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Original Research

Vegetation Response to Juniper Reduction and Grazing Exclusion in Sagebrush-Steppe Habitat in Eastern Oregon☆

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ABSTRACT

Western juniper expansion is one of the largest threats to conserving sagebrush steppe ecosystems in the northwestern United States. Juniper expansion has degraded the sagebrush steppe by altering fire regimes and outcompeting shrubs and herbaceous vegetation for limited resources. We characterized the effect of juniper removal in a severely degraded sagebrush steppe habitat for 3 yr following juniper cutting. In addition, we measured the effect of low-intensity seasonal grazing on plant community recovery through cattle exclusion treatments. We monitored plant community composition (exotic annual grasses, preferred grasses, preferred forbs, and shrubs); fuel loads; and juniper recruitment in a factorial design of juniper removal and grazing exclusion. We found that although there were significant differences between cut and uncut juniper treatments, there were no consistent trends across all 3 yr. Our results suggest that other factors, such as timing of precipitation, may also have strong short-term effects on plant community composition. We detected no significant grazing effects during the study period, suggesting the current grazing regime is appropriate for the area. The cutting of juniper increased total fuel loads and herbaceous fuel loads. Compared with open interspace, a twofold increase in juniper seedlings and saplings was detected beneath juniper piles, which will act as sources for future juniper encroachment.

Introduction

The sagebrush (*Artemisia* spp.) steppe is currently threatened due to altered fire regimes, a legacy of historical overgrazing, understory invasion of non-native annual grasses, and the expansion of juniper (Noss et al., 1995; Knick et al., 2003; Davies et al., 2011). In the northwest range of big sagebrush steppe, the largest potential threat, western juniper (*Juniperus occidentalis*), has vastly expanded its range at a rate that is historically unprecedented (Miller et al., 1994; Miller and Wigand, 1994). This expansion has contributed to the degradation of sagebrush steppe ecosystems at mid to high elevation across western North America, primarily by decreasing the frequency of fire (Miller and Rose, 1999) and outcompeting shrubs and other understory plants for limited resources (Miller et al., 2000; Roundy et al., 2014b). Altered fire regimes further contribute to ecosystem degradation through increased erosion of top soils via runoff (Pierson et al., 2007, 2010) and by creating conditions congruent for exotic plant invasions (Brooks et al., 2004).

In eastern Oregon, <3% of western juniper are presettlement trees (i.e., >120 years old) (USDA-BLM, 1990) and the area of juniper woodland has increased from approximately 600 000 ha in the 1930s to 2.6 million ha in the 1990s (Azuma et al., 2005). This increase in junipers along with its cascading effects has negatively impacted some wildlife populations (see Bombaci and Pejchar, 2016 for a review), leading many sagebrush-dependent plant and animal species to be labeled as species of conservation concern (Dobkin and Sauder, 2004; Wisdom et al., 2005; Baruch-Mordo et al., 2013). Additionally, improper grazing management and the invasion by non-native grasses such as cheatgrass (*Bromus tectorum* L.), medusahead (*Taeniatherum caput-medusae* [L.] Nevski), and ventenata (*Ventenata dubia* [Leers] Coss.) have further decreased wildlife habitat (Miller and Eddleman, 2001; Rottler et al., 2015). In response to the loss of sagebrush steppe ecosystems, land managers have deployed approaches to restore ecological communities affected by these altered disturbance factors and regimes. Reduction of juniper density, specifically killing or mechanically removing post-settlement juniper trees, has been one commonly used management action, and short-term recovery (<3 yr) in juniper reduction treatments has recently been well studied. Increased soil water and nitrogen

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availability has been linked to increased preferred vegetation (e.g., native perennial grasses and forbs) cover and biomass compared with that of exotic grasses (Bates et al., 2000, 2002; Eddleman, 2002). A recent study (Williams et al., 2017) from the SageSTEP project (McIver and Brunson, 2014) found that juniper removal increased tall grass cover without increasing cheatgrass cover but did not affect shrub cover 3 and 6 yr post treatment. However, a previous study by Miller et al. (2014) on the same cut plots did not find any difference in perennial grass and forb density in the first 3 yr after treatment, suggesting vegetation response is delayed.

Grazing management in the sagebrush steppe has varied considerably through time. Early postsettlement grazing by European settlers was characterized by inappropriately high stocking rates, resulting in degraded rangelands across much of the sagebrush steppe (Covington et al., 1994). In 1934, the Taylor Grazing Act initiated regulation of livestock on public lands to prevent further degradation. Beginning in the 1940s, rotational grazing, seasonal rest periods, and reduced stocking rates were implemented to improve range quality. Modern grazing practices continue to improve in order to balance the needs of wildlife, native vegetation communities, and the socioeconomics of livestock production, and thus management regimes range widely in type, intensity, seasonality, and therefore, impacts to vegetation communities and wildlife habitats (Strand et al., 2014). Because domestic livestock grazing continues across the sagebrush steppe and overlaps with areas of juniper expansion and subsequent treatment, data are critically needed to fully understand the potentially interactive effects of grazing and juniper reduction treatments on invasion/recovery dynamics between annual grasses and native plants. The current lack of long-term data and of data on potential interactions may be resulting in the inappropriate management of juniper (Belsky, 1996).

