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Response of Conifer-Encroached Shrublands in the Great Basin to Prescribed Fire and Mechanical Treatments

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Abstract

In response to the recent expansion of piñon and juniper woodlands into sagebrush-steppe communities in the northern Great Basin region, numerous conifer-removal projects have been implemented, primarily to release understory vegetation at sites having a wide range of environmental conditions. Responses to these treatments have varied from successful restoration of native plant communities to complete conversion to nonnative invasive species. To evaluate the general response of understory vegetation to tree canopy removal in conifer-encroached shrublands, we set up a region-wide study that measured treatment-induced changes in understory cover and density. Eleven study sites located across four states in the Great Basin were established as statistical replicate blocks, each containing fire, mechanical, and control treatments. Different cover groups were measured prior to and during the first 3 yr following treatment. There was a general pattern of response across the wide range of site conditions. There was an immediate increase in bare ground and decrease in tall perennial grasses following the fire treatment, but both recovered by the second or third growing season after treatment. Tall perennial grass cover increased in the mechanical treatment in the second and third year, and in the fire treatment cover was higher than the control by year 3. Nonnative grass and forb cover did not increase in the fire and mechanical treatments in the first year but increased in the second and third years. Perennial forb cover increased in both the fire and mechanical treatments. The recovery of herbaceous cover groups was from increased growth of residual vegetation, not density. Sagebrush declined in the fire treatment, but seedling density increased in both treatments. Biological soil crust declined in the fire treatment, with no indications of recovery. Differences in plant response that occurred between mechanical and fire treatments should be considered when selecting management options.

Key Words: sagebrush, cheatgrass, nonnative species, piñon–juniper, restoration, Utah juniper, western juniper, single-needle piñon, resilience

INTRODUCTION

Since the 1860s, several species of piñon and juniper have expanded into grassland, sagebrush-steppe, and aspen communities, increasing 125–625% in the central and northern portions of the Great Basin and river basins to the north and west (Cottam and Stewart 1940; Adams 1975; Burkhardt and Tisdale 1976; Tausch et al. 1981; Tausch and West 1988; Miller and Rose 1995; Tausch and West 1995; Gedney et al. 1999; Miller and Rose, 1999; Wall et al. 2001; Johnson and Miller 2006; Weisberg et al. 2007; Miller et al. 2008). In response to these recently formed conifer-encroached shrublands, private landowners and public agencies have treated large areas across the interior West by removing trees with prescribed fire and mechanical methods. The rationale for tree removal has included fuel reduction, restoration of sage-steppe communities and watersheds, and enhanced forage and wildlife habitat. Unfortunately, plant-community response to tree removal is not always consistent or predictable, and succession may not move in a desirable direction following treatment (Miller et al. 2013). Successional trajectories following tree removal in conifer-encroached shrublands can range from a progression toward native shrub-steppe communities (Tausch and Tueller 1977; Everett and Sharrow 1985a; Skousen et al. 1986; Stager and Klebenow 1987; Rose and Eddleman 1994; Bates et al. 2000, 2006, 2007; Coultrap et al. 2008) to no change in native understory vegetation (Everett and Sharrow 1985a; Yorks et al. 1994; Bates et al. 2006; Bristow 2010), to large increases in invasive annuals, at least during the first few years after tree removal (Barney and Frischknecht 1974; Koniak 1985; Skousen et al. 1986; Bates et al. 2007).

The initial stages of succession following treatment may be largely dependent on pre-existing conditions. Several studies have proposed a successional model following piñon and juniper removal that progresses from native and/or nonnative annuals to perennial grass-forb to perennial grass-forb shrub to young trees occupying a site, and eventually to mature woodland (Erdman 1970; Barney and Frischknecht 1974; Koniak 1985; Skousen et al. 1989). However, vegetation

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dominating the initial phase of succession has been reported to depend upon pre-existing plant composition (Bunting 1985) and can start with a strong response of native perennials (Everett and Sharrow 1985b; Bates et al. 2000, 2007). Conversely, where native herbaceous vegetation is depleted the potential for recovery is limited (Yorks et al. 1994) and invasive annuals may continue to dominate many years after disturbance (Koniak 1985). This may result in a shift of conifer-encroached shrublands to new steady states dominated by introduced annuals and biennials following treatment or wildfire (Pellant and Hall 1994; Tausch 1999; Holmes and Miller 2010; Miller et al. 2011).

Piñon- and juniper-encroached shrublands occur over a wide range of environmental conditions and are represented by continually changing gradients in climate, elevation, aspect, slope, geology, soils, and disturbance regimes (West et al. 1978, 1998; Gedney et al. 1999; Miller et al. 2000, 2005, 2011, 2013). This makes current region-wide recommendations difficult and possibly unreliable. However, generalized state-and-transition models that account for specific site attributes (e.g., soil moisture/temperature regimes) may link sites with similar attributes, enabling managers to apply results from one location to another (Bestelmeyer et al. 2009; Chambers et al. 2013; Miller et al. 2013, 2014; Chambers et al. 2014). A key question to address is—Are there predictable patterns in understory response to conifer removal? Although numerous studies have evaluated plant response to tree removal, most studies have been conducted at one location and lack comparability because they vary widely in predisturbance plant composition, the kind and severity of disturbance (e.g., wildfire versus prescribed fire treatment), posttreatment disturbance, and site attributes. A single study conducted at one time and in one or a few places cannot describe the general patterns of plant succession following tree removal across the wide range of environmental variables that characterize conifer encroached shrublands. And there have been surprisingly few studies that made replicated side-by-side comparisons between mechanical and prescribed fire treatments (Miller et al. 2013). The study by Everett and Sharrow (1985a), which evaluated mechanical tree removal across central Nevada, is one of the few multilocational studies conducted in the Intermountain West. Their results showed a distinct pattern in vegetation response across a large area in central Nevada. The results from regional studies that cover a meaningful portion of site variation greatly enhance our ability to evaluate the response of different plant and ground-cover groups under a wide range of conditions, and thus allow us to develop general state-and-transition models. Such studies can describe patterns of variation in plant successional trajectories in conifer-encroached shrublands across a large region, evaluate the consistency of response, identify attributes that may be linked to the response, and allow for the evaluation and extrapolation of results from one site to another. In addition, studies that combine before and after measurements with unmanipulated controls across multiple sites allow for the evaluation of treatment effects in the context of both temporal and spatial variation (Carpenter 1990).

The Sagebrush Treatment Evaluation Project (McIver et al. 2010) set up such a region-wide study to evaluate patterns of understory plant succession after tree removal and to identify important attributes for the development of predictive models. Study sites were located in conifer-encroached shrublands across a broad geographic area in the northern Great Basin and basins to the north and west. Each study site was a complete block of prescribed fire, mechanical, and control treatments, applied over relatively large plots (> 8 ha). We asked three questions relative to understory plant response to tree removal by fire and mechanical treatments: 1) how do different plant and ground-cover groups respond across a relatively wide range of tree-encroached sites, 2) how consistent was the response to tree removal across a wide range of sites, and 3) do understory cover groups respond differently to removal of juniper and/or piñon by fire versus mechanical treatments during the first 3 posttreatment years?

METHODS AND MATERIALS

Study Area

Eleven sites were selected across a broad geographical area in Utah, Nevada, California, and Oregon, encompassing a wide range of environmental conditions (Fig. 1) (McIver et al. 2010). Criteria used to select sites were 1) the dominant shrub was *Artemisia tridentata* Nutt., 2) there was no evidence that these stands had recently been dominated by old-growth juniper or piñon pine (absence of large stumps and/or logs), 3) soils were loams, 4) native grasses and forbs were present in the understory, and 5) introduced species were present but not a dominant component (observed cover of nonnatives was equal to or less then native herbs). The sites represented four different cover types, each reflecting a different dominant tree species (Table 1). Four study sites were located in western juniper (*Juniperus occidentalis* Hook.) in California and Oregon, three sites in singleleaf piñon (*Pinus monophylla* Torr. & Frém.) and Utah juniper (*Juniperus osteosperma* [Torr.] Little) in Nevada, and in Utah, two sites in Utah juniper, and two sites in Utah juniper and Colorado piñon (*Pinus edulis* Engelm). The sites are located within five climate-based major land resource areas (MLRAs), which include the Malheur High Plateau, Klamath Basin, Upper Snake River, Central Nevada Basin and Range, and Salt Lake Basin (US Department of Agriculture–Natural Resources Conservation Service [USDA-NRCS] 2011) (Table 1; Fig. 1). Sites vary considerably in elevation, topography, soils, current vegetation, and climate (Table 1). Soil temperature and moisture regimes ranged from warm–mesic to cool–frigid and aridic–xeric to xeric, respectively. Tree canopy cover within and across the 11 sites varied widely within and across study sites (see figure 4 in Roundy et al. 2014a) from relatively open (< 5%) to closed (> 20%). The predominant shrubs and grasses present on the sites prior to treatment are listed in Table 1.

