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Success of Seeding Native Compared with Introduced Perennial Vegetation for Revegetating Medusahead-Invaded Sagebrush Rangeland☆,☆☆

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ABSTRACT

Millions of hectares of Wyoming big sagebrush (*Artemisia tridentata* Nutt. subsp. *wyomingensis* Beetle &Young) rangeland have been invaded by medusahead (*Taeniatherum caput-medusae* [L.] Nevski), an exotic annual grass that degrades wildlife habitat, reduces forage production, and decreases biodiversity. Revegetation of medusahead-invaded sagebrush plant communities is necessary to restore ecosystem services. Disagreement, however, exists over whether to seed native or introduced perennial species to revegetate communities after controlling medusahead. Though native species generally do not establish as well as introduced species, interference from co-seeded introduced species has often been attributed to the limited success of natives. The potential for seeding natives to revegetate communities after medusahead control is relatively unknown because they have been largely co-seeded with introduced species. We compared the results of seeding native and introduced perennial species after controlling medusahead with prescribed burning followed with an imazapic herbicide application at five sites. Perennial bunchgrass cover and density were 5- and 10-fold greater in areas seeded with introduced compared with native species 3 years post seeding. Furthermore, exotic annual grass cover and density were less in areas seeded with introduced compared with native species. Seeded introduced and native shrubs largely failed to establish. High perennial bunchgrass density (15 individuals · m^{-2}) in areas seeded with introduced species in the third year post seeding suggests that the succession trajectory of these communities has shifted to becoming perennial dominated. Average perennial bunchgrass density of 1.5 individuals · m^{-2} with seeding native species will likely not limit medusahead and appears to already be converting back to exotic annual grass-dominated communities. These results suggest that seeding introduced compared with native species after medusahead control will likely be more successful. Our results also imply that if natives are selected to seed after medusahead control, additional resources may be necessary to recontrol medusahead and repeatedly sow native species.

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Introduction

Medusahead (*Taeniatherum caput-medusae* [L.] Nevski) is an exotic annual grass invading sagebrush (*Artemisia* L.) steppe rangelands. Medusahead is a serious conservation problem, and its spread has severe implications for many ecosystem services (Nafus and Davies, 2014). Biodiversity and native plant abundance decline exponentially with increasing medusahead (Davies, 2011). Litter accumulations from medusahead can increase wildfire frequency (Torell et al., 1961; D'Antonio and Vitousek, 1992; Davies and Svejcar, 2008). These effects can substantially degrade habitat for native wildlife (Davies and Svejcar, 2008). Medusahead is also a substantial threat to livestock production in sagebrush rangelands. Hironaka (1961) estimated that medusahead invasion reduced livestock forage production by up to 80%. The effects of medusahead on native ecosystems are escalating because medusahead is rapidly spreading from established infestations (Duncan et al., 2004; Davies, 2008). Hence, there is a critical need to revegetate medusahead-

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invaded rangelands to restore ecosystem services and function and reduce the rate of spread of medusahead.

To revegetate medusahead-invaded rangelands, medusahead must first be successfully controlled to allow seeded species to establish (Young, 1992; Davies, 2010; Nafus and Davies, 2014). Though a variety of methods have been used to control medusahead, the best control has generally been achieved by integrating control treatments (Nafus and Davies, 2014). In the sagebrush steppe, prescribed burning followed with pre-emergent herbicide application can successfully control medusahead (e.g., Kyser et al., 2007; Davies, 2010; Davies and Sheley, 2011; Sheley et al., 2012). After medusahead is controlled, perennial vegetation must often be seeded to limit medusahead reinvasion and dominance and restore ecosystem services and function (Nafus and Davies, 2014). The most successful revegetation method has been to wait 1 year after pre-emergent herbicide application to seed to reduce the risk of nontarget herbicide damage to seeded species (Davies et al., 2014c).

