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Long-Term Effects of Revegetation Efforts in Annual Grass–Invaded Rangeland ☆

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ABSTRACT

Invasive annual grasses, such as medusahead (*Taeniatherum caput-medusae* [L.] Nevski), have invaded tens of millions of hectares of the sagebrush ecosystem. These invasions severely reduce ecosystem goods and services provided, as well as increase the probability of frequent, large wildfires. Revegetation of invasive annual grass–invaded rangeland with perennial bunchgrasses is critical to reversing these negative consequences. Short-term evaluations of revegetation efforts have shown promising results. However, long-term evaluations of revegetation efforts in medusahead-invaded rangelands are lacking, so it remains unknown if revegetation attempts in these invaded rangelands have persistent effects. We evaluated the effects of controlling medusahead with prescribed burning and imazapic application followed 1 yr later with drill-seeding large perennial bunchgrasses at two seeding rates (medium and high) for more than a decade post seeding. Large perennial bunchgrass cover and density was $>$ 16- and $>$ 4-fold greater in revegetation treatments compared with the untreated control 11 yr after seeding, respectively. Invasive annual grass abundance was ~twofold greater in the untreated control compared with the revegetation treatments. These results suggest that revegetation efforts in medusahead-invaded rangelands can have persistent ecological benefits (increased perennials and decreased invasive annuals). The high seeding rate resulted in more perennial bunchgrass and less invasive annual grass compared with the medium seeding rate over the duration of the study, suggesting that high seeding rates may be needed to maximize benefits. Revegetation of medusahead-invaded rangelands can have long-lasting effects, though high establishment of perennial bunchgrasses is likely necessary for success.

Introduction

Invasive annual grasses and associated increased fire frequency are serious threats to the integrity of the sagebrush (*Artemisia* L.) ecosystem and wildlife dependent upon it (Mack 1981; D'Antonio and Vitousek 1992; Crawford et al. 2004; Balch et al. 2013). Annual grasses have invaded tens of millions of hectares of the 62 million ha sagebrush ecosystem in western North America (Meinke et al. 2009; Bradley et al. 2018; Smith et al. 2022). This problem is intensifying as invasive annual grasslands are replacing perennial-dominated communities, often sagebrush communities, at a rate of ~200 000 ha annually in the Great Basin (Smith et al. 2022). Invasive annual grasses are highly competitive with native perennial vegetation, especially at the seedling stage, which can result in the exclusion of native plants (Melgoza et al. 1990; Nasri and Doescher 1995; Rafferty and Young 2002; Humphrey and Schupp 2004). These annual grass invasions decrease biodiversity, reduce the quality and reliability of livestock forage, negatively impact native wildlife, and increase the risk of frequent, large wildfires (Brooks et al. 2004; Crawford et al. 2004; Davies 2011; Crist et al. 2023).

Medusahead (*Taeniatherum caput-medusae* [L.] Nevski) is one of the problematic invasive annual grasses in the sagebrush ecosystem (Young 1992; Nafus and Davies 2014). Invasion by medusahead is concerning because it develops a thick, persistent thatch layer that is a barrier to native plant establishment and increases the continuity of highly flammable fine fuel (Torell et al. 1961; Young et al. 1972; Young 1992). Medusahead is also competitive with native vegetation (Hironaka and Sindelar 1975; Goebel et al. 1988; Leffler et al. 2011). These factors lead to a loss of na-

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tive plant species and the development of a near monoculture of medusahead that does not meet the habitat needs of sagebrush-associated wildlife (Davies and Svejcar 2008; Davies 2011; USFWS 2013). Medusahead is also an inferior forage for livestock because it has high silica content and sharp awns (Hironaka 1961; Torell et al. 1961).

Revegetation of medusahead- and other invasive annual grass–dominated rangeland with a depleted perennial vegetation understory is needed to restore ecosystem goods and services. Most successful revegetation efforts control medusahead with a preemergent herbicide, followed by seeding with perennial bunchgrasses 1 yr later to limit herbicide damage to seeded species (Davies 2010; Kyser et al. 2013; Davies et al. 2015). Burning before preemergent herbicide application is a recommended practice (Davies 2010; Davies and Sheley 2011; Davies et al. 2015) because the annual grass thatch layer can intercept high levels of preemergent herbicides, preventing them from reaching the soil (Clark et al. 2019). Burning the thatch layer also improves the site for seeding (Davies 2010; Davies et al. 2015). Though short-term (2–3 yr) success has been accomplished with these methods (e.g., Davies 2010; Davies and Sheley 2011; Sheley et al. 2012b), long-term (~10 yr) effects are unknown. Longer term success with revegetation efforts in medusahead-invaded rangelands is not expected to be high as medusahead is expected to return to pre-treatment levels in 2–3 yr (James et al. 2015). However, revegetation efforts can be successful at limiting medusahead abundance for up to 5 yr post treatment (e.g., Davies and Boyd 2018). The key to successfully limiting medusahead and other invasive annual grasses after annual grass control is to establish abundant perennial vegetation (Davies and Johnson 2017). When perennial vegetation fails to establish after invasive annual grass control, annual grasses redominate the plant community (Monaco et al. 2005; Davies et al. 2014). Even if perennial vegetation initially establishes, it may not persist over longer time periods with climate fluctuations and competition from invasive annual grasses. Thus, it is unknown if revegetation efforts in medusahead-invaded rangelands are effective over the long term.

