Environmental Management (2014) 53:1015–1022
DOI 10.1007/s00267-014-0255-5

#799

Restoration of Mountain Big Sagebrush Steppe Following Prescribed Burning to Control Western Juniper

K. W. Davies · J. D. Bates · M. D. Madsen · A. M. Nafus

Received: 16 September 2013 / Accepted: 13 February 2014 / Published online: 27 February 2014

Abstract Western juniper (*Juniperus occidentalis* ssp. *occidentalis* Hook) encroachment into mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana* (Rydb.) Beetle) steppe has reduced livestock forage production, increased erosion risk, and degraded sagebrush-associated wildlife habitat. Western juniper has been successfully controlled with partial cutting followed by prescribed burning the next fall, but the herbaceous understory and sagebrush may be slow to recover. We evaluated the effectiveness of seeding perennial herbaceous vegetation and sagebrush at five sites where juniper was controlled by partially cutting and prescribed burning. Treatments tested at each site included an unseeded control, herbaceous seed mix (aerially seeded), and the herbaceous seed mix plus sagebrush seed. In the third year post-treatment, perennial grass cover and density were twice as high in plots receiving the herbaceous seed mix compared to the control plots. Sagebrush cover and density in the sagebrush seeded plots were between 74- and 290-fold and 62- and 155-fold greater than the other treatments. By the third year after treatment, sagebrush cover was as high as 12 % in the sagebrush seeded plots and between 0 % and 0.4 % where it was not seeded. These results indicate that aerial seeding perennial herbaceous vegetation can accelerate the recovery of perennial grasses which likely stabilize the site. Our results also suggest that seeding mountain big sagebrush after prescribed burning encroaching juniper can rapidly recover sagebrush cover and density. In areas where sagebrush habitat is limited, seeding sagebrush after juniper control may increase sagebrush habitat and decrease the risks to sagebrush-associated species.

Keywords Aerial seeding · *Artemisia tridentata* · Fire · Habitat · Recovery · Sage-grouse

K. W. Davies (✉) · J. D. Bates · M. D. Madsen
USDA – Agricultural Research Service, 67826-A Hwy 205, Burns, OR 97720, USA
e-mail: kirk.davies@oregonstate.edu

A. M. Nafus
Oregon State University, Burns, OR, USA

Introduction

The sagebrush ecosystem has been identified as one of the most imperiled in the United States (Noss and others 1995) and is currently being fragmented and degraded at an alarming rate by multiple stressors (Davies and others 2011). The loss of sagebrush habitat has resulted in more than 350 sagebrush-associated plants and animals being identified as species of conservation concern (Suring and others 2005; Wisdom and others 2005). Some sagebrush obligate species have experienced severe declines in their populations and range. Sage-grouse (*Centrocercus urophasianus*) range has been reduced to about one-half its original distribution (Schroeder and others 2004) and this species has experienced range-wide decline for the past 60 years (Connelly and Braun 1997; Braun 1998; Connelly and others 2004). The decline in sage-grouse has largely been attributed to the loss of sagebrush habitat (Aldridge and others 2008).

Mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana* (Rydb.) Beetle) plant communities, one of the most productive sagebrush communities, are being encroached by juniper (*Juniperus* L.) and/or piñon pine species (*Pinus* L.). In the northern Great Basin and Columbia Plateau, western juniper (*Juniperus occidentalis* ssp. *occidentalis* Hook) has increased from 0.3 million ha to 3.5 million ha since the 1870s (Miller and others 2000). Most of the expansion of western juniper has been into mountain big sagebrush communities (Miller and others 2000, 2005; Johnson and Miller 2006). Historically, western juniper was restricted to fire-safe sites (Miller and Rose 1995; Miller and Tausch 2001); however, with European settlement, fire return intervals have been lengthened, allowing encroachment into more productive plant communities (Burkhardt and Tisdale 1969; Miller and Rose 1995). As western juniper cover increases, sagebrush is lost, herbaceous production and diversity decreases, and runoff and erosion potential increases (Miller and others 2000; Bates and others 2005; Pierson and others 2007). Western juniper encroachment is detrimental to sagebrush obligate wildlife species because of the loss of sagebrush, decreases in herbaceous vegetation, and increased predation risk (Connelly and others 2000; Miller and others 2005).