Although all sites are classified as cold desert, weather patterns differ markedly across this geographic range. Sites in California and Oregon have a Pacific Maritime climate, with nearly all precipitation originating in the Pacific Ocean and falling between November and June. This area, which includes the western juniper cover type, lies north of the polar front gradient where temperatures are cooler, summer precipitation lower, and winter precipitation higher (Mitchell 1976). In contrast, sites in Nevada and Utah are located south of the

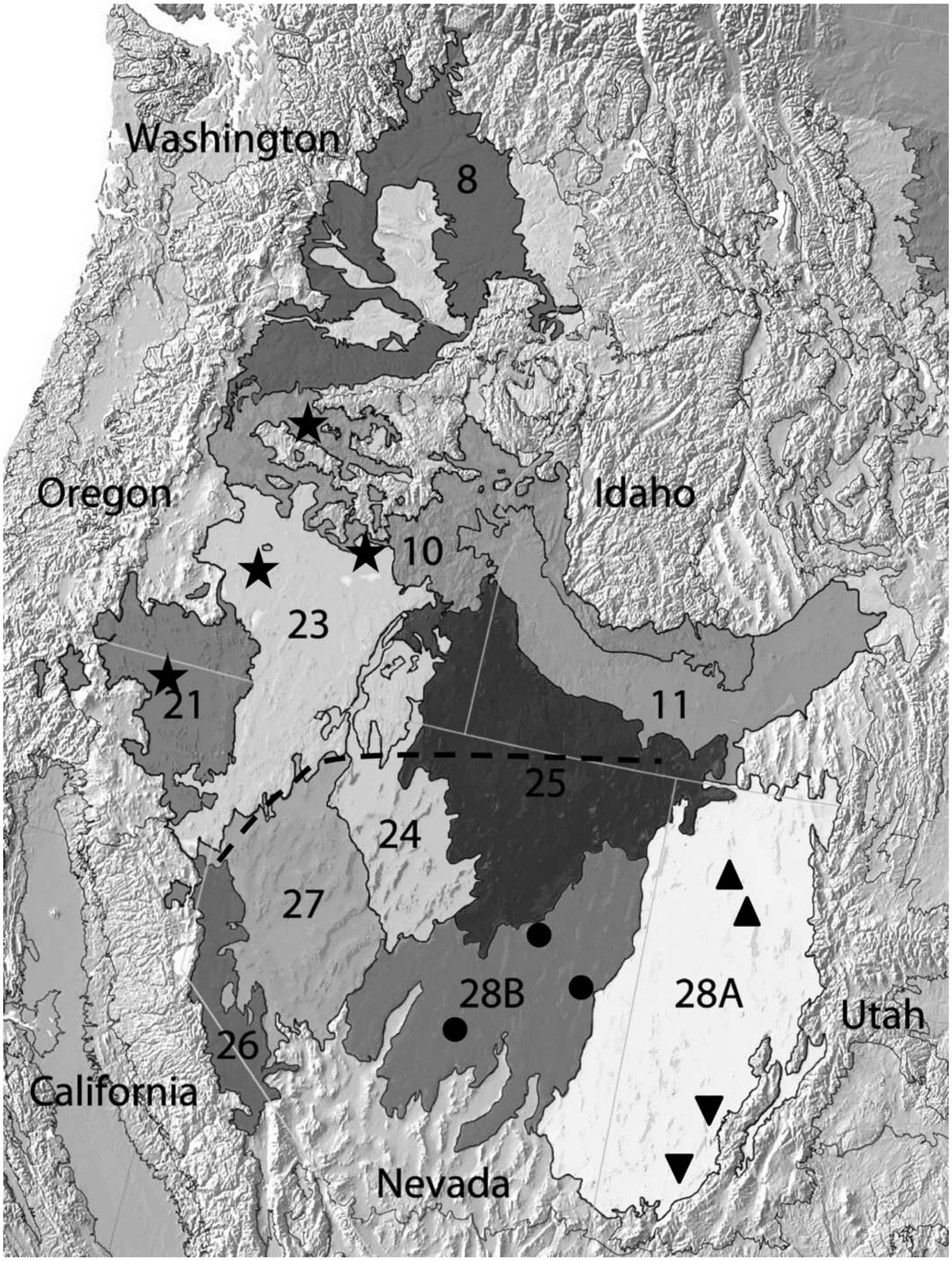

Figure 1. Location of the 11 sites across four major land resource area (MLRA), which include 10=Upper Snake River (USR), 21=Klamath Basin (KB), 23=Malheur High Plateau (MHP), 28B=Central Nevada Basin and Range (CNBR), 28A=Salt Lake Basin (SLB). The black dashed line separates the sagebrush steppe (north) and Great Basin sagebrush shrub (south) (West 1983). Sites with different tree cover groups are ★=western juniper, ●=two-needle piñon, ▲=Utah juniper, and ▼= Utah juniper+Colorado juniper (derived from US Department of Agriculture Natural Resources Conservation Service, 2011 by Eugénie MontBlanc, University of Nevada, Reno, NV).

polar front gradient, and have a more Continental climate, with less precipitation falling from November to June, and relatively more summer rains originating from the Gulf of Mexico, usually in July or August. However, the summer monsoonal influence for study sites in this portion of the study area is weak compared to regions lying to the south (Colorado Plateau and the southwest Hot Deserts).

Site Selection and Experimental Design

Treatment-plot layout was a randomized complete block, with each of the 11 study sites representing a block or replicate. Each replicate block was measured during the growing season immediately prior to treatment application and for three growing seasons immediately following. At each of the 11 study sites, three 8–20-ha treatment plots were established, and

Table 1. List of 11 sites within four tree cover types, major land resource area (MLRA) soil temperature/moisture regimes, dominant species in shrub/herb layer, soils, elevation, slope and aspect, and year treatments were applied. MLRAs: KB indicates Klamath Basin; MHP, Malheur High Plateau; CNBR, Central Nevada Basin and Range; SLB, Salt Lake Basin; and USR, Upper Snake River. Idaho fescue (*Festuca idahoensis* Elmer), bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) Á. Löve), needle and thread (*Hesperostipa comata* [Trin. & Rupr.] Barkworth), Thurber's needlegrass (*Achnatherum thurberianum* [Piper] Barkworth), muttongrass (*Poa fendleriana* [Steud.] Vasey), squirreltail (*Elymus elymoides* [Raf.] Swezey), and curlleaf mountain mahogany (*Cercocarpus ledifolius* Nutt.).

