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Longer-Term Evaluation of Sagebrush Restoration After Juniper Control and Herbaceous Vegetation Trade-offs☆

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ABSTRACT

Degradation of shrublands around the world from altered fire regimes, overutilization, and anthropogenic disturbance has resulted in a widespread need for shrub restoration. In western North America, reestablishment of mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* [Rydb.] Beetle) is needed to restore ecosystem services and function. Western juniper (*Juniperus occidentalis* ssp. *occidentalis* Hook) encroachment is a serious threat to mountain big sagebrush communities in the northern Great Basin and Columbia Plateau. Juniper trees can be controlled with fire; however, sagebrush recovery may be slow, especially if encroachment largely eliminated sagebrush before juniper control. Short-term studies have suggested that seeding mountain big sagebrush after juniper control may accelerate sagebrush recovery. Longer-term information is lacking on how sagebrush recovery progresses and if there are trade-offs with herbaceous vegetation. We compared seeding and not seeding mountain big sagebrush after juniper control (partial cutting followed with burning) in fully developed juniper woodlands (i.e., sagebrush had been largely excluded) at five sites, 7 and 8 yr after seeding. Sagebrush cover averaged ~30% in sagebrush seeded plots compared with ~1% in unseeded plots 8 yr after seeding, thus suggesting that sagebrush recovery may be slow without seeding after juniper control. Total herbaceous vegetation, perennial grass, and annual forb cover was less where sagebrush was seeded. Thus, there is a trade-off with herbaceous vegetation with seeding sagebrush. Our results suggest that seeding sagebrush after juniper control can accelerate the recovery of sagebrush habitat characteristics, which is important for sagebrush-associated wildlife. We suggest land manager and restoration practitioners consider seeding sagebrush and possibly other shrubs after controlling encroaching trees where residual shrubs are lacking after control.

Introduction

Restoration is needed across hundreds of millions of hectares of degraded wildlands to restore ecosystem function and services. Shrub restoration is underrepresented, with research and effort more focused on grasses and trees. However, the need for shrub restoration is becoming increasingly recognized in wildlands in Europe (Medina-Roldán et al., 2012), Africa (Linstadter and Baumann, 2013), Australia (Wong et al., 2007), North America (Davies et al., 2011), and Asia (Li et al., 2013). Restoration of shrubs around the world is essential because of overexploitation, altered fire regimes, and mismanagement (Han et al., 2008; Sasaki et al., 2008; Bedunah et al., 2010; Medina-Roldán et al., 2012; Linstandter and Baumann, 2013). Many shrubs are keystone species that provide critical ecosystem services (Prevéy et al., 2010; Fonseca et al., 2012; van Zonneveld et al., 2012).

Mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* [Rydb.] Beetle) is a shrub restoration priority in western North America (Davies et al., 2017a). Restoration of mountain big sagebrush is needed because of widespread conifer (*Juniperus* L. and *Pinus* L. species) encroachment caused by decreased fire frequency, historical overstocking of livestock, increasing atmospheric CO_2, and favorable climatic conditions for conifer growth (Tausch et al., 1981; Miller and Wigand, 1994; Knapp and Soulé, 1998; Miller et al., 2005). In the northern Great Basin and Columbia Plateau, the prevailing conifer encroaching into mountain big sagebrush communities is western juniper (*J. occidentalis* ssp. *occidentalis* Hook), having increased from 0.3 million to 3.5 million ha since the 1870s (Miller et al., 2000). Western juniper encroachment results in the loss of sagebrush, decreases in herbaceous production and diversity, and increases in erosion and runoff risk (Miller et al., 2000; Bates et al., 2005; Pierson et al., 2007). Juniper encroachment also reduces the retention of snow, increases evapotranspiration loss, and alters the timing of water availability (Kormos et al., 2017). Loss of sagebrush, decreases in herbaceous vegetation, and more predator perches with juniper encroachment negatively affect many sagebrush-associated wildlife including greater sage-grouse, a species of

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conservation concern (Connelly et al., 2000; Miller et al., 2005; Baruch-Mordo et al., 2013). Because of these ecological and productivity impacts, restoration of juniper-encroached mountain big sagebrush rangeland is a land management priority (Miller et al., 2005; Davies et al., 2011; Baruch-Mordo et al., 2013).