The effects of different seeding rates on perennial bunchgrass establishment and subsequent effects on medusahead are also relatively unknown, especially over longer time frames. Higher than average seeding rates may be needed in revegetation of invasive plant-infested rangelands (Jacobs et al. 1996; Sheley et al. 1999). Higher seeding rates may be particularly important in revegetation efforts in medusahead-invaded rangelands because medusahead often begins reinvading treated areas within a year or two after control treatments (James et al. 2015). If a higher seeding rate increases the abundance of seeded species, it may limit medusahead by reducing resources available to it during reinvasion. In a short-term study investigating revegetation of medusahead-invaded rangelands, a high (25.0 kg•ha^{-1} pure live seed [PLS]) compared with medium (13.2 kg•ha^{-1} PLS) seeding rate increased the biomass, but not the density of seeded species (Sheley et al. 2012a). However, the effect of seeding rate on medusahead was inconclusive because of the short-term nature of the study. Thus, long-term evaluations of medium and high seeding rates after medusahead control are warranted.

The objective of this study was to evaluate the long-term (> 10 yr) effects of annual grass control with burning and imazapic application followed with two perennial bunchgrass seeding rates on plant community composition in medusahead-invaded rangelands. We hypothesized that 1) controlling medusahead followed by seeding perennial bunchgrasses would increase bunchgrass density and cover and decrease invasive annual grass density and cover, and 2) a high compared with medium seeding rate would result in greater bunchgrass density and cover and less invasive annual grass density and cover for more than a decade.

Methods

Study area

The study was located between Juntura, Riverside, and Crane, Oregon in southeastern Oregon, United States. Five medusahead-invaded rangelands that were separated by up to 30 km were used as study sites. Potential natural vegetation at the study sites would have been Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis* [Beetle & A. Young] S.L. Welsh)–bunchgrass steppe before annual grass invasion. Before the initiation of the study, plant communities were near monocultures of medusahead with few residual bunchgrasses and perennial forbs, no shrubs, and, infrequently, some cheatgrass (*Bromus tectorum* L.) and non-native annual forbs. Cover of large perennial bunchgrass, Sandberg bluegrass (*Poa secunda* J. Presl), and perennial forbs at the start of the study was 0.09, 0.06, and 0.56%, respectively. Invasive annual grass and annual forb cover at this time was 18.29% and 0.20%, respectively. Climate was cool, wet winters and hot, dry summers, typical of the northern Great Basin. Average long-term (1991-2020) annual precipitation at the study sites was between 247 and 255 mm (PRISM Climate Group 2023). The year perennial bunchgrasses were seeded, crop-year precipitation (October 2011–September 2012) averaged 75% of the long-term average. Annual precipitation ranged from 66% to 131% of the long-term average over the duration of the study (Fig. 1). Elevations at the study sites were between 970 and 1 050 m above sea level. Study sites occurred on aspects varying from northeast to west with slopes from 0° to 12°. Soil texture at study sites varied from clay loam to loam. Cattle were excluded from the study sites for the duration of the study with barbed wire fences. In contrast, wildlife were not restricted from the study sites.

Experimental design and measurements

A randomized complete block design with five blocks (sites) was used to compare treatments. Treatments were 1) untreated control (Control) and burning, followed by imazapic application and 1 yr later seeded with perennial bunchgrasses at 2) medium seeding rate (Herb-Med) and 3) high seeding rate (Herb-High). Treatments were randomly assigned to one of three 30×50 m plots in each block. In September 2010 before imazapic application, treatment plots were fall-prescribed burned using strip-head fires to improve herbicide-soil contact and prepare the seedbed for seeding. During burns, air temperature was between 15°C and 30°C, relative humidity varied from 20% to 50%, and average wind speeds ranged from 1 km·h^{-1} to 8 km·h^{-1}. Ten to 11 d after burning, imazapic was applied at 87.5 g ai·ha^{-1} using a utility terrain vehicle (UTV)-mounted boom sprayer with a nozzle height of 60 cm and a tank pressure of 207 kPa. During herbicide application, air temperature was from 7°C to 16°C, and average wind speed varied from 0 h^{-1} to 5 km·h^{-1}. In October 2011, imazapic treated plots were drill-seeded with Siberian wheatgrass (variety Vavilov) and crested wheatgrass (variety Hycrest) at 10.8 (medium seeding rate) or 21.6 kg·ha^{-1} PLS (high seeding rate) with equal proportions by weight of each bunchgrass species. Untreated control plots were not seeded. Siberian and crested wheatgrass seeds were thoroughly mixed together and then seeded using a Versa-Drill (Kasco, Inc, Shelbyville, IN) pulled behind a UTV with drill rows spaced 23 cm apart. The medium seeding rate is commonly used in revegetation efforts in annual grass–invaded rangelands. Siberian and crested wheatgrass were selected for this study because they are common revegetation species in this region.