Cover type, study site (state), MLRA, soil regimes	Current native vegetation (PPT zone mm)	Soils	Elevation range	Slope/aspect	Year treated
Western Juniper					
Blue Mountain Site (CA), MHP, frigid/xeric	Mountain big sage/Idaho fescue–Sandberg bluegrass–bluebunch wheatgrass (300–400)	Loamy, mixed, frigid, lithic to Pachic Haploxerolls	1 500–1 700 m	5% N	2007
Bridge Creek (OR), USR, mesic/aridic–xeric	Basin big sage/bluebunch wheatgrass–Sandberg bluegrass (250–300)	Sandy loam, mesic Typic Haploxerolls to Frigid Torriorthents	800–900 m	25% NW	2006
Devine Ridge (OR), MHP, frigid/xeric	Mountain big sage/Sandberg bluegrass–Thurber's needlegrass–Idaho fescue (300–400)	Loamy–skeletal, mixed, superactive, frigid Lithic Haploxerolls	1 600–1 700 m	0–8% W	2007
Walker Butte (OR), MHP, frigid/xeric	Mountain big sage/Idaho fescue–Thurber's needlegrass–squirreltail (250–300)	Ashy, glassy, frigid vitritorrandic Durixerolls	1 400–1 500 m	Flat	2006
Pinyon–Utah juniper					
Marking Corral (NV), CNBR, cool mesic/aridic–xeric	Wyoming big sage/Thurber's needlegrass (300–400)	Loamy–skeletal, mixed, superactive, mesic Argidic Durixerolls	2 300–2 400 m	6–20% NW, NE, SE	2006
Seven Mile (NV), CNBR, cool frigid/xeric	Mt. mahogany–mountain big sage/bluebunch wheatgrass–muttongrass (12–16)	Loamy–skeletal, mixed, frigid Lithic & Typic Argixerolls	2 300–2 500 m	6–15% NW , E, SE	2007
South Ruby (NV), CNBR, warm frigid/xeric	Wyoming big sage–bitterbrush/bluebunch wheatgrass–Sandberg bluegrass–Thurber needlegrass (300–400)	Loamy, mixed, superactive, mesic, shallow Haploduridic Durixerolls	2 100–2 200 m	8–30% all aspects	2008
Utah juniper					
Greenville Bench (UT), SLB, warm frigid/xeric	Wyoming big sage/needle and thread–bluebunch wheatgrass (300–350)	Decca, very gravelly to cobbly sandy loam	1 750–1 850 m	2–28% N	2007
Stansbury Mountains (UT), SLB, cool mesic/aridic–xeric	Mountain big sage–antelope bitterbrush/bluebunch wheatgrass (300–350)	Loamy–skeletal, mixed, mesic Aridic Calcixerolls	1 700–1 850 m	8–30% W	2007
Utah juniper–Colorado pinyon					
Scipio (UT) SLB, warm mesic/aridic	Wyoming big sage/bluebunch wheatgrass (250–300)	Loamy–skeletal, mixed, mesic, shallow Calcic Petrocalcids	1 700–1 800 m	2–28% W	2007
Onaqui (UT) SLB, warm mesic/aridic–xeric	Wyoming big sage/bluebunch wheatgrass (250–300)	Loamy–skeletal, carbonatic, mesic, shallow Aridic Petrocalcic Palexerolls	1 700–2 100 m	2–30% E	2006

at each site we placed three treatment plots on a similar ecological site (e.g., similar topographic position, soils, and vegetation). Information collected across treatment plots included elevation, aspect, slope, topographic position, microtopography (concave, convex, flat), plant association (if known), current vegetation (dominant species in each vegetation layer), and soils. Treatment plots were fenced to exclude cattle grazing where necessary, because grazing could not be consistently controlled across the 11 study sites.

Treatments

Three treatments were applied at each of the 11 study sites by the government agency having jurisdiction for the site. One of three treatments was randomly selected to be applied in each treatment plot across the replicate block: control, mechanical, and fire. Treatments were applied at each site in the same year to form a statistical replicate block. Because of the size of the broadcast fire treatment and the number of locations, we were unable to apply treatments to all replicate blocks in the same year (Table 1). Limitations included a combination of weather, clearances, and smoke. Treatments were applied in 2006, 2007, and 2008, forming a staggered-start design (Loughin 2006). A staggered-start design alleviates the effects of starting an experiment under the same set of climate conditions, which limits the extrapolation of results. Fire was applied between August and October and mechanical treatments between September and November. The mechanical treatment consisted of cutting all trees > 2m tall and leaving them on the ground. The fire treatment was a broadcast burn ranging from low to moderate severity across all sites based on fuel consumption characteristics (Parson et al. 2010; Miller et al. 2013, 2014). The reduction of tree canopies in the fire treatment and mechanical treatments averaged 86% (range 63–100%) and 99% (range 96–100%), respectively, across the 11 study sites (see Roundy et al. 2014a), indicating that treatments were effective in accomplishing targeted tree removal goals. The burn treatment resulted in > 90% reduction in shrub cover with 50–75% shrub skeletons remaining. Greater than 80% of the plots were burned with fine surface fuels consumed or charred with little to no white ash on the surface except directly beneath the tree canopies. Most 100-h and greater fuels were charred but not consumed. The tree canopy was reduced to < 5% across all burned plots.

Vegetation Measurements

To compare vegetation response, nine cover groups were measured within each treatment plot of each replicate block in 15 randomly selected permanent 0.1-ha (33 × 30 m) subplots. All subplot corners were marked with steel stakes and UTM coordinates recorded. Measurements were collected the year prior (year 0) to and in the first 3 yr (year 1 to year 3) following treatment. With the exception of tree canopy, measurements of all cover groups were collected along five permanent 30-m transects placed at 2, 7, 15, 23, and 28 m along the 30-m baseline of each subplot. Plant and ground cover groups were sampled with the use of the point-intercept method along each transect (Herrick et al. 2009). Points were sampled at 0.5-m intervals along each of the five 30-m transects for a total of 300 points per subplot and 4 500 points per treatment plot.

Native vascular plant cover groups included total shrubs, sagebrush (*Artemisia* L.), tall perennial grasses, short perennial grasses (*Poa secunda* J. Presl.), perennial forbs, and forbs that sage-grouse (*Centrocercus urophasianus*) consume (e.g., *Crepis* sp., *Phlox longifolia*, *Microsteris gracilis,* etc.). The nonnative plant cover group included both non-native grasses (primarily *Bromus tectorum*) and nonnative forbs. Ground surface cover groups included bare ground, litter, and biological soil crusts. With the exception of bare ground, foliar cover of each cover group was recorded as a single hit at each point if the point came into contact with that group. However, shrub canopy cover rather than foliar cover was measured by recording a hit as a direct contact or the point falling within the live canopy perimeter. More than one cover group could be recorded at a single point, but each group could only have a maximum of one hit per point. Bare ground was recorded only if it was the first and only hit.

Density was measured in year 0 through year 3 for tall perennial grasses, nonrhizomatous perennial forbs, and shrub species < 50 mm in height in ¼-m^2 quadrats along the 7-, 15-, and 23-m transects, placed at every odd meter (n=45/subplot). Along these same three transects, all shrub species > 50 mm in height within 1 m of the transect (2 m × 30 m) were counted.

To measure the reduction in tree canopies to treatment, pretreatment live canopy cover of all trees > 0.5 m in height was recorded within each subplot by measuring the longest and perpendicular crown diameters. Crown area (A) for each tree was calculated with the use of the equation: $A = \pi\ (D1^*D2)/4$, where D1 is the longest and D2 the perpendicular to D1 canopy diameter. Total tree canopy cover was estimated by summing the crown area of each tree in the subplot. The same technique was used in the third posttreatment year to measure the reduction in tree canopy following treatment.

Statistical Analysis

Year was handled as time-since-treatment (0, 1, 2, and 3 yr). To evaluate potential treatment differences, we compared cover and density of the different cover groups across treatments within each year. Year 0 (baseline) data were analyzed to see if the randomization resulted in apparent preapplication differences. Subsequent years were analyzed as described below. We analyzed all 11 study sites for both year 0 and year 1. For year 2, data for 10 sites were analyzed because one site had only one posttreatment year (South Ruby). For year 3 data, nine sites were analyzed because at the Stansbury study site we lost all three treatments to a wildfire before year 3.

For the proportional cover response generalized linear mixed models (PROC GLIMMIX in SAS 9.2) were used with logit link. Binomial-type variation and over dispersion were accounted for with R-side residual variation (quasilikelihood). For each of the 3 yr after application of the treatments, both the year x and year 0 proportional cover were included as two responses for each experimental unit, so that treatment effects could be assessed for each treatment over time on the change in (logit of) cover since year zero (treatment-by-time interaction). Whenever such treatment effects were detected, regional effects (both additive and interactions) were added to the model, to see if there was evidence that the treatment effect (treatment-by-time) differed between regions (region by treatment by time). If not, then the study-area–averaged change since year 0 for each treatment was estimated and compared between treatments in pairwise comparisons. There were occasional convergence problems with this model, which were handled in two ways: 1) by removing sites with very little information about treatment effects (because of a preponderance of zero cover responses), and/or 2) increasing the convergence criterion (PCONV=option) to as high as 1E-5. When the logit link is used, the inferences regarding treatment differences are made on the logit scale, which can then be interpreted as multiplicative changes in the odds ($=P/[1-P]$). In Table 2, within-treatment comparisons across years with values > 1.0 indicate an increase in posttreatment cover and < 1.0 a decrease in posttreatment cover relative to year 0. For among-treatment comparisons within years (e.g., FI/ME), odds ratios > 1.0 indicate that the magnitude of increase was either larger or the decrease smaller for the treatment in the numerator compared to the treatment in the denominator. For example: nonnatives in year 3 significantly increased in both the FI (4.630) and ME (2.762) treatments compared to year 0. However, the difference in magnitude of increase was significantly greater in the FI than ME treatment (FI/ME, 4.630/2.762 = 1.6756). Although values are based on a logit scale, ratios approximate the magnitude of change of mean values.

