) 40 years after treatment.

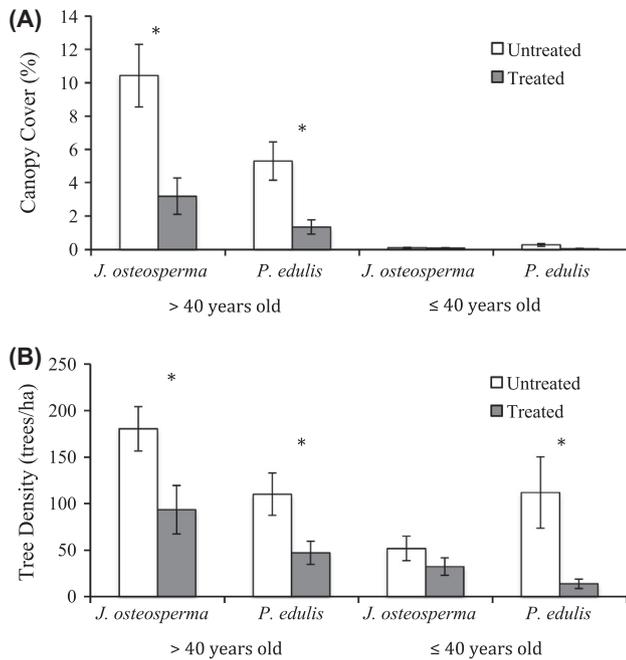


Fig. 5. Differences between treated (chained and seeded) and untreated sites in *J. osteosperma* and *P. edulis* canopy cover (a) and tree density (b) among trees >40 years old (i.e. trees that likely established prior to treatment) and trees ≤40 years old (i.e. trees that likely established post-treatment) at Grand Staircase-Escalante National Monument, Utah. Data are means ± 1 SE and only include the 12 sites that were treated between 1963 and 1971. An asterisk denotes significant differences between treated and untreated sites, with $\alpha = 0.05$.

While there was a trend of lower tree density and basal area among sites that had been chained more recently (i.e. 1980s) (Table 2), there was no strong relationship between year of treatment and difference in tree density or basal area among paired treated and untreated sites (linear regression, both $R^2 < 0.08$ and $P > 0.4$; Table 2). The lack of a relationship is likely due to the slow regeneration rate of *P. edulis* and *J. osteosperma* and may also be a function of the relatively low sample size given our paired sampling design and the high heterogeneity of P–J woodlands.

Unlike *J. osteosperma*, *P. edulis* regeneration may be negatively affected by chaining treatments, and consequently, treated woodlands may become increasingly *J. osteosperma* dominated. There was over 50% less recruitment of *P. edulis* during the last 40 years in treated sites compared to untreated sites among the sites treated between 1963 and 1971 (paired Wilcoxon test, $P = 0.01$; Fig. 5b), while there was no difference in *J. osteosperma* recruitment among treated and untreated sites (paired Student's *t* test, $P = 0.3$; Fig. 5b). Additionally, while *P. edulis* dominance among trees that were greater than 40 years old (i.e., trees that established prior to treatment) was similar in treated (mean ± SE: $36 \pm 7\%$) and untreated (mean ± SE: $35 \pm 7\%$) sites (paired Student's *t* test, $P > 0.8$; Fig. 5b), *P. edulis* dominance was significantly lower in treated sites (mean ± SE: $20 \pm 7\%$) as compared to untreated sites (mean ± SE: $63 \pm 9\%$) among trees that were less than 40 years old (i.e., trees that likely established after treatment) (paired Wilcoxon test, $P = 0.02$; Fig. 5b). Further, across all sites, there were over 10 times fewer *P. edulis* seedlings and saplings (BTD < 5 cm) in the treated sites as compared to the untreated sites (Fig. 6), whereas there was little difference in *J. osteosperma* seedling and sapling abundance (Fig. 6). These results support the idea that *J. osteosperma* is an earlier successional species that has higher establishment following disturbances compared to species of piñon (*P. monophylla* and *P. edulis*) (Barney and Frischknecht, 1974; Chambers et al., 1999). The differential recruitment response of *P. edulis* and *J. osteosperma* to chaining and other

