Rangeland Ecology & Management 101 (2025) 43–53

Contents lists available at ScienceDirect

Rangeland Ecology & Management

journal homepage: www.elsevier.com/locate/rama

Response of Cheatgrass and Other Vegetation to Proactive Management Using Two Consecutive Fall Herbicide Applications in High-Elevation Sagebrush Steppe☆,☆☆

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ARTICLE INFO

Article history:
Received 15 May 2024
Revised 11 February 2025
Accepted 26 February 2025

Key Words:
Bromus tectorum
Defend the core
High-elevation
Imazapic
Sagebrush conservation
Weed management

ABSTRACT

Nonnative plants are one of the foremost concerns to public and private land managers. To help guide managers, conservation and nonnative plant management methodologies have been blended to help prioritize management under the “defend the core” management framework. This approach emphasizes the need to control low-abundance infestations of nonnative plants before they become costly to manage and result in irreversible ecosystem alterations. However, few studies have explored the impacts and efficacy of nonnative plant management when target species occur in low abundance. Our study focused on one of the most prolific invaders in the western United States, cheatgrass (*Bromus tectorum*). We found that at the northern edge of cheatgrass’s historical range, two consecutive fall applications of the herbicide imazapic (Plateau; BASF Corporation, Research Triangle Park, NC) controlled low-abundance populations of cheatgrass for three or more years. However, cheatgrass reductions did not result in an increase in native plant richness, native plant abundance, or native perennial grass productivity, and ultimately, cheatgrass returned to pretreatment abundances in one of two areas. As a result, we conclude that imazapic can be an effective tool to control cheatgrass when it occurs at low abundance in high-diversity plant communities, i.e., “core” sagebrush areas, within the Middle Rockies ecoregion. To defend the core, we recommend herbicide as a tool within a broader adaptive and integrated management plan that includes alterations to factors that managers can control, e.g., stocking rates, timing, placement of grazing infrastructure, and off-road travel. Lastly, we suggest that monitoring should be conducted throughout the management process so managers can adapt to the real-time responses of plant communities.

Introduction

For more than 25 yr, proactive management of nonnative plants has been suggested for rangelands (Sheley 1995; DiTomaso 2000). Proactive management frameworks, such as early detection and rapid response (hereafter EDRR; United States Geologic Survey 2024), are promoted for their potential ability to reduce the impact of nonnative plants before populations become uncontrollable and impart landscape-scale impacts (Crist et al. 2019; Doherty et al. 2022), which could result in significant management costs (Sheley and Smith 2012; Mealor et al. 2013). For example, the “defend the core” management framework is being promoted within the sagebrush (*Artemisia spp.* L.) biome to prioritize the conservation of core sagebrush areas. This framework prioritizes the conser-

☆ This research was funded by the Montana Noxious Weed Trust Fund Grant (MDA-2019-013) and the National Institute of Food and Agriculture (NIFA), U.S. Department of Agriculture (USDA) under award number 2023-38640-39571 through the Western Sustainable Agriculture Research and Education program under project number WSARE SW20-915/GW20-204. J.M., C.Z., and L.J.R. are supported by the National Institute of Food and Agriculture, USDA Hatch (MONB00359, MONB00397, and MONB00363, respectively). The USDA is an equal opportunity employer and service provider. Any opinions, findings, conclusions, or recommendations expressed in this publication are those of the author(s) and do not necessarily reflect the view of the USDA.

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https://doi.org/10.1016/j.rama.2025.02.006
1550-7424/Published by Elsevier Inc. on behalf of The Society for Range Management.

vation of core areas of the sagebrush biome, that is, areas with low levels of invasion rather than highly invaded and degraded areas. Core conservation areas are defined by a combination of sagebrush species and subspecies, sagebrush cover, native plant understory cover, lack or low levels of nonnative annual grass cover, lack of conifer encroachment, and little human modification (Doherty et al. 2022).

Despite the call for land managers to adopt a proactive strategy to control new and low-abundance nonnative plant populations, little empirical documentation of the effectiveness of such an approach exists. For example, researchers' and managers' current understanding of nonnative annual grass management and the response of sagebrush plant communities to management, e.g., herbicide, has generally been from highly infested areas (e.g., Mangold et al. 2013). Wood and Mealor (2022) is one of the few studies that have documented the response of plant communities to management across a nonnative annual grass gradient. Evaluating how plant communities respond to management in areas with low nonnative annual grass abundance is an important knowledge gap to address because land managers are currently being prompted to adopt a proactive management strategy and defend core areas of the sagebrush biome from nonnative plant invaders like cheatgrass (*Bromus tectorum* L.).

Cheatgrass is a widespread nonnative winter annual grass that has been present for nearly two centuries in the American West (Mack 1981) and, as a result, represents one of the primary nonantive plant threat to the sagebrush biome (Evans and Young 1977; Young and Clements 2009; Remington et al. 2021; Doherty et al. 2022). Cheatgrass often establishes following different types of disturbance such as increased land development (Requena-Mullor et al. 2023), surface disturbance (Lavin et al. 2013), improper grazing by domestic livestock and feral horses (Mack 1981; Knapp 1996; Davies and Boyd 2019), and climate change (Compagnoni and Adler 2014; Blumenthal et al. 2016). However, the primary mechanism by which cheatgrass imparts landscape level change is by increasing the frequency of wildfire (Young et al. 1987; Brooks et al. 2004; Balch et al. 2013; Chambers et al. 2016; Pilliod et al. 2017), which can result in reduced perennial grass abundance, which is important for wildlife and cattle forage (Young and Evans 1978; Reisner et al. 2013) and competition with other vegetation, particularly perennial grasses (De Stefano et al. 2024). However, fire does not necessitate cheatgrass expansion or increased establishment (Smith et al. 2023). Increased fire frequency can reduce the cover of woody shrubs, notably species of sagebrush (Coates et al. 2016), which provide critical habitat for wildlife, including Greater sage-grouse (*Centrocercus urophasianus*) and Brewer's sparrow (*Spizella breweri*; Connelly et al. 2000; Williams et al. 2011).

Historically, the lower Snake River Plain, North Central Basin and Range, and the Central Basin and Range ecoregions of the western United States have been most impacted and well-studied with regard to cheatgrass invasion and management (Omernik 1987; Chambers et al. 2007, 2016; Maestas et al. 2016). However, recent attention has been given to the Middle Rockies ecoregion due to projected habitat suitability and the occurrence of cheatgrass (Bradley et al. 2018; Smith et al. 2022; Chambers et al. 2023a). Interestingly, cheatgrass does not appear to have the same ecosystem impacts in the upper Snake River Plain and Middle Rockies as it does in the other ecoregions. For example, there is little evidence for increased wildfire frequency, i.e., a positive fire feedback cycle in the Middle Rockies (Taylor et al. 2014; Seipel et al. 2018). Instead, it appears that physical disturbance (e.g., roads and other infrastructure) plays a greater role in native plant community structure than cheatgrass and its associated fire feedback cycle (Lavin et al. 2013; Seipel et al. 2018). These findings are likely due to the colder and wetter climate of the Middle Rockies compared with the other lower and drier elevation ecoregions (e.g., the North and Central Basin and Range) (Chambers et al. 2014, 2019, 2023a). For example, native plant productivity, increased soil moisture, and cooler temperatures have been correlated with higher resilience and resistance of the sagebrush biome and are key characteristics of mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) communities common to the Middle Rockies (Chambers et al. 2014, 2023a). As a result, mountain big sagebrush communities are often less degraded by nonnative annual grasses; under the defend the core management framework, these areas should be targeted for management. Thus, there is a need for an increased understanding of the impact that nonnative annual grass management, e.g., herbicide, will have on core areas when nonnative annual grasses are at low abundance.

