

RESEARCH ARTICLE

Pre-fire grazing and herbicide treatments can affect post-fire vegetation in a Great Basin rangeland

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Funding information

National Institute of Food and Agriculture, U.S. Department of Agriculture award within the Western Integrated Pest Management Program, Grant/Award Number: 2013-34103-21325

Handling Editor: Kevin Mganga

Abstract

1. Management of wildfire associated with spread of the highly invasive annual grass *Bromus tectorum* (cheatgrass) is a critical need in the western U.S.
2. We investigated the utility of coupling common rangeland management strategies pre-fire to modify post-fire plant community outcomes. We used a long-term, large-scale experiment to test the separate and combined effects of pre-fire targeted grazing (spring and fall), native plant seeding (seeding rate, seed coating and spatial seeding arrangement) and herbicide (glyphosate followed by 2 years of imazapic) on post-fire plant community outcomes in a highly invaded system in the Great Basin, U.S.
3. We found grazing and herbicide effects were consistent across cheatgrass biomass, count, and cover. Spring grazing reduced cheatgrass more effectively than fall grazing; however, this effect was detected primarily outside of the seeding treatments. Herbicide overall and in conjunction with grazing reduced cheatgrass and fuel loads. Among seeding treatments, seed mixtures proved more effective than monocultures for reducing both cheatgrass count and cover, particularly when combined with low seed rate. However, many seeding approaches resulted in higher cheatgrass dominance, and thus higher fuel loads.
4. This work suggests that effects of pre-fire herbicide for reducing cheatgrass abundance can persist post-fire. Grazing, however, might not produce consistent results, and season of grazing can affect outcomes. Employing pre-fire management strategies to interrupt the cheatgrass-fire cycle may have utility. Some of our treatments were able to reduce cheatgrass abundance after fire, but despite our intensive interventions, we did not find a strategy that led to full restoration of native perennial species.

KEYWORDS

Bromus tectorum, cheatgrass, downy brome, grazing, greenstrips, seeding, wildfire

Elise S. Gornish and Jessica S. Guo contributed equally to this manuscript.

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1 | INTRODUCTION

Climate change, fuel accumulations and anthropogenic activities are interacting to change fire dynamics in many rangeland systems worldwide. While some regions are experiencing reduced fire and subsequent woody plant increases (Wei et al., 2021), others are experiencing new levels or larger extents of high frequency, high intensity, wildfires (Kodandapani et al., 2004; Miller et al., 2009; Singleton et al., 2019). Although periodic fire is within the historic range of variability for many terrestrial systems, and can have desirable impacts on plant succession and biodiversity, nutrient cycling, and habitat for disturbance-oriented species (DellaSala & Hanson, 2015), an increased frequency of broad-scale, high intensity fires can have extreme socio-economic and ecological impacts (e.g. Reid et al., 2016; Xu et al., 2020). The environmental effects of fire are also profound and can include an increase in erosion, pollution of local water sources, and changes in wildlife disease dynamics (Albery et al., 2021; Smith et al., 2011). In arid rangeland systems lacking a historical fire regime characterized by frequent fires, wildfire has been particularly detrimental to biodiversity due to its enhancement of aggressive invasive plant species, resulting in a cascade of negative effects on extant native plant communities and the wildlife that depend on them (Balch et al., 2013).

Total exclusion is not a feasible nor a desirable approach to fire management (Doerr & Santín, 2016), so pre-fire vegetation management is often prioritized for managing fire intensity, and subsequent post-fire plant response. Managers employ a variety of approaches pre-fire in an attempt to modify fire outcomes, such as the removal of combustible biomass through thinning, overstory removal, and prescribed burning with varying levels of success (Kalies et al., 2010). Herbicides have also been used as a means of reducing fuel loads and fuel connectivity, particularly in areas where roads provide a potential ignition source or to protect infrastructure in wildland-urban interfaces (Pellent, 1990). In rangelands, however, these approaches might not be feasible at broad scales due to inherent challenges of controlling prescribed fire in highly connected landscapes with abundant fuels. Targeted grazing, which is defined as the strategic use of grazing animals with the timing, duration, and intensity needed to achieve a particular vegetation outcome (Frost & Launchbaugh, 2003), holds promise for fire management in rangeland systems.