In Wyoming big sagebrush (*Artemisia tridentata* Nutt. subsp. *wyomingensis* Beetle & A. Young) communities, there is disagreement over whether introduced (non-native) or native species should be used for revegetation projects. The introduced bunchgrasses, crested (*Agropyron cristatum* (L.) Gaertn. and *A. desertorum* [Fisch. Ex Link] Schult) and Siberian wheatgrass (*A. fragile* [Roth] P. Candargy), are commonly seeded after wildfires (Eiswerth et al., 2009; Knutson et al., 2014) and exotic annual grass control (Davies et al., 2010; Nafus and Davies, 2014). These introduced bunchgrasses are often selected instead of native bunchgrasses because they are more available, cost less, and establish better (Arredondo et al., 1998; Eiswerth et al., 2009; Boyd and Davies, 2010). Crested wheatgrass has also been selected because it is highly competitive and can be seeded around infestations to limit the spread of medusahead (Davies et al., 2010). However, established native bunchgrasses also limit medusahead establishment. Davies (2008) found medusahead establishment was negatively correlated with native bunchgrass density in sagebrush rangelands. The use of introduced species in revegetation efforts in sagebrush communities is also controversial (Davies et al., 2011). Crested wheatgrass often forms a dense monoculture (Pyke, 1990) and is highly competitive with native vegetation (Heinrichs and Bolton, 1950; Schuman et al., 1982; Gunnell et al., 2010). This can result, at least in the short-term, in plant communities that do not provide as high of quality habitat for native wildlife as native-dominated plant communities (Reynolds and Trost, 1981; McAdoo et al., 1989). Thus there is a desire to use native vegetation when revegetating Wyoming big sagebrush communities.

The potential for revegetating medusahead-invaded sagebrush rangeland with native species is relatively unknown because it has usually been co-seeded with introduced bunchgrasses or not seeded at all. Native bunchgrasses have been co-seeded with introduced bunchgrasses after medusahead control (Davies, 2010) and after fires (Boyd and Davies, 2010; Davies et al., 2013a; Knutson et al., 2014). Introduced bunchgrasses establish better than native bunchgrasses when they are co-seeded, but introduced bunchgrasses may have limited the success of co-seeded natives (Boyd and Davies, 2010; Knutson et al., 2014). Determining suitability of native vegetation to revegetate medusahead-invaded rangelands is important because the selection of plant materials to be seeded is crucial for successful revegetation. Davies et al. (2014b) found that revegetation success varied considerably by plant materials selected to seed after medusahead control with some seeded herbaceous functional groups largely failing to establish (forbs) and other groups (perennial grasses) establishing well. To evaluate the effectiveness of using native vegetation and to provide land managers with important information to assist them in selecting plant materials for revegetating medusahead-invaded sagebrush rangeland, it is critical to compare seeding native with introduced vegetation.

The objective of this study was to compare seeding commercially available native and introduced perennial vegetation (bunchgrasses and shrubs) after controlling medusahead with prescribed burning and pre-emergent herbicide application. We hypothesized that 1) perennial bunchgrass and shrub density and cover would be greater when seeding introduced compared with native vegetation and, subsequently, 2) exotic annual grass density and cover would be less when seeding introduced compared with native vegetation.

Methods

Study Area

The study was located in southeastern Oregon between Crane and Juntura, OR, in medusahead-invaded Wyoming big sagebrush rangeland. Study sites ranged in elevation from 972 to 1052 m above sea level and were separated by up to 30 km. Slopes were relatively flat to 12° with northeast, southwest, and west aspects depending on study site. Climatic conditions were representative of the northwestern Great Basin with most precipitation occurring in the winter and early spring and with typically hot and dry summers. Long-term (1981–2010) average annual precipitation was between 249 and 258 mm (PRISM Climate Group, 2014). Crop year (October–September) precipitation was 75%, 91%, and 79% of the long-term average in 2011–2012, 2012–2013, and 2013–2014, respectively (PRISM Climate Group, 2014). Soils ranged from clay loam to loam among study sites. The potential natural vegetation of sites was Wyoming big sagebrush-bunchgrass steppe. Before control treatments, vegetation at study sites was a near-monoculture of medusahead with a few (<0.4 plants$\cdot$m^{-2}) residual bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) Á. Löve) or other native bunchgrass individuals. Sagebrush had been lost from the plant communities from previous wildfires. Livestock were excluded from study sites for the duration of the study with a four-strand barbwire fence. Wildlife species were not excluded from study sites.

