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SETBACKS AND SURPRISES

Attempting to restore mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) four years after fire

Kirk W. Davies[1,2], Jon D. Bates[1], April Hulet[3]

Restoration of shrubs is needed throughout the world because of altered fire regimes, anthropogenic disturbance, and overutilization. The native shrub mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle) is a restoration priority because of its value to wildlife in western North America. One of the principal threats to mountain big sagebrush is encroachment by western juniper (*Juniperus occidentalis* ssp. *occidentalis* Hook) and other conifers. Fire is frequently applied to control juniper; however, sagebrush recovery after fire can be variable. Seeding sagebrush postfire can hasten sagebrush recovery; however, seeding is not always necessary. Therefore, it may be advantageous to monitor postfire recovery to determine if seeding is needed. The effect of seeding sagebrush several years after fire is unknown. We evaluated the efficiency of seeding mountain big sagebrush four years after fire-controlled junipers at five sites. Sagebrush cover (<0.5%) and density (<0.07 plants/m^2) was low in seeded plots and did not differ from unseeded controls in the three postseeding years. We conclude that seeding sagebrush four years after fire did not accelerate sagebrush recovery. We speculate that seeded sagebrush failed to establish because of competition from herbaceous vegetation that had four years to recover after fire. Although it would be beneficial to seed sagebrush only when needed, our results suggest postponing seeding until monitoring has determined that recovery is inadequate may not be advisable. We suggest researchers investigate methods to improve predicting sagebrush recovery to allow for seeding, when needed, before the first postfire growing season.

Key words: burning, conifer control, *Juniperus*, recovery, seeding, shrubs

Implications for Practice

- Waiting to seed shrubs until natural recovery is determined to be inadequate may not be a viable option as recovering herbaceous vegetation may limit seeded shrubs.
- Efforts to restore shrubs should probably occur prior to the first growing season after disturbance to improve likelihood of success.
- Competition for herbaceous vegetation may be a major barrier to successful shrub establishment and should be considered during restoration planning.
- Research should focus on developing methods to predict the need for shrub restoration immediately after disturbances.

Introduction

Restoration of sagebrush (*Artemisia* L.) and other shrubs around the world is needed because of mismanagement, overexploitation, and altered fire regimes (Han et al. 2008; Sasaki et al. 2008; Bedunah et al. 2010; Medina-Roldán et al. 2012; Linstadter & Baumann 2013). Shrub restoration is critical because many shrubs, including sagebrush, are keystone species that provide vital ecosystem services (Prevéy et al. 2010; Fonseca et al. 2012; van Zonneveld et al. 2012). Resources for restoration are limited; thus, it may be advantageous to wait several years after disturbances to determine if natural shrub recovery will occur. Then, if shrub recovery is inadequate, implement restoration efforts.

In western North America, mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle) is a restoration priority, largely due to widespread conifer (*Juniperus* L. and *Pinus* L. species) encroachment. Conifer encroachment of mountain big sagebrush communities coincided with European settlement (Miller & Wigand 1994; Miller & Rose 1995). Expansion and infilling of conifers has been attributed to historical overstocking of livestock, decreased fire frequency, increasing atmospheric CO_2, and favorable climatic conditions (Tausch et al. 1981; Miller & Wigand 1994; Knapp & Soulé 1998; Miller et al. 2005). The decrease in fire frequency has allowed conifers to expand from historically fire-safe areas into areas with historically shorter fire-return intervals (Miller & Wigand 1994; Gruell 1999; Miller & Rose 1999; Weisberg et al. 2007).

Author contributions: KWD, JDB, AH conceived and designed the study, and carried out the experiments; KWD analyzed the data and wrote the manuscript; KWD, JDB, AH edited the manuscript.

[1]USDA—Agricultural Research Service, Eastern Oregon Agricultural Research Center, 67826-A Highway 205, Burns, OR 97720, U.S.A.
[2]Address correspondence to K. W. Davies, email kirk.davies@oregonstate.edu
[3]University of Idaho, Department of Forest, Rangeland, and Fire Science, 875 Perimeter Drive MS 1133, Moscow, ID U.S.A.