Herbicide is the most common and widespread method of nonnative plant control (Radosevich et al. 2007; Mangold et al. 2018). For the last 23 yr, imazapic has been used to control nonnative annual grasses with mixed success. For example, reductions in cheatgrass were found to be site- and rate-dependent, varying from 10% to 60% cheatgrass cover ..in the first growing season following a fall application of imazapic in salt desert scrub and Wyoming big sagebrush plant communities (Morris et al. 2009). In contrast, cheatgrass frequency was reduced for 3–4 yr compared with the nonsprayed treatment following an early October application of imazapic in the Columbia Basin (Elseroad and Rudd 2011). Mangold et al. (2013, 2015) assessed the efficacy of imazapic timing and different rates across 24 sites in Montana to inform land managers of best practices and found that postemergent, fall application of imazapic at 105 g · active ingredient (ai) · ha^{-1} was most successful in this region (Middle Rockies). Since then, two consecutive yearly postemergent applications of imazapic have been recommended to exhaust cheatgrass seedbanks and avoid nontarget impacts to native vegetation because the perennial vegetation has senesced at this time in Montana (Personal communication with Mangold). This fall application strategy, at the low end of the label recommended rate, avoids most nontarget impacts. Nontarget impacts are of great concern in areas that contain a higher proportion of native plants that are considered of high quality to wildlife, i.e., core sagebrush areas.

Cost is also an important factor in developing a nonnative plant management strategy. In the United States alone, billions of dollars are spent annually to control and manage nonnative plant species (Ditomaso et al. 2010). As a result, it is paramount that land managers be provided with clear information on the efficacy of herbicide treatments, including both reductions in target species and duration of control, as well as the response of the native plant community.

Our study sought to provide land managers with timely and critical information about the efficacy of herbicide treatments to control low-abundance cheatgrass and any effects on the native plant community. The study was conducted in a high-elevation sagebrush landscape within the Middle Rockies ecoregion, i.e., core sagebrush habitat. In this research, we assess 1) the efficacy of two consecutive postemergence fall herbicide (imazapic) applications to reduce cheatgrass abundance over a 4-yr period and 2) the response of the a) plant community composition and b) perennial grass productivity. We hypothesized that the two applications of imazapic would result in a short-term (1–2 growing seasons) reduction of cheatgrass abundance, after which cheatgrass would return to prior abundances. Additionally, we hypothesized that the reduction in cheatgrass abundance would result in short-term increases in plant species richness, the proportion of native vegetation, and perennial grass productivity.

Table 1
Site locations (centroid coordinates) and soil texture descriptions for each site in the Antelope Peak (AP) and Eastern Sandhills (ES) areas, Centennial Valley, MT, USA. Site 16 in the AP region encompassed two distinct soil types; as a result, both are listed below. A soil ribbon test was conducted to approximate the soil texture in the field (Arshad et al. 1997).

Region	Site	Latitude	Longitude	Elevation (meter/feet)	Soil texture
AP	16	44.65958603	−111.9954618	2089/6853	Sandy clay loam
AP	15	44.66054363	−112.0015361	2135/7004	Gravelly sand loam
AP	14	44.65813905	−112.0055019	2131/6991	Silty clay
AP	11	44.65858126	−112.0165624	2176/7139	Silty clay
AP	3	44.65876424	−112.0404955	2168/7112	Gravelly silt loam
AP	2	44.66065154	−112.044243	2176/7139	Silty clay loam
AP	21	44.65637509	−112.0095762	2293/7522	Silty loam
AP	22	44.67905552	−112.0189539	2381/7811	Silty loam
ES	1	44.68076541	−111.6899827	2137/7011	Sand
ES	2	44.67944641	−111.6886901	2101/6893	Sand
ES	5	44.69461276	−111.7088441	2169/7116	Sand
ES	6	44.65939491	−111.6707083	2057/6748	Sand

Methods

Study area

Our study was conducted in the Middle Rockies ecoregion, specifically in the Centennial Valley of southwestern Montana, USA, a 65 000-ha high-elevation : (2000–2500m above sea level) mountain big sagebrush steppe valley typified by cold winters and short, cool summers (Cutting et al. 2019). The Centennial Valley has been utilized for cattle grazing since the 1870s (Vivion 1968). At the time of this study, the average stocking rate across the research area ranged from 1.2 to 2.0 ha per adult cow (453 kg weight), depending on the property and grazing permit (Cutting et al. 2019). Two areas characterized by distinct soils and dominant vegetation exist within the Centennial Valley's foothills. The Antelope Peak (AP) area is comprised of primarily clay/loam soil, while the Eastern Sandhills (ES) are composed of mostly sandy soil (Table 1). The 30-yr average annual precipitation is comparable between both AP (181 mm) and ES (190 mm) areas (Table 2). The flora of the AP area is dominated by a mixture of sagebrush and shrub types: mountain big sagebrush (*Artemisia tridentata* spp. *vaseyana* Nutt.), three-tip sagebrush (*Artemisia tripartite* Rydb.), fringed sagebrush (*A. frigida* Wild.) and rabbitbrush (*Chrysothamnus nauseosus* Nutt.; Culver 1994), and native perennial bunchgrasses, namely needle and thread grass (*Hesperostipa comata* Trin. & Rupr.) and Idaho fescue (*Festuca idahoensis* Elmer; Cutting et al. 2019). The ES area is dominated by thickspike wheatgrass (*Elymus lanceolatus* Scribn. & J.G. Sm.), silver leaf phacelia (*Phacelia hastata* Douglas ex Lehm.), mountain big sagebrush, basin big sagebrush (*A. tridentata* ssp. *tridentata*), and Idaho fescue (Culver 1994). Across both areas and nearby mountains, approximately 487 plant species (~21% of Montana's native flora) have been identified, including regional endemic species and those occurring at their range extents (Culver 1994). Twelve steep, south-facing hillslopes, averaging 4 ha each, were selected as sites. Each site consisted of an area that was infested by cheatgrass. Explicitly, the pretreatment (2017) cheatgrass cover was 13% in the AP area and 8% in the ES area.

Experimental treatment

In early October 2017 and 2018, after cheatgrass had emerged and was at the 2–3 leaf stage, the herbicide imazapic was aerially broadcast at 105 g · ai · ha^{-1} with a carrier rate of 28 L of water per hectare, 25 PSI pressure, and an aerial speed of 65–70 kmph by helicopter. The low average shrub cover, 4% in AP and 3% in ES, suggests minimal refuge for cheatgrass was provided by shrubs. At each site, paired 10 m^2 plots were established; one was

Table 2
Average seasonal precipitation (mm) for the Antelope Peak (AP) and Eastern Sandhills (ES) areas, Centennial Valley, MT, USA. Precipitation estimates were averaged across sites within each area (AP and ES). Nonbold text indicates the seasonal precipitation, and bold text indicates the departure from the 30-yr average (bottom), with positive values indicating above-average precipitation and negative values below-average precipitation. Data were downloaded from the PRISM climate group (https://prism.oregonstate.edu/explorer/).