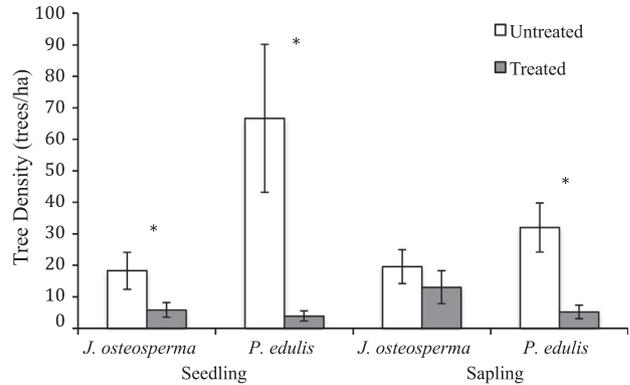


Fig. 6. Differences between treated (chained and seeded) and untreated sites in *J. osteosperma* and *P. edulis* seedling (BTD < 2.5 cm) and sapling (BTD < 5 cm and ≥ 2.5 cm) densities at Grand Staircase-Escalante National Monument, Utah. Data are means ± 1 SE and an asterisk denotes significant differences between treated and untreated sites, with $\alpha = 0.05$.

disturbances may be due to their differential abilities to establish and survive outside of the canopy of trees and shrubs. *J. osteosperma* is more drought tolerant than *P. edulis* (Linton et al., 1998; McDowell et al., 2008; Mueller et al., 2005), and while 90% of *P. edulis* juveniles are found underneath the canopy of trees and shrubs, significantly fewer (70%) *J. osteosperma* juveniles are found underneath the canopy of trees and shrubs (Redmond and Barger, 2013). Therefore, the decline in tree and shrub cover associated with the chaining method may have negatively affected *P. edulis* establishment more than *J. osteosperma* establishment due to reduced tree and shrub canopy cover.

Our results suggest that the chaining and seeding treatment method may result in an increase in *J. osteosperma* dominance, which could negatively impact wildlife and communities that rely on *P. edulis* for forage, fuel wood, and habitat (Brown et al., 2001). Additionally, there has been high drought-related mortality of *P. edulis* in certain areas across its range since 2002 (Breshears et al., 2005; Clifford et al., 2011), especially among reproductively mature trees (Floyd et al., 2009). There have also been recent declines in *P. edulis* cone production associated with increasing temperatures (Redmond et al., 2012). Thus, both recent mortality and decreased reproduction could further reduce *P. edulis* as a co-dominant in P–J woodlands.

3.8. Changes in surface fuel loads

While there were no differences between the treated and untreated sites in smaller diameter (1–10 h) surface fuels or 1000 h rotten surface fuels (Table 3), there were nearly two times more 100 h fuels and over five times more 1000 h fuels in the treated sites (Table 3). These results are consistent with other studies that

Table 3

Average (±1 SE) fuel loads (metric t ha⁻¹) in treated (chained and seeded) and untreated sites at Grand Staircase-Escalante National Monument, Utah.

Fuel component	Untreated	Treated
1 h	0.01 (0.00) ^a	0.01 (0.00) ^a
10 h	0.30 (0.02) ^a	0.39 (0.04) ^a
100 h	1.6 (0.2) ^a	3.0 (0.3) ^b
1–100 h	1.9 (0.2)	3.3 (0.3)
1000 h sound	3.1 (1.5) ^a	17.3 (5.9) ^b
1000 h rotten	10.9 (2.8) ^a	12.7 (3.2) ^a

Different letters denote significant differences between treated and untreated sites, with $\alpha = 0.05$.