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In the Columbia Plateau and northern Great Basin, western juniper (*Juniperus occidentalis* ssp. *occidentalis* Hook) is the prevailing conifer encroaching into sagebrush communities and has increased from 0.3 million to 3.5 million hectares since the 1870s (Miller et al. 2000). Western juniper expansion has primarily occurred in mountain big sagebrush and other productive plant communities (Burkhardt & Tisdale 1969; Miller & Rose 1995; Miller et al. 2005). As juniper cover increases diversity and forage production decrease, erosion and runoff potential increase and sagebrush is lost from the community (Miller et al. 2000; Bates et al. 2005; Pierson et al. 2007). Loss of sagebrush and more predator perches with juniper encroachment negatively impact sagebrush-associated wildlife, such as greater sage grouse (Connelly et al. 2000; Miller et al. 2005; Baruch-Mordo et al. 2013). Therefore, restoration of juniper-encroached sagebrush rangeland is a management priority (Miller et al. 2005; Pierson et al. 2007; Bates et al. 2011; Baruch-Mordo et al. 2013).

Western juniper can be effectively and cost-efficiently controlled with prescribed fire or partial cutting (felling one-fourth to half of mature juniper to increase surface fuels) followed by prescribed fire (Bates et al. 2011; Davies et al. 2014). Burning often results in more complete control of western juniper than mechanical treatments because fire kills more juniper seedlings and small juveniles and reduces the seed bank as well as generally being less expensive (Miller et al. 2005). Juniper control with fire is also predicted to maintain sagebrush dominance longer than mechanical treatments (Boyd et al. 2017). However, mountain big sagebrush recovery after burning western juniper can be variable. Estimates of mountain big sagebrush recovery after fire range from 15 to 100 years (Baker 2006; Ziegenhagen & Miller 2009; Nelson et al. 2014) and may be especially slow if juniper encroachment has significantly reduced sagebrush prior to burning (Bates et al. 2005; Davies et al. 2014). Expediting sagebrush recovery may be needed because it is a critical habitat element for sagebrush-associated species (Crawford et al. 2004; Shipley et al. 2006; Aldridge et al. 2008) that are of conservation concern because of the loss of sagebrush habitat (Suring et al. 2005).

Seeding mountain big sagebrush after using prescribed fire to control juniper can greatly accelerate sagebrush recovery (Davies et al. 2014; Davies & Bates 2017). Sagebrush cover can be up to 12% in three years on seeded sites, while cover on unseeded (natural recovery) sites was less than 0.5% (Davies et al. 2014). Fairly rapid natural (unseeded) recovery of sagebrush, however, can occur on some sites, eliminating the need for costly seeding (Davies & Bates 2017). Variability in natural recovery of sagebrush can be caused by climate the first few years after fire (Ziegenhagen & Miller 2009; Nelson et al. 2014) or the availability of viable sagebrush seed, particularly in juniper-encroached communities (Bates et al. 2014; Davies et al. 2014). Clearly, it would be more cost effective to only seed mountain big sagebrush when natural recovery will be inadequate to meet management goals. One method to accomplish this task may be to assess natural sagebrush recovery several years after fire to determine if seeding sagebrush is needed to achieve sagebrush cover and density objectives. However, the outcome of seeding mountain big sagebrush several years after juniper control with fire is unknown. Seeding mountain big sagebrush after controlling juniper with prescribed burning has received limited attention and these studies (Davies et al. 2014; Davies and Bates 2017) only evaluated only fall seeding in the same year as the fire.

The purpose of this study was to investigate if seeding shrubs could be postponed until after natural recovery was determined to be inadequate. In this study, we evaluated seeding mountain big sagebrush four years after prescribed fire was used to control encroaching western juniper on sites where sagebrush recovery was limited. We hypothesized that seeding mountain big sagebrush would accelerate sagebrush recovery (greater sagebrush cover and density) compared with natural (unseeded) recovery.