Restoration of mountain big sagebrush requires controlling encroaching conifers. Western juniper is commonly controlled with cutting, prescribed burning, and combinations of cutting and burning. Burning generally results in more complete and longer-term control of juniper than cutting because cutting often fails to completely control juniper seed sources, seedlings, and small juveniles (Miller et al., 2005; Boyd et al., 2017). Prescribed burning or partial cutting (cutting one-fourth to one-half of mature trees to increase dry fuel) with burning is also generally more cost efficient than cutting (Miller et al., 2005; Boyd et al., 2017). Burning kills most remaining sagebrush, and because mountain big sagebrush does not resprout, sagebrush recovery is dependent on the site's seedbank and seed dispersal from adjacent unburned areas (Ziegenhagen and Miller, 2009).

Sagebrush is often assumed to recover naturally after conifer control (Barney and Frischknecht, 1974; Tausch and Tueller, 1977; Skousen et al., 1989); however, the rate of recovery is variable. Sagebrush recovery after fire is estimated to take from 15 to 100+ yr (Baker, 2006; Ziegenhagen and Miller, 2009; Nelson et al., 2014). These estimates were derived from areas dominated by sagebrush before burning. The rate of recovery may be even slower in areas where sagebrush has largely been excluded from the community by juniper encroachment as the seed banks in these communities are likely sagebrush seed limited (Bates et al., 2005; Davies et al., 2014). Therefore, it may be valuable to expedite sagebrush recovery after prescribed burning in areas dominated by juniper as sagebrush is a crucial habitat component for sagebrush-associated wildlife (Crawford et al., 2004; Shipley et al., 2006; Aldridge et al., 2008).

Broadcast seeding mountain big sagebrush after juniper control with prescribed burning can accelerate short-term sagebrush recovery (Davies et al., 2014; Davies and Bates, 2017). Sagebrush restoration success, however, varies by site characteristics (Davies and Bates, 2017) and seeding failure may occur when herbaceous competition is high (Davies et al., 2017a). The long-term effect of broadcast seeding sagebrush or not seeding sagebrush after prescribed burning encroaching junipers is unknown. This information is needed to evaluate the effects of this treatment. Furthermore, Davies et al. (2014) raised questions about the effect of seeding sagebrush on herbaceous vegetation, speculating that as sagebrush cover increased, the herbaceous vegetation would decrease. Determining the long-term effects of seeding sagebrush after juniper control on sagebrush recovery and herbaceous vegetation is needed for developing well-informed management decisions.

The purpose of this study was to determine the long-term effects of seeding mountain big sagebrush after western juniper was controlled with partial cutting followed by prescribed burning on sagebrush recovery and herbaceous vegetation. We hypothesized that seeding sagebrush would accelerate sagebrush recovery (i.e., greater sagebrush cover and density) and decrease herbaceous vegetation cover and density.

Materials and Methods

Study Area

The study was conducted in southeastern Oregon approximately 80 km southeast of Burns, Oregon at five study sites. Before western juniper encroachment, study sites were mountain big sagebrush—bunchgrass communities (NRCS, 2012). Juniper encroachment had largely eliminated sagebrush from the plant communities by the time juniper control treatments were implemented. Before treatment, juniper cover ranged from 29% to 43% and understories were composed of perennial grasses and forbs. Common perennial grasses were Idaho fescue (*Festuca idahoensis* Elmer), bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] A. Löve), and Sandberg bluegrass (*Poa secunda* J. Presl). Study sites were Loamy 12-16 PZ, North Slope 12-16 PZ, and Droughty Loam 11-13 PZ Ecological Sites (NRCS, 2012). Elevation at the study sites ranged from 1 746 to 1 808 m above sea level, and slopes varied between 5% and 10% with aspects ranging from northeast to southwest. Long-term (1981 − 2010) average annual precipitation was 447 mm (PRISM, 2016). Climate of the study sites was typical of the northwestern Great Basin with cool, wet winters and warm, dry summers. Cattle were excluded from the study sites 1 yr before the treatments and then for 2 yr following seeding. Cattle grazing for the remainder of the study was moderate use (~40% utilization by weight) and alternated between late spring and late summer use.