To evaluate the consistency of treatment response of a specific cover group across study sites, we compared the relative change in the control treatment with the relative change in the fire treatment or mechanical treatment between year 0 and year 3 based on the logit scale. We report the proportion of study sites that expressed a positive increase, no change, or decrease compared to the control.

Differences in density were analyzed each year with the use of repeated-measures ANOVA for a randomized complete block design. Main effects were years within treatments and treatments within years. Data were tested for normality with the use of SAS univariate procedure. When treatment interactions were found to be significant (*P* <hairsp;0.05), the Tukey test was used to determine significant differences between treatments.

RESULTS

Cover

Cover groups in treatment plots within replicate blocks were relatively similar (year 0, Table 3), resulting in no significant differences between treatment plots prior to treatment (Table 2). In addition, the consistency of covariate significance and lack of significant treatment × covariate interactions for pretreatment data across replicate blocks (Roundy et al. 2014a) allows us to assume differences among cover groups for year 1 to year 3 were a result of treatment effects.

Bare Ground. In year 0 bare ground ranged from 10% to 49% across study sites and averaged 29% (Table 3). In year 1 bare ground was significantly higher in the fire-treatment plots compared to year 0 and was greater than in the mechanical and control treatments (Table 2). By year 2, bare ground declined in the fire treatment to levels similar to the control, but remained higher than both the mechanical treatment in year 2 and the pretreatment year (year 0). By year 3 bare ground in the fire-treatment plots did not differ from the mechanical and control treatments or year 0. The mechanical treatment had the least amount of bare ground in year 2 but was similar to the other two treatments by year 3. Changes in bare ground reported above occurred in 90% of the sites.

Total Shrub and Sagebrush Cover. Prior to treatment, total shrub cover averaged 11.5% (ranging from 3% to 24%) across study sites. Following treatment, shrub cover was significantly lower in the fire-treatment plots than in the mechanical or control treatments and year 0 in all three posttreatment years (Table 2). Shrub cover increased twofold in the fire treatment between year 1 (1.3%) and year 3 (3%), but still remained well below the other two treatments and year 0. An increase in shrub cover in the fire-treatment plots between year 1 and year 3 occurred in all sites. Shrub cover in the control and mechanical treatments did not differ over the 3 yr; however, shrub cover in the mechanical treatment was significantly greater in year 3 than year 0, which occurred in 78% of the plots.

Sagebrush cover, which accounted for 68% of total shrub cover, followed a similar pattern as total shrubs, and was reduced to levels in the treatment below the other treatments (Table 2). Fire treatment initially reduced sagebrush cover from 7.6% to 0.5%, but cover increased to 1% by year 3. There was no significant difference in sagebrush cover between mechanical and control treatments or between year 3 and year 0 because of high variability among sites. However, measured cover was 20% greater in the mechanical treatment in year 3 compared to year 0.

Tall and Short Perennial Grass Cover. Tall perennial grass cover averaged 11% (range 1–37%) across the sites in year 0 (Table 3). Cover in the fire treatment decreased 36% in year 1 compared to year 0 and was significantly less than cover in the mechanical or control treatments (Table 2). The decline in tall perennial grass cover in year 1 across the fire treatment occurred in 82% of the sites. In year 2, tall perennial grass cover recovered in fire-treatment plots to similar percentages as the control and year 0 (70% of the sites). By year 3 cover was higher on the fire treatment than in the control. This occurred in 90% of the sites. In the mechanical treatment tall perennial grass cover was significantly greater in year 2 and year 3 compared to year 0 (increasing 150%) and the control (Table 2). The greater increase in perennial grasses in the mechanical relative to changes in the control treatment in year 3 occurred in 89% of the sites.

Short perennial grass cover, which averaged 5.5% across sites prior to treatment, did not differ among treatments or years (Table 2).

Perennial Forbs and Forbs Used by Sage-Grouse. Perennial forb cover increased in the fire-treatment and mechanical plots relative to the control in year 2 and year 3 (Table 2). Cover in the fire and mechanical treatments was also greater in year 3 than year 0, and this occurred in 90% and 78% of the sites, respectively. Cover of forbs used by sage-grouse was higher in the fire treatment than year 0 and the control treatment in all years (Table 2). In the mechanical treatment cover was higher in year 2 and year 3 compared to both year 0 and the control treatment. The relative increase in sage-grouse forbs in both the fire and mechanical treatments compared to the control was consistent across all sites.

Nonnative Forbs and Annual Grasses. Cover of nonnative herbs did not increase in the fire treatment or mechanical treatments in year 1 compared to year 0 or the control treatment (Table 2), on 90% of the sites. However, nonnative herb cover was significantly higher in the fire treatment in year 2 and year 3 and in the mechanical treatment during year 3 than in year 0. Cover of nonnatives was also greater in the mechanical treatment in year 2 and year 3 compared to the control. However, nonnative herb cover in the fire treatment in the last 2 yr was greater than in either the mechanical or control treatments. The greater abundance of nonnative herb cover in the fire treatment compared to mechanical and control treatments occurred on 66% and 100%, respectively of the sites.

Table 2. Mean percent pre- and posttreatment cover for groups in each treatment followed by the standard error in parentheses. Odds ratios (year *x*/year 0), based on a logit scale of the relative change in cover of each plant group between pretreatment (year 0) and posttreatment (year *x*) within treatments; and the comparison of magnitude of relative change between two treatments (year 0 trt1/year *x* trt1)/(year 0 trt2/year *x* trt2) within years. *P* values follow each value in parentheses. Values followed by an asterisk denote a significant difference of varying strength of evidence (*some evidence, $0.01 <= P < 0.05$; **strong evidence, $0.001 <= P < 0.01$; ***very strong evidence, $P < 0.001$) across years within a single treatment and treatment comparisons (e.g., FI/ME) within years. CO indicates control; FI, fire; and ME, mechanical.