Western juniper encroachment in mountain big sagebrush plant communities has been categorized into three developmental phases (Miller and others 2005). Phase I is characterized by sagebrush being the dominant over-story species with few juniper trees present. Phase II occurs when western juniper and sagebrush co-dominate the over-story. Phase III occurs when western juniper dominates the over-story, sagebrush is largely lost from the community, and herbaceous production and diversity decreases. A common treatment used in Phase II and Phase III has been to cut 25 to 50 % of the mature trees to create a more continuous fuel bed and then to prescribe burn the target area the next fall when the cut juniper is dry (i.e., partial cutting and prescribed burning). This generally results in more complete control of western juniper than cutting alone, because cutting without burning often fails to control juniper seedlings and small juveniles (Miller and others 2005). Bates and others (2011) reported that partial cutting and burning successfully controlled western juniper and herbaceous vegetation increased after the burn. After controlling juniper with prescribed burning, seeding may be needed to accelerate recovery (Sheley and Bates 2008). However, information regarding the effects of seeding is limited. In Idaho, using a small plot study on one ecological site, Sheley and Bates (2008) reported that seeding perennial herbaceous vegetation after partial cutting and burning juniper may accelerate recovery and prevent exotic plant invasions. Thus, further testing of seeding herbaceous vegetation after juniper control at multiple ecological sites is warranted.

Sagebrush often recovers after western juniper and other conifer control (Barney and Frischknecht 1974; Tausch and Tueller 1977; Skousen and others 1989), but the rate of recovery can be slow when sagebrush densities are low prior to treatment (Bates and others 2005). Baker (2006) estimated that natural recovery of mountain big sagebrush after fire takes on average 35–100 years. With the current wide-spread loss of sagebrush habitat and associated precipitous decline in sagebrush obligate wildlife species (Connelly and others 2000; Crawford and others 2004; Knick and others 2003; Davies and others 2011), waiting several decades or more for natural sagebrush recovery after controlling juniper with prescribed fire may not always be practical. Sagebrush recovery is often slow because sagebrush seeds remain viable for only a year or two (Young and Evans 1989; Wijayratne and Pyke 2009) and only disperses a few meters from parent plant (Young and Evans 1989). In late Phase II and Phase III juniper woodlands, sagebrush recovery may be even slower after fire because the sagebrush seed bank may be depleted as a result of the severe decline in sagebrush with increasing juniper cover. Seeding mountain big sagebrush after prescribed burning may accelerate sagebrush recovery; however, it has not been evaluated. Many landscapes encroached by western juniper are too rugged to use ground-based seeding equipment; thus, seeding operations associated with juniper control projects are often relegated to aerial broadcast methods. Aerial broadcast seeding is generally considered one of the least successful methods for revegetating sagebrush rangelands, but may be successful in mountain big sagebrush communities because they generally occur in cooler and less arid environments than many other sagebrush communities.

The purpose of this research project was to determine if the recovery of mountain big sagebrush plant communities after juniper control could be expedited by aerial seeding perennial herbaceous vegetation and mountain big sagebrush. We hypothesized that these treatments would increase the cover and density of seeded species. We speculated that natural recovery of sagebrush in late Phase II and Phase III juniper woodlands is constrained by limited sagebrush seed in the seed bank and thus, seeding sagebrush would significantly accelerate sagebrush recovery.

Materials and Methods

Study Area

The study was conducted on Steens Mountain in southeastern Oregon approximately 80 km southeast of Burns, OR (lat 42° 33′ 36″N, long 118° 19′ 12″ W). All study sites would have been mountain big sagebrush-bunchgrass communities prior to western juniper encroachment (NRCS 2012). Prior to prescribed burning, the plant communities were dominated by western juniper with an understory of perennial grasses and forbs. Mountain big sagebrush had been largely excluded, as evident by sagebrush skeletons, by juniper encroachment. Juniper encroachment at the study sites was in late Phase II and Phase III (Miller and others 2005). The study sites included Loamy 12-16 PZ, North Slope 12-16 PZ, and Droughty Loam 11-13 PZ Ecological Sites (NRCS 2012). Elevation among study sites ranged from 1,746 to 1,808 m above sea level. Climate across the study area is typical of the northern Great Basin with cool wet winters and dry hot summers. Crop year precipitation (Oct. 1–Sept. 30) was 99, 100, 150, and 78 % of the long-term average in 2009, 2010, 2011, and 2012 at the Burns, OR (Western Regional Climate Center 2012). Livestock grazing was excluded 1 year prior to treatments and for the duration of the study. Wildlife had unrestricted access to the study plots, but little evidence of ungulate and other wildlife use was detected.