Herbaceous vegetation was measured in mid-June 2012 through 2022 along four parallel 45-m transects located 5 m apart in each treatment replicate. Herbaceous canopy cover by species was vi-

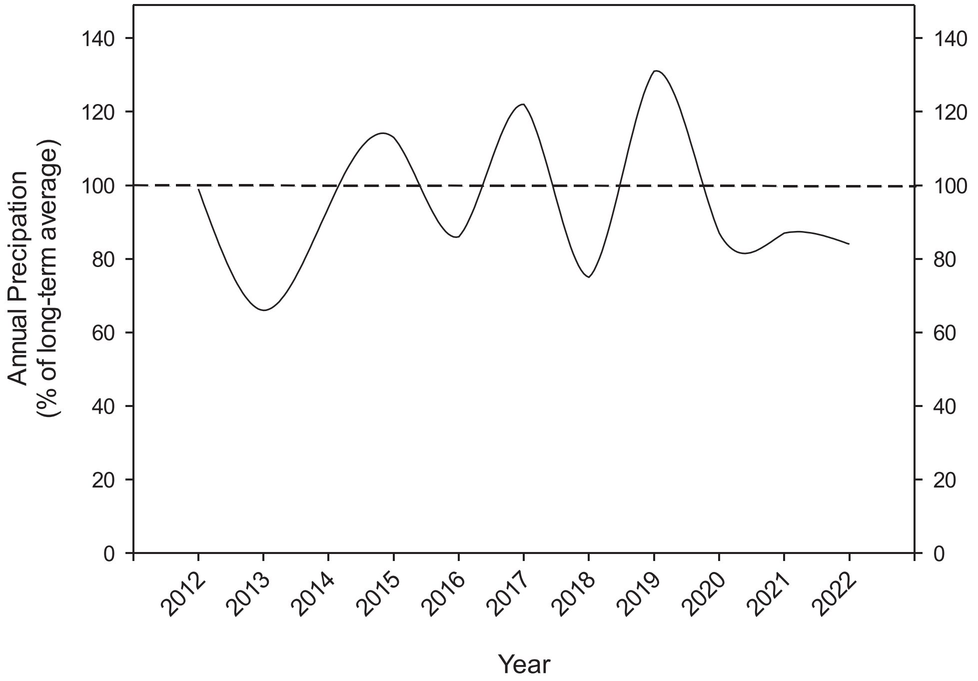

Figure 1. Annual precipation as a percent of the long-term (1991–2020) average for the study sites from 2012 to 2022.

sually estimated in 0.2-m^2 quadrats located at 3-m intervals on each 45-m transect, resulting in 15 quadrats per transect and 60 quadrats per treatment replicate. Bare ground and litter cover were also visually estimated in these quadrats. Quadrats were divided into 5%, 10%, 25%, and 50% sections using marking along their edges to improve the accuracy of visual cover estimates. Density of perennial herbaceous species was measured by counting individuals rooted inside the 0.2-m^2 quadrats. Density of annual species was measured by counting all individuals rooted inside a permanently marked 10% section of the 0.2-m^2 quadrats. No shrubs occurred in any of the treatment replicates, hence shrubs were not measured.

Statistical analysis

Repeated measures analyses of variances (ANOVAs) using the mixed-model method (PROC MIXED SAS v. 9.4) were used to investigate treatment effects. Random variables were block and block by treatment interactions, and year was the repeated variable in the analyses. Treatment was treated as a fixed variable in analyses. When there was a treatment-by-year interaction, we also analyzed the final year individually to determine the treatment effect at the conclusion of the study. Data were square root transformed when assumptions of ANOVAs were not met. Text and figures report nontransformed data. Appropriate covariance structure for analyses was selected using the Akaike's information criterion (Littell et al. 1996). Compound symmetry was selected for all models. Vegetation was categorized into five groups for analyses: large perennial bunchgrasses, Sandberg bluegrass, invasive annual grasses, perennial forbs, and annual forbs. The large perennial bunchgrass group was predominately Siberian and crested wheatgrass. Sandberg bluegrass was analyzed separate from the other perennial grasses because it is smaller in stature, develops earlier, and responds differently to management and disturbances (McLean and Tisdale 1972; Yensen et al. 1992; Davies et al. 2021). The invasive annual grass group was largely composed of medusahead with some cheatgrass. The annual forb group was predominantly nonnative species (> 90%). Treatment means were separated with the LS function (SAS v. 9.4), reported with standard errors, and were considered different at $P \leq 0.05$.