Experimental Design and Measurements

We used a randomized block design with five blocks (sites) to evaluate the long-term effects of seeding mountain big sagebrush after controlling juniper with partial cutting followed with prescribed fire. Blocks varied in soil, topography, and understory vegetation composition. One yr before prescribed burns were applied, one-half of mature junipers were cut down with chainsaws to increase dry ground fuel to carry fire across each block and across the surrounding area as part of a larger (6 475-ha) rangeland restoration project conducted by the Bureau of Land Management. Blocks were > 200 m from the edge of the restoration project. For a block to be selected for inclusion in the study, it had to have uniform soil, topography, and vegetation across a large enough area for both treatments. In late September 2009 over a 10-d period, blocks were prescribe burned with head fires, resulting in 100% mortality of juniper trees. In each block, treatments were randomly applied to one of two 15 × 30 m plots. Treatments were seeded with perennial herbaceous vegetation (Herb) ($n = 5$) and seeded with perennial herbaceous vegetation and mountain big sagebrush (Sage + Herb) ($n = 5$). The Herb treatment serves as the control treatment because exotic annual grass dominance is a serious risk when fire occurs in fully developed juniper woodlands (Bates et al., 2014; Davies et al. in press). Herbaceous species were aerially seeded in early November 2009 with a fixed-wing aircraft. Herbaceous seeding was performed as part of the larger rangeland restoration project. The herbaceous seed mix consisted of 1.6 $kg \cdot ha^{-1}$ Idaho fescue (*Festuca idahoensis* Elmer), 3.0 $kg \cdot ha^{-1}$ Sherman big bluegrass (*Poa ampla* Merr.), 1.3 $kg \cdot ha^{-1}$ Oahe intermediate wheatgrass (*Thinopyrum intermedium* [Host] Barkworth & D.R. Dewey), 3.6 $kg \cdot ha^{-1}$ Manchar smooth brome (*Bromus inermis* Leyss.), 2.2 $kg \cdot ha^{-1}$ Paiute orchardgrass (*Dactylis glomerata* L.), 0.6 $kg \cdot ha^{-1}$ Maple Grove Lewis flax (*Linum lewisii* Pursh), and 0.3 $kg \cdot ha^{-1}$ Ladak alfalfa (*Medicago sativa* L.). Mountain big sagebrush was seeded immediately after herbaceous vegetation was seeded using a hand-cranked broadcast seeder at a rate of 1.8 kg $PLS \cdot ha^{-1}$. Herbaceous seeding was applied across the burned juniper woodland, and sagebrush was seeded across the 15 × 30 m plot in each block.

Shrub density and herbaceous cover and density were measured in July 2016 and 2017 (seventh and eight yr post seeding). Shrub cover was measured in July from 2010 through 2017 (first through eighth yr post seeding). Vegetation was sampled along five, 25-m transects spaced at 2-m intervals in each 15 × 30 m plot. Herbaceous cover and density, bare ground, biological soil crust, and litter were measured in 0.2-m^2 quadrats located along the 25-m transects at 2-m intervals (12 quadrats per transect). Cover was visually estimated, and density was determined by counting plants rooted inside the quadrats. Rhizomatous species density was estimated by dividing the quadrats into quarters and counting the quarters that contained the species. Shrub cover was measured by species using the line-intercept method (Canfield, 1941) along the five 25-m transects. Shrub density was measured by species by overlaying each of the five, 25-m transects with a 2 × 25 m belt transect. All shrubs rooted in the belt transects were counted.

Statistical Analyses

Repeated measures analysis of variance (ANOVA) using the mixed models procedure (Proc Mixed) in SAS v. 9.4 (SAS Institute Inc., Cary, NC) was used to compare treatments. Year was the repeated variable. Block and block-by-treatment interactions were considered random effects. Covariance structure was determined using Akaike's Information Criterion (Littell et al., 1996). Data that violated ANOVA assumptions were square root transformed before analyses. All data presented are in their original dimensions (i.e., nontransformed). Significance level for all tests was set at $P \le 0.05$. Response variable means were reported with standard errors. For analyses, herbaceous cover and density were separated into five groups: Sandberg bluegrass (*Poa secunda* J. Presl), large perennial grasses, annual grasses (cheatgrass [*Bromus tectorum* L.] was the primary annual grass detected), perennial forbs, and annual forbs. Groups are used to combine species that respond similarly to disturbances and have similar growth characteristics to reduce data to improve presentation and analysis (Boyd and Bidwell, 2002). Sherman big bluegrass, though sometimes classified as a variety of Sandberg bluegrass, was considered a large perennial grass in the analyses because it is larger and matures later than the common Sandberg bluegrass in this ecosystem. Shrubs were separated into sagebrush and other shrubs for analyses.