Much research has been devoted to understanding the effects of livestock grazing on rangeland plant communities. However, despite a logical link between targeted grazing and fire management, there is a small but growing pool of research dedicated to understanding how wildfire and pre-fire grazing might interact (Davies et al., 2016, 2021; Ridder et al., 2022; Watson et al., 2021). Grazing is expected to be useful for reducing fuel loads at key times of the year while also shifting plant species composition and competition dynamics. For example, grazing can potentially increase the presence and distribution of fire resilient native plant species by controlling the biomass of highly flammable invasive grasses, reducing litter, thatch, and invasive species seed banks (Papanastasis, 2009; Perryman et al., 2020), which can reduce competitive interactions between

invasive and native species. Subsequently, these characteristics can significantly impact wildfire behaviour and the reestablishment of biomass after a fire (Davies et al., 2017; Gutman et al., 2001). Existing research highlights the utility of using season-long moderate grazing to influence post-fire vegetation outcomes (Kupfer & Miller, 2005), but little research has been done on post-fire effects of targeted grazing strategies (but see Diamond et al., 2012). Understanding how pre-fire targeted grazing might affect post-fire plant communities is critical for developing effective rangeland management strategies.

Pre-fire targeted grazing is not the only understudied management strategy that might hold promise for modifying wildfire outcomes. Seeding of native species, for example, is a common post-fire management strategy to reestablish functional plant communities (e.g. Kulpa et al., 2012). However, pre-fire seeding (i.e. seeding native plants in areas that may be vulnerable to future fire) could have critical impacts on fire behaviour and post-fire vegetation outcomes (Bowman-Prideaux et al., 2021). Seeding and subsequent establishment of native species can be an effective way to minimize dominance of high biomass invasives and enhance the likelihood that fire tolerant natives establish post-fire (e.g. Epanchin-Niell et al., 2009). This occurs because plants with fire tolerant traits, such as rapid growth post-fire (e.g. Pilon et al., 2021) can be leveraged with this management approach. Since pre-fire seeding can be used to achieve a variety of management goals simultaneous to wildfire outcome management (Eastburn et al., 2018), research that articulates which seeding designs are most likely to result in desired postfire plant communities is critical for rangeland managers.

We focused our work in the Great Basin, US, where the invasion of annual species, including the highly pervasive cheatgrass *Bromus tectorum* L., an invasive annual grass that suppresses the growth of keystone native plant species (Kainrath et al., 2021), has a well-documented relationship with wildfire (Balch et al., 2013; D'Antonio & Vitousek, 1992). Cheatgrass germinates after the first significant fall rains, which can be as early as September or as late as January, and typically flowers in April and May, and sets seeds by June or early July (Rice et al., 1992), which then drop from the plant. After seed dispersal, biomass senescens and turns into fine fuels, which are flammable and can also build up into a robust layer of litter over time (Pilliod et al., 2017). Most of the dominant native plant species in this system are perennials, and many of these species also germinate in the fall with the first rains, but more importantly, perennial species are able to green up and begin growing rapidly when the first rains or snows fall (Knapp, 1996). Native perennial systems typically have larger interspaces, which reduces fire spread, and other characteristics, such as remaining green longer through the season or investing in below-ground biomass that does not produce copious fuels (Melgoza & Nowak, 1991), which reduces the probability of fire in intact ecosystems (Wright & Klemmedson, 1965). Due to the severity of cheatgrass invasion in the Great Basin, managers have invested considerable resources to control the grass. Since fire plays a significant role in the persistence of cheatgrass (Knapp, 1996), developing an integrated strategy for weed control could form the basis for fire management.