Precipitation (mm)

Antelope Peak

Growing season	Fall	Winter	Spring	Summer	Annual
2017	64.4	31.6	60.6	15.7	172.3
	23.4	**−12.7**	**4.3**	**−24.2**	**−9.1**
2018	50.5	52.9	55.4	34.2	193.1
	6.6	**4.8**	**−3.3**	**−5.6**	**2.5**
2019	47.6	68.7	37.5	35.8	189.5
	6.6	**24.4**	**−18.7**	**−4**	**8.2**
2020	23.3	42.8	43.3	42.6	152
	−17.7	**−1.5**	**−13**	**2.8**	**−29.4**
2021	53.4	57.7	40.3	28.7	180.1
	12.4	**13.4**	**−16**	**−11.2**	**−1.3**
2022	53.4	46.4	62.4	42.3	204.5
	12.4	**2.1**	**6.1**	**2.4**	**23.1**

Eastern Sandhills

Growing season	Fall	Winter	Spring	Summer	Annual
2017	70.8	39.9	68.5	16.4	195.6
	64.2	**35.1**	**71.8**	**22.0**	**193.1**
2018	55.1	60.8	61.4	32.0	209.3
	7.5	**−7.9**	**23.9**	**−3.8**	**19.8**
2019	56.6	67.8	41.9	36	202.3
	12.7	**19.7**	**−16.8**	**−3.8**	**11.8**
2020	25.4	49.6	50.8	44.4	170.3
	−18.4	**1.5**	**−7.9**	**4.6**	**−20.2**
2021	58.1	65.5	46.2	31.8	201.6
	14.3	**17.4**	**−12.5**	**−8**	**11.1**
2022	58.1	40.7	70.3	48.6	217.7
	14.3	**−7.4**	**11.6**	**8.8**	**27.2**

30-yr average

Area	Fall	Winter	Spring	Summer	Annual
AP	41	44.3	56.3	39.9	181.4
ES	43.9	48.1	58.7	39.8	190.6

sprayed, and the other was not sprayed. The nonsprayed control was achieved by placing a square plastic tarp (3.16 m × 3.16 m; 10 m^2) on the ground before herbicide application and removing it within 24 h in both years. Sprayed and adjacent nonsprayed plots were separated by 1 m. These plots were permanently marked for subsequent sampling.

Vegetation sampling

Vegetative canopy cover of all plant species and nonvegetative cover (rock, bare ground, and litter) were sampled using five Daubenmire frames (20 cm × 50 cm; 0.1 m^2; Daubenmire 1959) that were randomly placed within each 10 m^2 sprayed and nonsprayed treatment area. Species were identified and recorded using the nomenclature and authority of the Manual of Montana Vascular Plants (Lesica 2022) and the United States Department of Agriculture Plant Database (United States Department of Agriculture 2024). Additionally, we used the four-letter United States Department of Agriculture Plant Database codes for Figure 3 and 4 (United States Department of Agriculture 2024). Cover was estimated to the nearest 1%. Yearly monitoring was initiated in mid-June and concluded around 1 July, a time that corresponds with peak plant species richness during the growing season.

Every frame was surveyed annually in both the nonsprayed and sprayed treatments from 2019 to 2022. Additionally, cheatgrass cover alone was recorded in the nonsprayed treatment in 2017 and 2018. The total sample size of the nonsprayed treatments recorded over 6 yr was 240 frames (5 frames per site, 8 sites, 6 yr) for the AP area and 120 for the ES area (5 frames per site, 4 sites, 6 yr). The sprayed treatment plots were monitored post herbicide application (4 yr), and the sample size was 160 in the AP area (5 frames per site, 8 sites, 4 yr) and 80 in the ES area (5 frames per site, 4 sites, 4 yr).

Perennial grass biomass collection

To correlate native perennial grass cover with its biomass, we harvested the five most abundant perennial grass species based on estimated cover: needle and thread grass, thickspike wheatgrass, bluebunch wheatgrass (*Pseudoroegneria spicata* Pursh.), Sandberg bluegrass (*Poa secunda* J. Presl.), and Idaho fescue. Each species was clipped at ground level at the end of the cover sampling campaign (second week of July), occurring near the time of peak standing crop. Clipped biomass was dried for 72 h at 49°C and weighed to the nearest 0.1 g at the Montana State University Plant Growth Center (Bozeman, MT, USA).

Statistical analysis

Cheatgrass abundance was found to be zero-inflated (53% zeroes), so we employed a hurdle model with the "pscl" package (Zeileis et al. 2008). The hurdle model assumed a negative binomial distribution and included herbicide (sprayed/nonsprayed), area (AP/ES), and year as fixed effects. To test for differences in cheatgrass abundance between preherbicide and 4 yr postherbicide, a Dunnett's test was conducted using the "emmeans" package (Lenth 2024). The "pscl" package did not support random effects for the hurdle model; as a result, sites were combined for this analysis.

Plant species richness was modeled using a linear mixed-effects model with Poisson distribution with the "glmmTMB" package (Brooks et al. 2017). The model used herbicide, area, and year as explanatory fixed effects variables with site included as a random effect. Cheatgrass was omitted from the species richness analysis because it was a target species for removal and could artificially create differences between sprayed and nonsprayed treatments. Seven native forb genera were not identifiable to the species or life-cycle level and were removed from the analysis. No other species were removed from the analysis. Rank abundance was calculated by summing the cover of each species for each year, herbicide, and area combination using the "BiodiversityR" package (Kindt and Coe 2005). The top five most abundant plant species from each area, year, and treatment were plotted graphically to aid the discussion of the effects of treatment on plant community composition. The top five species were chosen because average species richness varied between 4 and 5 species per 0.1 m^2 frame.

When cover for each species was observed and estimated in the field, the total cover was allowed to exceed 100% of the frame due to species overlapping canopies. As a result, we analyzed the proportional cover of each species relative to the total cover of the frame. Differences in the proportion of functional groups were assessed for treatments and area. The seven vegetative functional groups included introduced (nonnative) annual forb, introduced (nonnative) annual grass, introduced (nonnative) perennial forb, native annual forb, native perennial forb, native shrubs, and native perennial grass, and three nonvegetative groups (litter, rock, and bare ground) were also evaluated. Functional group cover was calculated by summing the cover of each group and dividing by the total cover for each treatment and area. Proportions were then multiplied by 100 and expressed as percentages; this was done to aid in the interpretation and discussion of results. The proportion of vegetative and nonvegetative functional group abundance was modeled using a linear mixed-effects model with a beta distribution in the "glmmTMB" package (Brooks et al. 2017). The model used herbicide, area, and year as explanatory fixed effects, with site included as a random effect.