The purpose of this study was to determine the individual and interactive effects of juniper reduction and exclusion of cattle grazing over multiple years on a degraded sagebrush steppe community. To achieve this, we collected plant community data 1 yr before treatment and for 3 yr following juniper treatment. We hypothesized that the combined effect of juniper reduction and elimination of cattle grazing would result in the largest increases in native plants, while the controls (no juniper cutting and grazing allowed) would show no difference in or continued degradation of the plant community when compared with pretreatment data. Furthermore, we hypothesized that intermediate treatments (no juniper reduction but grazing eliminated or juniper reduction but grazing allowed) would show intermediate levels of native rehabilitation. As second-tier objectives of this study, we monitored the biomass (fuel load) of herbaceous material and juniper duff within our study plots each year, as well as juniper recruitment to quantify the effects of management treatments on fire potential and future juniper re-encroachment. We hypothesized that that biomass would be highest in ungrazed and cut plots due to the increase of plants and shrubs, and we expected ungrazed-uncut plots would have no change in biomass post treatment. Lastly, we hypothesized that downed juniper skeletons would protect juniper seedlings from the elements and herbivory, as well as provide microsites of higher soil moisture, thus increasing density of seedlings underneath them than in the open (between-tree) interspace.

Methods

Study Location

This study was conducted from 2012 to 2016 in the Phillip W. Schneider Wildlife Area (PWSWA) during the months of May and June. The PWSWA is 21 014 ha of primarily sagebrush-steppe habitat owned by the ODFW and located between the Ochoco and Malheur National Forests just south of Dayville, Oregon. Our study site was located in the Flat Creek area at the northeast portion of the PWSWA (~119.45 E, 42.42 N) in the foothills of the Aldrich Mountains at approximately 1 750 m. The study site has a semiarid climate with a 30-yr mean precipitation of 43.22 cm annually (PRISM Climate Group, 2004). During the study period, annual precipitation was 43.18 cm in 2012, 48.18 cm in 2014, 40.37 cm in 2015, and 37.97 cm in 2016 (PRISM Climate Group, 2004). Historically, this site would have been occupied by a sagebrush steppe plant community composed of big sagebrush (*Artemisia tridentata* Nutt.), low sagebrush (*Artemisia arbuscula* Nutt.) antelope bitterbrush (*Purshia tridentata* [Pursh] DC.), green rabbitbrush (*Chrysothamnus viscidiflorus* [Hook.] Nutt.), rubber rabbitbrush (*Ericameria nauseosa* [Pall. Ex Pursh] Nesom and Baird), lupine (*Lupinus* L. spp.), fescue (*Festuca* L. spp.), bottlebrush squirreltail (*Elymus elymoides* [Raf.] Swezey), and various wheatgrasses (*Elymus* L. spp. and *Pseudoroegneria* [Nevski] Á. Löve spp.). However, the PWSWA was historically overgrazed and most wildfires were suppressed (Oliver et al., 1994; Powell, 2008). This resulted in juniper expansion, spread of invasive grasses, and significant loss of topsoil (Oregon Department of Fish and Wildlife, 2006), leaving the site in phase II of juniper succession, in which shrubs and trees codominate the landscape, with a thinning shrub layer (Miller et al., 2005). Currently, ODFW is implementing a deferred rest-rotation grazing program within the two pastures of the Flat Creek area. Both pastures are about 800 ha and together allow for 13.3 ha per animal unit month (AUM). Flat Creek was last rested in 2013. Cattle have been allowed to graze during parts of May-June in both pastures every year since, but precise dates vary yearly. The study site includes several species of planted non-natives (crested wheatgrass; *A. cristatum,* small burnet: *Sanguisorba minor*); invasive grasses (cheatgrass, medusahead, and ventenata); and encroaching western juniper. In 2011 ODFW designated the Murderer's Creek Unit as a Mule Deer Initiative unit where ODFW wanted to focus efforts on improving mule deer populations (Oregon Department of Fisheries and Wildlife, 2011). Within the Murderer's Creek Unit, PWSWA was identified as critical wintering range for mule deer and in response ODFW is actively attempting to improve the wildlife area in efforts to reestablish viable wintering habitat for mule deer and other sage-steppe — dependent species while maintaining grazing and recreational uses of the land.