Using a five-year field study, we asked: What are the effects of pre-fire targeted grazing (spring and fall), native species seeding and herbicide (a more traditional fire mitigation technique) on the abundance (biomass, cover and density) of cheatgrass and native plants after fire? Herbicide is an extremely effective weed control method, but expense, logistical challenges of application, non-target effects, and need for repeated application make it a difficult strategy to apply at large spatial scales (Merriam et al., 2006; Pellent, 1990). Thus, a primary goal of this project was to determine if grazing or seeding treatments could result in comparable outcomes in the plant community. After our grazing, herbicide and seeding treatments had been designed and implemented, a wildland fire opportunistically burned our experimental plots and surrounding vegetation, and thus our results reflect outcomes from realistic fire timing and intensity. Earlier pre-fire results indicate that grazing treatments reduced standing biomass (dominated by invasive species) by 30%–50%, herbicide treatments reduced biomass even further, and targeted spring grazing in combination with seeding of diverse mixes at high rates could result in the establishment of native perennial grasses (Porensky et al., 2018). However, pre-fire litter cover, invasive species cover and invasive species density were not significantly impacted by our grazing or seeding treatments (Porensky et al., 2018). Grazing treatments can, however, affect seedbank density and fire intensity (Diamond et al., 2012), and therefore, we expected pre-fire targeted grazing treatments to result in reduced cheatgrass post-fire. We also expected pre-fire herbicide treatments to reduce fire intensity and therefore reduce cheatgrass abundance postfire, and we expected pre-fire restoration treatments, including higher seed rates, seed coatings, use of competitive native grasses and spatially segregated planting arrangements, to enhance coverage of native plant species, ultimately resulting in reduced fire intensity and lower postfire cheatgrass abundance.

2 | MATERIALS AND METHODS

2.1 | Experimental design

Our experimental plots were located in Northern Nevada, on the TS Ranch (40.843–40.895 N, 116.509–116.554 W). We did not need permission for fieldwork. This former sagebrush-steppe community is heavily invaded by annual species, with the highly competitive cheatgrass dominating the plant community, along with lower densities of introduced (non-native) forbs, including *Sisymbrium altissimum* L., *Lepidium perfoliatum* L., *Salsola tragus* L., *Ceratocephala testiculata* (Crantz) Roth and *Chorispora tenella* (Pall.) DC. The ~1450 m elevation study site is located on a gently sloped alluvial fan, with loamy soils; annual average precipitation at the site is 250 mm, and temperatures are cold in the winter (average daily temperature in January is -3°C) and warm in the summer (average daily temperature of 22°C in July; PRISM Climate Group, 2022). Historic plant communities likely included a mixture of native perennial shrubs, grasses, and forbs, including sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis*

Beetle & Young), basin wildrye (*Leymus cinereus* (Scribn. & Merr.) Á. Löve), squirreltail (*Elymus elymoides* (Raf.) Swezey), sandberg bluegrass (*Poa secunda* J. Presl) and a suite of native forbs; some of these native grass and annual forb species were still present even in the highly-invaded state (Porensky et al., 2018). The area used for this study has been used as grazing land since the 1800s, with current management as a cattle operation, including private land used for cattle production (this study), areas of irrigated alfalfa, and surrounding Bureau of Land Management seasonal pastures.

In 2014 we initiated a broad-scale experiment to test the separate and combined effects of targeted grazing, native plant seeding, and herbicide on plant community outcomes (Figure S1), with replicated treatments arranged in nine large (18.2 ha) paddocks (Figure S1), as described in Porensky et al. (2018). Briefly, seasonal (fall or spring) applications of short-duration and high intensity targeted grazing were implemented with the goal of reducing cheatgrass standing biomass and seedset by grazing during time periods when this winter annual should be most vulnerable. Paddocks experiencing either fall or spring targeted grazing were compared to ungrazed plots, with one of these three treatments randomly assigned to one of three paddocks in three blocks, for three replicates per grazing treatment. Targeted grazing was applied with the goal of leaving ~112 kg/ha of standing crop in each paddock after implementation. Due to high variability in vegetation growth across seasons and years, the number of animals used to achieve our targeted vegetation outcome varied through time. During the first year, 25 cows (25 animal units) grazed the three paddocks assigned to the fall grazing treatment for 7–9 days each between October 15 and November 9, 2015, and 29 yearlings (22 animal units) grazed the three spring-grazed paddocks for 9–11 days each between April 5 and May 4, 2016. Grazing treatments were repeated the subsequent year, after wet conditions in spring 2016 led to ample forage growth. In fall 2016, 50 head of 6–10 year old dry cows (50 animal units) grazed the three fall grazing paddocks for 17–22 days each between October 31 and December 30, 2016. In spring 2017, 105 cows (105 animal units) grazed the three spring grazing paddocks for 7–11 days each between April 10 and May 8, 2017. Utilization data are presented in Porensky et al. (2018).