Methods

Study Area

Study sites were located on Steens Mountain approximately 80 km southeast of Burns, OR, U.S.A. Study sites were mountain big sagebrush plant communities encroached by western juniper prior to burning. Juniper woodland development prior to treatment was classified as Phase III (juniper dominated) based on criteria in Miller et al. (2005) and the understory was dominated by native perennial bunchgrasses and forbs and sagebrush had largely been lost. Common perennial grasses included Idaho fescue (*Festuca idahoensis* Elmer), bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) A. Löve), and Sandberg bluegrass (*Poa secunda* J. Presl). Elevation of study sites ranged from 1,708 to 1,863 m above sea level, slopes were between 5 and 35%, and aspects were north and west. The 30-year (1981–2010) average annual precipitation was 447 mm (PRISM 2016). Annual precipitation was 83 and 85% of the 30-year long-term average in 2014 and 2015 and 81% of long-term average in 2016 for the 5 months prior to sampling (PRISM 2016).

Experimental Design

A randomized complete block design with five blocks (study sites) was used to evaluate seeding mountain big sagebrush four years after prescribed fire-controlled western juniper. Each study site consisted of two 50 × 100-m plots with a 2-m buffer between plots. Treatments were an unseeded control and seeded with mountain big sagebrush. In 2008, 50% of mature juniper trees were felled with chainsaws to provide sufficient ground fuel to carry prescribed fire across study sites. Each study site was prescribed burned with a head-fire ignited with drip torches between 15 and 25 September, 2009. Fires resulted in 100% mortality of western juniper and nonsprouting shrubs. Burned sites were allowed to naturally recover (i.e. not seeded) after fires for the next four years. Sagebrush was broadcast seeded with a handheld seeder at 500 PLS/m^2 in November of 2013 (four years after burning) in the seeded plots. Mountain big sagebrush seed were locally collected (within 75 km of study sites) at similar elevations. Percent live seed were estimated using the Petri dish germination method (Meyer & Monsen 1991).

Vegetation characteristics were measured in late June and early July of 2014, 2015, and 2016 along three parallel 50-m transects spaced 10 m apart in the center of each plot. Foliar cover of herbaceous vegetation by species, bare ground, and litter were estimated in 0.2-m^2 quadrats placed at 3-m intervals along each 50-m transect (starting at 3 m and ending at 45 m). Cover estimates were aided by markings segmenting quadrats into 1, 5, 10, 25, and 50%. Herbaceous vegetation density was measured by species by counting all plants rooted in the 0.2-m^2 quadrats. Shrub canopy cover was estimated by species using the line-intercept method (Canfield 1941) on each 50-m transect. Shrub density was measured by species by counting each shrub rooted inside a 2 × 50-m belt transect positioned over each 50-m transect.

Statistical Analyses

Repeated measures analysis of variances (ANOVAs) using the PROC MIXED procedure in SAS 9.4 (PROC MIXED SAS Institute, Inc., Cary, NC, U.S.A.) was used to determine the effects of seeding mountain big sagebrush on plant community characteristics. Year was the repeated variable, and treatment was considered a fixed variable in models. Treatment, year, block, and treatment × year were used as explanatory variables in models. Compound symmetry covariance structure was selected based on Akaike's information criterion (Littell et al. 1996). Data that violated assumptions of ANOVAs were log transformed to better meet assumptions. Original (i.e. nontransformed) data were presented in the text and figures. Herbaceous vegetation was separated into five functional groups for analyses: Sandberg bluegrass, perennial grasses, exotic annual grasses, perennial forbs, and annual forbs. Sandberg bluegrass was treated as an individual functional group because it matures earlier and differs in its response to disturbances compared with other native perennial grasses in the sagebrush ecosystem (McLean & Tisdale 1972; Yensen et al. 1992). The exotic annual grass group was primarily comprised of cheatgrass (*Bromus tectorum* L.). Total herbaceous vegetation cover was the summation of herbaceous functional groups. Shrubs were separated into sagebrush and other shrubs for analyses. The other shrub group was comprised of species that re-sprout after fire. Treatment means were considered different at $\alpha = 0.05$ and were reported with standard errors (SE).