Experimental Design and Data Collection

In 2012 ODFW created six, 1-ha blocks within the study site, averaging 226 m between each block (min = 150 m). Blocks were placed nonrandomly, so they had similar north-facing aspects and slope (~4%). Blocks were initially divided in half (50 × 100 m), with one-half being fenced (ungrazed) and the other half remaining open (grazed). The exclosures were made with barbed wire fencing and were approximately 1.25 m in height. Barbed wire keeps cattle and resident feral horses out of the study plots but does not prevent mule deer *(Odocoileus hemionus)* or elk *(Cervus canadensis)* from entering. The blocks were then divided again (creating 50 × 50 m plots), in which all juniper were mechanically felled (cut) in one-half while in the other juniper were allowed to remain (uncut). Juniper cutting occurred March — April 2013 with chainsaws, and trees were left where they fell. This created four treatment plots within each block: grazed-cut (GC), grazed-uncut (GUC), ungrazed-cut (UGC), and ungrazed-uncut (UGUC) for a total of 24 study plots, with 6 replicates (1 per block) of each treatment type. Plots were randomly assigned within each block.

Within each plot, a single permanent 30-m transect was established in 2012 by randomly selecting a starting point and then semirandomly selecting a compass direction (i.e., transects could not exit the plot) to determine the end point. Start and end points were marked with 1.27-cm rebar stakes. These transects were used to collect annual reference data on percent cover of herbaceous and woody plants and approximate the number of shrubs within the plot, shrub cover, amount of downed woody material, and biomass. In May to early June of each sample year, we estimated point cover by recording the species of plants intersected by the transect at 0.5-m intervals and then divided by 60 (total number of points along the transect; Elzinga et al., 1998). Plants were divided into seven categories: exotic grasses, preferred grasses, preferred forbs, exotic forbs, preferred plants, juniper, and litter. Exotic grasses and forbs were defined as any non-native grass or forb not intentionally planted at the site (i.e., invasive species). Preferred grasses and forbs included all native grass and forb species, as well as non-native species planted at the site as forage, while preferred plants included both the preferred grasses and preferred forbs. We classified any nonliving herbaceous plant material as litter. Exotic forbs were not encountered in the study site, so they were not included in data analysis. We calculated shrub percent cover by calculating the percent of the transect that was intersected by live shrub canopy. Shrub cover was considered continuous as long as there were no gaps ≥ 10 cm. To quantify shrub abundance, we counted shrubs within 3-m belt transects on the right side (looking from start to end point) of our 30-m transect, identifying to species every shrub rooted within the belt. We collected aboveground biomass by placing 1 × 1 m plots on the left side of the transect and cutting all nonwoody plant material to the ground. Living plant material that was cut was divided into preferred herbs (preferred grasses plus forbs) and exotic herbs (non-native plants). Nonliving biomass (i.e., litter) was divided into herbaceous litter and juniper litter. We collected all biomass material in paper bags, labeled by type, plot number, and date and then dried at 60°C for 72 hr. After 72 hr, we removed dried materials from the oven and immediately weighed them to 0.01 g.

A 26 700-ha wildfire in 2014 burned much of the PWSWA but did not affect the Flat Creek location. To monitor effects of that fire in a parallel study, the Flat Creek location was used as an unburned control. As such, additional fuels parameters (downed wood, surface fuels) were collected in 2015 and 2016. In order to quantify woody biomass within each plot, all downed woody material that intersected the transect was enumerated (Brown, 1974). We also collected the biomass of the litter and duff layers in the aforementioned 1 × 1 m plots, to determine the amount of surface fuel within the plots. The duff was divided into two categories: herbaceous litter and juniper duff.

To assess the potential role of downed juniper as juniper reinvasion "islands," we compared juniper recruitment underneath downed trees and in the between-tree interspaces. Beginning in 2016, 4 yr post cut, we measured the crown (length × width) of five randomly chosen downed junipers within each cut plot and counted the number of juniper seedlings and saplings within the area of the crown. We then picked a random direction from the downed juniper and counted the number of juniper saplings in a 4 × 2 m interspace area. We then divided the number of saplings by the area measured to obtain the number of saplings per square meter.

Data Analysis

We performed all of our data analyses in R (R Development Core Team, 2017). Most of our data were collected within a blocked repeated measures design, and thus we used linear mixed models in those instances. Treatment (GC, GUC, UGC, UGUC) and year (as a factor) were the independent variables, block and plot were random effects within the model, and we used a continuous autoregressive (corCAR1) structure to account for the time (year) autocorrelation. We ran models using the *lme* function within the *nlme* package (Pinheiro et al., 2017). To conduct multiple comparisons of significant linear mixed models, we used the *glht* function within the *multcomp* package using Tukey's all-pair comparisons set at $\alpha = 0.05$ (Hothorn et al., 2008). For data that were not collected multiple years (i.e., some biomass and sapling recruitment), we used analyses of variance to test for differences among treatments. Because sapling recruitment was only characterized in cut treatments, we used location (interspace or juniper skeleton) and grazed/ungrazed as the independent variables. Data were natural log transformed when necessary to meet assumptions of the model.