Experimental Design and Measurements

A randomized complete block design with five sites (blocks) was used to compare treatments. Treatments after medusahead was controlled were 1) seeded with commercially available native perennial vegetation (Native-Seeded) or 2) seeded with commercially available introduced perennial vegetation (Introduced-Seeded). Each treatment was applied to one of two 30 × 50 m plots separated by a 2-m buffer at each block. Medusahead was controlled by prescribed fall burning followed with a fall imazapic application. Prescribed burning occurred in late September 2010 using strip-head fires ignited with drip torches. During prescribed burns wind speed varied from 0 to 6 km · hr^{-1}, relative humidity ranged from 21% to 48%, and air temperature varied from 14–29°C. Burns were nearly complete across plots with 95% of the medusahead litter and other fuels being consumed. Imazapic was applied within 2 weeks of burning at 87.5 g · ai · ha^{-1} using a UTV-mounted seven-nozzle boom spray with a nozzle height of 0.6 m from the ground and a tank pressure of 207 kPa. During imazapic application wind speed varied from 0 to 5 km · hr^{-1} and air temperature ranged from 7–16°C. One year after imazapic application treatment plots were seeded. The seed mix for the Native-Seeded treatment consisted of bottlebrush squirreltail (*Elymus elymoides* (Raf.) Swezey) (variety unspecified), bluebunch wheatgrass (variety Anatone), and Wyoming big sagebrush. The seed mix for the Introduced-Seeded treatment consisted

of crested wheatgrass (variety Hycrest), Siberian wheatgrass (variety Vavilov), and forage kochia (*Bassia prostata* [L.] A. J. Scott). All seed used for this study was purchased from Granite Seed Company (Lehi, UT). Perennial grasses were drill seeded using a Versa-Drill (Kasco, Inc, Shelbyville, IN) with drill rows spaced 23 cm apart in early October of 2011 at 21.6 kg · ha^{-1} pure live seed with equal proportions by weight of each bunchgrass species. Perennial bunchgrass species were mixed together before drill seeding for both treatments. Forage kochia and sagebrush were broadcast seeded with a nonautomated fertilizer flinger (hand-cranked broadcaster) at 3.4 kg · ha^{-1} pure live seed in December 2011.

Herbaceous cover and density were measured in June in 2012, 2013, and 2014, the first, second, and third growing season after seeding. Vegetation sampling occurred along four parallel 45-m transects spaced 5 m apart in each treatment plot. Herbaceous canopy cover was estimated by species in 0.2-m^2 quadrats located at 3-m intervals on each 45-m transect, resulting in 60 quadrats per treatment plot. Quadrats were divided into 1%, 5%, 10%, 25%, and 50% segments to increase the accuracy of cover estimates. Bare ground, biological soil crust, and litter cover were also estimated in the 0.2-m^2 quadrats. Herbaceous density was measured by species by counting all plants rooted inside of the 0.2-m^2 quadrats. Shrub cover by species was estimated using the line-intercept method (Canfield, 1941) along the four, 45-m transects. Shrub density by species was determined by counting all shrubs rooted in 2 × 45-m belt transects laid over the 45-m transects.

Statistical Analysis

Treatment effects were estimated using repeated measures ANOVAs with years as the repeated factor in PROC MIX SAS v. 9.2 (SAS Institute Inc., Cary, NC). Treatment was considered a fixed variable, and random variables were site and site by treatment interactions. Covariance structure for each analysis was selected using Akaike's Information Criterion (Littell et al., 1996). Data were square-root or log-transformed when assumptions of ANOVA were violated. Figures and text report nontransformed (i.e., original) data. Treatment differences were considered significant at $P \leq 0.05$. Means were reported with standard errors in the text and figures. Herbaceous cover and density were grouped into five groups for analyses: perennial bunchgrasses, Sandberg bluegrass (*Poa secunda* J. Presl), perennial forbs, exotic annual grasses, and annual forbs. Sandberg bluegrass was treated as a separate group because it is much smaller in stature and matures considerably earlier than other perennial bunchgrasses in the sagebrush ecosystem. The exotic annual grass group was predominately composed of medusahead with some cheatgrass (*Bromus tectorum* L.).