Experimental Design

A randomized complete block design with five blocks was used to evaluate seeding sagebrush and perennial herbaceous vegetation after western juniper control with partial cutting and prescribed fire. The year prior to prescribed burning, 50 % of the mature junipers were cut (felled with chainsaws) to provide sufficient ground fuel to carry a prescribed fire across the blocks. The blocks were then prescribed burned between the 15th and 25th of September 2009, resulting in 100 % mortality of remaining juniper trees. Soil, aspect, topography, and vegetation characteristics varied among blocks. Slope was 5–10 % with aspect ranging from southwest to northeast among blocks. Blocks were 0.5–3 km from each other. Treatments were: unseeded control (CONTROL), seeded with perennial herbaceous vegetation (SEED), and seeded with perennial herbaceous vegetation and mountain big sagebrush (SEED + SAGE). At each block, the three treatments were randomly assigned to one of three 15 × 30 m plots with 2 m buffers between them. Perennial herbaceous vegetation was aerially seeded the first week of November 2009 using a fixed wing aircraft. Plastic tarps were laid over the control plots just 1 day prior to aerial seeding to keep them from being seeded. Tarps were removed within 2 days after aerial seeding was completed. Sagebrush was broadcast seeded with a non-automated centrifugal flinger fertilizer spreader (hand-cranked broadcaster) to simulate aerial seeding immediately after herbaceous seeding. The perennial herbaceous seed mix consisted of 1.6 kg ha^{-1} Idaho fescue (*Festuca idahoensis* Elmer), 3.0 kg ha^{-1} Sherman big bluegrass (*Poa ampla* Merr.), 1.3 kg ha^{-1} Oahe intermediate wheatgrass (*Thinopyrum intermedium* (Host) Barkworth & D.R. Dewey), 3.6 kg ha^{-1} Manchar smooth brome (*Bromus inermis* Leyss.), 2.2 kg ha^{-1} Paiute orchardgrass (*Dactylis glomerata* L.), 0.6 kg ha^{-1} Maple Grove Lewis flax (*Linum lewisii* Pursh), and 0.3 kg ha^{-1} Ladak alfalfa (*Medicago sativa* L.). This seed mix, consisting of native and introduced species, is the seed mix that the Bureau of Land Management uses after partial cutting and prescribed burning juniper-encroached mountain big sagebrush communities in this region. Mountain big sagebrush was seeded at 1.8 kg PLS ha^{-1}.

Measurements

Vegetation cover and density was measured in July of the first, second, and third years (2010, 2011, and 2012) after seeding. Five, 25 m transects spaced at 2 m intervals were used to sample vegetation. Herbaceous cover and density by species, bare ground, and litter were measured in 60, 0.2 m^2 quadrats. Cover was visually estimated to the nearest 1 % and density was determined by counting plants rooted inside the 0.2 m^2 quadrats. Density of rhizomatous species was estimated by dividing the quadrats into quarters and counting the quarters that contained the rhizomatous species. The quadrats were located at 2 m intervals along the 25 m transects (12 quadrats per transect). Sagebrush cover was measured using the line-intercept method (Canfield 1941) along the five 25 m transects. Sagebrush density was measure by overlaying each of the five, 25 m transects with a 2 × 25-m belt transect. All sagebrush plants rooted in the belt transects were counted.

Statistical Analyses

Repeated measures analysis of variance (ANOVA) using the mixed models procedure (Proc Mix) in SAS v. 9.2 (SAS Institute Inc., Cary, NC) were used to determine the influence of aerial seeding herbaceous vegetation and broadcast seeding sagebrush on response variables. Fixed variables were treatments, year, and their interactions. Site and site by treatment interactions were considered random effects. Covariance structure was determined using Akaike's Information Criterion (Littell and others 1996). When a significant treatment effect or treatment by year effect was found, data were also analyzed within each year using ANOVA. Fisher's LSD was used to separate means. Data that violated assumptions of normality were log-transformed prior to analyses. All data presented graphically are in their original dimensions (i.e. non-transformed). Significance level for all tests was set at $P \leq 0.05$. 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, exotic annual grasses (cheatgrass (*Bromus tectorum* L.) was the only annual grass detected), perennial forbs, and annual forbs. Sherman big bluegrass, though sometimes classified as a variety of Sandberg bluegrass, was considered a large