Results

Before treatments, the mean percent cover of exotic grasses (39%) and preferred grasses (32%) did not statistically differ among the plots ($t = -1.35$, $P = 0.18$); however, there was much more variance in the exotic grasses (SD = 24%) than in preferred grasses (SD = 11%). The exclusion of low-intensity seasonal grazing had no effect on either shrub cover, shrub abundance, percent cover, or biomass for any category of plants, nor was there any interaction among grazing, year, or juniper cutting (Table 1). There was a cut × year effect on percent cover of exotic grasses ($F_{3,59} = 4.7$, $P = 0.005$), forbs ($F_{3,59} = 5.8$, $P = 0.008$), preferred plants ($F_{3,59} = 3.8$, $P = 0.015$), and shrubs ($F_{3,59} = 3.3$, $P = 0.025$) but not preferred grasses ($F_{3,59} = 2.1$, $P = 0.106$). Herbaceous (except preferred grasses) and shrub cover increased every year in cut plots after treatment until 2016 (see Fig. 1). For all five categories of plant cover, there were no statistical differences in the percent cover of plants pretreatment or 3 yr post treatment between the cut and uncut plots. There was no consistent trend over time, but instead there was year-to-year variation (see Fig. 1). There was a cut × year effect on exotics with uncut treatments having lower exotic grass cover than cut treatments in post-treatment years ($F_{3,59} = 4.7$, $P = 0.005$). (See Fig. 2.)

Shrub cover increased in cut plots in both 2015 and 2016 post treatment (cut × year effect; $F_{3,59} = 44.42$, $P < 0.01$). There was a cut × year effect for shrub cover ($F_{3,59} = 3.3$, $P = 0.02$; Fig. 3), such that there was a delay in treatment response until the second post-treatment year, where shrub percent cover was approximately 60% higher in cut plots than uncut plots (see Fig. 3). No bitterbrush occurred within the belt transects, and while rabbitbrush was detected, it was not numerous enough to allow analysis (mean = 0.75 shrubs per plot). Only two species were numerous enough to warrant statistical analysis on occurrence within belt transects: big sagebrush and broom snakeweed *(Gutierrezia sarothrae)*. There was no change in sagebrush density between years, nor in juniper reduction or grazing treatments ($F_{7,17} = 0.76$, $P = 0.63$). Broom snakeweed was only found in 2016 and did not differ between treatments ($F_{1,3} = 0.85$, $P = 0.48$).

There was a cut and a year effect for aboveground biomass of exotic plants, preferred plants, and total aboveground biomass (see Table 1). However, there were not any cut × year effects for exotic plants ($F_{2,39} = 1.9$, $P = 0.16$), preferred plants ($F_{2,39} = 0.66$, $P = 0.52$), or total biomass ($F_{3,59} = 1.2$, $P = 0.32$). Generally, cut plots had higher aboveground biomass than uncut plots where juniper remained. Neither total aboveground herbaceous biomass (exotics plus preferred) nor preferred biomass differed between pretreatment and 3 yr post treatment (Table 2).

There were no treatment effects on the amount of herbaceous litter biomass (see Table 1), but there was a year effect ($F_{1,18} = 48.83$, $P < 0.001$) with litter being higher in 2016 (mean = 532 kg/ha) than 2015 (mean = 175 kg/ha), the only 2 yr the data were collected. Juniper litter did not differ between treatments or years (see Table 1) with a mean of 4 374 kg/ha across plots and years. Downed wood differed between cut treatments ($F_{1,1} = 7.84$, $P = 0.01$) with a mean of 17 837 kg/ha in cut plots and 2 490 kg/ha in uncut plots. There was no difference in downed woody material between grazing treatments ($F_{1,1} = 0.34$, $P = 0.57$) or plot (grazed × cut; $F_{1,3} = 1.59$, $P = 0.22$). Lastly, there were approximately two times more juniper saplings underneath downed juniper skeletons than in the interspace (mean = 0.07/m^2 vs. 0.03/m^2; $t = 2.45$, $P = 0.016$) and grazing did not have any effect on sapling density ($t = 0.3$, $P = 0.76$).

Discussion

One purpose of juniper cutting is to maintain the current shrub community, whereas other methods (e.g., broadcast burning) may eliminate established shrubs and require longer times to recover after treatment. Within the duration of our study, there was little post-treatment

Table 1
Analysis of variance results of linear mixed-effect models (lme). lme models were formulated as y ~ cut · grazed · year, where *y* is the measured variable (percent cover or biomass). The cut variable is whether the plot had juniper cut or not, grazed is whether the plot was open to grazing or remained ungrazed during the study, and the year variable covers the 4 yr of the study (2012, 2014–2016). Exotic grasses were defined as any non-native grass not intentionally planted at the site (i.e., invasive species). Exotic plants included both exotic grasses and exotic forbs. Preferred grasses included all native grass species and non-native species planted as forage, while preferred plants included both preferred grasses and forbs. Herbaceous litter was all litter within the 1 m^2 plot. Values shown in bold represent models that had significant *P* values at $\alpha = 0.05$