Results

Perennial bunchgrass cover was greater in the Introduced-Seeded compared with the Native-Seeded treatment (Fig. 1A; $P = 0.049$). In 2014, perennial bunchgrass cover was fivefold greater in the Introduced-Seeded treatment compared with the Native-Seeded treatment. Perennial bunchgrass cover increased in both treatments from 2012 through 2014 ($P = 0.009$). Exotic annual grass cover was twofold to fivefold greater in the Native-Seeded treatment compared with the Introduce-Seeded treatment (Fig. 1B; $P = 0.042$). Exotic annual grass cover increased with time since seeding in both treatments ($P = 0.009$). Sandberg bluegrass, perennial forb, and annual forb cover did not differ between treatments (Fig. 1C, D, and E; $P = 0.264$, 0.731, and 0.299, respectively) or among years ($P = 0.839$, 0.290, and 0.146, respectively). Sandberg bluegrass cover was low in both treatments with an average cover of less than 0.5%. Biological soil crust cover did not differ between treatments or among years (Fig. 1F; $P = 0.182$ and 0.062, respectively). Biological soil crust cover was low throughout the study with its average not exceeding 0.2%. Bare ground did not differ between treatments (Fig. 1G; $P = 0.465$) but decreased from 2012 to 2014 ($P = 0.001$). We did not find evidence that litter differed between treatments (Fig. 1H; $P = 0.293$). Litter varied by year ($P = 0.015$) with it generally being greatest in the last year (2014) of the study. Forage kochia (0 ± 0% cover) did not establish in any plots, and sagebrush (0.025 ± 0.023% cover) only established a few plants in two of the five treatment plots; thus there was no difference in shrub cover between treatments ($P = 0.373$). Shrub cover also did not vary among years ($P = 0.376$). The interaction between treatment and year was not significant for any measured cover response variable ($P > 0.05$).

Perennial bunchgrass density was 6- to 11-fold greater in the Introduced-Seeded treatment compared with the Native-Seeded treatment (Fig. 2A; $P = 0.009$). Though the year effect was not significant when both treatments were analyzed together ($P = 0.071$), perennial bunchgrass density in the Introduced-Seeded treatment increased from 8.6 ± 1.8 individuals · m^{-2} in 2012 to 15.0 ± 3.7 individuals · m^{-2} in 2014. In the Native-Seed treatment, perennial bunchgrass density was unchanged between 2012 (1.4 ± 0.4 individuals · m^{-2}) and 2014 (1.5 ± 0.8 individuals · m^{-2}). Exotic annual grass density was 2- to 6-fold greater in the Native-Seeded treatment compared with the Introduced-Seeded treatment (Fig. 2B; $P = 0.026$) and increased with time since treatment ($P = 0.010$). Sandberg bluegrass density was 2- to 5-fold greater in the Introduced-Seeded treatment than the Native-Seeded treatment (Fig. 2C; $P = 0.020$) but did not vary among years ($P = 0.651$). Perennial forb (data not shown) and annual forb density (Fig. 2D) did not vary between treatments ($P = 0.275$ and 0.393, respectively) or among years ($P = 0.407$ and 0.203, respectively). Shrub density did not differ between Introduced-Seeded (0 ± 0.00 individuals · m^{-2}) and Native-Seeded (0.006 ± 0.004 individuals · m^{-2}) treatments and did not vary among years ($P = 0.260$). The interaction between treatment and year was not significant for any measured density response variable ($P > 0.05$).

Discussion

In support of our hypotheses, seeding commercially available introduced compared with commercially available native perennial vegetation after medusahead control resulted in greater perennial bunchgrass cover and density and less exotic annual grass cover and density. However, contrary to the prediction of our first hypothesis, shrub cover and density did not vary between treatments. The magnitude of differences between seeding natives or introduced species was large, as exemplified by the 10-fold greater density of perennial bunchgrasses in the introduced compared with native seeded plots in 2014, the third year post seeding. Our results suggest that it may, at least when site and climatic conditions are similar to our study, not be effective to seed natives after medusahead control in Wyoming big sagebrush habitat because of the limited establishment of native perennial vegetation. After medusahead control, it is critical to establish perennial vegetation to limit reinvasion and dominance of medusahead and improve ecosystem services and function (Nafus and Davies, 2014). Similar to our results, other authors have reported that introduced bunchgrasses generally have higher establishment rates than native bunchgrasses in sagebrush communities (Robertson et al., 1966; Hull, 1974; Boyd and Davies, 2010). However, the success of establishing native plants from seed increases in big sagebrush communities with increasing precipitation and elevation (Davies et al., 2011; Davies et al., 2014a). Using local seed sources for native species may also improve establishment but needs to be tested.