Within each paddock, five restoration treatments were randomly assigned to ten 20×60 m plots (0.12 ha each), with two replicates per treatment per paddock (Figure S1). Treatments included: herbicide only (two plots), or herbicide followed by seeding with one of four native grass treatments (eight plots). Native grass treatments followed a 2×2 factorial design that manipulated the following factors: seeding rate (1× and 2× rate), and spatial seeding arrangement (monoculture strips or mixed seedings; mono vs. mix). Twelve unseeded, unsprayed control plots were also monitored in each block. See Porensky et al. (2018) for more details.

When seeding into sites that are highly invaded with cheatgrass, some sort of seedbank reduction is needed in order to facilitate plant establishment, as the competitive nature of this plant can lead to extremely low establishment of seeded species in areas with extensive cheatgrass (e.g. Baughman et al., 2016; Clements

et al., 2022). Further, resident vegetation can compete with seeded species. Thus, all plots except for unseeded controls were initially sprayed at recommended rates with glyphosate, a broad-spectrum herbicide, in April of 2014, prior to seeding (840 g/ha). Herbicide only plots were resprayed at recommended rates in April of 2015 and 2016 with imazapic (420 g/ha), an herbicide that can have stronger effects on cheatgrass than native species (e.g. Burnett & Meador, 2015). In 2016, spraying occurred just prior to grazing in spring-grazed paddocks. Plots assigned to seeding treatments were not resprayed and were seeded with a rangeland drill in October and November 2014, a full year before grazing treatments were implemented. The native grass seedings included four native perennial grasses, seeded at the following 1× rates, which followed rangeland restoration guidelines for each species: *Elymus elymoides* (ELEM), 3.05 pure live seeds (PLS) kg/ha; *Elymus trachycaulus* (ELTR), 4.34 PLS kg/ha; *Poa fendleriana* (POFE), 0.65 PLS kg/ha; *Poa secunda* (POSE), 0.56 PLS kg/ha. We also seeded one native annual grass, *Vulpia microstachys* (VUMI), at 1× rate of 0.61 PLS kg/ha. In mixture plots, the entire 20×60 m plot was seeded with a mixture of all five grass species, at either 1× (low) or 2× (high) rates. In monoculture plots, each species was seeded on its own (at low or high rates) in 4×20 m adjacent strips. For a given seeding rate, each 20×60 m monoculture and mixture plot received the same total amount of seed per species; only the spatial arrangement of seeds within the plot differed. Finally, within each seeded plot, we included both a coated and uncoated seed treatment. Specifically, we applied a seed coating to all species seeded within a 20×20 m subplot, which was randomly located within each main plot. Seeds of each species were coated with a non-ionic alkyl terminated block copolymer surfactant coating based on C1–C4 alkyl ethers of methyl oxirane-oxirane copolymers (Aquatrols Corporation of America). This surfactant has been used to increase the wettability of water-repellent soils (Fernelius et al., 2017; Kostka, 2000) but also improves plant drought tolerance in wettable soils by reducing the time it takes for a root to rehydrate and by decreasing plant transpiration rates (Ahmed et al., 2018). We hypothesized that increased drought tolerance provided by the seed coating could produce more vigorous plants with greater survival.

The entire site burned in the Roosters Comb fire, an ~88,000 ha wildland fire that burned both public and private lands in July 2017. We did not assess fire severity. No grazing, seeding, or herbicide treatments were applied post-fire.

2.2 | Data collection

All plots were sampled from May 31–June 30, 2019, approximately 2 years post-fire. Spring of 2019 was unusually wet (Figure 1); May precipitation (81.1 mm) was more than twice the 30-year average (34.9 mm). In the water year immediately following the fire (2018), annual precipitation was 208 mm and just below the 1991–2020 normal (218 mm), while 2019 experienced precipitation of 363 mm, about 66% above average (Figure S1). For cheatgrass cover and