	% cover				Odds ratios (*P* value)		
Treatment	Year 0	Year 1	Year 2	Year 3	Year 1	Year 2	Year 3
Bare ground							
CO	27 (3.68)	28.5 (3.79)	31.6 (4.07)	30.1 (4.04)	1.092 (0.5611)	1.267 (0.0122)	1.181 (0.1903)
FI	28 (3.76)	43.9 (4.48)	33.4 (4.18)	29.9 (4.03)	2.024 (< 0.0001)***	1.330 (0.0024)**	1.093 (0.4728)
ME	28.3 (3.78)	25.3 (3.54)	24.7 (3.57)	25.3 (3.68)	0.876 (0.3913)	0.853 (0.0972)	0.897 (0.3967)
FI/ME	—	—	—	—	2.310 (< 0.0001)***	1.560 (0.0015)**	1.219 (0.2131)
ME/CO	—	—	—	—	0.803 (0.1799)	0.673 (0.0046)**	0.760 (0.0930)
FI/CO	—	—	—	—	1.854 (0.0004)***	1.049 (0.6999)	0.926 (0.6098)
Shrubs							
CO	9.4 (1.79)	9.3 (1.79)	8.8 (1.71)	9.9 (1.90)	0.985 (0.9096)	0.901 (0.4504)	1.021(0.8022)
FI	11.6 (2.11)	1.3 (0.42)	1.8 (0.53)	3.0 (0.76)	0.101 (< 0.0001)***	0.130 (< 0.0001)***	0.251 (< 0.0001)***
ME	10.1 (1.90)	9.5 (1.81)	10.4 (1.96)	12.3 (2.25)	0.923 (0.5274)	1.000 (0.9986)	1.199 (0.0277)*
FI/ME	—	—	—	—	0.109 (< 0.0001)***	0.130 (< 0.0001)***	0.209 (< 0.0001)***
ME/CO	—	—	—	—	0.937 (0.6887)	1.110 (0.5661)	1.174 (0.1097)
FI/CO	—	—	—	—	0.102 (< 0.0001)***	0.145 (< 0.0001)***	0.246 (< 0.0001)***
Sagebrush							
CO	6.1 (1.21)	5.9 (1.19)	5.6 (1.14)	5.7 (1.19)	0.979 (0.8099)	0.925 (0.4149)	1.033 (0.7203)
FI	7.6 (1.47)	0.5 (0.20)	0.7 (0.25)	1.0 (0.33)	0.066 (< 0.0001)***	0.087 (< 0.0001)***	0.128 (< 0.0001)***
ME	6.6 (1.31)	5.9 (1.19)	6.6 (1.31)	8.3 (1.61)	0.891 (0.1717)	0.967 (0.7012)	1.135 (0.1419)
FI/ME	—	—	—	—	0.074 (< 0.0001)***	0.090 (< 0.0001)***	0.113 (< 0.0001)***
ME/CO	—	—	—	—	0.910 (0.3842)	1.045 (0.7220)	1.099 (0.3987)
FI/CO	—	—	—	—	0.067 (< 0.0001)***	0.094 (< 0.0001)***	0.124 (< 0.0001)***
Tall perennial grass							
CO	11.3 (2.51)	11.0 (2.45)	10.1 (2.32)	10.0 (2.32)	0.967 (0.7934)	0.873 (0.4613)	0.852 (0.3325)
FI	10.2 (2.30)	6.5 (1.57)	10.4 (2.36)	13.1 (2.89)	0.608 (0.0015)**	0.967 (0.8564)	1.245 (0.1918)
ME	11.5 (2.55)	11.6 (2.57)	17.3 (3.57)	17.2 (3.59)	1.002 (0.9904)	1.563 (0.0241)*	1.499 (0.0265)*
FI/ME	—	—	—	—	0.606 (0.0004)***	0.619 (0.0013)**	0.831 (0.0684)
ME/CO	—	—	—	—	1.036 (0.7496)	1.789 (0.0002)***	1.760 (< 0.0001)***
FI/CO	—	—	—	—	0.628 (0.0008)***	1.107 (0.4553)	1.462 (0.0014)**
Short perennial grass							
CO	4.3 (1.37)	4.2 (1.35)	4.5 (1.45)	4.7 (1.56)	0.976 (0.7546)	1.046 (0.8981)	1.093 (0.7436)
FI	4.1 (1.33)	3.4 (1.12)	3.5 (1.17)	3.9 (1.34)	0.829 (0.5548)	0.853 (0.0679)	0.951 (0.8142)
ME	4.0 (1.29)	3.5 (1.15)	4.9 (1.54)	4.7 (1.56)	0.875 (0.208)	1.225 (0.0455)	1.175 (0.1423)
FI/ME					0.947 (0.8874)	0.623 (0.0274)*	0.809 (0.2047)
ME/CO					0.897 (0.7455)	1.171 (0.8212)	1.075 (0.8891)
FI/CO					0.849 (0.6152)	0.815 (0.4865)	0.870 (0.7441)
Perennial forb							
CO	3.1 (0.64)	3.5 (0.70)	2.8 (0.61)	3.8 (0.79)	1.088 (0.5576)	1.263 (0.4570)	1.110 (0.5475)
FI	3.2 (0.66)	3.6 (0.71)	4.5 (0.90)	6.5 (1.25)	1.063 (0.6747)	1.898 (0.0912)	1.908 (0.0013)**
ME	2.6 (0.55)	3.3 (0.66)	3.9 (0.79)	4.5 (0.92)	1.230 (0.1835)	2.381 (0.0099)*	1.898 (0.0017)*
FI/ME	—	—	—	—	0.864 (0.2953)	0.797 (0.2000)	1.006 (0.9710)
ME/CO	—	—	—	—	1.130 (0.3975)	1.886 (0.0013)**	1.710 (0.0052)**
FI/CO	—	—	—	—	0.977 (0.8456)	1.503 (0.0135)*	1.719 (0.0027)**
Sage-grouse forbs							
CO	1.5 (0.41)	1.7 (0.45)	1.6 (0.43)	2.0 (0.55)	1.130 (0.5301)	1.011 (0.9630)	1.284 (0.3546)
FI	2.0 (0.52)	3.0 (0.74)	6.0 (1.38)	6.3 (1.48)	1.558 (0.0264)*	3.165 (< 0.0001)***	3.300 (0.0002)***
ME	1.5 (0.42)	2.0 (0.53)	2.8 (0.71)	3.4 (0.87)	1.325 (0.1552)	1.949 (0.0076)**	2.299 (0.0034)**
FI/ME	—	—	—	—	1.176 (0.2837)	1.623 (0.0166)*	1.436 (0.1000)
ME/CO	—	—	—	—	1.172 (0.3334)	1.928 (0.0069)**	1.791 (0.0392)*
FI/CO	—	—	—	—	1.379 (0.03421)*	3.130 (< 0.0001)***	2.571 (0.0008)***

Table 2. Continued.

Treatment	% cover				Odds ratios (*P* value)		
	Year 0	Year 1	Year 2	Year 3	Year 1	Year 2	Year 3
Non-native forbs and annual grasses							
CO	2.4 (1.46)	1.5 (0.97)	1.2 (0.74)	1.7 (1.09)	0.648 (0.2608)	0.511 (0.0073)**	0.766 (0.4169)
FI	3.4 (2.04)	1.9 (1.18)	5.5 (3.26)	12.6 (6.88)	0.541 (0.1215)	1.730 (0.0188)*	4.630 (0.0009)***
ME	2.1 (1.28)	1.6 (1.02)	2.7 (1.65)	5.2 (3.10)	0.754 (0.4560)	1.333 (0.1801)	2.762 (0.0092)**
FI/ME	—	—	—	—	0.7174 (0.0252)*	1.2987 (0.0391)*	1.6756 (0.0030)**
ME/CO	—	—	—	—	1.1639 (0.2936)	2.6076 (< 0.0001)***	3.6101 (< 0.0001)***
FI/CO	—	—	—	—	0.5058 (0.1997)	3.3875 (< 0.0001)***	6.0496 (< 0.0001)***
Litter							
CO	52.6 (3.60)	54.3 (3.58)	55.4 (3.68)	54.4 (3.80)	1.068 (0.6005)	1.109 (0.4345)	1.074 (0.6343)
FI	50.7 (3.61)	37.4 (3.40)	43.3 (3.66)	47.6 (3.82)	0.695 (0.0107)*	0.727 (0.0241)*	0.865 (0.7158)
ME	50.0 (3.61)	58.1 (3.52)	61.6 (3.54)	62.3 (3.62)	1.385 (0.0194)*	1.548 (0.0038)**	1.610 (0.0063)**
FI/ME	—	—	—	—	0.502 (< 0.0001)***	0.469 (< 0.0001)***	0.537 (0.0006)***
ME/CO	—	—	—	—	1.296 (0.0166)*	1.396 (0.0084)**	1.499 (0.0055)**
FI/CO	—	—	—	—	0.651 (0.0004)***	0.656 (0.0014)**	0.805 (0.3279)
Biological soil crust							
CO	5.6 (2.96)	4.2 (2.30)	2.9 (1.60)	3.0 (1.69)	0.748 (0.0778)	0.487 (0.0429)*	0.547 (0.0351)*
FI	4.8 (2.59)	1.5 (0.87)	0.8 (0.44)	0.8 (0.49)	0.305 (< 0.0001)***	0.146 (< 0.0001)***	0.162 (< 0.0001)***
ME	4.4 (2.39)	2.8 (1.56)	2.1 (1.2)	2.0 (1.14)	0.629 (0.0098)**	0.451 (0.0287)*	0.452 (0.0128)*
FI/ME	—	—	—	—	0.485 (< 0.0001)***	0.325 (< 0.0001)***	0.359 (< 0.0001)***
ME/CO	—	—	—	—	0.841 (0.0887)	0.926 (0.6332)	0.827 (0.4050)
FI/CO	—	—	—	—	0.408 (< 0.0001)***	0.301 (< 0.0001)***	0.297 (< 0.0001)***

Litter and Biological Crusts. In the fire treatment litter cover was 26% lower in year 1 than year 0 and remained significantly lower in year 2 than year 0 and the control plots (Table 2). However, in year 3 litter cover in the fire treatment did not differ from the control or year 0. In year 3 litter had recovered to prefire treatment levels in 45% of the sites. Still, litter cover in the fire-treatment plots remained lower than the mechanical treatment throughout the entire measurement period.

There was a gradual temporal decline in biological soil crust in the control treatment between year 0 and year 3. However, cover was consistently lower in the fire treatment compared to mechanical and control treatments in all three posttreatment years (Table 2). There was no difference in biological soil crust between control and mechanical treatments in all 3 yr.