Measurement	Variable	Exotic grass		Preferred grass		Forbs		Preferred plants		Shrubs	
Percent cover	Cut	$\mathbf{F_{1,59} = 6.3}$	$\mathbf{P = 0.02}$	$F_{1,59} = 1.3$	$P = 0.26$	$\mathbf{F_{1,59} = 6.1}$	$\mathbf{P = 0.02}$	$\mathbf{F_{1,59} = 5.9}$	$\mathbf{P = 0.02}$	$\mathbf{F_{1,59} = 44.4}$	$\mathbf{P < 0.01}$
	Grazed	$F_{1,21} = 0.01$	$P = 0.89$	$F_{1,21} = 1.4$	$P = 0.25$	$F_{1,21} = 0.03$	$P = 0.85$	$F_{1,21} = 0.03$	$P = 0.85$	$F_{1,21} = 0.12$	$P = 0.73$
	Year	$\mathbf{F_{1,59} = 36.4}$	$\mathbf{P < 0.01}$	$\mathbf{F_{1,59} = 4.3}$	$\mathbf{P = 0.01}$	$\mathbf{F_{1,59} = 34}$	$\mathbf{P < 0.01}$	$\mathbf{F_{1,59} = 6.8}$	$\mathbf{P < 0.01}$	$\mathbf{F_{1,59} = 5.8}$	$\mathbf{P < 0.01}$
	Full interaction	$F_{1,59} = 0.9$	$P = 0.41$	$F_{1,59} = 0.68$	$P = 0.57$	$F_{1,59} = 1.1$	$P = 0.36$	$F_{1,59} = 1.2$	$P = 0.29$	$F_{1,59} = 0.69$	$P = 0.56$
		Exotic plants		Preferred plants		Total biomass		Herbaceous litter		Juniper litter	
Biomass	Cut	$\mathbf{F_{1,39} = 7.0}$	$\mathbf{P = 0.01}$	$\mathbf{F_{1,39} = 14.9}$	$\mathbf{P < 0.01}$	$\mathbf{F_{1,39} = 4.2}$	$\mathbf{P = 0.05}$	$F_{1,39} = 0.09$	$P = 0.76$	$F_{1,16} = 0.16$	$P = 0.69$
	Grazed	$F_{1,21} = 1.0$	$P = 0.32$	$F_{1,21} = 0.01$	$P = 0.25$	$F_{1,21} = 0.86$	$P = 0.36$	$F_{1,21} = 0.46$	$P = 0.50$	$F_{1,21} = 0.01$	$P = 0.76$
	Year	$\mathbf{F_{1,39} = 15.4}$	$\mathbf{P < 0.01}$	$\mathbf{F_{1,39} = 44.0}$	$\mathbf{P < 0.01}$	$\mathbf{F_{1,39} = 10.2}$	$\mathbf{P < 0.01}$	$\mathbf{F_{1,39} = 48.8}$	$\mathbf{P < 0.01}$	$F_{1,16} = 1.5$	$P = 0.24$
	Full interaction	$F_{1,39} = 0.44$	$P = 0.64$	$F_{1,39} = 0.88$	$P = 0.43$	$F_{1,39} = 0.51$	$P = 0.68$	$F_{1,39} = 0.07$	$P = 0.79$	$F_{1,16} = 0.73$	$P = 0.40$

establishment of shrub species important to wildlife (i.e., big sagebrush and bitterbrush). The shrubs did not change in abundance over the course of the study (sagebrush, see Fig. 3) or were not present within the study site (bitterbrush), despite being locally abundant nearby, or our sampling occurred too soon post treatment to detect changes. Disturbance-adapted and early successional shrubs, primarily broom snakeweed, had increased shrub cover starting 2 yr post treatment. The 60% increase in total shrub cover was likely due to an increase in soil water availability. Roundy et al. (2014b) showed that soil water availability increased after tree reduction for up to 4 yr compared with untreated control plots at similar western juniper sites. Unlike broom snakeweed and rabbitbrush, both sagebrush and bitterbrush are later seral species and a change in abundance may not be seen for 5 − 8 yr post treatment (Tausch and Tueller, 1977) or longer (Bates et al., 2005; Bates and Davies, 2017). Although there is no short-term effect on sagebrush abundance, the persistence of sagebrush in the study plots provides a seed source that may lead to a longer-term recovery of sagebrush from seed (Allen and Nowak, 2008).