Results

Sagebrush cover varied by the interaction between treatment and year (Fig. 1; $P < 0.001$). Sagebrush cover increased over time in the Sage + Herb treatment but remained low in the Herb treatment. By the end of the study, sagebrush cover was 30× greater in the Sage + Herb compared with Herb treatment. Other shrub cover was similar between the Sage + Herb (0.41 ± 0.31%) and Herb (0.31 ± 0.25%) treatments ($P = 0.598$). Sandberg bluegrass cover did not differ between treatments (Fig. 2; $P = 0.158$) but was greater in 2017 than 2016 ($P < 0.001$). Perennial grass cover was less in the Sage + Herb compared with Herb treatment (see Fig. 2; $P = 0.011$) and greater in 2016 than 2017 ($P < 0.001$). Annual grass cover did not differ between treatments (see Fig. 2; $P = 0.267$). Annual grass cover was greater in 2016 compared with 2017 ($P = 0.014$). Perennial forb cover was similar between treatments (see Fig. 2; $P = 0.492$) and years ($P = 0.063$). Annual forb cover was less in the Sage + Herb compared with Herb treatment (see Fig. 2; $P = 0.012$) and decreased from 2016 to 2017 ($P < 0.001$). Total herbaceous cover was 1.6−2.0× greater in the Herb compared with Sage + Herb treatment in 2016 and 2017, respectively (see Fig. 2; $P = 0.047$), and less in 2017 compared with 2016 ($P < 0.001$). Bare ground, litter, and biological soil crust cover did not differ between treatments (see Fig. 2; $P = 0.256$, 0.634, and 0.387, respectively). Bare ground and biological soil crust cover did not differ between years ($P = 0.388$ and 0.143, respectively). Litter was greater in 2017 than 2016 ($P = 0.006$).

Sagebrush density was greater in the Sage + Herb (0.800 ± 0.200 plants·m^{-2}) compared with Herb (0.020 ± 0.008 plants·m^{-2}) treatment in the seventh and eighth yr after seeding (Fig. 3; $P = 0.047$) but did not differ between yr ($P = 0.656$). Other shrub density averaged < 0.05 individuals·m^{-2} and was similar between treatments (see Fig. 3; $P = 0.412$) and yr ($P = 0.197$). Sandberg bluegrass, perennial grass, and annual grass densities were similar between treatments (see Fig. 3; $P = 0.263$, 0.509, and 0.514, respectively). Sandberg bluegrass density was similar between yr ($P = 0.434$). Perennial grass density was greater in 2017 than 2016 ($P = 0.002$). Annual grass density was greater in 2016 than 2017 ($P = 0.028$). We did not find evidence that perennial forb density differed between treatments (see Fig. 3; $P = 0.051$) or yr ($P = 0.215$). Annual forb density was 2.5 times greater in the Herb compared with Sage + Herb treatment (see Fig. 3; $P = 0.030$) and greater in 2016 than 2017 ($P = 0.003$).