The biomass of the dominant native perennial grass species was modeled using linear models. The model estimates were then used to predict the biomass from each year of the study using the observed cover. We then tested each species' biomass response to herbicide, area, and year. Differences between the predicted mean biomass for species, area, year, and herbicide were compared using a post hoc pairwise test in the "emmeans" package (Lenth 2024). All statistical analyses were conducted in the software R (version 4.2.2, R Core Team 2022).

Results

Precipitation trend

Overall, we observed similar trends in departure from the 30-yr average precipitation in the AP and ES areas, though the extent of these departures differed in magnitude within each area (Table 2). A single difference in pattern occurred in winter 2020, during which time the ES area received less precipitation than average, and the AP area received more (Table 2). Spring precipitation was consistently drier than average in both AP and ES areas from 2019 to 2021 and wetter than average in 2022 (Table 2). Fall precipitation was also consistent between areas and wetter than average in 2019, 2021, and 2022 and drier than average in 2020 (Table 2). On an annual basis, the AP area was wetter than average in 2019 and 2022 and drier than average in 2020 and 2021 (Table 2). The ES area was wetter than average in 2019, 2021, and 2022 and drier than average in 2020 (Table 2).

Cheatgrass cover

Preherbicide cheatgrass cover was 13% in the AP area and 8% in the ES area. Cheatgrass cover responded differently depending on herbicide treatment, area, and year ($p = 0.039$; Fig. 1; Table S1; available online at xxx). Herbicide reduced the cover of cheatgrass in the AP area for 3 yr following the two fall herbicide applications. Mean cheatgrass cover ranged from < 1% to 12% in the sprayed treatment and was lower than in the nonsprayed treatment until the fourth year (2022), at which time there was no difference between the two (Fig. 1). In the nonsprayed treatment, cheatgrass cover ranged from 5% to 20% over 4 yr (2019–2022), declining naturally due to poor growing conditions from 2019 to 2021 before increasing in the fourth year (Fig. 1; Table S2; available online at xxx). In the ES area, cheatgrass cover was also reduced by herbicides. Mean cheatgrass cover in the sprayed treatment ranged from < 1% to 1% over 4 yr and remained lower than in the nonsprayed treatment for the duration of the study (Fig. 1; Table S2). In the nonsprayed treatment, the mean cheatgrass cover ranged from 3% to 6% over 4 yr (Table S2).

Plant richness and rank abundance

A total of 124 native and nonnative plant species were observed across the study. The overall mean species richness increased from 3.4/0.1 m^2 in the first year to 4.7 in the fourth year ($p < 0.0001$; Fig. 2; Table S3; available online at xxx). There were no differences detected in species richness between areas ($p = 0.259$) or herbicide treatment ($p = 0.358$). Native perennial forbs were the most species-rich (65) functional group, followed by native perennial grasses (18), native shrubs (13), native annual forbs (11), nonnative annual forbs (5), nonnative perennial forbs (2), and nonnative annual grass (1).

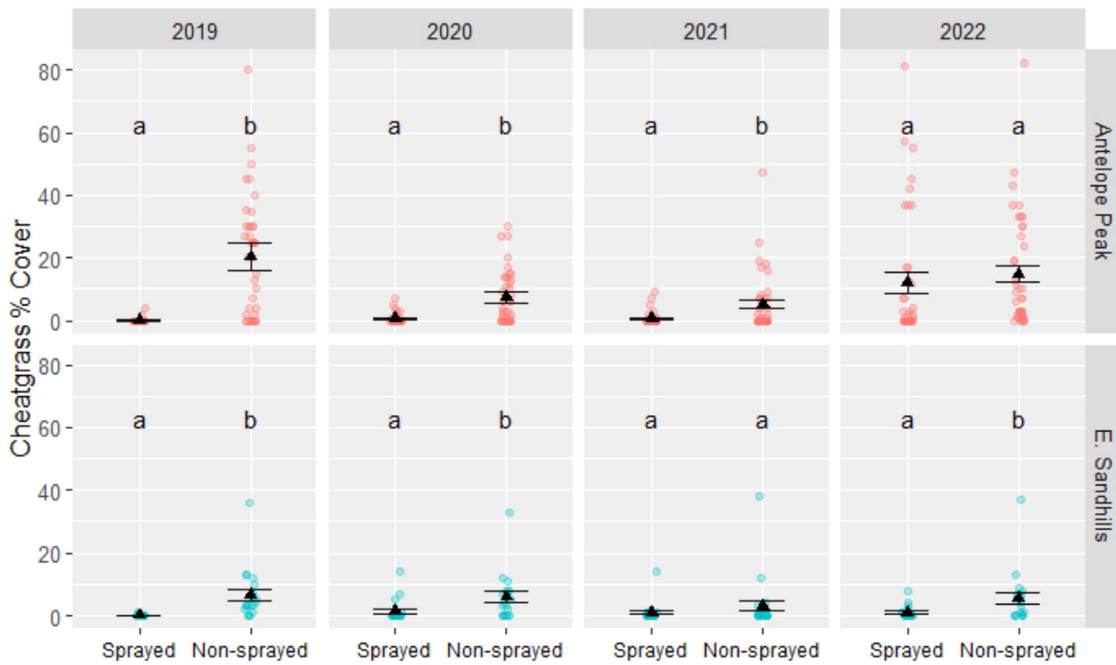

Figure 1. Estimated cheatgrass cover 4 yr following herbicide application during the fall of 2017 and 2018 for the sprayed and nonsprayed treatments in the Antelope Peak and Eastern (E.) Sandhills areas, Centennial Valley, MT, USA. The mean cheatgrass cover before herbicide application (2017) was 13% and 8% in the Antelope Peak and E. Sandhills areas, respectively. Within each area (Antelope Peak, E. Sandhills), groups that share a letter(s) are statistically similar ($p > 0.05$). Black triangles are the means, error bars are constructed using standard errors, and the points are the jittered observed cover values.