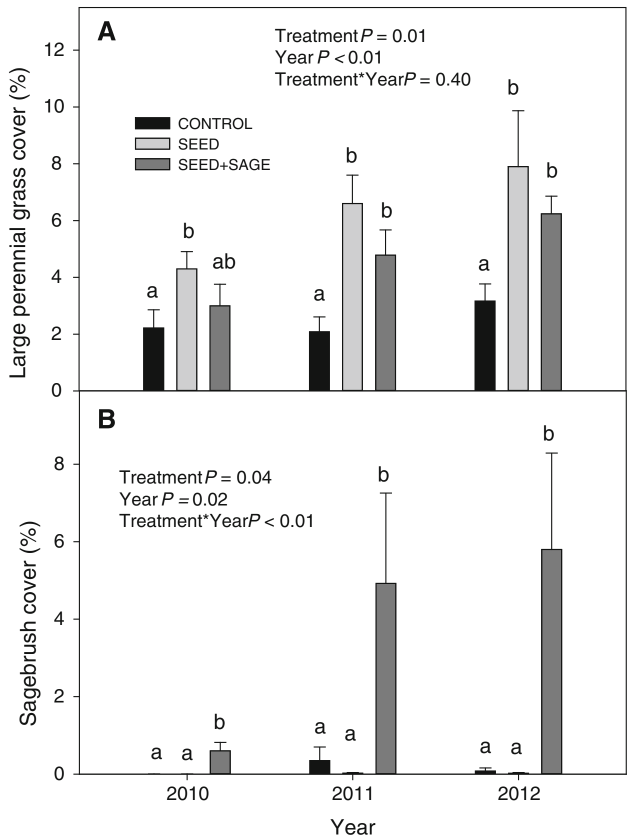

Fig. 1 Large perennial grass **a** and sagebrush **b** cover after partial cutting and prescribed burning western juniper encroached mountain big sagebrush communities that were not seeded (CONTROL), seeded with a herbaceous seed mix (SEED), or seeded with a herbaceous seed mix plus sagebrush (SEED + SAGE). Plots were seeded in the fall of 2009 and monitored for 3 years (2010, 2011, and 2012). *Different letters* indicate a significant difference ($P \leq 0.05$) between treatments in that year

perennial grass in the analyses because it is larger and matures later than the common Sandberg bluegrass in this ecosystem.

Results

Cover

Large perennial grass cover was greater in the SEED + SAGE and SEED treatments compared to CONTROL treatment (Fig. 1a; $P = 0.03$ and <0.01, respectively). By the third year post-seeding, large perennial grass cover was 2.0 and 2.5-fold greater in the SEED + SAGE and SEED treatments compared to the CONTROL treatment, respectively. We did not find evidence that large perennial grass cover differed between the SEED + SAGE and SEED treatments ($P = 0.22$). Large perennial grass cover increased with time since treatment ($P < 0.01$). Sagebrush cover increased in the SEED + SAGE treatment over time, but remained relatively static in the SEED and CONTROL treatments (Fig. 1b; $P < 0.01$). The SEED + SAGE treatment had greater sagebrush cover than the SEED and CONTROL treatments ($P = 0.03$ and 0.02, respectively). In 2012, sagebrush cover was 74- and 290-fold greater in the SEED + SAGE treatment compared to the CONTROL and SEED treatments, respectively. In 2012, sagebrush cover among sites where it was seeded ranged from a low of 1 % to a high of 12 %. Most plots where sagebrush was not seeded did not have any sagebrush cover and the unseeded plot with highest sagebrush cover had only 0.4 %. Sandberg bluegrass and perennial forb cover did not vary by treatment, year, or their interaction ($P > 0.05$). Perennial forb cover was relatively high across all treatment plots with 9.2 ± 3.2, 8.8 ± 2.3, and 7.7 ± 1.7 % in the CONTROL, SEED, and SAGE + SEED plots in 2012, respectively. Annual grass, annual forb, litter and total herbaceous cover varied by year ($P < 0.01$), with no apparent trend. Annual grass, annual forb, litter and total herbaceous cover did not vary by treatment or the interaction between treatment and year ($P > 0.05$). Bare ground did not vary by treatment ($P = 0.15$) or the interaction between treatment and year ($P = 0.99$). Bare ground decreased in all treatments from the first (57 %) to the second (42 %) and third (42 %) years after burning ($P < 0.01$).

Density

The SEED + SAGE and SEED treatments had 1.7- and 2.2-fold greater large perennial grass density compared to the CONTROL treatment (Fig. 2a; $P = 0.02$ and < 0.01, respectively). We did not find evidence that large perennial grass density varied between the SEED + SAGE and SEED treatments ($P = 0.09$). Large perennial grass density did not vary by year or the interaction of treatment and year ($P = 0.98$ and 0.93, respectively). Sagebrush density varied by the interaction between treatment and year (Fig. 2b; $P < 0.01$). Sagebrush density increased in the SEED + SAGE treatment almost 10-fold between the first and second year after seeding, but remained relatively unchanged in the SEED and CONTROL treatments. In 2012, sagebrush density was 62- and 155-fold greater in the SEED + SAGE treatment compared to the SEED and CONTROL treatments. In the third year after seeding, sagebrush density in the SEED + SAGE treatment ranged from 0.11 to 2.97 plants·m^{-2} depending on block. In the plots not seeded with sagebrush, sagebrush was not detected at three of the blocks, and on the remaining two blocks densities were low (<0.06 plants·m^{-2}). Sandberg bluegrass and perennial forb density did not vary by treatment, year, or their interaction ($P > 0.05$). Annual