Results

Sandberg bluegrass, perennial grass, and exotic annual grass cover did not vary between treatments (Fig. 1; $F_{[1,5]} = 1.27$, 3.44, and 0.52, $p = 0.311$, 0.123, and 0.505). Sandberg bluegrass and perennial grass cover varied by year ($F_{[2,20]} = 4.24$ and 3.62, $p = 0.029$ and 0.045), but no trend was apparent. Exotic annual grass cover did not vary by year ($F_{[2,20]} = 2.51$, $p = 0.106$). Perennial forb, annual forb, and total herbaceous cover did not differ between seeded and unseeded control plots (Fig. 1; $F_{[1,5]} = 0.41$, 0.97, and 0.00, $p = 0.552$, 0.370, and 0.988). Perennial and annual forb cover generally decreased over time ($F_{[2,20]} = 7.48$ and 6.17, $p = 0.004$ and 0.008). Total herbaceous cover was similar among years ($F_{[2,20]} = 0.58$, $p = 0.568$). Bare ground and litter cover were similar between treatments (Fig. 1; $F_{[1,5]} = 0.73$ and 0.56, $p = 0.431$ and 0.487), but varied by year ($F_{[2,20]} = 19.84$ and 79.50, $p < 0.001$). In the last sampling year, bare ground was 53–68% and litter was 150–270% of prior years. The interaction between treatment and year was not significant for any measured cover variable ($p > 0.05$).

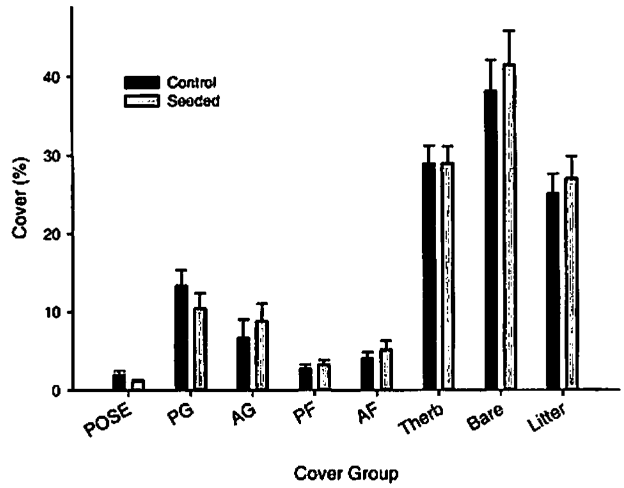

Figure 1. Cover (mean + SE) of cover groups in the control and seeded treatments summarized for the 3-year sample period. POSE, Sandberg bluegrass; PG, perennial grasses; AG, exotic annual grasses; PF, perennial forbs; AF, annual forbs; Therb, total herbaceous; Bare, bare ground; and Litter = ground litter. No significant differences between treatments were detected ($p > 0.05$).

Density of Sandberg bluegrass, perennial grass, and exotic annual grass did not differ between seeded and control plots (Fig. 2; $F_{[1,5]} = 0.64$, 0.40, and 1.33, $p = 0.48$, 0.554, and 0.301). Perennial grass density varied by year ($F_{[2,20]} = 5.92$, $p = 0.010$), but no apparent trend was evident. Exotic annual grass density generally increased over time ($F_{[2,20]} = 5.84$, $p = 0.010$). Perennial forb and annual forb density were similar between treatments (Fig. 2; $F_{[1,5]} = 2.58$ and 1.44, $p = 0.169$ and 0.285). Perennial forb density was generally greater in the second compared with the first and third sampling years ($F_{[2,20]} = 4.42$, $p = 0.026$). Annual forb density did not differ among years ($F_{[2,20]} = 2.54$, $p = 0.112$). The interaction between treatment and year was not significant for any measured density variable ($p > 0.05$).