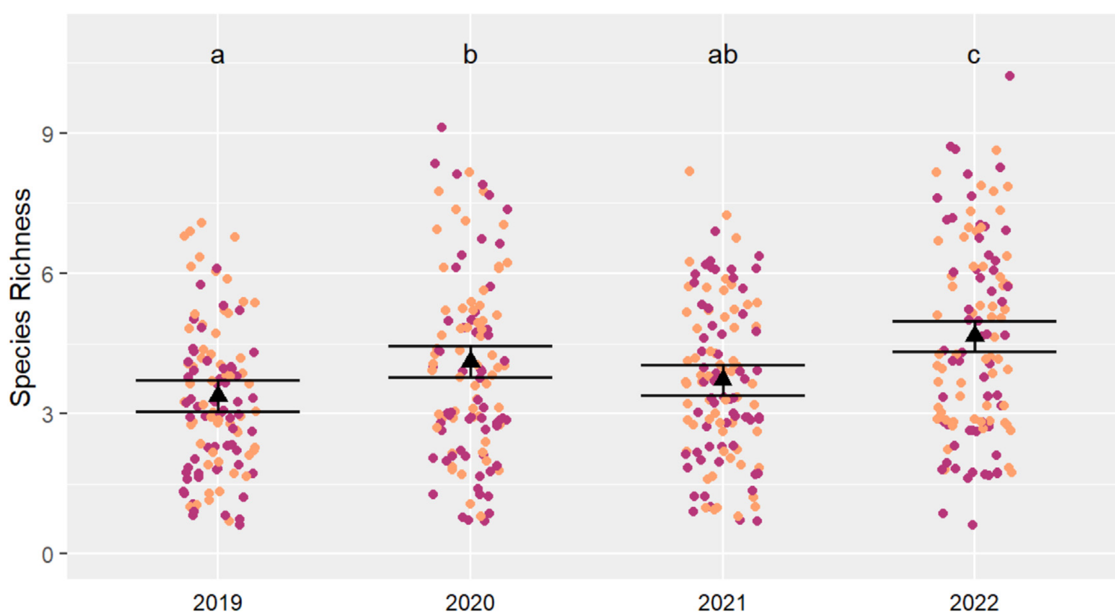

Figure 2. Estimated mean species richness (0.1 m^2) for the 4 yr postherbicide application in the Centennial Valley, MT, USA. The mean species richness varied by year but not by region or treatment. Note that cheatgrass is omitted from this analysis. The observations differ by treatment in color: orange is the sprayed treatment, and purple is the nonsprayed treatment. Black triangles are the means, and error bars are constructed using the standard error of the mean. Groups that share a letter(s) are statistically similar at an alpha level of 0.05.

Over the duration of the study, the top five most abundant plant species were generally native and nonnative forb species (Figs. 3 and 4). In the AP sprayed plant community, cheatgrass did not become the most abundant species until the final (fourth) year of the study. This was in contrast to the AP nonsprayed plant community, where cheatgrass was the first- or second-ranked species across the duration of the study. Interestingly, cheatgrass never became the most abundant species in the ES-sprayed plant community but was the most abundant in the nonsprayed plant community (Fig. 4).

Four years following the final herbicide treatment, the AP sprayed community's most abundant species consisted of cheatgrass, the nonnative annual forb desert alyssum (*Alyssum desertorum* Stapf.), and the native perennial forbs nineleaf biscuitroot (*Lomatium triternatum* [Pursh] J.M. Coult. & Rose), Indian paintbrush (*Castilleja* spp. Mutis ex L.f.), and Yellowstone draba (*Draba incerta*

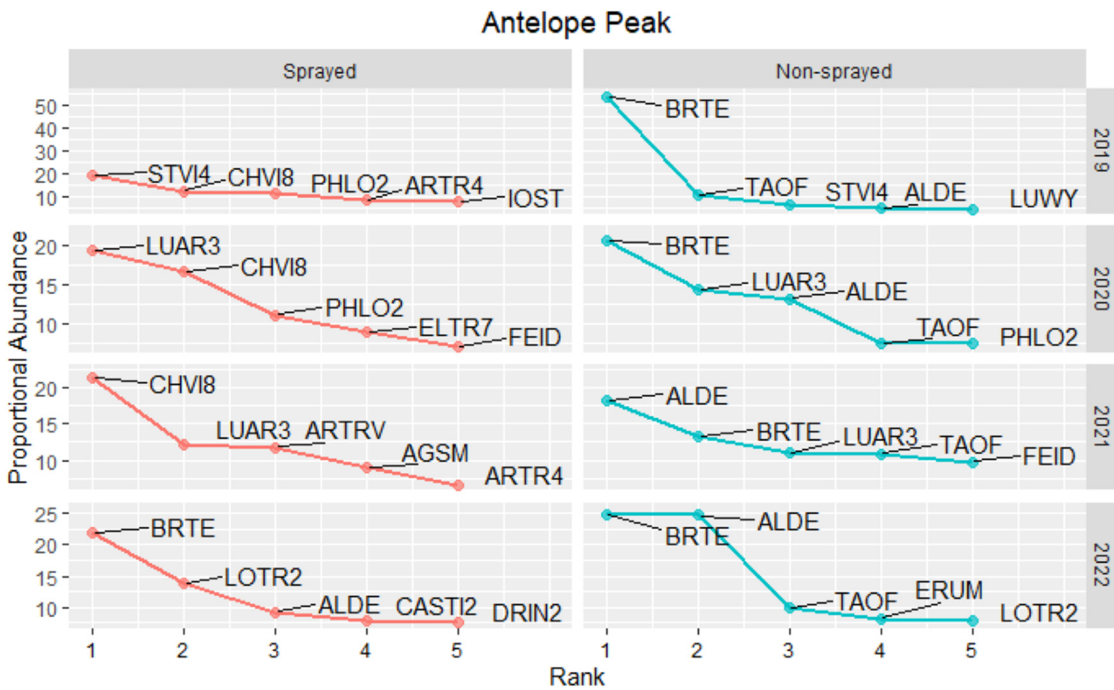

Figure 3. Proportional abundance of the top five species in sprayed and nonsprayed treatments of the Antelope Peak plant community over the 4 yr postherbicide application. Note that steeper lines indicate a less even distribution of abundance among the top five species. Species are represented by four letter codes as follows: AGSM: western wheatgrass (*Agropyron smithii* Rydb.); AGSP: bluebunch wheatgrass (*P. spicata*); ALDE: desert alyssum (*A. desertorum*; nonnative); ARTR4: threetip sagebrush (*Artemisia tripartite* Rydb.); ARTRV: mountain big sagebrush (*A. tridentata* var. *vaseyena*); BRTE: cheatgrass (*B. tectorum*; nonnative); CASTI2: Indian paintbrush (*Castilleja* sp. Mutis ex L. f.); CHVI8: yellow rabbitbrush (*Chrysothosanthmus visidiflorus* Hook. Nutt); CIUN: wavyleaf thistle (*C. undulatum*); DRIN2: Yellowstone draba (*Draba incerta* Payson); ELTR7: slender wheatgrass (*Elymus trachycaulus* Link Gould ex Shinners); ERUM: sulphur-flower buckwheat (*Eriogonum umbellatum* Torr.); FEID: Idaho Fescue (*F. idahoensis*); LOTR2: nineleaf biscuitroot (*L. triternatum*); LUAR3: silvery lupine (*Lupinus argentius* Pursh); LUWY: Wyeth's lupine (*Lupinus wyethii* S. Watson); PHLO2: longleaf phlox (*Phlox longifolia* Nutt.); PODO4: Douglas' knotweed (*Polygonum douglasii* Greene); POSE: Sandberg bluegrass (*Poa secunda* J. Presl); STCO4: needle and thread (*H. comata*); STVI4: green needlegrass (*Stipa viridula* Trin); TAOF: common dandelion (*Taraxacum officinale* F.H. Wigg.; nonnative).

Payson), in declining order (Fig. 3). The AP nonsprayed community was comprised of cheatgrass; two nonnative forbs, desert alyssum and common dandelion (*Taraxacum officinale* F.H. Wigg.); and the native perennial forbs, sulphur-flower buckwheat (*Eriogonum umbellatum* Torr.) and nineleaf biscuitroot, in declining order (Fig. 3). The ES sprayed plant community's most abundant species, in decreasing order, were desert alyssum; the native perennial forb, wavyleaf thistle (*Cirsium undulatum* [Nutt.] Spreng); the native perennial grass, needle and thread grass; the native annual forb, Douglas' knotweed (*Polygonum douglasii* Greene); and nineleaf biscuitroot (Fig. 4). The ES nonsprayed community was comprised of cheatgrass, nineleaf biscuitroot, the native perennial grass, muttongrass (*Poa fendleriana* Steud. Vasey), desert alyssum, and wavyleaf thistle, again in declining order.