found an increase in surface fuel loads following mechanical treatments that did not involve fire (Stephens and Moghaddas, 2005; Stephens et al., 2009). The high amount of 100 h and 1000 h surface fuels present 40 years after treatment demonstrates one of the differences of not physically removing, burning, or grinding up (i.e. mastication) killed trees, as wood decays slowly in these semi-arid ecosystems (Jacobi et al., 2005). These larger diameter surface fuels may increase habitat complexity and benefit ground-dwelling arthropods (Clifford et al., 2008; Grove, 2002) and may also alter fire behavior by increasing the length of time fire heats the soil surface (Clifford et al., 2008). However, despite the increased surface fuel loads (Table 3), the probability of catastrophic wildfires may not be higher in treated areas given the significantly lower canopy cover (Table 2; Clifford et al., 2008). There was no relationship between year since treatment and surface fuel loads ($R^2 = 0.09$, $P = 0.15$; Table 2), which highlights the slow decay rate of wood in semi-arid ecosystems.

4. Conclusion

Tree-reduction treatments involving chaining and seeding have been applied to over 169,000 ha of P–J woodlands across the Colorado Plateau over the past half-century (<http://www.mpcer.nau.edu/pj/pjwood/>) and are still used as a management technique. These treatments clearly have long-term (40 year) effects on ecosystem dynamics that need to be taken under consideration when developing future management strategies.

4.1. Management-intended long-term (40 year) effects of past chaining treatments

- Past chaining treatments effectively increased perennial grass cover, and thus, increased forage production. The increase in perennial grass cover was predominately due to the increase in *A. cristatum*, a nonnative bunchgrass that was seeded following treatment.
- Past chaining treatments effectively reduced tree cover. While trees were present in all treated sites, their populations had significantly lower densities in treated sites as compared to untreated sites. Additionally, we found slow tree regeneration in our treated sites (40 year old trees were less than 7 cm in basal trunk diameter), highlighting the slow growth of *P. edulis* and *J. osteosperma* in these semi-arid woodlands of the Colorado Plateau.
- While previous research found a decline in understory diversity immediately following chaining (O'Meara et al., 1981), our results suggest past chaining treatments did not affect species diversity in the long-term.
- No long-term treatment effects on the abundance of *B. tectorum*, a common invasive species, were detected. This may be partly due to the low invasibility of *B. tectorum* in our study area as well as competitive exclusion by *A. cristatum* and other perennial grasses (Chambers et al., 2007).

4.2. Unintended long-term (40 year) effects of past chaining treatments

- Treated areas had higher bare mineral soil cover and reduced biocrust cover, which may be due to the direct effects of chaining and seeding or due to post-treatment grazing patterns.
- Past chaining treatments may result in more *J. osteosperma* dominated (as compared to *P. edulis* dominated) woodlands. Treated areas had over 10 times fewer *P. edulis* seedlings and saplings than untreated areas, whereas there was no difference in *J. osteosperma* seedling and sapling density. Further, within

the 12 sites that were treated between 1963 and 1971, *P. edulis* dominance was 69% lower among trees that recruited in the last 40 years in the treated areas as compared to the untreated areas.

- Past chaining treatments increased nonnative species cover. This increase was predominately due to *A. cristatum*, a nonnative species that was seeded.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.foreco.2013.05.020>.