Results

Cover

Large perennial bunchgrass cover varied by the interaction between treatment and year (Fig. 2A; $P < 0.001$). The magnitude of the differences between the untreated control and revegetation treatments varied over time. At the conclusion of the study, perennial bunchgrass cover was 16.3- and 20.7-fold greater in the Herb + Med and Herb + High treatments compared with the Control treatment, respectively ($P < 0.001$). Perennial bunchgrass cover was on average greater in Herb + High compared with the Herb + Med treatment ($P < 0.001$). At the end of the study, large perennial bunchgrass cover was 1.3-fold greater in the Herb + High compared with the Herb + Med treatment ($P < 0.001$). Sandberg bluegrass cover did not vary among treatments ($P = 0.554$) but varied among years ($P = 0.024$). Sandberg bluegrass cover was low in all treatments (Control = 0.07% ± 0.02%, Herb + Med = 0.09% ± 0.02%, and Herb + High = 0.21% ± 0.07%). Invasive annual grass cover response to treatments was influenced by year (Fig. 2B; $P = 0.004$). Invasive annual grass cover was greater in the untreated control compared with the revegetation treatments, but the magnitude of difference was greatest in the first couple of years after treatment. At the conclusion of the study, annual grass cover was 2.2- and 2.3-fold greater in the untreated control compared with the Herb + Med and Herb + High treatments, respectively ($P = 0.023$ and 0.016, respectively). Average across all years, invasive annual grass cover was greater in the Herb + Med compared with the Herb + High treatment ($P = 0.018$); however, in the final year of the study we did not detect a difference ($P = 0.873$). Perennial forb cover did not vary among treatments ($P = 0.483$) but varied among years ($P < 0.001$). Annual forb cover response to treatment was influenced by year (Fig. 2C; $P < 0.001$). Annual forb cover was on average 3.2-fold to 3.3-fold greater in the untreated control than the Herb + Med and Herb + High treatments, respectively. How-