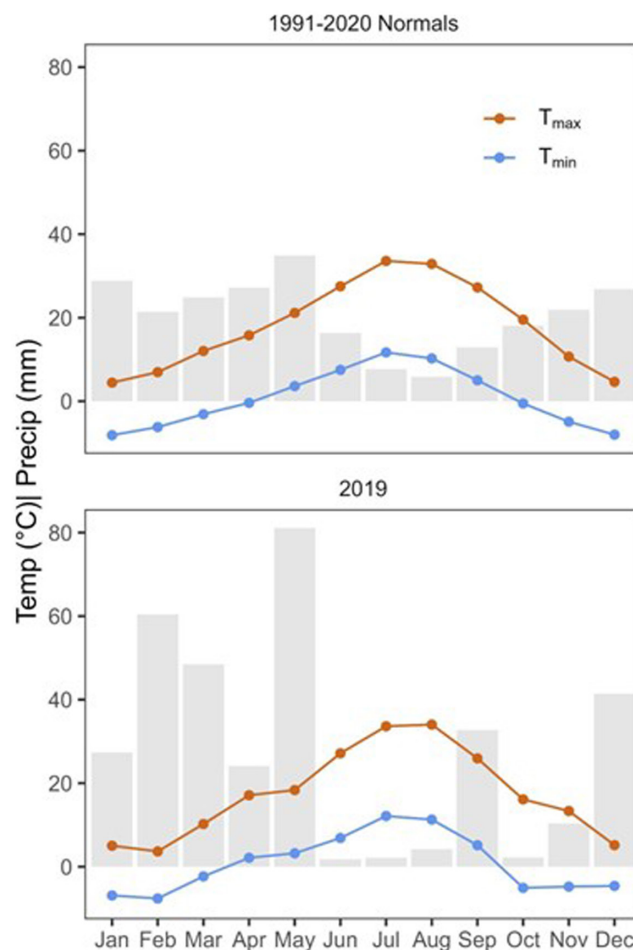


FIGURE 1 Monthly air temperature and precipitation from 1991–2020 normals (top) and 2019 (bottom). Field data were collected between May 31–June 30. Data are from PRISM climate layers at 4 km resolution (PRISM Climate Group, 2022).

density, we sampled a total of 72 locations per paddock (denoted by red and black squares in Figure S1). These included: (1) Monoculture seeded plots, low and high seed rates: 10 locations per plot, including one coated seed and one uncoated seed strip for each of the five planted species; (2) Mixture seeded plots, low and high seed rates: 4 locations, including two with coated seed and two with uncoated seed; (3) Herbicide plots: 2 locations; and (4) Unseeded controls: 12 locations located randomly within each paddock but outside of planted plots (>15 and <50 m from any planted plot).

At each sampling location, we visually estimated percent plant foliar cover by functional group within a 1×1 m quadrat using a continuous integer scale from 0%–100%. Functional groups included native forb, non-native (introduced) forb, cheatgrass, native grass, and standing dead vegetation (i.e. vegetation that grew in the previous growing season). Species diversity within each functional group was low across the site, and cover patterns were relatively homogenous across space within each plot. In a 10×10 cm sub-quadrat located in the NE corner of the 1×1 m quadrat, we counted the number of cheatgrass individuals. If cheatgrass count was <5 individuals, we

upgraded the sub-quadrat area to 25×25 cm. We recorded the size of area sampled and subsequently converted all density estimates to individuals per m².

For standing biomass, we clipped all aboveground biomass in one 50×50 cm sub-quadrat at half of the unseeded control locations (six locations per paddock). We did not clip in herbicide plots or seeded plots. We separated clipped biomass into four functional group categories: cheatgrass, native grasses, forbs (native+introduced) and standing dead vegetation that grew in the previous growing season. The latter functional group was rare and not analysed further.

2.3 | Analysis

Biomass, density, and cover were analysed in a Bayesian framework. Biomass of major plant functional groups (cheatgrass, forbs, and native grasses) were measured only for the control plots and assessed with a multivariate single-factor ANOVA testing the effect of grazing. The remaining variables (cheatgrass count and cover of each functional group) were analysed in an ANOVA framework at three levels, with each model including fixed effects and two-way interactions (Figure S2). In Model I, the entire dataset was analysed with a two-factor ANOVA for the effects of grazing (ungrazed, fall, spring) and restoration treatments (control, herbicide, grass seeding). In Model II, the native grass plots were analysed with a four-factor ANOVA for the effects of spatial arrangement (mixture, monoculture), seed rate (low, high), seed coat (uncoated, coated), and grazing (ungrazed, fall, spring). In Model III, the monoculture plots were analysed in a four-factor ANOVA for the effects of species identity, seed rate, seed coat, and grazing.