References

- Aro, R.S., 1971. Evaluation of pinyon–juniper conversion to grassland. *Journal of Range Management* 24, 188–197.
- Barger, N.N., Herrick, J.E., Zee, J., Belnap, J., 2006. Impacts of biological soil crust disturbance and composition on C and N loss from water erosion. *Biogeochemistry* 77, 247–263.
- Barger, N.N., Adams, H.D., Woodhouse, C., Neff, J.C., Asner, G.P., 2009. Influence of livestock grazing and climate on pinyon pine (*Pinus edulis*) dynamics. *Rangeland Ecology & Management* 62, 531–539.
- Barney, M.A., Frischknecht, N.C., 1974. Vegetation changes following fire in the pinyon–juniper type of west-central Utah. *Journal of Range Management* 27, 91–96.
- Bates, J.D., Miller, R.F., Svejcar, T., 2005. Long-term successional trends following western juniper cutting. *Rangeland Ecology & Management* 58, 533–541.
- Belnap, J., 1996. Soil surface disturbances in cold deserts: effects on nitrogenase activity in cyanobacterial-lichen soil crusts. *Biology and Fertility of Soils* 23, 362–367.
- Belnap, J., Gillette, D.A., 1998. Vulnerability of desert biological soil crusts to wind erosion: the influences of crust development, soil texture, and disturbance. *Journal of Arid Environments* 39, 133–142.
- Beymer, R.J., Klopatek, J.M., 1992. Effects of grazing on cryptogamic crusts in pinyon–juniper woodlands in Grand Canyon National Park. *American Midland Naturalist* 127, 139–148.
- BLM, 2008. Integrative vegetation management handbook. Bureau of Land Management, Handbook H-1740, Rel. 1–1714.
- Breshears, D.D., Cobb, N.S., Rich, P.M., Price, K.P., Allen, C.D., Balice, R.G., Romme, W.H., Kastens, J.H., Floyd, M.L., Belnap, J., Anderson, J.J., Myers, O.B., Meyer, C.W., 2005. Regional vegetation die-off in response to global-change-type drought. *Proceedings of the National Academy of Sciences* 102, 15144.
- Brown, J.K., 1974. Handbook for inventorying downed woody material. USDA For. Serv. Gen. Tech. Rep. INT-16, pp. 24.
- Brown, J.H., Whitham, T.G., Ernest, S.K.M., Gehring, C.A., 2001. Complex species interactions and the dynamics of ecological systems: long-term experiments. *Science* 293, 643–650.
- Chambers, J.C., Vander Wall, S.B., Schupp, E.W., 1999. Seed and seedling ecology of piñon and juniper species in the pygmy woodlands of western North America. *The Botanical Review* 65, 1–38.
- Chambers, J.C., Roundy, B.A., Blank, R.R., Meyer, S.E., Whittaker, A., 2007. What makes great basin sagebrush ecosystems invulnerable by *Bromus tectorum*? *Ecological Monographs* 77, 117–145.
- Clary, W.P., Jameson, D.A., 1981. Herbage production following tree and shrub removal in the pinyon–juniper type of Arizona. *Journal of Range Management* 34, 109–113.
- Clifford, M.J., Rocca, M.E., Delph, R., Ford, P.L., Cobb, N.S., 2008. Drought induced tree mortality and ensuing bark beetle outbreaks in southwestern pinyon–juniper woodlands. In: *Ecology, Management, and Restoration of piñon–Juniper and Ponderosa Pine Ecosystems: Combined Proceedings of the 2005 St. George, Utah*