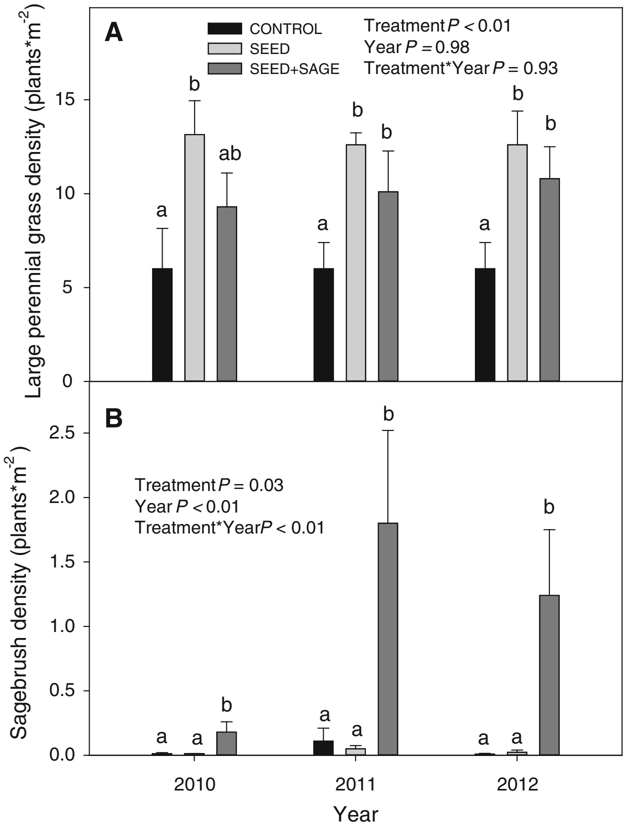

Fig. 2 Large perennial grass **a** and sagebrush **b** density after partial cutting and prescribed burning western juniper encroached mountain big sagebrush communities that were not seeded (CONTROL), seeded with a herbaceous seed mix (SEED), or seeded with a herbaceous seed mix plus sagebrush (SEED + SAGE). Plots were seeded in the fall of 2009 and monitored for 3 years (2010, 2011, and 2012). *Different letters* indicate a significant difference ($P \leq 0.05$) between treatments in that year

grass and annual forb density increased with time since burning ($P < 0.01$), but did not vary by treatment or by the interaction between treatment and year ($P > 0.05$).

Discussion

We achieve our objective of evaluating if the recovery of mountain big sagebrush plant communities after partial cutting and prescribed burning encroaching juniper could be expedited by aerial seeding perennial herbaceous vegetation and mountain big sagebrush. Aerial seeding after juniper control accelerated herbaceous vegetation recovery by approximately doubling large perennial grass cover and density. Rapid recovery of this functional group likely stabilizes the plant community as established perennial grasses can greatly limit exotic plant invasion (Clausnitzer and others 1999; Davies 2008) and decrease erosion potential after juniper control (Pierson and others 2007). Although not all the seeded perennial grasses are native, we consider the large increase in perennial grasses as evidence that seeding promoted faster recovery of the herbaceous understory because perennial grasses are the dominant herbaceous functional group in mountain big sagebrush steppe communities (Davies and Bates 2010). Similar to our results, Sheley and Bates (2008) also reported broadcast seeding perennial grasses after partial cutting and prescribed burning juniper increased perennial grass cover and density. Our results suggest that natural recovery of the herbaceous understory will probably occur, but may be slow. We did not see any change in perennial grass or forb density in the CONTROL treatment over the course of the study, thus we cannot estimate when perennial herbaceous vegetation density would naturally recover. When controlling juniper with partial cutting and prescribed burning, Bates and others (2011) estimated that natural recovery of perennial grass cover and density would take 6–8 years.

The lack of a perennial forb response with seeding was probably the result of limited establishment of seeded species. No alfalfa was detected in any of the treated plots and only a few Lewis flax plants were found in each plot. There were some places outside of the sampling plots that were aerially seeded where alfalfa and Lewis flax established at higher densities. Sheley and Bates (2008) reported that western yarrow and Lewis flax established well when seeded after prescribed burning western juniper. Differences between sites and seeded species probably explain the difference in perennial forb response in their study compared to our study. The relatively high perennial forb cover at our study sites also suggests that seeding perennial forbs may not have been needed in this area.

The lack of a treatment effect on exotic annual grass cover and density was probably due to most sites not having a significant annual grass presence regardless of treatment. Therefore, our study was not a robust test for determining the efficacy of aerial seeding for limiting exotic annual grasses. When only evaluating the two sites that had a significant annual grass presence (>0.5 % cover), annual grass cover and density averaged 2.7- and 3.8-fold less when aerially seeded (3.5 ± 1.0 % and 73 ± 21 plants·m^{-2}), as compared to unseeded (9.3 ± 1.0 % and 281 ± 63 plants·m^{-2}) areas in the third year after treatment, respectively. These results imply that it may be important to seed after juniper control where exotic annual grasses are a threat. Our results also suggest that post-juniper control seeding will not always be necessary to limit exotic annual grass invasion as annual grasses are not an issue at all sites. Research determining the variability in annual grass invasion risk after prescribed burning encroaching western juniper trees would be valuable to land managers as they attempt to restore sagebrush steppe communities.