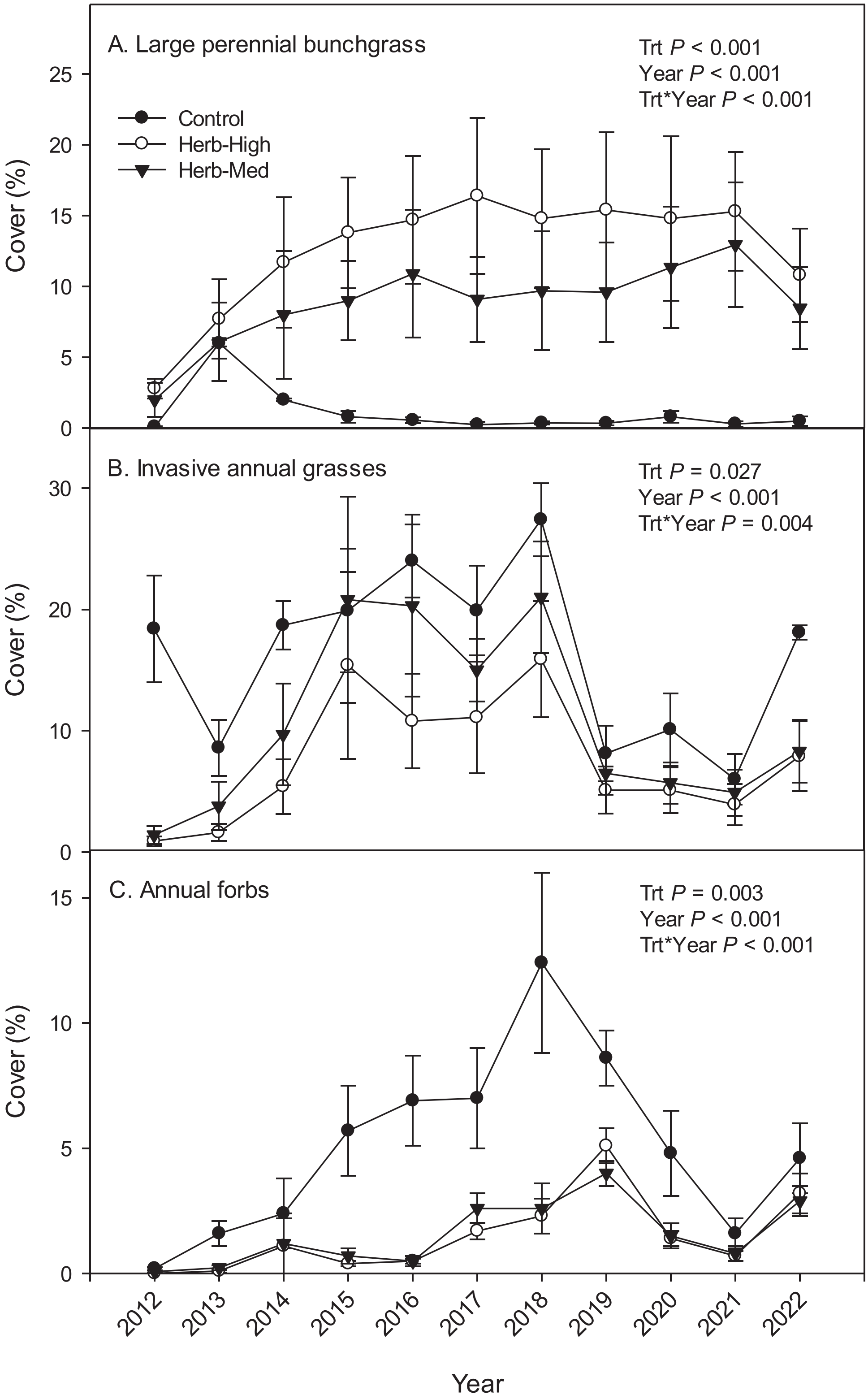

Figure 2. Cover (mean ± standard error) of large perennial bunchgrasses, invasive annual grasses, and annual forbs across treatments for 11 yr after seeding. Control indicates untreated control; Herb-Med, burning followed by imazapic application and 1 yr later seeded with perennial bunchgrasses at a medium seeding rate; Herb-High, burning followed by imazapic application and 1 yr later seeded with perennial bunchgrasses at a high seeding rate.

ever, by the end of the study, annual forb cover was similar between the Control and the Herb + High and Herb + Med treatments ($P = 0.354$ and 0.263). Annual forb cover did not differ between the Herb + High and Herb + Med treatments ($P = 0.978$). Bare ground and litter (predominately annual grass litter) cover were influenced by the interaction between treatment and year (Fig. 3A and 3B; $P < 0.001$). Bare ground was less, and litter cover was on average greater in the untreated control compared with the Herb + High and Herb + Med treatments ($P < 0.001$ and < 0.001, respectively), though in the final year of the study we did not detect a difference

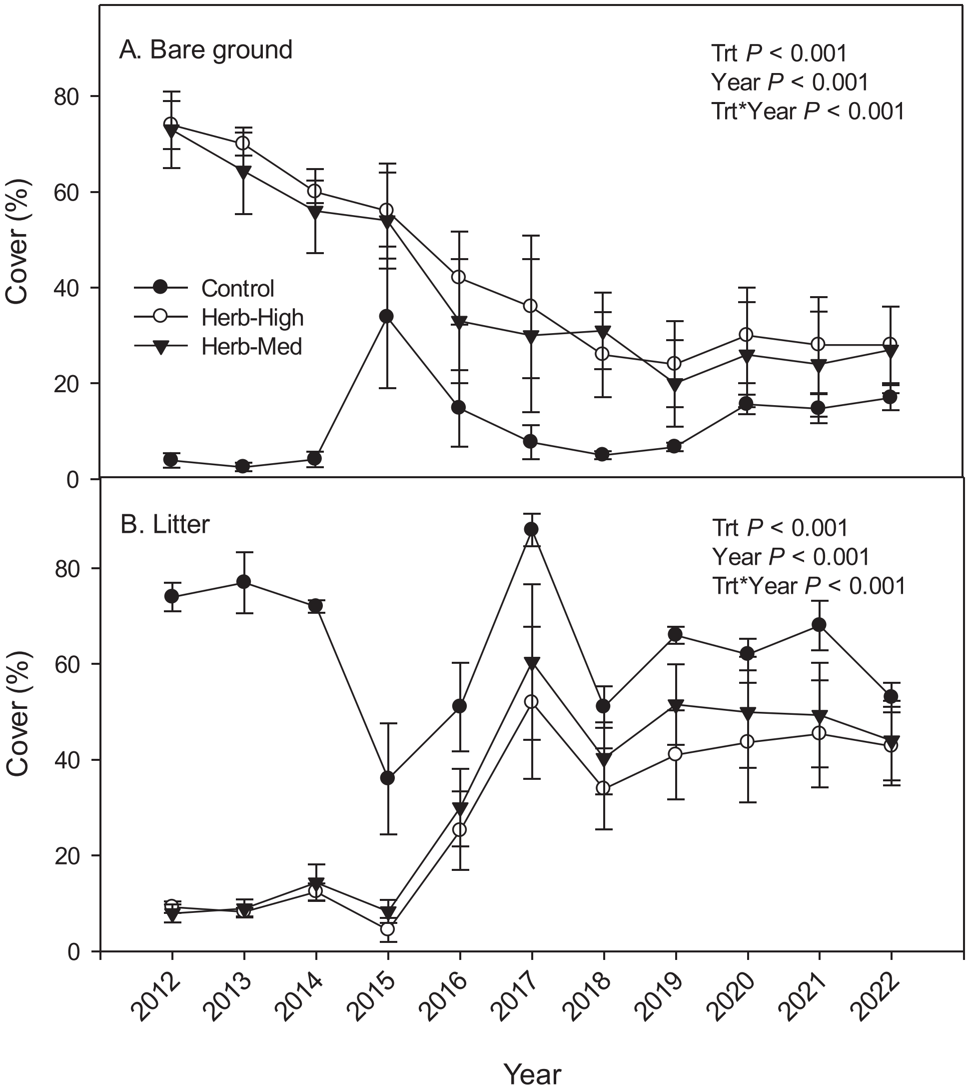

Figure 3. Bare ground and litter cover (mean ± standard error) across treatments for 11 yr after seeding. Control indicates untreated control; Herb-Med, burning followed by imazapic application and 1 yr later seeded with perennial bunchgrasses at a medium seeding rate; Herb-High, burning followed by imazapic application and 1 yr later seeded with perennial bunchgrasses at a high seeding rate.

between the untreated control and other treatments ($P > 0.100$). Bare ground and litter cover were similar between the Herb + High and Herb + Med treatments ($P = 0.436$ and 0.364, respectively).