- and 2006 Albuquerque, New Mexico Workshops. Proceedings RMRS-P-51, pp. 39–51.
- Clifford, M.J., Cobb, N.S., Buenemann, M., 2011. Long-term tree cover dynamics in a piñon-juniper woodland: climate-change-type drought resets successional clock. *Ecosystems* 14, 949–962.
- Cox, R.D., Anderson, V.J., 2004. Increasing native diversity of cheatgrass-dominated rangeland through assisted succession. *Rangeland Ecology & Management* 57, 203–210.
- Crall, A.W., Newman, G.J., Stohlgren, T.J., Jarnevich, C.S., Evangelista, P., Guenther, D., 2006. Evaluating dominance as a component of non-native species invasions. *Diversity and Distributions* 12, 195–204.
- Davenport, D.W., Breshears, D.D., Wilcox, B.P., Allen, C.D., 1998. Viewpoint: Sustainability of piñon-juniper ecosystems: a unifying perspective of soil erosion thresholds. *Journal of Range Management* 51, 231–240.
- Despain, D.W., 1989. Radial Growth Relationships in Utah juniper (*Juniperus osteosperma*) and Pinyon Pine (*Pinus edulis*). University of Arizona, Tuscon, AZ, p. 153.
- Farmer, M.E., Harper, K.T., Davis, J.N., 1999. The influence of anchor-chaining on watershed health in a juniper-pinyon woodland in central Utah. In: Proceedings, Ecology and Management of Pinyon-Juniper Communities within the Interior West. Proceedings RMRS-P-9. Ogden, UT. USDA Forest Service, Rocky Mountain Research Station 299301.
- Floyd, M.L., Clifford, M., Cobb, N.S., Hanna, D., Delph, R., Ford, P., Turner, D., 2009. Relationship of stand characteristics to drought-induced mortality in three southwestern piñon-juniper woodlands. *Ecological Applications* 19, 1223–1230.
- Gifford, G.F., 1973. Runoff and sediment yields from runoff plots on chained piñon-juniper sites in Utah. *Journal of Range Management* 26, 440–443.
- Grove, S.J., 2002. Saprophytic insect ecology and the sustainable management of forests. *Annual Review of Ecology and Systematics* 33, 1–23.
- Henderson, D.C., Naeth, M.A., 2005. Multi-scale impacts of crested wheatgrass invasion in mixed-grass prairie. *Biological Invasions* 7, 639–650.
- Hill, M.O., 1973. Diversity and evenness: a unifying notation and its consequences. *Ecology* 54, 427–432.
- Jacobi, W.R., Kearns, H.S.J., Johnson, D.W., 2005. Persistence of pinyon pine snags and logs in southwestern Colorado. *Western Journal of Applied Forestry* 20, 247–252.
- Leffler, A.J., Monaco, T., James, J., 2011. Morphological and physiological traits account for similar nitrate uptake by crested wheatgrass and cheatgrass. *Natural Resources and Environmental Issues*, 17.
- Lesica, P., DeLuca, T.H., 1996. Long-term harmful effects of crested wheatgrass on Great Plains grassland ecosystems. *Journal of Soil and Water Conservation* 51, 408–409.
- Linton, M.J., Sperry, J.S., Williams, D.G., 1998. Limits to water transport in *Juniperus osteosperma* and *Pinus edulis*: implications for drought tolerance and regulation of transpiration. *Functional Ecology* 12, 906–911.
- McDowell, N., Pockman, W.T., Allen, C.D., Breshears, D.D., Cobb, N., Kolb, T., Plaut, J., Sperry, J., West, A., Williams, D.G., Yezpe, E.A., 2008. Mechanisms of plant survival and mortality during drought: why do some plants survive while others succumb to drought? *New Phytologist* 178, 719–739.
- Miller, M.E., 2008. Broad-scale assessment of rangeland health, Grand Staircase-Escalante National Monument, USA. *Rangeland Ecology & Management* 61, 249–262.
- Miller, R.F., Rose, J.A., 1999. Fire history and western juniper encroachment in sagebrush steppe. *Journal of Range Management* 52, 550–559.
- Miller, R.F., Tausch, R.J., McArthur, E.D., Johnson, D.D., Sanderson, S.C., 2008. Age Structure and Expansion of piñon-Juniper Woodlands: a Regional Perspective in the Intermountain West. Res. Pap. RMRS-RP-69. Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. 15p.
- Monsen, S.B., Stevens, R., Shaw, N.L., 2004. Grasses, in: Restoring Western Ranges in Wildlands, USDA Forest Service Gen. Tech. Rep. RMRS-GTR-136. US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO., pp. 295–424.