Our results suggest that seeding mountain big sagebrush after using partial cutting and prescribed fire to control western juniper can greatly accelerate the recovery of sagebrush cover and density at least in sites similar to those included in this study. By the third year after treatment, two of the five areas seeded with sagebrush had approximately 12 % sagebrush cover. Sagebrush cover varied considerably among the sagebrush seeded plots, ranging from 1 to 12 % 3 years post-treatment. However, 1 % sagebrush cover was still an improvement over natural recovery since sagebrush was not present in the unseeded plot in the same block. Sagebrush density in the sagebrush seeded plots was on average higher than the density that Davies and Bates (2010) reported for relatively intact mountain big sagebrush communities. Thus, sagebrush density was on average fully recovered by the second year after seeding. Low sagebrush recruitment (one sagebrush·115 m^{-2}) in the unseeded plots suggests that there was limited viable sagebrush seed in the seed bank. Late Phase II and Phase III juniper woodlands have largely excluded sagebrush from the plant communities; therefore, seed input would be limited. What little sagebrush seed that may have been in the seed bank was probably not viable because it would be at minimum a year old as burning occurred prior to seed maturity of the current seed crop. Young and Evans (1989) reported that big sagebrush seed remained viable for only 6 months under field conditions. Similarly, Wijayratne and Pyke (2009) determined that after 2 years, big sagebrush seed near the soil surface was no longer viable. Little, if any, sagebrush that established in our plots was from seeds that dispersed from outside burned areas as dispersal of sagebrush seed is limited to a few meters from the parent plant (Young and Evans 1989). The sagebrush that did occur in plots where sagebrush was not seeded may have been from seed that dispersed from the sagebrush seeded plots. Thus, these plant communities are likely sagebrush seed limited without seeding.

We observed seeded sagebrush plants producing seed by the second year after seeding. This suggests that the sagebrush seeded areas could serve as a seed source for unseeded areas as well as providing additional recruitment potential in seeded areas. We speculate that even lower sagebrush seeding rates than used in this study may be successful because seeded sagebrush will start producing seed in a few years. However, lower rates will probably increase the length of time for sagebrush to recovery. In contrast, herbaceous recovery may be accelerated with lower sagebrush abundance. We did not observe a significant effect of seeding sagebrush on herbaceous vegetation in the first 3 years of our study; however, as sagebrush cover continues to increase it will likely limit herbaceous vegetation. Increases in sagebrush have been reported to decrease the herbaceous understory (Cook and Lewis 1963; Rittenhouse and Sneva 1976; McDaniel and others 2005). Longer term research is needed to determine how accelerated mountain big sagebrush recovery influences the herbaceous understory. Managers will have to weigh the cost of sagebrush seed against speed of sagebrush recovery and the interaction between sagebrush and herbaceous vegetation to determine the seeding rate that best meets their needs within budget constraints.

Our results are in stark contrast to attempts to broadcast seed Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young). Wyoming big sagebrush has generally failed to establish when broadcast seeded after disturbance (Lysne and Pellant 2004; Davies and others 2013). Although weather conditions were probably favorable for establishment with a cool, wet spring in 2010, subsequent broadcast seeding of mountain big sagebrush in the following 2 years was similarly successful (K. W. Davies, unpublished data). Higher elevation and greater precipitation in mountain big sagebrush communities compared to Wyoming big sagebrush communities probably account for the greater seeding success. Thus, the recruitment of sagebrush in mountain big sagebrush steppe may not be as episodic or as dependent on yearly climate conditions compared to Wyoming big sagebrush communities. Additional research determining where aerial seeding is likely to be successful and not successful is needed to improve the ability of land managers to effectively and efficiently restore sagebrush plant communities. This is especially true in sagebrush communities at lower elevation than those in ours study and also in sagebrush communities with more exotic annual grasses present prior to treatments.

Our results suggest that seeding mountain big sagebrush after partial cutting and prescribed burning western juniper can improve sagebrush-associated wildlife species habitat. With large reductions in sagebrush habitats (Knick and others 2003; Schroeder and others 2004; Davies and others 2011) and severe declines in sagebrush obligate wildlife species such as sage-grouse (Connelly and Braun 1997; Braun 1998; Connelly and others 2004) this research provides an important example of management that can be applied to mitigate and possibly reverse these declines. Our research suggests that western juniper encroached sagebrush steppe similar to our study area under similar climatic conditions may be restored in a relatively short time period with western juniper control followed by seeding sagebrush.

A limitation to our study was that the first year and second year after seeding were average to above average precipitation years. Our results may not have been so favorable or we may have even had a complete seedling failure if seeding had occurred during a drought. Additional research is needed to determine aerial seeding success in drier years.