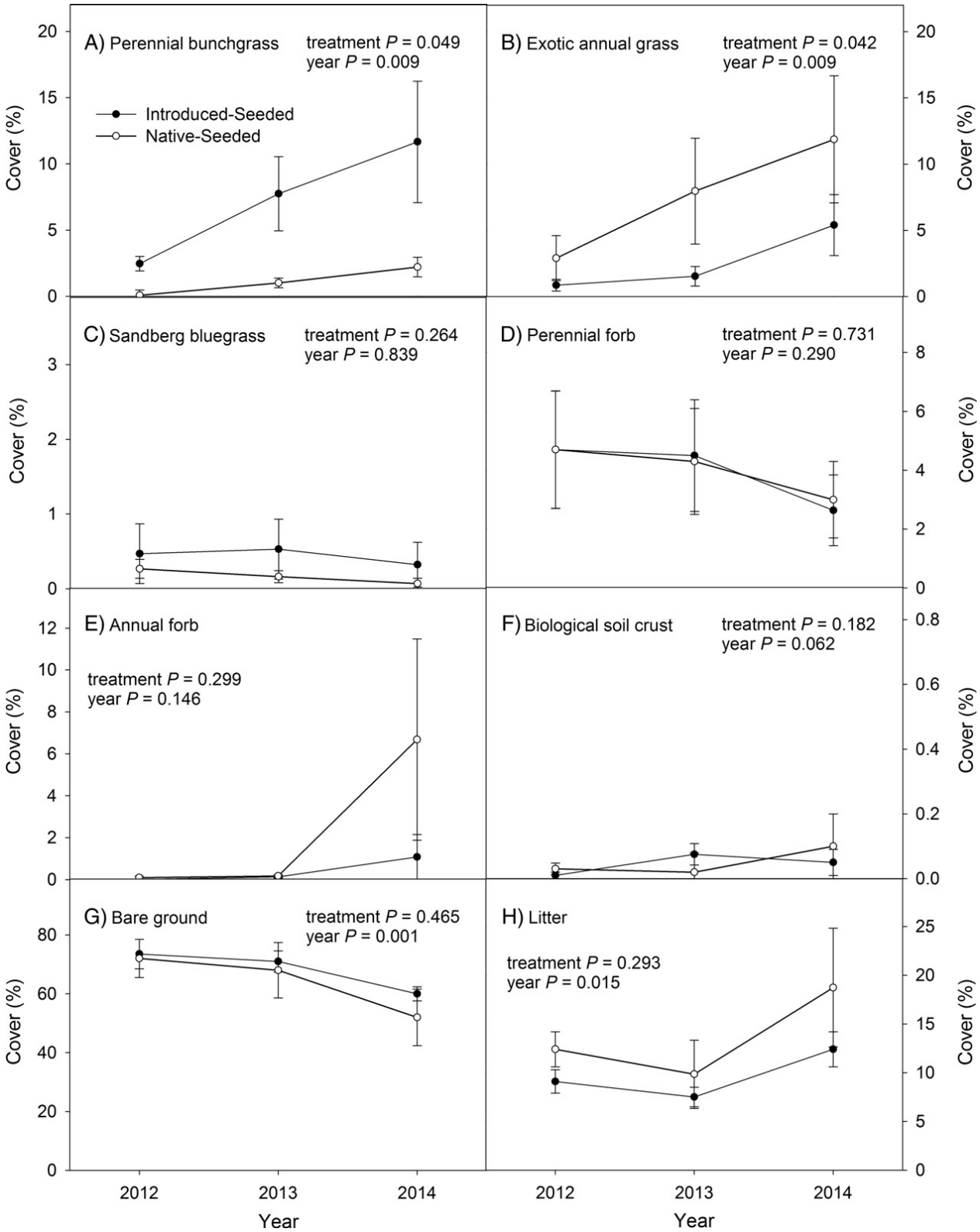

Fig. 1. Cover group values (mean ± S.E.) in Introduced-Seeded and Native-Seeded treatments used to revegetate medusahead-invaded sagebrush communities after medusahead control.