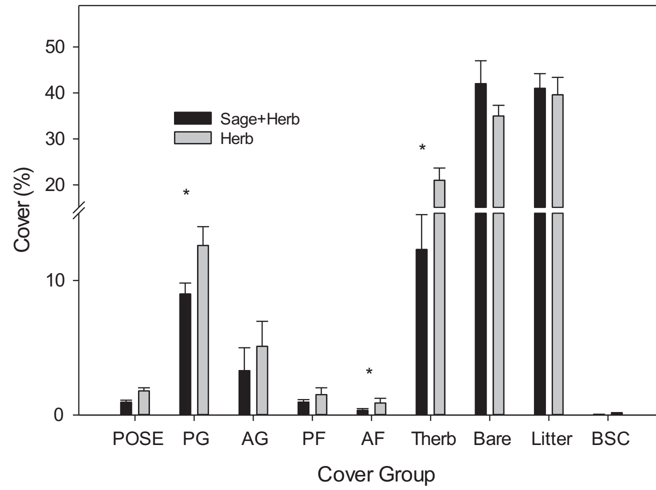

Figure 2. Group cover (mean + S.E.) in Sage + Herb and Herb treatments averaged from 2016 and 2017 in southeastern Oregon. POSE indicates Sandberg bluegrass; PG, large perennial bunchgrass; AG, annual grass; PF, perennial forb; AF, annual forb; Therb, total herbaceous; Bare, bare ground; Litter, ground litter; BSC, Biological soil crust. Asterisk (*) indicates significant difference ($P \le 0.05$) between treatments.

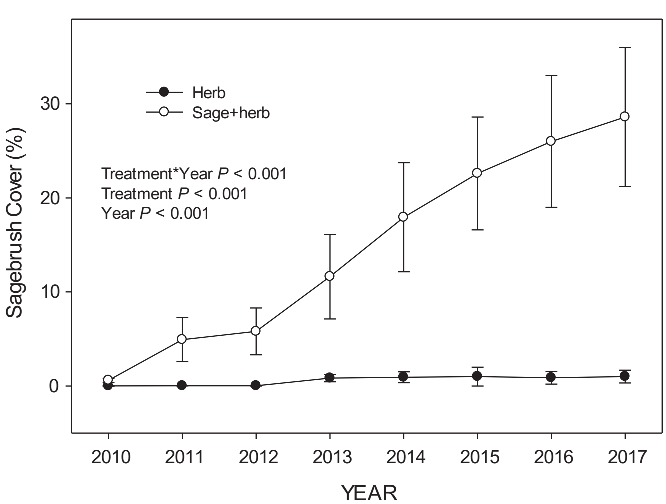

Figure 1. Sagebrush cover (mean ± S.E.) in the Herb and Sage + Herb treatments from 2010 to 2017 (first through the eighth yr post seeding) in southeastern Oregon.

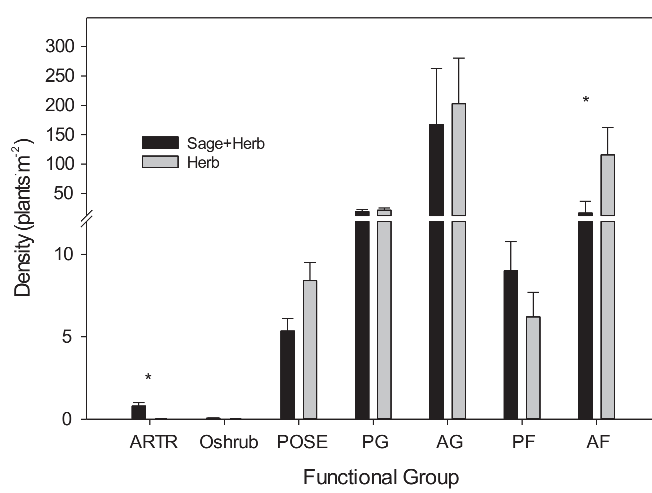

Figure 3. Functional group density (mean + S.E.) in Sage + Herb and Herb treatments averaged from 2016 and 2017 in southeastern Oregon. ARTR indicates mountain big sagebrush; Oshrub, Other shrubs; POSE, Sandberg bluegrass; PG, large perennial bunchgrass; AG, annual grass; PF, perennial forb; AF, annual forb. Asterisk (*) indicates significant difference ($P \le 0.05$) between treatments.

Discussion

Our results suggest that seeding shrubs after encroaching trees are controlled may be a viable strategy to restore shrubs. Our results also suggest this may provide long-term shrub dominance and highlights that there will likely be a trade-off with herbaceous vegetation. These results may be informative for shrub restoration in many ecosystems as trees are encroaching shrublands in North America (Tausch et al., 1981; Callaway and Davis, 1998; Miller and Rose, 1999), Australia (Rundel et al., 2014), Africa (Homes and Cowling, 1997; Rundel et al., 2014), and South America (Sarasola et al., 2006; Langdon et al., 2010). Results from this study supported our hypothesis that seeding sagebrush would accelerate its recovery as both sagebrush cover and density were substantially greater 8 yr post seeding in the sagebrush-seeded areas compared with areas not seeded. Results partially supported our hypothesis that seeding sagebrush would decrease herbaceous vegetation cover and density. Several herbaceous groups and total herbaceous cover were less with sagebrush seeding. Densities of herbaceous vegetation were similar between treatments, except annual forb density was less in sagebrush-seeded plots. Seeding sagebrush did not appear to influence other shrubs, but they were not abundant at the study sites and were largely species that resprout after fire.