Juniper cutting increased cover of all herbaceous plants, with exotic grasses, forbs, and total preferred plants higher in cut plots than uncut plots in yr 1 and 2 post treatment. Yr 2015 and 2016, however, were drought years (NDMC, USDA, and NOAA, 2017), and cover of all herbaceous plants was lower by 2016, with no differences between cut and uncut treatments (see Fig. 1). Similar short-term responses of the

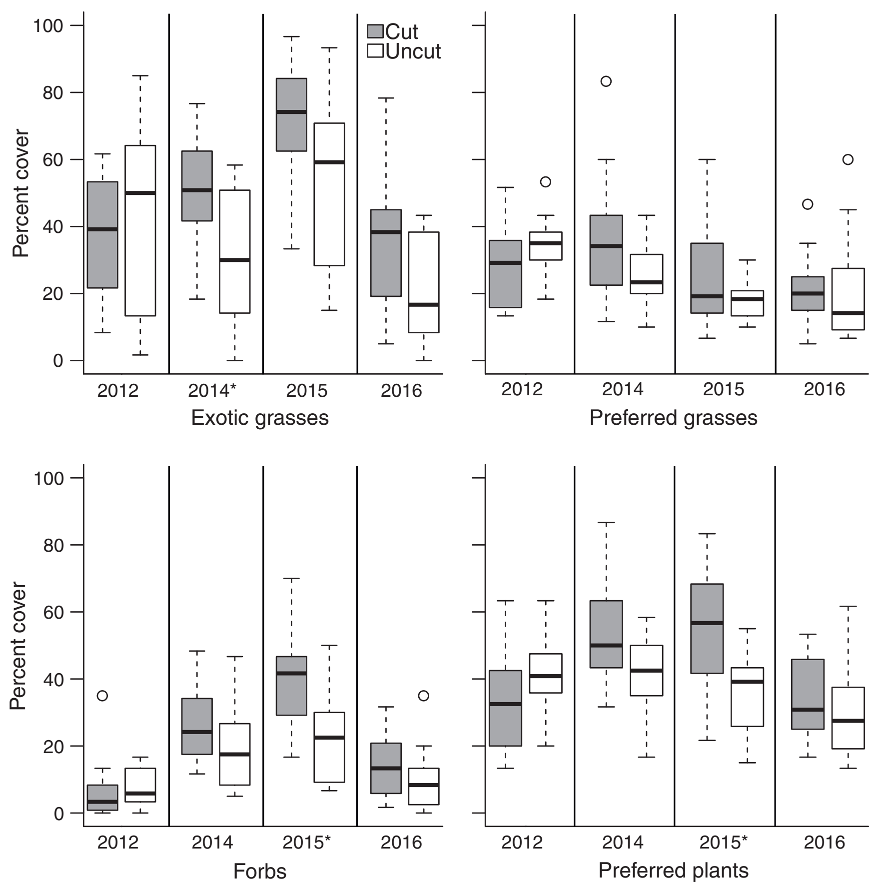

Figure 1. Median percent cover of herbaceous plants per cut treatment at Philip W. Schneider Wildlife Area, Dayville, Oregon, United States from 2012 to 2016. Data are represented as box plots showing median *(lines within boxes)*, first and third quartiles *(boxes)*, and 1.5 interquartile range *(whiskers)*. Data outside the 1.5 interquartile range are represented as *open circles*. *White boxes* represent uncut plots, and *gray boxes* represent cut plots. *Asterisks* next to the years represent significant results of post-hoc analysis ($\alpha = 0.05$) following mixed-model analysis of transformed data (when necessary) for differences between treatments within each treatment year.