Conclusions

Our study provides evidence that sagebrush habitat can be restored in some late Phase II and Phase III western juniper-encroached mountain big sagebrush community by partial cutting followed by prescribed burning, and seeding herbaceous species and sagebrush. The applicability of our results to lower elevation or more exotic annual grass-invaded sites is unknown as our study was conducted in relatively high elevation juniper-encroached sagebrush communities with limited exotic annual grass presence prior to treatment. In addition, the first 2 years of our study were average to above average precipitation years and thus, our study needs replicated in drier years to determine the success of aerial seeding mountain big sagebrush communities after juniper control across a broader climatic spectrum. There will be variability in success across the landscape as demonstrated in our study and seeding may not be as successful under less favorable climatic conditions. However, overall we found that seeding doubled cover and density of perennial grasses, and increased sagebrush cover and density 74- to 290-fold and 62- to 155-fold, respectively. These results suggest that seeding herbaceous species and sagebrush after prescribed burning can limit opportunities for invasive plants. Seeding herbaceous vegetation, however, is not always needed. Our results suggest that sagebrush recovery with seeding may be adequate to provide sage-grouse habitat at some sites in as little as 3 years after partial cutting and prescribed burning to control juniper. We suggest that land managers should consider seeding mountain big sagebrush after controlling western juniper to accelerate the recovery of sagebrush habitat.

Acknowledgments The authors thank the Burns District-Bureau of Land Management for allowing this research to be conducted on lands they administer and for providing some of the funds used to carry out this study. Additional funding for this research project was provided by the USDA—Agricultural Research Service. The authors also thank Julie Garner, Sarah Fitzpatrick, Brandi Carlon, Rodney Johnson, Savannah Duff, Alanna Riggs, Tim Rowe, and Anne Crecelius for assisting with data collection. We also are grateful to Dustin Johnson and Brenda Smith for reviewing earlier versions of this manuscript. We appreciate the thoughtful reviews by the anonymous reviewers and Editorial Board Member.

Disclaimer Mention of a proprietary product does not constitute a guarantee or warranty of the product by the USDA or the authors and does not imply its approval to the exclusion of other products that may also be suitable. USDA is an equal opportunity provider and employer.

References

Aldridge CL, Nielson SE, Beyer HL, Boyce MS, Connelly JW, Knick ST, Schroeder MA (2008) Range-wide patterns of greater sage-grouse persistence. Divers Distrib 14:983–994

Baker WL (2006) Fire and restoration of sagebrush ecosystems. Wildl Soc Bull 34:177–185

Barney M, Frischknecht N (1974) Vegetation changes following fire in the pinyon-juniper type of west-central Utah. J Range Manage 27:91–96

Bates JD, Miller RF, Svejcar TJ (2005) Long-term successional trends following western juniper cutting. Range Ecol Manage 58: 533–541

Bates JD, Davies KW, Sharp RN (2011) Shrub-steppe early succession following invasive juniper cutting and prescribed fire. Environ Manag 47:468–481

Braun CE (1998) Sage grouse declines in western North America: what are the problems? Proc Western Assoc Fish Wildl Agencies 78:139–156

Burkhardt JW, Tisdale EW (1969) Nature and successional status of western juniper vegetation in Idaho. J Range Manag 22:264–270

Canfield RH (1941) Application of the line interception methods in sampling range vegetation. J For 39:388–394

Clausnitzer DW, Borman MM, Johnson DE (1999) Competition between *Elymus elymoides* and *Taeniatherum caput-medusae*. Weed Sci 47:720–728

Connelly JW, Braun CE (1997) Long-term changes in sage grouse *Centrocercus urophasianus* populations in western North America. Wildl Biol 3:229–234

Connelly JW, Schroeder MA, Sands AR, Braun CE (2000) Guidelines to manage sage-grouse populations and their habitat. Wildl Soc Bull 28:967–985

Connelly JW, Knick ST, Schroeder MA, Stiver SJ (2004) Conservation assessment of greater sage-grouse and sagebrush habitats. Western Association of Fish and Wildlife Agencies. US Government Documents (Utah Regional Depository). Paper 73. http://digitalcommons.usu.edu/govdocs/73

Cook CW, Lewis CE (1963) Competition between big sagebrush and seeded grasses on foothill ranges in Utah. J Range Manag 16:245–250

Crawford JA, Olson RA, West NE, Mosley JC, Schroeder MA, Whitson TD, Miller RF, Gregg MA, Boyd CS (2004) Ecology and management of sage-grouse and sage-grouse habitat. J Range Manag 57:2–19

Davies KW (2008) Medusahead dispersal and establishment in sagebrush steppe plant communities. Range Ecol Manag 61:110–115

Davies KW, Bates JD (2010) Vegetation characteristics of mountain and Wyoming big sagebrush plant communities in the northern Great Basin. Range Ecol Manag 63:461–466