Density

Shrubs. Prior to treatment application, density of shrub seedlings (< 50 mm) and established plants (> 50 mm) averaged 0.12 m^{-2} and 0.58 m^{-2}, respectively, and did not differ among pretreated plots (year 0) (Table 4). Nearly 50% of established shrub density was sagebrush. Bitterbrush (*Purshia tridentata* [Pursh] DC.) and rabbitbrush (*Chrysothamnus* sp. Nutt.) accounted for a large portion of the remaining shrubs. Density of total established shrubs was reduced in the fire treatment by nearly 75%. Densities were similar between control and mechanical treatments in all posttreatment years and were not different than year 0. However, density of total established shrubs in the fire treatment was significantly lower than in the control or mechanical plots for all three posttreatment years ($P < 0.0001$). There was no difference among total shrub seedlings between treatments or years. For sagebrush seedlings there was a significant increase in densities in the mechanical treatment in year 2 compared to year 0 and the control treatment. By year 3 densities in both the mechanical and fire treatment were higher than year 0 and the control.

Perennial Tall Grasses and Forbs. There was no evidence of a treatment effect on tall perennial grass densities nor was there a significant change over time (Table 4). For perennial forbs there was no clear evidence of a treatment effect on density. There appeared to be an increase in the fire treatment in year 1 and year 2 compared to the control and mechanical treatment. However, the large degree of variation may have masked this treatment effect.

DISCUSSION

Our sites ranged from cool–mesic Wyoming big sagebrush to cool–frigid Mountain big sagebrush communities occupying sandy loam to loam soils. Precipitation zones primarily ranged from 250–300 to 300–350 mm, which fall into the wetter end of aridic and drier end of xeric moisture regimes. All sites had greater than 5% cover of perennial grasses, with the exception of Seven Mile in Nevada. Nonnative herbs ranged from < 1% cover to codominant with native perennial herbs. Managers typically deal with this range of variation both within and across treatment areas. Although the 11 study sites included a wide range of variation, there was a fairly consistent pattern in response for some cover groups across all sites following fire and mechanical treatments. In the first year, neither fire nor

Table 3. Percent cover of the different cover groups within treatment plots for each site (replicate-block) in the years before (year 0) and third year (year 3) after treatment. CO indicates control; FI, fire; and ME, mechanical. Because of the wildfire at the Stansbury site in the fall of year 2, cover data reported is for year 2, and only year 1 was collected for the South Ruby site. Tree canopy cover for the individual sites is reported by Roundy et al. (2014a, 2014b).

		Bare ground		Shrubs		Tall perennial grass		Short perennial grass		Perennial forb		Sage-grouse forbs		Nonnative forbs and annual grasses		Litter		Biological soil crust	
Site	Treatment	Year 0	Year 3	Year 0	Year 3	Year 0	Year 3	Year 0	Year 3	Year 0	Year 3	Year 0	Year 3	Year 0	Year 3	Year 0	Year 3	Year 0	Year 3
Western juniper																			
Blue Mountain	FI	16.4	22.1	16.9	7.1	19.6	17.8	7.3	7.3	4.2	15.1	3.2	6.9	3.6	8.8	64.3	52.7	3.1	0.1
	CO	13.8	23.5	14.7	16.7	35.0	21.0	3.7	5.5	2.4	4.4	1.5	2.6	1.3	0.9	66.4	60.1	5.1	2.0
	ME	12.7	15.4	21.0	26.9	28.7	26.9	5.6	6.8	4.0	8.0	1.4	4.0	1.7	4.0	67.7	65.6	9.0	0.5
Bridge Creek	FI	25.5	21.5	3.6	0.9	22.0	17.5	3.5	8.8	1.5	2.5	1.0	1.7	2.3	27.4	49.9	49.9	26.0	15.8
	CO	16.5	16.7	7.3	6.0	29.2	23.3	6.3	8.6	2.1	1.6	1.5	0.8	4.8	8.8	53.3	61.7	28.7	34.5
	ME	20.5	14.1	6.2	7.6	26.9	32.2	5.4	8.3	2.0	3.0	1.1	2.2	2.6	13.8	52.3	64.1	24.1	33.0
Devine Ridge	FI	23.1	28.7	12.7	3.0	7.6	8.7	14.6	4.8	2.3	1.3	4.1	3.5	7.6	14.5	54.5	35.0	5.7	0.6
	CO	21.5	28.6	10.5	11.7	6.7	5.6	13.6	7.6	3.5	3.8	4.8	2.5	2.8	0.6	57.2	47.8	13.2	10.4
	ME	20.0	18.7	11.6	13.6	10.0	11.8	10.4	7.8	3.0	3.2	3.1	2.6	1.6	2.1	61.6	60.7	9.9	3.6
Walker Butte	FI	35.4	38.6	1.7	0.7	31.4	27.6	0.2	0.1	2.0	2.5	2.0	10.6	0.0	3.6	43.2	29.5	4.3	0.2
	CO	37.5	43.5	3.0	1.7	24.9	18.6	0.6	0.1	2.9	2.6	1.9	2.2	0.0	0.1	43.6	43.2	6.8	6.9
	ME	32.7	31.7	10.0	7.7	36.9	31.0	0.3	0.0	2.6	2.0	3.9	4.9	0.6	1.7	32.6	47.3	4.8	4.0
Piñon–Utah juniper																			
Marker Corral	FI	47.1	39.8	14.1	4.4	8.7	17.6	3.1	1.8	5.2	7.3	3.1	16.6	0.0	1.9	43.4	64.4	0.4	0.0
	CO	40.1	33.7	8.1	10.1	8.3	8.3	1.2	1.2	3.6	2.6	1.6	1.3	0.0	0.0	59.3	61.6	1.7	0.4
	ME	43.9	35.6	13.1	19.3	9.3	19.6	4.1	1.7	2.1	4.1	1.1	3.4	0.2	2.2	50.0	70.5	0.8	0.4
Seven Mile	FI	48.9	61.5	13.1	3.4	0.8	0.8	0.7	0.5	4.4	6.9	2.4	7.5	0.0	0.0	39.9	56.8	1.1	0.0
	CO	44.0	30.6	10.0	12.5	3.1	2.6	1.0	3.6	4.6	5.8	1.6	5.5	0.0	0.0	43.1	59.7	0.4	0.4
	ME	42.9	48.6	11.4	14.4	1.9	2.2	3.4	7.1	3.5	6.6	2.2	7.4	0.0	0.0	45.4	65.6	0.2	0.0
South Ruby (Year 1)	FI	30.6	21.9	7.6	3.8	6.3	13.8	5.8	13.9	5.9	15.7	2.3	6.6	2.2	1.2	55.4	66.2	0.0	0.0
	CO	34.1	35.6	7.3	10.8	10.8	16.7	3.7	11.1	5.5	9.4	1.2	4.1	0.7	0.8	55.2	64.1	0.1	0.0
	ME	38.2	38.3	7.5	10.0	7.8	13.5	3.0	6.2	8.4	12.1	3.6	7.0	0.8	0.1	49.2	73.1	0.0	0.0
Utah juniper																			
Greenville Bench	FI	42.0	32.1	15.0	5.6	5.3	11.2	4.2	6.5	4.4	5.0	3.9	11.9	1.3	9.5	40.4	41.3	5.2	0.8
	CO	43.7	41.6	9.8	8.3	3.0	5.1	2.2	4.2	2.4	2.4	2.0	1.9	4.3	2.5	43.4	43.8	2.7	1.6
	ME	45.0	34.4	11.0	12.5	4.3	12.2	1.6	1.3	1.4	3.9	1.0	3.2	1.8	4.4	43.0	47.2	4.5	1.2
(Year 2)	FI	9.6	8.3	37.3	0.5	11.7	17.2	11.0	9.7	11.0	9.7	0.6	4.5	43.4	68.8	71.9	56.9	6.1	0.0
	CO	14.0	16.4	22.7	21.0	14.0	12.2	17.7	21.3	17.7	21.3	1.8	4.0	29.3	25.5	66.0	72.2	12.2	0.9
	ME	13.1	4.9	11.9	10.7	12.7	33.2	14.3	17.7	14.3	17.7	0.9	4.2	30.0	46.3	65.6	81.3	13.9	0.9
Utah juniper–Colorado piñon																			
Scipio	FI	13.1	15.2	13.5	1.4	6.8	13.6	4.9	2.1	5.3	16.9	0.2	2.2	44.8	48.0	59.1	50.9	33.9	3.5
	CO	16.2	23.5	15.3	14.6	6.6	7.9	5.8	7.3	7.7	10.3	0.3	1.1	41.4	10.4	57.1	58.2	27.9	12.7
	ME	23.5	13.8	12.8	12.4	3.1	12.6	2.4	4.0	2.1	7.6	0.1	2.3	37.6	38.4	51.6	69.3	16.1	8.2
Onaqui	FI	34.5	24.7	5.8	0.8	15.0	13.2	5.6	5.7	0.9	2.9	0.5	0.9	10.1	45.1	36.0	32.5	22.6	5.6
	CO	35.0	42.4	6.1	5.7	13.3	9.8	7.2	6.3	1.2	1.8	0.8	1.1	8.2	10.6	32.4	35.5	24.1	12.4
	ME	39.7	28.7	6.1	7.4	11.5	18.1	8.4	7.6	1.0	4.2	0.4	1.7	7.3	16.9	31.2	48.8	17.7	7.1

Table 4. Mean density across sites for each treatment over time followed by the standard error in parentheses. Shrubs > 50 mm were considered established, and those < 50 mm seedlings. CO indicates control; FI, fire; and ME, mechanical. Upper-case letters indicate a significant difference between treatments within years and lower-case letters significant differences within treatments across years at $P < 0.05$.