Seeding sagebrush after juniper control in communities where juniper had largely excluded sagebrush appears to be critical to restoring sagebrush habitat in a timely manner. Eight yr after western juniper control, sagebrush cover averaged about 1% in unseeded areas. In contrast, sagebrush cover averaged ≈30% in sagebrush-seeded plots 8 yr after seeding. Restoring sagebrush communities is a priority (Davies et al., 2011) considering that sagebrush only occupies about 56% of its historical range (Knick et al., 2003; Schroeder et al., 2004). Our results suggest that seeding sagebrush should be incorporated into juniper control projects where sagebrush has largely been eliminated, and an accelerated return to sagebrush dominance is a management objective. Likely, these communities have a limited sagebrush seedbank because of a lack of seed input and a relatively short seed viability period. Big sagebrush seed remains viable for 6 months (Young and Evans, 1989) to 2 yr (Wijayratne and Pyke, 2012) under field conditions near the soil surface. Seeding sagebrush may also be critical for rapid recovery in these communities because sagebrush seed does not disperse long distances. Sagebrush seed dispersal is typically limited to a few meters from parent plants (Young and Evans, 1989). Thus, if the sagebrush seed bank is limited and seed sources are not in close proximity, it may take decades or longer for sagebrush recovery without seeding.

Recovery of sagebrush is essential for sagebrush-associated wildlife, including sage-grouse, a ground nesting bird, which is a conservation concern across the western United States and Canada (Crawford et al., 2004). Sage-grouse diet in the winter can be almost exclusively sagebrush leaves, and sagebrush communities are critical for every stage of their life cycle (Connelly et al., 2000; Crawford et al., 2004). Furthermore, the extirpation of local sage-grouse populations has been linked to the loss of sagebrush habitat (Aldridge et al., 2008). Our results suggest that by the fifth yr after seeding, average sagebrush cover (~18%) was sufficient for providing productive sage-grouse habitat, assuming other habitat requirements were met for all life stages based on habitat guidelines proposed by Connelly et al. (2000). Besides sage-grouse, > 350 sagebrush-associated species have been identified as species of conservation concern (Suring et al., 2005; Wisdom et al., 2005) that may benefit from sagebrush restoration. Mountain big sagebrush communities are also some of the most productive big sagebrush communities (Davies and Bates, 2010; Davies et al., 2011). Therefore, it would be quite valuable to restore sagebrush dominance to these communities for wildlife and other ecosystem services.

Similar to our current study, broadcast seeding mountain big sagebrush after control of juniper with prescribed fire was generally successful, though success was limited on south slopes (Davies and Bates, 2017). In contrast, the establishment of broadcast seeded mountain big sagebrush after fire was limited when herbaceous vegetation recovered before seeding sagebrush (Davies et al., 2017a). Our results likely differed from Davies et al. (2017a) because competition from herbaceous vegetation limited sagebrush establishment in their study, and in our current study many areas were largely devoid of herbaceous vegetation immediately after the burning when sagebrush was seeded. Sagebrush and other shrub establishment can be limited by competition from herbaceous vegetation (Schuman et al., 1998; Hall et al., 1999; Rinella et al., 2015, 2016).