- Mueller, R.C., Scudder, C.M., Porter, M.E., Talbot Trotter, R., Gehring, C.A., Whitham, T.G., 2005. Differential tree mortality in response to severe drought: evidence for long-term vegetation shifts. *Journal of Ecology* 93, 1085–1093.
- Natural Resource Conservation Service. 2006. Major Land Resources Area: Colorado and Green River Plateaus. Natural Resources Conservation Service. URL <<http://www.ut.nrcs.usda.gov/technical/technology/range/mlra35.html>> (accessed 11.3.11).
- Noson, A.C., Schmitz, R.A., Miller, R.F., 2006. Influence of fire and juniper encroachment on birds in high-elevation sagebrush steppe. *Western North American Naturalist* 66, 343–353.
- O'Meara, T.E., Haufler, J.B., Stelter, L.H., Nagy, J.G., 1981. Nongame wildlife responses to chaining of piñon-juniper woodlands. *The Journal of Wildlife Management* 45, 381–389.
- Ott, J.E., McArthur, E.D., Roundy, B.A., 2003. Vegetation of chained and non-chained seedlings after wildfire in Utah. *Journal of Range Management* 56, 81–91.
- Peet, R.K., 1974. The measurement of species diversity. *Annual Review of Ecology and Systematics* 5, 285–307.
- Pierson, F.B., Bates, J.D., Svejcar, T.J., Hardegree, S.P., 2007. Runoff and erosion after cutting western juniper. *Rangeland Ecology & Management* 60, 285–292.
- R Development Core Team. 2011. R: a Language for Statistical Computing. Vienna, Austria: R Foundation for Statistical Computing. URL: <<http://www.R-project.org>>.
- Redmond M.D., Barger N.N., 2013. Tree regeneration following drought and insect-induced mortality in piñon-juniper woodlands, *New Phytologist*, in press. <http://dx.doi.org/10.1111/nph.12366>
- Redmond, M.D., Forcella, F., Barger, N.N., 2012. Declines in piñon pine cone production associated with regional warming. *Ecosphere* 3, 120.
- Rippel, P., Pieper, R.D., Lymbery, Gordon A., 1983. Vegetational evaluation of piñon-juniper cabling in south-central New Mexico. *Journal of Range Management* 36, 13–15.
- Romme, W.H., Allen, C.D., Bailey, J.D., Baker, W.L., Bestelmeyer, B.T., Brown, P.M., Eisenhart, K.S., Floyd, M.L., Huffman, D.W., Jacobs, B.F., Miller, R.F., Muldavin, E.H., Swetnam, T.W., Tausch, R.J., Weisberg, P.J., 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon-juniper vegetation of the western United States. *Rangeland Ecology & Management* 62, 203–222.
- Skousen, J.G., Davis, J.N., Brotherson, J.D., 1989. Pinyon-juniper chaining and seeding for big game in central Utah. *Journal of Range Management* 42, 98–104.
- Smoliak, S., Dormaar, J.F., 1985. Productivity of russian wildrye and crested wheatgrass and their effect on prairie soils. *Journal of Range Management* 38, 403–405.
- Stephens, S.L., Moghaddas, J.J., 2005. Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. *Forest Ecology and Management* 215, 21–36.
- Stephens, S.L., Moghaddas, J.J., Edminster, C., Fiedler, C.E., Haase, S., Harrington, M., Keeley, J.E., Knapp, E.E., McIver, J.D., Metlen, K., Skinner, C.N., Youngblood, A., 2009. Fire treatment effects on vegetation structure, fuels, and potential fire severity in western US forests. *Ecological Applications* 19, 305–320.
- Tausch, R.J., Tueller, P.T., 1977. Plant succession following chaining of piñon-juniper woodlands in eastern Nevada. *Journal of Range Management* 30, 44–49.
- Tausch, R.J., West, N.E., Nabi, A.A., 1981. Tree age and dominance patterns in Great Basin piñon-juniper woodlands. *Journal of Range Management* 34, 259–264.
- Thompson, T.W., Roundy, B.A., McArthur, E.D., Jessop, B.D., Waldron, B., Davis, J.N., 2006. Fire rehabilitation using native and introduced species: a landscape trial. *Rangeland Ecology & Management* 59, 237–248.
- Walker, S.C., 1997. Species compatibility and successional processes affecting seeding of piñon-juniper types. In: Ecology and Management of Pinyon-Juniper Communities in the Interior West., Proc. RMRS-P-9. USDA Forest Service, Rocky Mountain Research Station, Provo, Utah, pp. 331–337.
- Wilcox, B.P., 1994. Runoff and erosion in intercanopy zones of piñon-juniper woodlands. *Journal of Range Management* 47, 285–295.
- Wilson, S.D., Gerry, A.K., 1995. Strategies for mixed-grass prairie restoration: herbicide, tilling, and nitrogen manipulation. *Restoration Ecology* 3, 290–298.