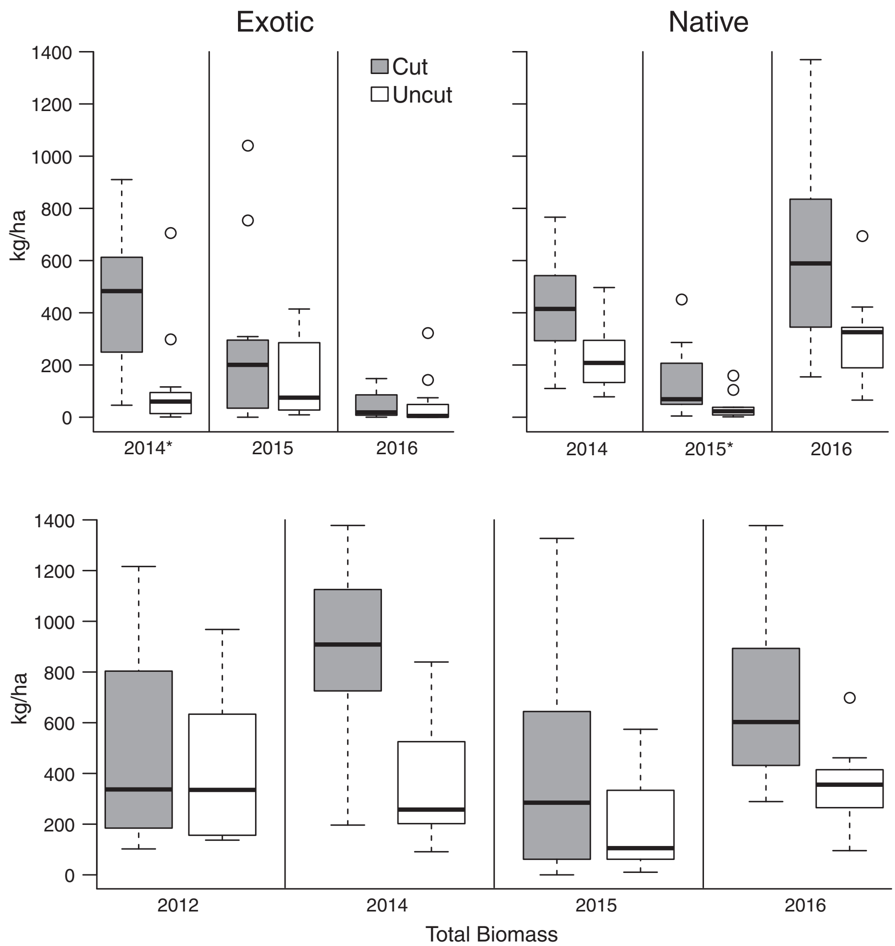

Figure 2. Median biomass (kg/ha) for each cut treatment by year at Philip W. Schneider Wildlife Area, Dayville, Oregon, United States from 2012 to 2016. Data are represented as box plots showing median *(lines within boxes)*, first and third quartiles *(boxes)*, and 1.5 interquartile range *(whiskers)*. Data outside the 1.5 interquartile range are represented as *open circles*. *White boxes* represent uncut plots, and *gray boxes* represent cut plots. *Asterisks* next to the years represent significant results of post-hoc analysis ($\alpha = 0.05$) following mixed-model analysis of transformed data (when necessary) for differences between treatments within each treatment year.

herbaceous plants to juniper cutting have been observed in other studies (Bates et al., 2000; Roundy et al., 2014a). Despite there being no increase in the cover of preferred grasses after juniper cutting, we did not observe overall dominance by exotic grasses. This may be because our study sites are at relatively high elevations and on north-facing slopes (i.e., cooler temperatures), which minimize the competitive advantage of non-native grasses (Chambers et al., 2007, 2014). The exception was 2015, when the exotic-to-preferred grass ratio was near 3:1. This large difference in percent cover followed a year with high precipitation (2014 = 48.18 cm, 30-yr mean = 43.22; PRISM Climate Group, 2004). This increase in exotic grass cover was likely a result of exotic grasses being better able to take advantage of pulse moisture availability and

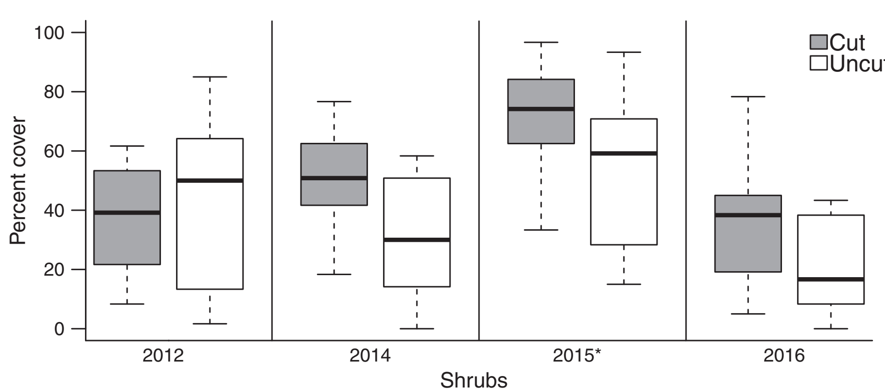

Figure 3. Median percent cover of shrubs in juniper reduction treatments at Philip W. Schneider Wildlife Area, Dayville, Oregon, United States from 2012 to 2016. Data are represented as box plots showing median *(lines within boxes)*, first and third quartiles *(boxes)*, and 1.5 interquartile range *(whiskers)*. Data outside the 1.5 interquartile range are represented as *open circles*. *White boxes* represent uncut plots, and *gray boxes* represent cut plots. *Asterisk* represents the results of post-hoc analysis ($\alpha = 0.05$) following mixed-model analysis of transformed data (when necessary) for differences between treatments within each treatment year.