Our results support the prediction by Davies et al. (2014) that seeding sagebrush after juniper control would decrease herbaceous vegetation as sagebrush cover increased over time. Seeding sagebrush reduced herbaceous vegetation cover likely through competition for limited resources as sagebrush competes with herbaceous vegetation for resources in this ecosystem (Robertson, 1947; Williams et al., 1991; Cook and Lewis, 1963). Increases in sagebrush generally decrease herbaceous vegetation (Cook and Lewis, 1963; Rittenhouse and Sneva, 1976; McDaniel et al., 2005), and reductions of sagebrush in sagebrush-dominated communities often result in several-fold increases in herbaceous vegetation (Mueggler and Blaisdell, 1958; Hedrick et al., 1966; Davies et al., 2007). We found no evidence that seeding sagebrush limited perennial herbaceous vegetation density, but we speculate that over time this may occur. Sagebrush appears to limit the recruitment of annual forbs, suggesting that it could also limit the recruitment of other herbaceous species.

Less herbaceous vegetation cover in areas seeded with mountain big sagebrush suggests lower forage production. Increases in herbaceous cover are often accompanied by an even greater increase in herbaceous biomass. For example, ~30% and 50% increases in total herbaceous and perennial grass cover equated to ~50% and 90% increases in total herbaceous and perennial grass biomass, respectively, when mountain big sagebrush stands were mechanically treated to reduce sagebrush (Davies et al., 2012a). Similar differences in magnitude of response to fire between herbaceous cover and biomass have also been reported in Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young) communities (Davies et al., 2007). Thus, it is logical to assume that a decrease in herbaceous vegetation cover likely translates to a decrease in herbaceous biomass with seeding sagebrush in the current study.

The effects of seeding mountain big sagebrush on herbaceous vegetation may have been reduced if sagebrush was seeded at a lower rate, resulting is less sagebrush establishment and cover. Sagebrush cover on our seeded plots was fairly high (~30%) by the end of the study, likely intensifying its effects on herbaceous vegetation. Mountain big sagebrush cover at our seeded plots was greater than the average of 23% cover for mountain big sagebrush communities across southeastern Oregon measured by Davies and Bates (2010). However, intact mountain big sagebrush communities at Hart Mountain in southeastern Oregon at similar elevations had approximately the same amount of sagebrush cover (Davies et al., 2012b). Varying seeding rates of mountain big sagebrush and potentially seeding patches should be evaluated to determine the best methods to achieve a variety of management objectives for habitat recovery and forage production.

Our results should not be applied to Wyoming big sagebrush communities. The establishment of mountain big sagebrush has been more successful than attempts to restore Wyoming big sagebrush. Wyoming big sagebrush has a high rate of establishment failure when broadcast seeded after disturbance (Lysne and Pellant, 2004; Davies et al., 2013), though there are exceptions (see Davies et al., 2018). Mountain big sagebrush seeding is likely more successful than seeding Wyoming big sagebrush because these communities are cooler, wetter, and more productive (Davies and Bates, 2010; Davies et al., 2011). Furthermore, after juniper control with fire there are many areas largely devoid of competing vegetation, in particular the former juniper canopy locations, with high resource availability (Bates and Davies, 2017;

Davies et al., 2017b) that likely aids establishment of mountain big sagebrush.

Implications

Seeding mountain big sagebrush after controlling encroaching western juniper with partial cutting and burning can accelerate the recovery of sagebrush. Seeding shrubs also have potential to be effective where other conifer species are invading sagebrush steppe and in other shrub ecosystems experiencing tree encroachment. In communities where sagebrush has been largely eliminated by juniper encroachment, seeding sagebrush likely restores sagebrush habitat several decades sooner than natural recovery. However, seeding sagebrush can reduce herbaceous vegetation cover, likely translating into reduced livestock forage production. Thus, there is a trade-off between habitat recovery and maximizing short-term forage production objectives with seeding mountain big sagebrush. However, considering the widespread decline of sagebrush-associated wildlife and the potential negative impact of a listing of one of these species under the Endangered Species Act of 1973 on livestock producers dependent on sagebrush communities for livestock forage, seeding sagebrush after juniper control may indirectly benefit livestock producers by limiting potential management restriction to public land grazing. This trade-off, however, will need to be evaluated case by case to ensure that multiple-management objectives will be met. Our results also suggest that prescribed burning of juniper in mountain big sagebrush communities should not be abandoned as a management practice because of concerns about the loss of sage-grouse and other sagebrush-associated wildlife habitat as sagebrush cover and density rapidly recover with seeding. We recommend that land managers and restoration practitioners consider seeding mountain big sagebrush after juniper control with fire where sagebrush habitat is limited.

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