Herbicide effect on the proportion of vegetative and nonvegetative cover

The proportion of total cover (vegetative and nonvegetative), discussed herein as percentages, was explained by the three-way interaction of functional group, treatment, and year ($p \leq 0.001$). Three (native perennial forb, nonnative annual grass, and nonnative annual forb) of seven vegetative and two (bare ground and litter) of three nonvegetative functional groups varied by treatments over the study duration (Fig. 5). Only native shrubs and rock did not differ between treatments or year (Table S5; available online at xxx).

Across both areas combined, the native annual and perennial forb functional groups comprised a larger proportion of the plant community than the native perennial grasses every year in both sprayed and nonsprayed communities (Fig. 5). Cheatgrass comprised 8% more of the nonsprayed plant community compared with the sprayed community in the first year ($p \leq 0.0001$), 4% in the second (2020) year ($p = 0.001$), and 5% in the fourth year postherbicide ($p = 0.002$). There was no difference in the proportion of cheatgrass (1%) between the sprayed and nonsprayed plant communities in the third year (2021; $p = 0121$). The native perennial forbs constituted 4% more of the sprayed community compared with the nonsprayed community in the first year ($p = 0.013$) and 5% more in the second year ($p = 0.006$), but no difference was found in the third ($p = 0.139$) and fourth years of study ($p = 0.787$; Table S5; Fig. 5). Nonnative annual forbs made up 2% more of the nonsprayed community compared with the sprayed community in the second year of the study only ($p = 0.029$; Table S5; Fig. 5).

In the first year following the second herbicide application, the sprayed community had a 9% greater proportion of litter ($p = 0.026$), but in the second ($p \leq 0.0001$) and third ($p = 0.001$) years there was a 15% and 13% greater proportion of litter, respectively, in the nonsprayed community. By the fourth year after herbicide applications, there was no difference in the proportion of litter between the two treatments (Fig. 5).

Perennial grass biomass

Of the five native perennial grass species, only Sandberg bluegrass showed a biomass response to area ($p = 0.012$). The other four species showed no change in biomass between sprayed and nonsprayed treatments over 4 yr or between areas. Due to the lack of treatment response, we combined all species biomass into one category, perennial grass, and tested for the response of perennial grass biomass to herbicide, year, and area.

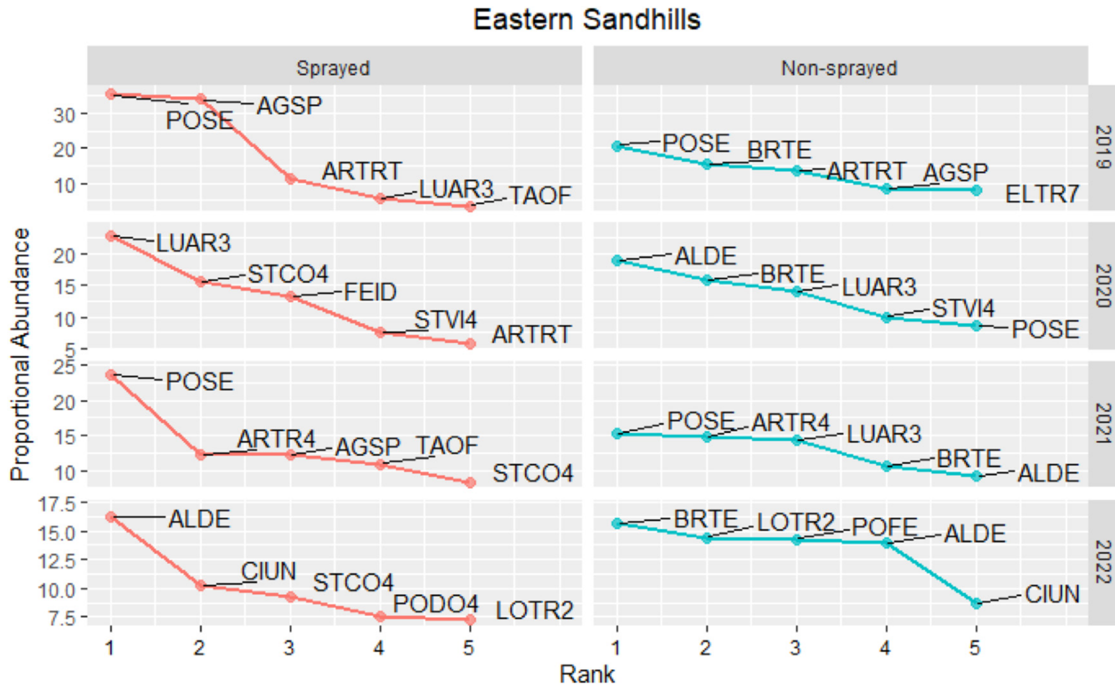

Figure 4. Proportional abundance of the top five species in sprayed and nonsprayed treatments of the Eastern Sandhills plant community over the 4 yr postherbicide application. Note that steeper lines indicate a less even distribution of abundance among the top five species. Species are represented by four letter codes as follows: AGSM: western wheatgrass (*Agropyron smithii* Rydb.); AGSP: bluebunch wheatgrass (*P. spicata*); ALDE: desert alyssum (*A. desertorum*; nonnative); ARTR4: threetip sagebrush (*Artemisia tripartite* Rydb.); ARTRV: mountain big sagebrush (*A. tridentata* var. *vaseyena*); BRTE: cheatgrass (*B. tectorum*; nonnative); CASTI2: Indian paintbrush (*Castilleja* sp. Mutis ex L. f.); CHVI8: yellow rabbitbrush (*Chrysothosanthmus visidiflorus* Hook. Nutt); CIUN: wavyleaf thistle (*C. undulatum*); DRIN2: Yellowstone draba (*Draba incerta* Payson); ELTR7: slender wheatgrass (*Elymus trachycaulus* Link Gould ex Shinners); ERUM: sulphur-flower buckwheat (*Eriogonum umbellatum* Torr.); FEID: Idaho Fescue (*F. idahoensis*); LOTR2: nineleaf biscuitroot (*L. triternatum*); LUAR3: silvery lupine (*Lupinus argentius* Pursh); LUWY: Wyeth's lupine (*Lupinus wyethii* S. Watson); PHLO2: longleaf phlox (*Phlox longifolia* Nutt.); PODO4: Douglas' knotweed (*Polygonum douglasii* Greene); POSE: Sandberg bluegrass (*Poa secunda* J. Presl); STCO4: needle and thread (*H. comata*); STVI4: green needlegrass (*Stipa viridula* Trin); TAOF: common dandelion (*Taraxacum officinale* F.H. Wigg.; nonnative).