	Density (no. m^{-2})			
Treatment	Year 0	Year 1	Year 2	Year 3
Established shrubs				
CO	0.58 (0.37)^{A}	0.49 (0.21)^{A}	0.48 (0.20)^{A}	0.56 (0.27)^{A}
FI	0.61 (0.39)^{Ba}	0.16 (0.16)^{Bb}	0.17 (0.17)^{Bb}	0.23 (0.19)^{Bb}
ME	0.56 (0.29)^{A}	0.51 (0.24)^{A}	0.56 (0.24)^{A}	0.67 (0.25)^{A}
Shrub seedlings				
CO	0.13 (0.16)	0.06 (0.09)	0.07 (0.08)	0.34 (0.61)
FI	0.10 (0.11)	0.04 (0.04)	0.11 (0.17)	0.51 (0.72)
ME	0.14 (0.29)	0.02 (0.03)	0.66 (1.82)	0.46 (0.64)
Sagebrush seedlings				
CO	0.07 (0.13)	0.02 (0.04)	0.02 (0.02)^{A}	0.09 (0.11)^{A}
FI	0.06 (0.10)^{a}	0.01 (0.02)^{a}	0.08 (0.16)^{Aa}	0.44 (0.63)^{Bb}
ME	0.07 (0.15)^{a}	0.01 (0.02)^{a}	0.60 (1.70)^{Bb}	0.41 (0.64)^{Bb}
Tall perennial grasses				
CO	6.4 (3.62)	7.4 (4.15)	6.3 (2.87)	6.2 (4.44)
FI	5.9 (2.48)	5.9 (2.97)	5.2 (2.82)	5.4 (2.13)
ME	7.8 (5.37)	7.9 (3.24)	8.1 (3.47)	7.6 (2.85)
Perennial forbs				
CO	4.2 (3.48)	7.6 (6.38)	4.8 (2.89)	8.6 (6.60)
FI	5.9 (4.44)	11.0 (8.72)	8.6 (9.02)	9.7 (7.90)
ME	6.2 (7.09)	7.1 (7.08)	5.9 (6.07)	8.5 (7.08)

mechanical treatments resulted in an immediate increase in native or nonnative herbaceous cover. In fact, in year 1 for the fire treatment there was a consistent and significant decline in tall perennial grasses, biological soil crusts, and litter in addition to woody vegetation, resulting in a significant increase in bare ground. The decline in tall perennial grasses in the first year is consistent with other studies conducted in the region (Blaisdell 1953; Countryman and Cornelius 1957; Harniss and Murray 1973; Young and Evans 1978; Everett and Ward 1984; West and Hassan 1985; Cook et al. 1994; Hosten and West 1994; Pyle and Crawford 1996; Bates et al. 2000; West and Yorks 2002; Seefeldt et al. 2007; Davies and Bates 2008; Ellsworth and Kauffman 2010; Rhodes et al. 2010). In a comprehensive literature review of fire effects on plants and soils in the Great Basin region the decline in tall perennial grass cover in the first year following fire was observed in 86% of the studies (Miller et al. 2013). In our study, tall perennial grass cover quickly recovered in the fire treatment in year 2 compared to year 0 and in year 3 was greater than the control. The majority of studies (90%) conducted in the region reported that recovery of tall perennial grasses occurred within the second or third year following fire (Miller et al. 2013). In the mechanical treatment tall perennial grass cover did not differ in year 1 but increased to percentages higher than year 0 and the control in year 2 and year 3. Even where tree cover was in the middle or upper phases of infilling, perennial grasses increased following cutting on these sites (Roundy et al. 2014a). Increases in perennial grasses following cutting of Utah juniper and singleleaf piñon have been reported in Nevada, where levels of this cover group were not depleted prior to treatment (Everett and Sharrow 1985a, 1985b). On the other hand, increases were not reported on treated sites where cover of this group was low (< 1%) (Bristow 2010; Koniak 1985). Sandberg bluegrass (short perennial grass) did not decline following fire treatment in year 1, nor did it increase in cover in year 1 to year 3 across either treatment. Others have reported little to no change in this species following treatment (Blaisdell 1953; Wright and Klemmedson 1965; Young and Evans 1978; Akinsoji 1988). However, an increase in Sandberg bluegrass cover was reported in the third postfire year in eastern Oregon (Davies et al. 2012).

Although we reported a significant increase in perennial forb cover following both treatments, the response reported in other studies in the region have been markedly inconsistent. A number of studies reported no change in perennial forb biomass during the first 1–5 yr following fire compared to levels in prefire treatment or nearby untreated plots (Cook et al. 1994; Hosten and West 1994; Fischer et al. 1996; West and Yorks 2002; Beck et al. 2008; Rhodes et al. 2010; Bates et al. 2011), whereas other studies have reported increases (Stager and Klebenow 1987; Martin 1990; Pyle and Crawford 1996; Wrobleski and Kauffman 2003; Bates et al. 2011). Site characteristics are likely a key attribute that influence the postfire response of this cover group. Perennial forbs did not increase on soils with mesic temperature and aridic (< 300-mm precipitation) moisture regimes on previous studies (Miller et al. 2013). However, perennial forbs did increase following treatment on frigid/xeric soils in 70% of previous studies. Our sites were on the cool end of mesic and moist end of aridic (250–300 mm), which may explain the positive response in perennial forbs. Mixed results in perennial forb response may also be attributed to variation in predisturbance composition and annual precipitation in the years following fire.

In our study, forbs used by sage-grouse increased with both prescribed fire (years 1–3) and mechanical treatment (years 2–3). However, this increase was largely a result of an increase in annual forbs known to be eaten by sage-grouse. This may be a short-term response, because these forbs are typically early successional. Rhodes et al. (2010) reported increases in sage-grouse annual food forbs the first year following fire, but levels declined to unburned levels in years 2–5.

We attributed the increase in tall perennial grass and perennial forb cover in year 2 and year 3 in both treatments to increased growth of residual plants that were present on the site prior to treatment, because increased cover was not accompanied by an increase in plant density. Others have reported that the initial response of cover and biomass for perennial grasses and forbs in the first and often second year following fire is primarily the result of growth from plants that survived the disturbance (Everett and Sharrow 1985b; West and Hassan 1985; Hassan and West 1986; Bates et al. 2000; Wehking 2002; Bates et al. 2009; Davies et al. 2009). The occurrence and relative abundance of resident perennial grasses and forbs on a site prior to treatment is therefore a more important driver of early and midsuccessional trajectories following disturbance than the seed bank (Koniak and Everett 1982; Everett and Sharrow 1985a; Allen et al. 2008; Pekas 2010).

Bare ground nearly doubled in the fire treatment in the first postfire year. However, it declined in year 2 and had nearly returned to prefire treatment levels by year 3. This was likely a result of increased cover in tall perennial grasses, forbs, nonnatives, and litter in the fire treatment in year 2 and year 3, compared to year 1. Litter cover in the fire-treatment plots also recovered to near prefire treatment levels by year 3 following treatment. The increase in litter can be important in the recovery of a site, by reducing negative soil water potentials and soil surface temperatures, which in turn can enhance seedling emergence (Evans and Young 1970; Chambers 2000). Biological soil crusts, however, did not show signs of recovery by year 3 in the fire-treatment plots. West and Hassan (1985) reported an 80% reduction in biological soil crusts following fire. Recovery of biological soil crusts can occur within a decade or can take centuries (Callison et al. 1985; Johansen 2001). Following a fire in Wyoming big sagebrush, Hilty et al. (2004) reported an increase in short mosses but a decline in tall mosses and lichens. The increase in bare ground and decline in biological soil crusts did not occur in the mechanical treatment, which did not differ from the control treatment. However, litter significantly increased in the mechanical treatment, compared to year 0 and the controls in all three posttreatment years. Although not measured, litter depth increased beneath the fallen tree canopies.