Plant community succession trajectories after medusahead control appear to be vastly different on the basis of whether native or introduced species were seeded. Similarly, Davies et al. (2014b) reported that revegetation success after medusahead control varied by seed mix composition. In our study, areas seeded with introduced species had relatively high densities of perennial bunchgrasses (15.0 ± 3.7 individuals · m^{-2}) 3 years after seeding, which suggests these areas will be dominated by perennial vegetation, which will likely break the exotic annual grass-fire cycle. In contrast, 3 years post seeding, perennial bunchgrasses density was low (1.5 ± 0.8

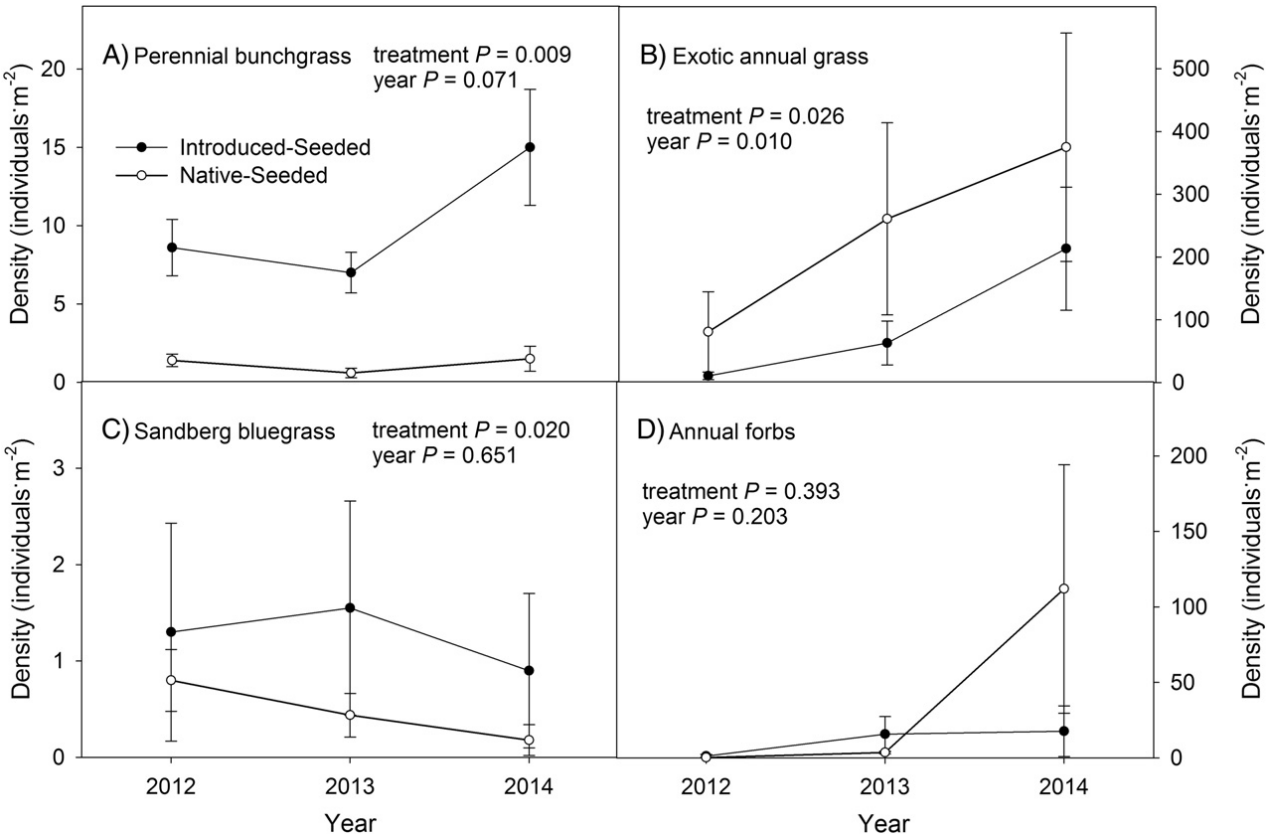

Fig. 2. Functional group density (mean ± S.E.) in Introduced-Seeded and Native-Seeded treatments used to revegetate medusahead-invaded sagebrush communities after medusahead control.