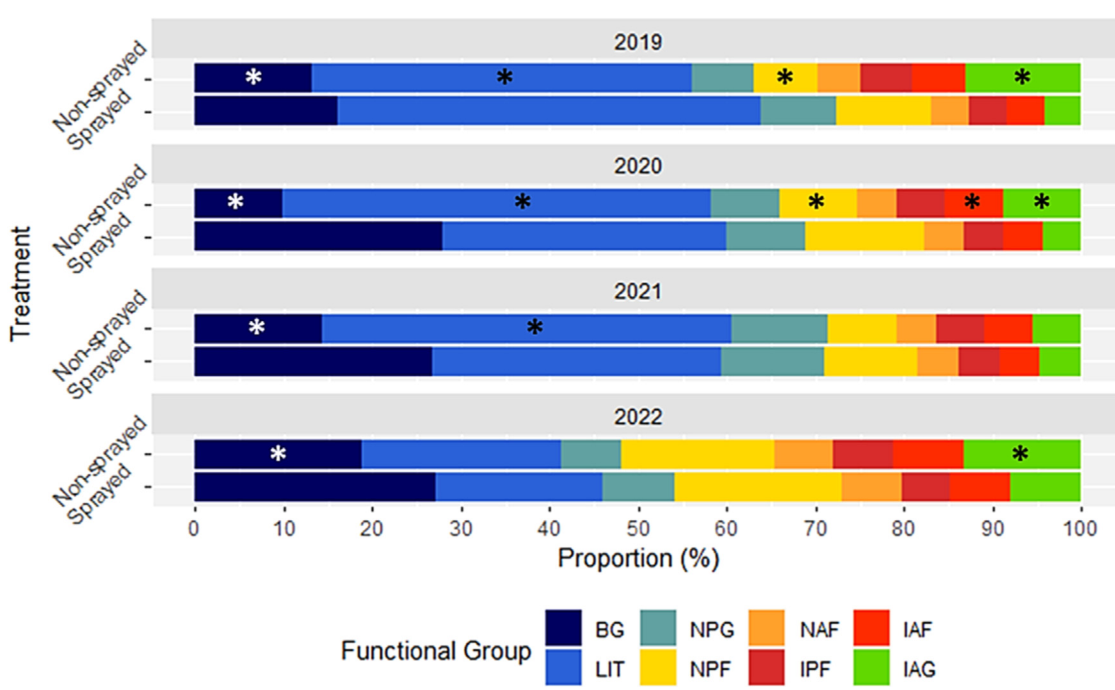

Figure 5. Mean proportional, expressed as a percentage, change in functional group cover across 4 yr postherbicide application between sprayed and nonsprayed treatments. The asterisks denote differences in the proportion of each functional group by treatment within each year using an alpha level of 0.05. The functional groups include introduced (nonnative) annual grass (IAG), introduced (nonnative) annual forb (IAF), introduced (nonnative) perennial forb (IPF), native annual forb (NAF), native perennial forb (NPF), native perennial grass (NPG), litter (LIT), and bare ground (BG). This figure omits the shrub functional group and rock cover type for clarity of display; neither differed in their proportion over time or between treatments.

Perennial grass biomass did not differ between sprayed and nonsprayed treatments ($p = 0.168$) or areas ($p = 0.488$). Perennial grass biomass did, however, differ over the 4 yr of the study ($p = 0.001$). The lowest combined mean perennial grass biomass (54.9 kg · ha ± 12.1 SE) was observed in the second year, and the highest in the final year (88.6 kg · ha ± 13.2 SE; $p = 0.027$); otherwise, there was no difference.

Discussion

An extensive body of literature warns of the negative impacts of cheatgrass, particularly when coupled with wildfire and subsequent conversion of shrublands to grasslands (Young and Evans 1978; Brooks et al. 2004; Young and Clements 2009; Germino et al. 2016). The transformation from shrublands to nonnative annual grass-dominated systems has led to widespread loss and degradation of the sagebrush biome. To avoid further transformation, land managers are being urged to be proactive. Early detection and proactive management of intact ecosystems with low cheatgrass occurrence has been proposed to reduce downstream costs of managing widespread and dense populations of nonnative species (Sheley and Smith 2012). Further, recent spatial analyses have identified core areas of the sagebrush biome for conservation and proposed a “defend the core” management framework, incorporating EDRR ideology (Doherty et al. 2022; Chambers et al. 2023b).

Our study provides vital information that will help to inform management decisions in core sagebrush areas that have low abundances of cheatgrass and intact, species-rich plant communities. We found that herbicide effectively reduced low-abundance populations of cheatgrass for 3–4 yr following two consecutive years of imazapic application. These management effects, while not permanent, lasted longer than expected compared with results from the Great Plains of central Montana (Orloff et al. 2015) and resembled the control length observed in the Columbia Plateau and North and Central Basin and Range (Elseroad and Rudd 2011; Pyke et al. 2014). We suspect the control observed in our study was due to multiple interacting factors: a low abundance of cheatgrass preherbicide applications, two consecutive fall herbicide applications, high native species richness and diversity, and poor cheatgrass growing conditions in some seasons following herbicide treatments at our sites in the Centennial Valley.

Explicitly, the combination of low preherbicide abundances of cheatgrass (13% in AP and 8% in ES) and 2 yr of consecutive postemergent applications of imazapic, which were timed late in the fall and coincided with a period of above-average precipitation, likely increased the emergence and germination of cheatgrass preceding treatment and contributed to the length of control observed. Further, herbicide efficacy was likely enhanced by poor growing conditions following application. For example, spring and fall precipitation has been correlated with cheatgrass biomass production and survival (Vermeire and Rinella 2020), and we observed spring precipitation values below the 30-yr average in both AP and ES areas over the course of the first three growing seasons and above average in the fourth growing season posttreatment. We also observed above-average fall precipitation in all years except 2020, which should have favored cheatgrass germination and emergence. However, if fall precipitation is a major driver of cheatgrass abundance, it did not manifest itself until the fourth year of our study, when cheatgrass abundance increased to pretreatment levels in the AP but not ES area. Importantly, it is also possible that low preherbicide cheatgrass cover levels were a contributing factor. The nonsprayed plant community in the ES had an average cheatgrass cover of 8%, and it was here that we observed 4 yr of continued suppression. Low levels (< 25%) of cheatgrass cover combined with a diverse, desirable native community can be considered a mild infestation, where eradication or high levels of control of cheatgrass are likely (Mealor et al. 2013).

Despite strong initial control, we observed the return of cheatgrass as a dominant member of the AP sprayed plant community in the fourth year of the study. This suggests that while initial control was effective, conditions that are conducive to cheatgrass remain in this area. No changes to site management occurred to accompany herbicide application, i.e., changes in grazing pressure or restoration seeding, and it is also possible that cheatgrass propagules entered the site from surrounding areas via wildlife, cattle, wind, or other vectors. Comparatively, cheatgrass did not return as a dominant member of the ES-sprayed community. Differences in cheatgrass response to management in relation to site differences have been observed elsewhere (e.g., Wood and Mealor 2022). Within the context of our study, this may be due to differences in soil type; for example, the ES has sandier soil than the AP area. We hypothesize that the high percentage of sand in the ES soil, coupled with cattle trampling, although at the same level as in the AP area, likely suppresses populations of cheatgrass in this area (Vivion 1968; Davis and Cutting 2014).