Density

Large perennial bunchgrass density varied among treatments (Fig. 4A; $P < 0.001$) and years ($P = 0.049$). At the conclusion of the study, bunchgrass density was 4.2- and 4.3-fold greater in the Herb + Med and Herb + High treatments compared with the untreated control, respectively. On average, large perennial bunchgrass density was 1.3-fold greater in the Herb + High compared with the Herb + Med treatment ($P = 0.020$); however, by the end of the study, large perennial bunchgrass density was only 3% greater in the Herb + High treatment. Sandberg bluegrass density was similar among treatments ($P = 0.500$) but varied among years ($P = 0.020$). Invasive annual grass density response to treatments varied by year (Fig. 4B; $P < 0.001$). Annual grass density was greater in the untreated control compared with the other treatments, but the magnitude of difference decreased after the first couple of years post treatment. However, annual grass density was still greater in the untreated control compared with Herb + Med and Herb + High treatments at the conclusion of the study ($P =$ 0.039 and 0.009, respectively). Invasive annual grass density was on average 1.3-fold greater in the Herb + Med compared with the Herb + High treatment ($P = 0.004$). At the conclusion of the study, we did not find evidence that annual grass density varied between the Herb + Med and the Herb + High treatments ($P = 0.395$), even though numerically it was 1.2-fold greater in the Herb + Med treatment. Perennial forb density did not vary among treatments ($P = 0.983$) but varied among years ($P < 0.001$). Annual forb density response to treatments varied by year (Fig. 4C; $P < 0.001$). Annual forb density was initially much greater in the untreated control compared with the other treatments, but by the end of the study it was similar among treatments ($P > 0.100$). Annual forb density was similar between the Herb + High and Herb + Med treatments ($P = 0.789$).

Discussion

Revegetation efforts in medusahead-invaded rangelands had positive effects on plant community composition for more than a decade. Prescribed burning before imazapic control of medusahead followed 1 yr later with seeding perennial bunchgrasses re-