2.3.1 | Biomass across plant functional groups and cheatgrass density

The biomass of major plant functional groups was assessed in the control plots ($n = 54$) for the effect of grazing. The observed biomass values of cheatgrass, forbs, and native grasses were assumed to follow a multivariate normal distribution where block was a random effect (Methods S1). Cheatgrass density was analysed in an ANOVA framework at the three levels (Figure S2). To account for overdispersion, observation-level random effects were nested within block random effects (Methods S1).

2.3.2 | Cheatgrass, forbs and native grass cover

Due to the low cover of native forbs, native and introduced forbs were combined for analysis. Plant cover of cheatgrass, forbs, and native grass were analysed separately in an ANOVA framework at three levels (Figure S2). A binomial-beta mixture distribution was implemented to account for absence observations (Bayes & Valdivieso, 2016), and block was treated as a random effect. For

cheatgrass and forbs, absences were uniformly low or zero such that a single probability of absence (ρ) was specified. As native grasses had higher and more variable rates of absence, we modelled ρ with the same ANOVA model as the mean cover (Methods S1).

2.3.3 | Model implementation and interpretation

The models (Methods S1) were implemented using JAGS 4.3.0 (Plummer, 2003) via the R package rjags in R 4.1.0 (R Core Team, 2020). Three independent chains were initialized with random starting values and run until convergence was achieved. Convergence was assessed with visual inspection and the Gelman-Rubin diagnostic (Gelman & Rubin, 1992). For biomass and cover models where block was the only random effect, each chain was run for 15,000 iterations and thinned by 5, resulting in 9000 total samples saved for posterior calculations. Cheatgrass density models nested two random effects (block and observation-level) to account for overdispersion, which required each chain to be run for 150,000 iterations and thinned by 50 to achieve 9000 relatively independent posterior samples.

Converged chains were combined and summarized as the posterior mean and central 95% credible interval (CI). If the 95% CI of a covariate did not overlap zero, the covariate was deemed significant. An effect was deemed marginally significant if a post-hoc, one-sided test showed a Bayesian p -value < 0.1 even when the 95% CI overlapped with zero. All data is available online (Guo et al., 2022).

3 | RESULTS

3.1 | Biomass in unseeded plots across plant functional groups

For ungrazed, unseeded control plots, the biomass of cheatgrass (31.0 g m⁻² [24.4, 37.6]) was one order of magnitude greater than that of native grasses (3.11 g m⁻² [1.49, 4.74]), with forbs exhibiting intermediate biomass (14.8 g m⁻² [7.62, 21.9], Figure 2a). Spring grazing significantly decreased cheatgrass biomass, with no significant effects on either forb or native grass biomass (Figure 2b), while fall targeted grazing significantly increased forb biomass (Figure 2b). Thus, the ratio of cheatgrass to forb biomass was lower under both grazing treatments than in ungrazed controls (Figure 2a). Native grass biomass negatively covaried with the biomass of the other functional groups while cheatgrass and forb biomass did not covary significantly (Figure 2c).

3.2 | Cheatgrass density

Across all surveyed plots, herbicide treatments in 2014–2016 significantly reduced post-fire cheatgrass density in 2019 by approximately 3700 m² relative to ungrazed controls (3890 m²);