Sagebrush and other shrub cover did not differ between seeded and unseeded control plots (Fig. 3A; $F_{[1,5]} = 0.58$ and 3.14, $p = 0.479$ and 0.137). Sagebrush cover did not vary among years ($F_{[2,20]} = 1.20$, $p = 0.321$). Other shrub cover increased approximately 280% from the first through the third sampling year ($F_{[2,20]} = 4.73$, $p = 0.021$) and in the third year ranged from 0 to 9% among sites. Sagebrush and other shrub density did not differ between treatments (Fig. 3B; $F_{[1,5]} = 0.44$

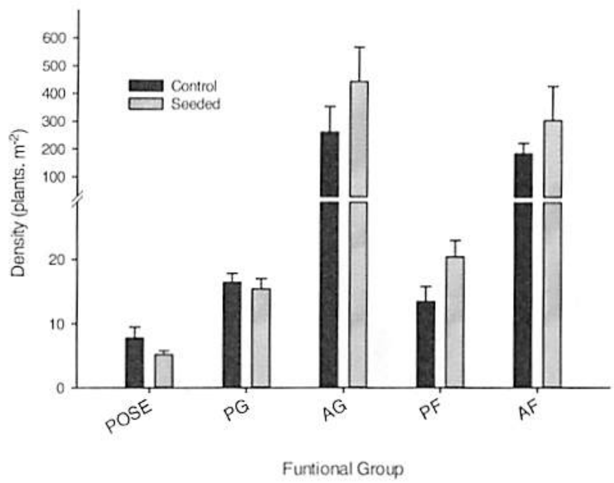

Figure 2. Density (mean + SE) of functional groups in the control and seeded treatments summarized for the 3-year sample period. POSE, Sandberg bluegrass; PG, perennial grasses; AG, exotic annual grasses; PF, perennial forbs; and AF, annual forbs. No significant differences between treatments were detected ($p > 0.05$).

and 0.01, $p = 0.538$ and 0.937) and did not vary among years ($F_{[2,20]} = 1.07$ and 1.42, $p = 0.361$ and 0.266). Sagebrush and other shrub cover and density was not influenced by the interaction between treatment and year ($p > 0.05$).

Discussion

Shrub recovery after fire can be highly variable (Ziegenhagen & Miller 2009; Nelson et al. 2014; Davies & Bates 2017) and, therefore, it may be advantageous to wait several years after fire to determine if shrub restoration is needed prior to expending limited resources on restoration. However, our results suggest that this may not be a viable restoration strategy as seeding mountain big sagebrush four years after prescribed fire-controlled encroaching junipers did not increase sagebrush cover or density. This was unexpected as other studies (Davies et al. 2014; Davies & Bates 2017) on similar sites generally found mountain big sagebrush seeded in the fall after prescribed fire had high establishment and rapid growth. However, establishment of mountain big sagebrush seeded on south slopes was limited (Davies & Bates 2017). Aspects in the current study were north and west, and thus, at the very least, we expected good establishment and growth on north aspects. However, sagebrush density (<0.07 plants/m^2) and cover (<0.05%) were low in seeded areas and did not differ from unseeded areas.

A common assumption when seeded native vegetation fails to establish is that precipitation was inadequate (James et al. 2011). However, it is not necessarily the causal factor for seeding failure in this situation. Annual precipitation was slightly below average in the first 2 years after seeding, but Davies and Bates (2017) reported successful establishment of seeded mountain big sagebrush (>0.6 plants/m^2) on north aspects with similar annual precipitation in the first 2 years postseeding. Precipitation clearly influences seeding success (Hardegree et al. 2011); however, near-average precipitation in high elevation, cool, moist mountain big sagebrush communities is unlikely to be the major factor limiting sagebrush seedling establishment.