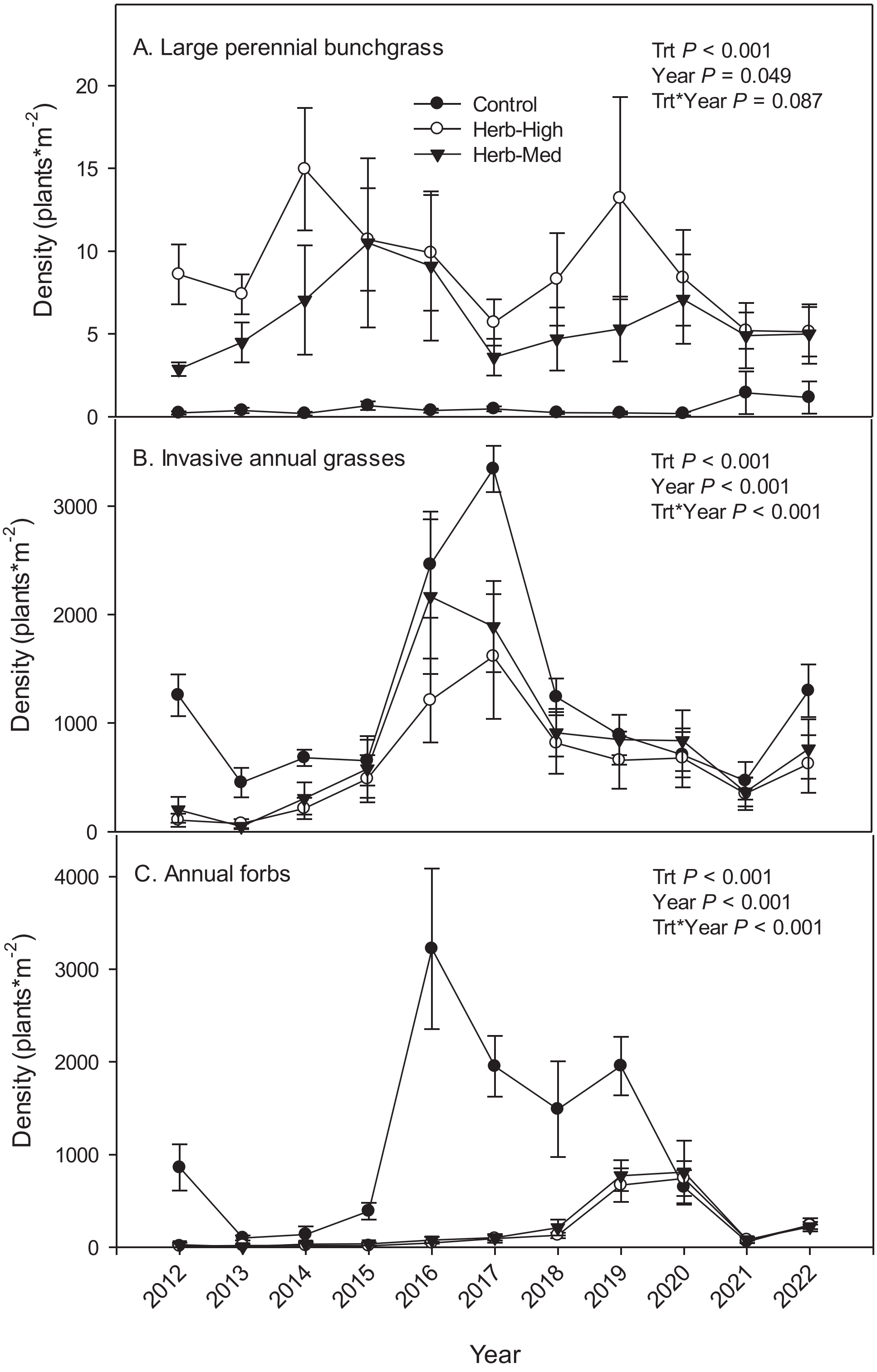

Figure 4. Density (mean ± standard error) of large perennial bunchgrasses, invasive annual grasses, and annual forbs across treatments for 11 yr after seeding. Control indicates untreated control; Herb-Med, burning followed by imazapic application and 1 yr later seeded with perennial bunchgrasses at a medium seeding rate; Herb-High, burning followed by imazapic application and 1 yr later seeded with perennial bunchgrasses at a high seeding rate.

sulted in lasting increases in large perennial bunchgrass cover and density. This, subsequently, limited invasive annual grasses and annual forbs (predominately non-native species). Shorter-term research has shown similar results with controlling medusahead with preemergent herbicides and seeding perennial bunchgrasses (e.g., Davies 2010; Sheley et al. 2012a; Davies et al. 2015, 2018). However, when perennial vegetation fails to establish after herbicide control of medusahead, invasive annual grasses redominate the plant community (Monaco et al. 2005; Davies et al. 2014; James et al. 2015). These results suggest that establishing perennial vegetation after invasive annual grass control is critical to having lasting desirable outcomes (i.e., increased perennial vegetation and decreased invasive annuals), unless enough residual perennial vegetation is present to substantially increase with the reduction in invasive annual grasses (e.g., Davies and Sheley 2011; Lazarus and Germino 2022; Davies et al. 2023).

Perennial bunchgrass seeding rate had long-lasting effects on invasive annual grasses and large perennial bunchgrasses, though in the final year annual grass cover and density were similar among seeding rates. Large perennial bunchgrass cover and density was greater at the high compared with the medium seeding rate, and this was generally associated with less invasive annual grass cover and density. Our results support speculation that seeding rates may need to be high to revegetate invasive plant–invaded rangelands (Jacobs et al. 1996; Sheley et al. 1999). Further supporting this, seeding rates needed to be high to limit invasive annual grasses after control treatments in a short-term study in Idaho (Schantz et al. 2019). Because invasive annual grasses are highly competitive with perennial bunchgrass seedlings (Nasri and Doescher 1995; Rafferty and Young 2002; Humphrey and Schupp 2004; Young and Mangold 2008), initial establishment from seed while annual grass competition is limited from herbicide treatments is likely critical to long-term outcomes. Thus, a high seeding rate may increase the likelihood that perennial bunchgrass seed is available when conditions are favorable for establishment and that enough bunchgrasses establish to limit annual grasses.

Though we didn't test if greater plant functional diversity in the seed mix would further limit invasive annual grasses, it would likely be worth pursuing in revegetation efforts. Perennial vegetation, in general, limits invasive annual grasses (Davies and Johnson 2017), thus we expect that seeding Sandberg bluegrass and likely shrubs and perennial forbs could further limit reinvasion by invasive annual grasses. Adding Sandberg bluegrass to revegetation seed mixes may be particularly valuable because it often grows in the interspace between large perennial bunchgrasses, where invasive annual grasses initially invade plant communities (Reisner et al. 2013; Rayburn et al. 2014). We observed this pattern of reinvasion in our treated plots, with medusahead establishing in the interspaces between large bunchgrasses. Sandberg bluegrass's resource acquisition patterns also overlap substantially with invasive annual grasses (James et al. 2008). More functionally diverse postrevegetation plant communities would also improve the habitat value for native wildlife (McAdoo et al. 1989; Kennedy et al. 2009). Thus, higher seeding rates and likely with more functionally diverse seed mixes could increase the resistance of the plant community to reinvasion and promote biodiversity.