individuals · m^{-2}) in areas seeded with native vegetation. This suggests that exotic annual grasses will redominate these areas. The density of perennial bunchgrasses is a fairly robust indicator of the resistance of these areas to exotic annual grass invasion and dominance. Davies (2008) reported that medusahead establishment was inversely correlated with perennial bunchgrass density. Perennial bunchgrasses are critical to limit exotic annual grasses in the sagebrush steppe (Chambers et al., 2007) because they overlap greatly with annual grasses in resource acquisition patterns (James et al., 2008). Perennial bunchgrasses are also the dominant herbaceous plant functional group in big sagebrush communities (Davies et al., 2006; Davies and Bates, 2010), further demonstrating their importance in limiting resources to exotic annual grasses.

The response of exotic annual grasses further implies that succession trajectories will vary between treatments and that the increase in perennial bunchgrasses in areas seeded with introduced species was likely already limiting exotic annual grasses. Exotic annual grass cover and density were initially near zero after the burn and herbicide treatment and have increased with time, but introduced seeded areas have much less annual grass cover and density compared with native seeded areas. At the end of the study, exotic annual grass cover was more than twofold greater in native seeded compared with introduced seeded areas. This suggests that seeding introduced species can limit exotic annual grasses at low elevations, which contradicts Knutson et al.'s (2014) findings that drill seeding only limits exotic annual grasses at higher elevations. Our results may have differed from Knutson et al. (2014) because they were comparing drill seeded areas with nearby areas that were not drill seeded for a variety of reasons (e.g., too rocky to drill, different land ownership, or cultural protection) and we randomly assigned treatments to similar areas. Preexisting differences for seeded and nonseeded areas in Knutson et al. (2014) may have resulted in inherent plant community differences between drill seeded and nonseeded areas that may have masked treatment effects. Another potential reason for differences between our study and Knutson et al. (2014) was that we were investigating seeding after exotic annual grass control and they evaluated post–fire seeding with no annual grass control.

Introduced and native species are often seeded together after wildfires (Boyd and Davies, 2010; Knutson et al., 2014) and exotic annual grass control (Davies, 2010). Seeding of introduced species is expected to limit the establishment of native species (Knutson et al., 2014). However, our results suggest that native bunchgrasses can fail to establish regardless of whether or not they were coseeded with introduced species. Therefore, it would be naïve to assume that the introduced species limited establishment of co-seeded native species in sagebrush communities where introduced and native species have been coseeded and few seeded native species are detected post seeding. Our results also imply that if native species are not coseeded with introduced species and native species failed to establish, these plant communities would likely be open for exotic annual grass invasion and dominance. However, when both native and introduced bunchgrasses successfully establish in the same community, introduced bunchgrasses may over time limit native bunchgrasses. Introduced bunchgrasses are often more competitive than natives (Heinrichs and Bolton, 1950; Schuman et al., 1982; Gunnell et al., 2010) and can significantly out recruit native species in sagebrush steppe communities (Nafus et al., 2015). Though high recruitment

of seeded introduced species may restrict native vegetation recruitment, it may be critical for limiting the expression of exotic annual grasses over time.

The general lack of establishment of sagebrush and forage kochia indicates that these species may be difficult to establish after medusahead control, though success likely varies considerably with differing site characteristics and postseeding weather. Similar to our results, others have found that seeded Wyoming big sagebrush and forage kochia often fail to establish. In Idaho, 23 out of 35 areas seeded with Wyoming big sagebrush had no recruitment of sagebrush (Lysne and Pellant, 2004). Sagebrush density and cover on the remaining seeded areas were low and not statistically different from unseeded areas (Lysne and Pellant 2004). However, Davies et al. (2013b) found that a few (0.07 individual · m^{-2}) Wyoming big sagebrush plants established from broadcasted seed in crested wheatgrass stands when crested wheatgrass was controlled with glyphosate. Similar to our results, Morris et al. (2009) found poor establishment of forage kochia in central Utah after controlling annual grasses with imazapic. Though forage kochia is sensitive to imazapic (Morris et al., 2009), we waited to seed until one year after imazapic application to reduce the risk of nontarget herbicide damage. However, forage kochia establishment without imazapic application in the northern Great Basin has had erratic results (Haferkamp et al., 1990; Sheley et al., 2007).