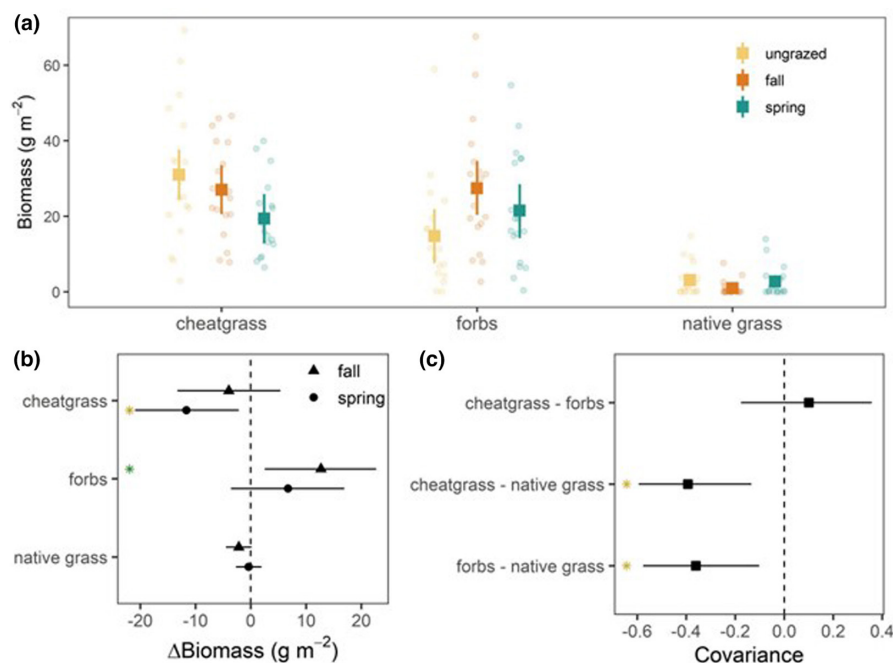


FIGURE 2 (a) Plant biomass as raw data (background) and modelled means (foreground) of unseeded control plots by plant functional group and across grazing treatments ($n = 54$). (b) The offset effects of each grazing treatment relative to the ungrazed control. (c) The covariance in biomass among the three functional groups. Foreground points and bars represent the posterior mean and central 95% credible interval. Significant positive (negative) variables are denoted by a green (gold) asterisk.