Table 2
Biomass of groups of species (kg/ha) in cut and uncut treatments by year plus/minus standard error. Exotic grasses were defined as any non-native grass not intentionally planted at the site (i.e., invasive species), while preferred plants included both the preferred grasses (defined as all native grass species and non-native species planted at the site as forage) and forbs. Values with different letters indicate whether values are statistically different between years (a−c for cut and x−z for uncut) based on a post-hoc analysis (α = 0.05). Daggers (†) represent data collected before juniper removal from plots. NA indicates data were not collected that year

Treatment	Yr	Exotics grasses kg • ha^{-1}	Preferred plants kg • ha^{-1}	Total biomass kg • ha^{-1}	Herbaceous litter kg • ha^{-1}	Juniper litter kg • ha^{-1}
Cut	2012†	NA	NA	480.13 ± 110.25 abc	NA	NA
	2014	459.31 ± 82.27 a	418.03 ± 58.22 a	877.34 ± 100.53 a	NA	NA
	2015	271.16 ± 91.47 a	137.83 ± 49.53 b	386.39 ± 117.04 b	50.76 ± 16.30 a	1147.87 ± 769.47 a
	2016	43.36 ± 14.61 b	618.68 ± 97.77 a	662.05 ± 91.39 c	576.9 ± 144.42 a	3240.31 ± 1607.10 a
Uncut	2012†	NA	NA	410.88 ± 80.64 x	NA	NA
	2014	122.47 ± 57.83 x	228.66 ± 38.44 x	351.14 ± 66.14 x	NA	NA
	2015	147.03 ± 44.03 x	40.72 ± 16.25 y	191.52 ± 55.14 y	298.78 ± 260.14 x	669.45 ± 299.67 x
	2016	48.33 ± 27.91 y	303.15 ± 47.78 x	351.47 ± 44.50 x	483.78 ± 89.44 x	11552.93 ± 6836.91 x

the higher nitrogen concentrations associated with increased precipitation (Jones et al., 2015) and tree cutting (Roundy et al., 2014b; Bates and Davies, 2017) Therefore, the increase of exotic grasses is not likely indicative of a long-term treatment effect.

The cutting of juniper increased the amount of downed woody biomass on the site. There was however, no difference between treatments in the amount of herbaceous biomass, which represents the fine surface fuels that carry fire (Scott and Burgan, 2005). Similar to other findings, there was a difference in herbaceous biomass between years coincident with patterns of precipitation (Bates et al., 2005).This reflects the yearly differences that we also measured in percent cover of herbaceous plants (see Fig. 1). There was an increase in cover in 2015 in all categories except preferred grasses. Juniper litter (duff) was a large source of fuel (mean = 4374 kg/ha) across all treatments and years, but there was no difference between treatments or years. This juniper litter was patchy in distribution, occurring under the canopies or around cut stumps, likely modifying potential fire behavior toward higher-severity burns in those areas because of the potential for longer-duration smoldering combustion (Covington and DeBano, 1990).

The removal method of juniper used for this study, cutting trees and allowing them to lie as they fall, appears to facilitate juniper germination and establishment, at least in the short term. Bates et al. (2005) showed that tree density returned to precut values 13 yr after juniper removal when left unmanaged. In the 3 yr post treatment in our study, we found twice as many juniper saplings per meter beneath juniper skeletons than in the interspace. Cut juniper may offer ideal establishment sites because of higher soil nutrient concentrations, greater soil water, and protection from herbivory (Chambers, 2001; Roundy et al., 2014b; Bates and Davies, 2017). Therefore, it is likely that these downed junipers are acting as nurse-plants and are islands for future juniper encroachment or reestablishment after juniper cutting. We observed that some other shrub species (primarily *Rosa* spp.) appear to establish better beneath juniper skeletons; however, we did not test for such a relationship due to small sample sizes.

Our results suggest that the current grazing regime is appropriate for the area. The regime of low stocking rates (17.5 ha/AUM), short duration of grazing, and rotating rest years did not cause any additional measurable detrimental effects to the plant community. While there were vegetation responses to juniper reduction treatments 1, 2, and 3 yr post treatment, there was not a consistent trend across years. In fact, much of the plant community did not differ from pretreatment communities 3 yr after treatment, despite deviating significantly in vegetation cover and biomass in yr 1 and 2 post treatment. Another consideration for this study is that all of the study plots were located on a north-facing aspect. These results may not be representative of south-facing aspects, where the microclimate tends to be warmer and drier. These results support the need for more studies that follow plant community responses over longer time frames (i.e., >3 yr; Bates et al., 2017; Williams et al., 2017) in order to determine the efficacy of management practices.

Implications

Our study shows that cutting postsettlement juniper, a common management strategy, increases herbaceous plant cover within 1 and 2 yr post treatment in phase II woodlands. Shrub cover also increases 2−3 yr post treatment but may be limited to early seral species in the short term. Additionally, our results demonstrate that low-intensity grazing, in a deferred rest rotation scheme, did not affect cover of herbaceous plants or shrubs during the first 3 yr after juniper reduction. This suggests that managers may have more flexibility in the rest rotation schedule, deferring rotation to juniper reduction treatments and extending rest periods on sites with additional disturbances (e.g., fire). However, the practice of cutting and leaving juniper skeletons on the landscape increased juniper recruitment. To prevent juniper reestablishment, managers may want to retreat managed areas, focusing on juniper skeletons where recruitment was twice as high than the interspace. Because succession and postdisturbance/treatment recovery in arid sagebrush steppe communities is slow, management interventions should be planned to match the temporal scale of ecosystem recovery.

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