Davies KW, Boyd CS, Beck JL, Bates JD, Svejcar TJ, Gregg MA (2011) Saving the sagebrush sea: an ecosystem conservation plan for big sagebrush plant communities. Biol Conserv 144:2573–2584

Davies KW, Boyd CS, Nafus AM (2013) Restoring the sagebrush component in crested wheatgrass-dominated communities. Range Ecol Manag 66:472–478

Johnson DD, Miller RF (2006) Structure and development of expanding western juniper woodlands as influenced by two topographic variables. Forest Ecol Manag 229:7–15

Knick ST, Dobkin DS, Rotenberry JT, Schroeder MA, Vander Haegen WM, Riper CV (2003) Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. Condor 105:611–634

Littell RC, Milliken GA, Stroup WW, Wolfinger RD (1996) SAS system for mixed models. SAS Institue Inc., Cary, p 633

Lysne CR, Pellant ML (2004) Establishment of aerially seeded big sagebrush following souther Idaho wildfires. Technical Bulletin 2004-01. Department of the Interior, Bureau of Land Management. Boise, p 14

McDaniel KC, Torell LA, Ochoa CG (2005) Wyoming big sagebrush recovery and understory response with tebuthiuron control. Range Ecol Manag 58:65–76

Miller RF, Rose JR (1995) Historic expansion of *Juniperus occidentalis* southeastern Oregon. Great Basin Nat 55:37–45

Miller RF, Tausch RJ (2001) The role of fire in juniper and pinyon woodlands: a descriptive analysis. In: Gailey KEM, Wilson TP (eds). Proceedings of the Invasive species workshop: the role of fire in the control and spread of invasive species. Tall Timbers Research Station, Misc Pub No. 11, Tallahassee, pp 15–30

Miller RF, Svejcar TJ, Rose JR (2000) Impacts of western juniper on plant community composition and structure. J Range Manag 53:574–585

Miller RF, Bates JD, Svejcar TJ, Pierson FB, Eddleman LE (2005) Biology, ecology, and management of western juniper. Oregon State University Agricultural Experiment Station, Technical Bulletin 152, p 77

Noss RF, LaRoe ET III, Scott JM (1995) Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. National Biological Service Biological Report 28. Washington, p 95

NRCS (2012) Soil Survey. http://soils.usda.fov/survey/. Accessed 1 Dec, 2012

Pierson FB, Bates JD, Svejcar TJ, Hardegree SP (2007) Runoff and erosion after cutting western juniper. Range Ecol Manag 60:285–292

Rittenhouse LR, Sneva FA (1976) Expressing the competitive relationship between Wyoming big sagebrush and crested wheatgrass. J Range Manag 29:326–327

Schroeder MA, Aldridge CL, Apa AD, Bohne JR, Braun CE, Bunnell SD, Connelly JW, Deibert PA, Gardner SC, Hilliard MA, Kobriger GD, McAdam SM, McCarthy CW, McCarthy JJ, Mitchell DL, Rickerson EV, Stiver SJ (2004) Distribution of sage-grouse in North America. Condor 106:363–376

Sheley RL, Bates JD (2008) Restoring western juniper (*Juniperus occidentalis*) infested rangeland after prescribed fire. Weed Sci 56:469–476

Skousen JG, Davis JN, Brotherson JD (1989) Pinyon-juniper chaining and seeding for big game in central Utah. J Range Manag 42:98–104

Suring LH, Rowland MM, Wisdom MJ (2005) Identifying species of conservation concern. In: Wisdom MJ, Rowland MM, Suring LH (eds) Habitat threats in the sagebrush ecosystem—methods of regional assessment and applications in the Great Basin. Alliance Communications Group, Lawrence, pp 150–162

Tausch RJ, Tueller PT (1977) Plant succession following chaining of pinyon–juniper woodlands in eastern Nevada. J Range Manag 30:44–49

Western Regional Climate Center (2012) Cooperative climatological data summaries. (http://www.wrcc.dri.edu/climatedata/climsum/) Accessed 14 Dec, 2012

Wijayratne UC, Pyke DA (2009) Investigating seed longevity of big sagebrush (*Artemisia tridentata*). Open-file report 2009-1146. USDI-USGS, p 26

Wisdom MJ, Rowland MM, Suring LH, Schueck L, Meinke CW, Knick ST (2005) Evaluating species of conservation concern at regional scales. In: Wisdom MJ, Rowland MM, Suring LH (eds) Habitat threats in the sagebrush ecosystem—methods of regional assessment and applications in the Great Basin. Alliance Communications Group, Lawrence, pp 5–24

Young JA, Evans RA (1989) Dispersal and germination of big sagebrush (*Artemisia tridentata*) seed. Weed Sci 37:201–206