We speculate that competition from herbaceous vegetation limited sagebrush establishment, especially because herbaceous vegetation had four years to recover and increase after fire-controlled juniper. Compared with successful seeding of mountain big sagebrush (Davies et al. 2014; Davies & Bates

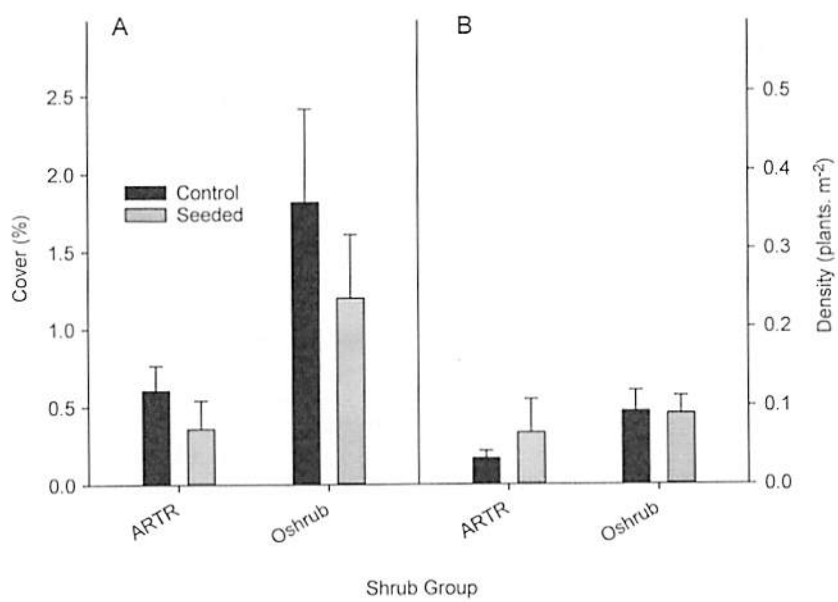

Figure 3. (A) Cover and (B) density (mean + SE) of shrub groups in the control and seeded treatments summarized for the 3-year sample period. ARTR, sagebrush and Oshrub, other shrubs. No significant differences between treatments were detected ($p > 0.05$).

2017), our current study had approximately 3.5-fold more cover and 2- to 27-fold greater abundance of perennial grasses in the first growing season after seeding sagebrush. Competition from perennial grasses and other vegetation has limited the establishment of seeded shrubs in other restoration efforts (Allen 1988; Schuman et al. 1998; Hall et al. 1999; Rinella et al. 2015, 2016). Competition from perennial grasses, in particular, may be a widespread constraint to shrub establishment in arid and semi-arid lands (Rinella et al. 2015). For example, grass competition has been established to decrease the growth and survival of several woody species (Midoko-Iponga et al. 2005; DeFalco et al. 2007; Dick et al. 2016). Reductions in herbaceous competition can accelerate shrubland recovery (Midoko-Iponga et al. 2005). Differences in herbaceous vegetation at the time of seeding, that is, greater competition in the current study, likely explains the contrasting results of our current study compared with prior studies.

Our results suggest that it may not be the best restoration strategy to postpone seeding sagebrush and potentially other shrubs until their recovery can be determined because postfire increases in herbaceous vegetation may limit establishment of seeded shrubs. In shrub ecosystems, this will potentially lead to unnecessary seeding of shrubs in some situations; however, the alternative of not being able to establish shrubs when needed is even less desirable. Improvements in predicting postfire recovery, however, would improve the efficiency of seeding. Different climatic conditions than experienced during our study may also allow successful establishment of sagebrush seeded several years after fire, but needs further investigation. We expect similar result with other shrub species, because competition from herbaceous vegetation can limit their establishment (Hall et al. 1999; Midoko-Iponga et al. 2005; Rinella et al. 2015). This suggests that restoration practitioners should consider and potentially mediate the effects of competition from herbaceous vegetation during shrub restoration. This may be particularly important if shrub restoration is delayed because of logistical or financial constraints.

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