One of the primary concerns in treating conifer-encroached shrublands is the potential increase of nonnative annual grasses and forbs. In this study, nonnative plant cover did not increase in year 1 in either treatment but increased significantly in year 2 and year 3. Several authors have reported increases in nonnative species during the early years following treatment of conifer encroached shrublands (Barney and Frischknecht 1974; Tausch and Tueller 1977; Quinsey 1984; Koniak 1985). However, the delayed response in year 1 has been reported by the majority of studies conducted in the region (Young and Evans 1978; West and Hassan 1985; Young and Miller 1985; Hassan and West 1986; Akinsoji 1988; Cook et al. 1994; Pyle and Crawford 1996; Bates et al. 2000; Davies et al. 2007; Mata-Gonzalez et al. 2007; Rowe and Leger 2011). The limited response of nonnative forbs and grasses in the first year after fire may be a result of reduced microsites available for germination and establishment (Hilty et al. 2003; Davies et al. 2009), reduced litter cover, and/or consumption of seed by fire (Young and Evans 1975; Hassan and West 1986; Humphrey and Schupp 2001; Allen et al. 2008). The magnitude of increase in nonnative cover in year 2 and year 3 was greatest in the fire-treatment plots, but was variable across our study sites and treatments. The three warmest sites in our study area had the highest cover of nonnative species. Others have reported greater increases in nonnative species on warmer compared to cooler sites (Chambers et al. 2014; Roundy et al. 2014a, 2014b). In the fire-treatment plots, cover of nonnatives was generally less (< 10%) on sites occupied by mountain big sagebrush (*Artemisia tridentata* Nutt. subsp. *vaseyana* [Rybd.] Beetle) on frigid soils compared to Wyoming big (*Artemisia t.* subsp. *wyomingensis* Beetle & Young) or basin big sagebrush (*Artemisia t.* subsp. *tridentata*) sites, which occupied mesic soils (> 30%). Prior studies indicate that growth and reproduction of the nonnative annual grasses that dominated in this study, especially cheatgrass (*Bromus tectorum* L.), are limited by cold temperatures in sagebrush ecosystems (Chambers et al. 2007). Although nonnative plant cover increased following both the fire and mechanical treatments, native plant cover in year 3 was 2.5–3.5 times greater in the fire treatment and mechanical treatments, respectively, compared to nonnative plant cover. Longer-term studies indicate that although nonnative annual grasses are likely to persist on warmer sites, their relative abundance over time depends on the cover and density of native perennial species, especially perennial herbs, and repeated occurrence of fire (Koniak and Everett 1982; Rew and Johnson 2010).

Recovery of sagebrush cover is a primary concern, as it provides valuable habitat for many wildlife species, particularly for sagebrush obligates (Connelly et al. 2004). As expected, fire significantly reduced cover and densities of sagebrush and other established shrubs, whereas the mechanical treatment had no initial effect on shrub cover. The number of years required for sagebrush canopies to recover to predisturbance levels can range from 15 to > 50 yr (Ziegenhagen 2004) and has been reported to average 30–35 yr in cooler, moister sagebrush cover types (Harniss and Murray 1973; Barney and Frischknecht 1974; Watts and Wamboldt 1996; Nelle et al. 2000; Ziegenhagen 2004; Nelson et al. 2014). However, compared to mountain big sagebrush, postfire recolonization of Wyoming big sagebrush has been reported to be very slow to nearly nonexistent (Ralphs and Busby 1979; Hosten and West 1994; West and Yorks 2002; Beck et al. 2008; Rhodes et al. 2010). Consistent with this finding, on our study higher sagebrush seedling densities occurred on the mountain big sagebrush sites on frigid soils compared to Wyoming and basin big sagebrush on mesic soils (Miller et al. 2013). In the mechanical treatment sagebrush seedling density was greater by year 2 than either the control or fire treatments, and in year 3 was higher in both the mechanical and fire treatments compared to densities in the control or year 0. However, this trend was not consistent across sites, ranging from 0 to 2 plants $\cdot$ m^{-2}. Establishment of new sagebrush plants during the first few years following disturbance is a primary factor in determining the rate of recolonization. This is particularly true in fire-treated areas, where initial density and cover of established shrubs are severely reduced (Ziegenhagen and Miller 2009). Sagebrush seedling establishment has been reported to be strongly related to soil moisture availability (Boltz 1994) and seed source (Ziegenhagen and Miller 2009). Ziegenhagen (2004) estimated that densities of mature mountain big sagebrush and bitterbrush, when growing in combination, typically ranged between 0.87 and 1.3 plants $\cdot$ m^{-2} on sites considered to be fully occupied. Davies and Bates (2010) reported densities of 1.1 plants $\cdot$ m^{-2} for mature mountain big sagebrush and 0.5 plants $\cdot$ m^{-2} for Wyoming big sagebrush across over 100 sites in southeast Oregon and northwest Nevada. Based on a sagebrush recovery model, densities of 0.26 sagebrush $\cdot$ m^{-2} in the third postfire year in northwestern Nevada were considered adequate for the stand to recolonize within 30 yr (Ziegenhagen and Miller 2009). In our study, sagebrush seedling densities in the third year following both the mechanical and fire treatments were over 0.40 plants $\cdot$ m^{-2}. However, density ranged from 0 to 2 plants $\cdot$ m^{-2} across our sites, indicating a wide range in the rate of recolonization.

Unlike herbaceous vegetation, sagebrush is solely dependent on seed banks or seed rain for recolonization.

MANAGEMENT IMPLICATIONS

This study indicates a relatively consistent pattern of vegetation response can be expected to occur the majority of the time within the range of study site conditions evaluated. These patterns and the differences between the burn and mechanical treatments have important implications for predicting management outcomes and selecting appropriate sites and treatments. When burning or mechanically removing piñon and or juniper an increase in both tall perennial grasses and perennial forbs can be expected across the majority of sites within the range of conditions evaluated in this study. However, an increase in nonnatives can also be expected, but will vary with sited conditions and type of treatment. The increase in nonnatives was greatest in burn treatment and on the warmer, drier sites (Chambers et al. 2014). Both treatments resulted in an increase in available soil water, emphasizing the importance of desirable residual vegetation that will reduce resource availability for invasive species (Roundy et al. 2014b). Elevated levels of nitrogen and the reduction of shrubs, tall perennial grasses, and biological soil crusts provides a greater opportunity for nonnatives to increase on a site following fire (Blank et al. 2007; Chambers et al. 2007) than after a mechanical treatment. Our results suggest that fire treatment is more likely to lower a treatment areas' resistance to invasives temporally, more than the mechanical removal of trees by cutting (Chambers et al. 2014). Thus, the overall composition and structure of the vegetation prior to treatment and soil moisture/temperature regimes are key site attributes that will influence recovery. This study suggests mechanical treatment will usually be a better option on warm dry sites, typically occupied by Wyoming big sagebrush. Advantages are less impact on the abundance of sagebrush and biological soil crusts, which recover very slowly on these sites. And the mechanical treatment usually resulted in a smaller increase in nonnative invasives. The greater abundance of biological soil crusts on warm dry compared to cool moist sites may result in their greater importance in the resilience and resistance to invasives on these more arid sites.

The primary limitation of this study is the limited number of years in which succession was evaluated following treatment. Three years was inadequate to evaluate the recolonization of sagebrush or determine long-term trends in perennial herbaceous species, the persistence of nonnatives, and the encroachment of conifers in both the fire and mechanical treatments. Indeed, long-term monitoring studies of succession following wild or prescribed fire and mechanical treatments of conifer-encroached shrublands, where time has not been substituted for space, are very limited in the Intermountain West (Miller et al. 2013). Only continued measurements of these sites will allow confident determination of potential changes in the abundance of introduced and native species in the future. In addition to the lack of long-term studies, few have reported the effects of repeated fires on plant succession.

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