Native perennial bunchgrasses, sagebrush, and forage kochia may have established in greater amounts under more favorable climatic conditions. The first year after seeding precipitation was approximately 75% of the long-term average (PRISM Climate Group, 2014). However, annual precipitation less than the long-term average is common in western United States (Diaz, 1983) and thus similar low establishment of native bunchgrasses, sagebrush, and forage kochia may be common at sites similar to those used in our study. Our study highlights the difficulty in establishing native perennial bunchgrasses, Wyoming big sagebrush, and forage kochia in low-elevation big sagebrush communities. The limited recruitment of native vegetation and forage kochia in these plant communities does not work well with the often "one-shot" attempt to revegetate them. In contrast, introduced perennial bunchgrasses appear to establish well under unfavorable conditions.

The establishment of introduced bunchgrasses in our study would be considered a successful revegetation effort because their density was almost 50% greater than the average density of bunchgrasses in relatively intact Wyoming big sagebrush communities (Davies and Bates, 2010). As plants grow larger, they may self-thin (Mueggler and Blaisdell, 1955); however, we expect that the areas seeded with introduced species will maintain a high density of bunchgrasses because it is already 3 years after seeding. Perennial bunchgrass cover in areas seeded with introduced species was also similar to slightly greater than the average reported by Davies et al. (2006) and Davies and Bates (2010) for relatively intact Wyoming big sagebrush communities. Additionally, bunchgrass density increased from the first and second year to the third year post seeding, further suggesting that seeding introduced bunchgrasses successfully revegetated medusahead-invaded sagebrush plant communities after medusahead control. This implies that new introduced bunchgrasses were being recruited in these plant communities. Davies (2010) had similar success with seeding crested wheatgrass and squirreltail after controlling medusahead with prescribed burning and imazapic application. However, Davies (2010) did not record an increase in perennial bunchgrass density after the first year post seeding. Davies (2010) only measured treatment effects for 2 years post treatment and the plots were relatively small (5 × 5 m) compared with the current study, which may explain why he did not record a similar increase in bunchgrass density over time.

Though we were successful at establishing introduced perennial bunchgrass after medusahead control, not all efforts have been as successful (e.g., Sheley et al., 2007; Kyser et al., 2013). Site, climate, plant material, and seeding technique differences likely explain differing levels of success. For example, Sheley et al. (2007) found a fourfold difference in Siberian wheatgrass cover between two sites in Oregon after medusahead control and seeding. Therefore though our results suggest that seeding introduced species after medusahead control will improve the likelihood of success over seeding natives, these results should not be misinterpreted to mean that seeding introduced species is without risk, as has been demonstrated by others (e.g., Sheley et al., 2007; Kyser et al., 2013).

Management Implications

Seeding introduced perennial bunchgrasses (crested and Siberian wheatgrass) after medusahead control with prescribed burning and imazapic application can successfully revegetate medusahead-invaded sagebrush communities. The successful establishment of introduced perennial bunchgrass appears to be limiting reinvasion of medusahead. In contrast, establishment of seeded native perennial bunchgrasses was too low to successfully revegetate these areas and they will likely be redominated by medusahead. On the basis of these differences, we suggest that medusahead-invaded sagebrush steppe revegetation projects in areas with site characteristics similar to our study sites use introduced perennial bunchgrasses or, if native bunchgrasses will be seeded, have resources available and plans for repeated sowing and control treatments because of the risk that the initial seeding effort may fail. As advances in technology improve establishment of seeded native bunchgrasses (Madsen et al., 2013), advantages of using introduced over native bunchgrasses may dissipate; however, currently introduced bunchgrasses appear to be more likely to establish than native bunchgrasses in low-elevation sagebrush communities. In addition, using local seed sources may improve the success when using natives but would need to be tested. Forage kochia and Wyoming big sagebrush establishment was unsuccessful, providing further evidence that these species only sporadically establish when seeded with current technologies and techniques. Considering that there are tens of millions of hectares invaded by exotic annual grasses in the Great Basin and surrounding area, which promotes more frequent and large wildfires (D'Antonio and Vitousek, 1992; Davies, 2011; Balch et al., 2013), and resources are limited for revegetation, we recommend that seeding introduced perennial bunchgrasses should be considered after exotic annual grass control and in areas at high risk of being invaded by exotic annual grasses.

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