The long-term nature of our study provides evidence that revegetation efforts have changed the trajectory of these plant communities from invasive annual grass–dominated to perennial bunchgrass–dominated or bunchgrass–annual grass codominated. In some years these communities appear to be perennial bunchgrass–annual grass codominated, especially in the medium seeding rate areas. However, increases in invasive annual grasses appear to be cyclic, likely increasing with weather favorable for annual grass growth, and decreasing in less favorable weather years. This fluctuation, compared with a steady increase, in invasive annual grass abundance and consistently less annual grass in revegetation areas compared with the untreated control is encouraging because it suggests that established perennial vegetation is limiting invasive annual grasses. This is indicative of a resilient system that is capable of retaining original conditions in the face of biotic and abiotic variability. These revegetation treatments are relatively small (30 × 50 m plots) and surrounded by near monocultures of medusahead, suggesting that these revegetation communities are likely experiencing substantially higher medusahead propagule pressure than would be expected in larger revegetation efforts typically applied by land managers. Most medusahead seeds disperse only short distances from parent plants (Davies 2008); thus, medusahead propagule pressure decreases as treatment size increases (assuming successful control of medusahead) because of longer distances to seed sources. High propagule pressure increases the success of invasive annual grasses (Davies 2008; Schantz et al. 2015). Thus, limiting invasive annual grasses in these small research plots for over a decade suggests that larger treatment areas, typical of land management practices, would likely be even more effective. Annual forb cover was also reduced for more than a decade with revegetation efforts. Substantially less invasive annual grass cover and density and annual forb cover in the revegetation treated areas compared with untreated control after more than a decade indicate that the establishment of large perennial bunchgrasses after annual grass control has shifted the plant community to a perennial-dominated system.

Large perennial bunchgrass density in the revegetation treatments fluctuated over time. Initially, it appeared that self-thinning or, potentially, increases in invasive annual grasses were reducing large perennial bunchgrass density after it peaked in the third yr post seeding. Self-thinning is often expected because as plants mature and grow larger, they compete with each other and may self-thin to a level that can be supported by the site (Mueggler and Blaisdel 1955). However, large perennial bunchgrass density increased after its initial decrease and then decreased again, showing two peaks and two troughs. These peaks and troughs may be associated with weather conditions. The two peaks in bunchgrass density are evidence that perennial bunchgrasses were recruiting new individuals into the community, which is essential to their persistence. Recruitment of dominant perennial plant groups, especially large perennial bunchgrasses in this ecosystem, is key to successful revegetation.

Litter cover was lower in revegetation treatments compared with untreated controls, likely favoring perennial bunchgrasses compared with invasive annual grasses. Litter on the soil surface can be a barrier to perennial plant establishment (Evans and Young 1970), though exceptions may exist (Wolkovick et al. 2009). In contrast, soil surface litter creates a microclimate that is conducive to invasive annual grass emergence and growth (Evans and Young 1970; Facelli and Pickett 1991; Newingham et al. 2007; Adair et al. 2008; Wolkovich et al. 2009). The increased likelihood of frequent fire with greater amounts of litter and its negative impacts on native perennials (D'Antonio and Vitousek 1992) further suggests less litter in the revegetation treatment areas is desirable. Because of the critical role perennial bunchgrasses play in these communities (Davies et al. 2006; Chambers et al. 2007) and the negative impact of litter on bunchgrasses, less litter in the revegetation treatments likely contributes to greater resilience and resistance (Chambers et al. 2014).

Sandberg bluegrass, shrubs, and perennial forbs were not influenced by the revegetation treatments. This suggests that the seedbank for these plants may need to be augmented with seeding to elicit increases in their abundance. Sandberg bluegrass, shrubs, and perennial forbs often do not increase with the control of medusahead and other invasive annual grasses (Davies 2010; Davies and Boyd 2018; Davies and Hamerlynck 2019; Davies et al. 2023). Adding these plant groups to the seed mix may increase their abundance in the postrevegetation plant communities and, as mentioned earlier, may further limit invasive species. However, the addition of Sandberg bluegrass, shrubs, and perennial forbs to revegetation seed mixes needs to be investigated. Optimal seeding rates and techniques need to be investigated, as well as determining which species and ecotypes should be used at different sites to achieve management objectives.

Implications

Control of invasive annual grasses with prescribed burning and imazapic application, followed 1 yr later with drill-seeding large perennial bunchgrasses, can have long-lasting positive outcomes in medusahead-invaded rangelands. These revegetation treatments shifted the plant communities from annual to perennial plant dominance. There is also evidence that perennial bunchgrasses in revegetation treatments were recruiting new individuals into the plant community, suggesting that these benefits will continue to persist. The high compared with medium seeding rate resulted in greater large perennial bunchgrass density and cover and generally less invasive annual grass cover and density. This suggests that higher seeding rates may be warranted. Nonseeded plant groups (Sandberg bluegrass, shrubs, and perennial forbs) did not increase with medusahead control, suggesting that they may need to be seeded to improve revegetation efforts. Reestablishing these plant groups would increase the value of these plant communities to native wildlife and may further limit invasive annual grasses. The results of this study suggest that the revegetation of medusahead-invaded rangelands can promote long-term restoration of ecosystem goods and services, though refinement of revegetation treatments may further improve forage production, habitat value, increase invasion resistance, and promote biodiversity.

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