Studies in areas of high degradation or high pretreatment cheatgrass abundance have observed an increase in native perennial grass abundance following reductions in cheatgrass (Davies and Sheley 2011; Kyser et al. 2013; Orloff et al. 2015; Wood and Mealor 2022) or a correlation between increasing nonnative annual grass cover, including cheatgrass, and decreasing native plant richness (Gasch et al. 2013; Bansal and Sheley 2016; De Stefano et al. 2024). In our study, with a low initial abundance of cheatgrass, we did not observe an increase in native perennial grass proportion or biomass or species richness between sprayed and nonsprayed treatments. This is consistent with the cheatgrass response from Wyoming (i.e., the southern-middle Rockies ecoregion), where it is suggested that preherbicide levels of cheatgrass may dictate the response of the native plant community following herbicide treatment (Wood and Mealor 2022). Specifically, Wood and Mealor (2022) found a minimal increase in native perennial grass biomass or species richness when herbicide was applied to areas with < 25% cheatgrass cover. Our results support this finding.

In addition, we assessed the proportional changes in functional groups in response to herbicide treatments. Both sprayed and nonsprayed communities were characterized by a dominant component of native annual and perennial forbs. Specifically, a greater proportion of native perennial forbs was observed in the sprayed compared with nonsprayed plant communities in the first 2 yr following treatment but not in the final 2 yr. The initial pulse of native perennial forbs following herbicide treatment could have temporary benefits for Greater sage-grouse, other wildlife, and pollinators (Gilgert and Vaughan 2011; Pennington et al. 2016; Cutting et al. 2024). One of the dominant native perennial forbs, nineleaf biscuitroot, is an important food source for sage-grouse chicks and adults (Luna et al. 2018).

Implications

This study provides evidence of the effects of managing low-abundance populations of cheatgrass in high-elevation geographies, identified as core sagebrush areas under the “defend the core” management framework (Remington et al. 2021; Doherty et al. 2022). We found that if low-abundance populations of cheatgrass are treated in the 2–3 leaf stage with 2 yr of imazapic herbicide in the fall, land managers should expect good cheatgrass control but without an associated increase in native plant abundance, species richness, or perennial grass production. While managing cheatgrass in such areas fits within nonnative plant management frameworks that emphasize proactive mitigation actions

(Sheley and Smith 2012; Mealor et al. 2013; Crist et al. 2019; Doherty et al. 2022), convincing land managers to do so without immediate, short-term returns on their investment through increased forage production or improved wildlife habitat may be difficult (Finnoff et al. 2007) but warranted (Smith et al. 2023; Boyd et al. 2024). However, those managers with the goal of maintaining the ecological function and structure of plant communities by proactively reducing cheatgrass cover could realize the utility of this approach. Understanding the tradeoff between the allocation of limited resources in areas where nonnative annual grass control results are obvious (e.g., increase in perennial grass production) vs. where those benefits may only be realized in a future scenario may not be abundantly clear.

Our results also suggest that native forbs make up the greatest herbaceous component of these core high-elevation mountain big sagebrush communities. As a result, in management situations similar to those observed in this study, which are highly representative of the core sagebrush habitat in the Middle Rockies and Upper Snake River Plain, we suggest that special care be taken to avoid nontarget impacts. Within the context of this study, we achieved this by using a herbicide with low soil persistence, applied at the suggested rate and at a time when cheatgrass was the most actively growing species in the plant community and highly susceptible to herbicide application (Mangold et al. 2013). Others have found that preemergent applications of imazapic in the North Central Basin and Range ecoregion have also proven successful (Davison and Smith 2007; Kyser et al. 2013). The differences in the response of cheatgrass and the native plant community to pre- and postemergence applications of imazapic highlight the importance of considering ecoregion and site characteristics when considering a management plan. Additionally, these differences support the value of small-scale herbicide efficacy plots and monitoring before expanding treatments to larger management scales so that nontarget impacts and herbicide effectiveness are fully understood.

Since our study was initiated, the active ingredient indaziflam (Rejuvra, Environmental Science U.S. LLC, Cary, NC) has become available for annual grass management. Indaziflam is a preemergent, cellulose-biosynthesis-inhibiting, nonselective herbicide that has exhibited multiple years of cheatgrass control from a single application (Sebastian et al. 2016; Clark et al. 2020; Courkamp et al. 2022a). While highly effective at controlling cheatgrass, a small number of studies have indicated nontarget impacts on native annual forbs both above ground (Meyer-Morey et al. 2021) and in the soil seed bank (Meyer-Morey et al. 2021; Courkamp et al. 2022b). Additionally, evidence from the North and Central Basin and Range suggests that indaziflam may reduce both native forb and planted grass seedlings (Shriver et al. 2024). Minimizing nontarget effects in both the short- and long-term was an important consideration in our study because the Centennial Valley is considered a core sagebrush habitat, i.e., comprised largely of native grasses and forbs and has a small nonnative plant component. Our results show that nontarget impacts were avoided, i.e., we did not observe a decrease in the proportion of native forbs nor species richness after two applications of imazapic. In addition to minimizing nontarget effects, we suspect that cost is also an important consideration of land managers when deciding between indaziflam (approximately \$114/ha) or imazapic (\$13/ha; Montana Department of Agriculture 2025). While indaziflam can provide multiple years of control, some land managers may not be able to afford the expense upfront. In such cases, annual imazapic applications over 2–3 yr can provide an option that delivers similar levels of control.

We suggest that cheatgrass populations be monitored before, during, and after herbicide application to confirm the efficacy and ecological and economic return on investment, as results are likely context-dependent. We have shown that using herbicide alone to treat cheatgrass occurring at low abundance in high-elevation sagebrush steppe does not ensure its long-term suppression under all conditions. Management will likely be more effective if the herbicide is used within an adaptive and integrated weed management strategy that considers potential drivers of habitat suitability that can be impacted by management decisions (Menalled et al. 2008; Boyd 2022; Maestas et al. 2022). For example, taking precautions to limit the movement of cheatgrass seed from infested areas and minimizing the amount of off-road or track driving (Rew et al. 2018), altering grazing pressure and timing (Eldridge et al. 2018), placement of grazing infrastructure, and following herbicide with restoration seeding when necessary (Clements et al. 2022).

Declaration of Competing Interest

None.

CRediT authorship contribution statement

Colter Mumford: Writing – original draft, Visualization, Validation, Software, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Jane Mangold:** Writing – review & editing, Validation, Resources, Methodology, Conceptualization. **John Winnie Jr.:** Writing – review & editing. **Kyle A. Cutting:** Writing – review & editing, Validation, Resources, Methodology, Investigation, Data curation, Conceptualization. **Catherine Zabinski:** Writing – review & editing, Validation, Funding acquisition. **Lisa J. Rew:** Writing – review & editing, Visualization, Validation, Supervision, Software, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Conceptualization.

Acknowledgments

The authors would like to thank the J Bar L Ranch (Andrew and Hilary Anderson) and Lee Martinell Co. (Allen and Yvonne Martinell), who provided access to their land as part of this study. Further, we would like to thank Red Rock Lakes National Wildlife Refuge, Bureau of Land Management-Dillon Field Office, U.S. Fish and Wildlife Service-Invasives Program, and The Nature Conservancy-Montana staff, notably James Waxe, for their support and facilitation of this project. Projects like this are not possible without the dedication and support of many field technicians.

Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.rama.2025.02.006.

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