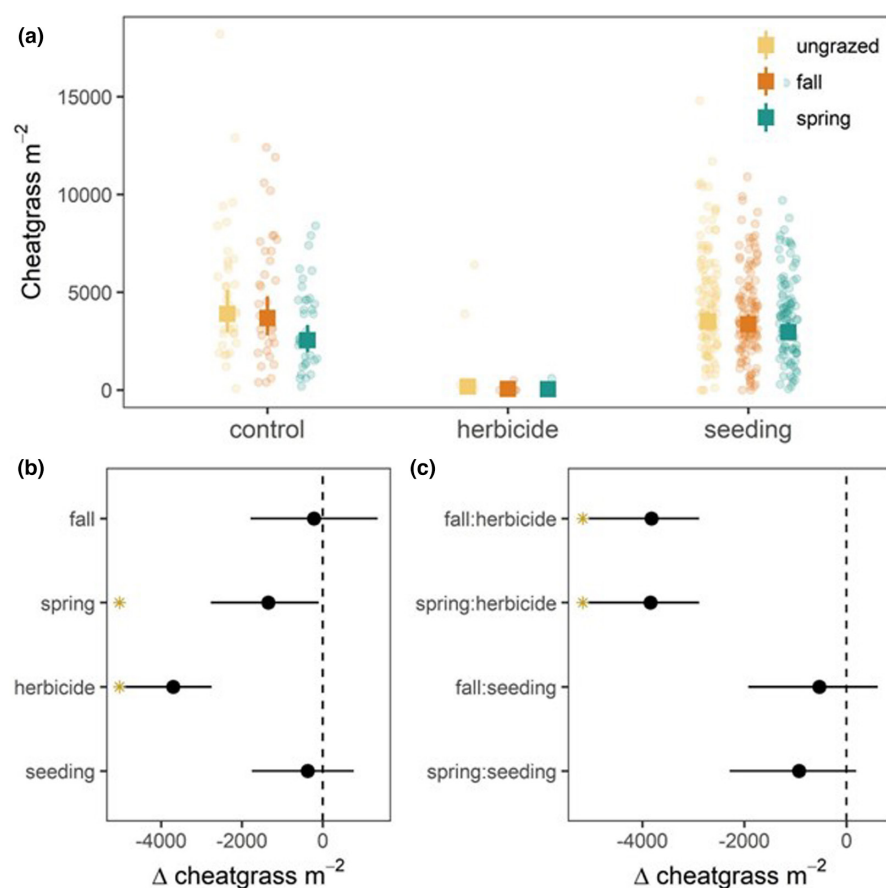


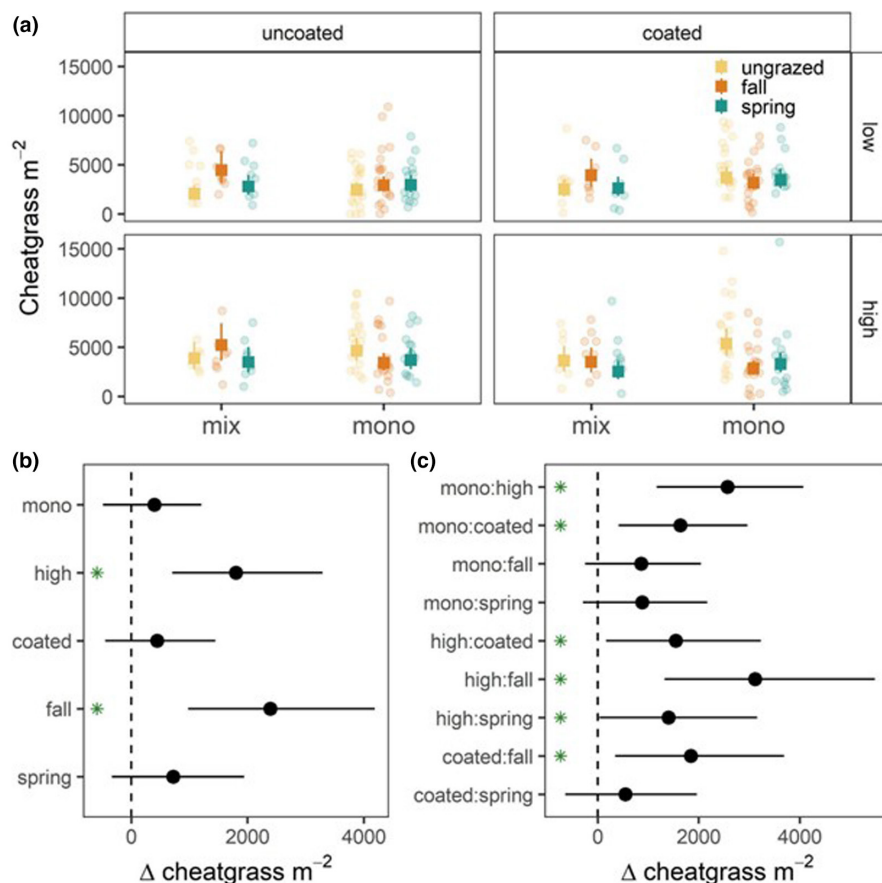
FIGURE 3 (a) Cheatgrass density as raw data (background) and modelled means (foreground) for all plots across grazing and plot type treatments (Model 1, $n = 458$). Main effect (b) and two-way interaction (c) offsets relative to the ungrazed and untreated control ($3890 \text{ cheatgrass m}^{-2}$ [2950, 5120]). Foreground points and bars represent the posterior mean and central 95% credible interval. Significant negative effects are denoted by a gold asterisk.

independently, spring grazing decreased cheatgrass density by 1350 m^2 (Figure 3a,b). The combined effect of herbicide with either spring or fall grazing reduced post-fire cheatgrass density by about 3800 m^2 (Figure 3c). The main effect of seeding did not significantly reduce cheatgrass density (Figure 3a), but when combined with

spring grazing, cheatgrass density was marginally reduced by 929 m^2 (Bayesian $p = 0.055$).

Within the seeded plots, the lowest cheatgrass density (2066 m^2) was found in the reference treatment: ungrazed mixture plots planted with uncoated seeds at low seed rates (Figure 4a).

FIGURE 4 (a) Counts of cheatgrass as raw data (background) and modelled means (foreground) for seeded plots across spatial arrangement, seed rate, seed coat, and grazing treatments (Model II, $n = 316$). Main effect (b) and two-way interaction (c) offsets relative to the mixture, low seed rate, uncoated, and ungrazed control (2060 cheatgrass m^{-2} [1430, 2880]). Foreground points and bars represent the posterior mean and central 95% credible interval. Significant positive effects are denoted by a green asterisk.



High seed rates and fall grazing significantly increased cheatgrass density, while monoculture spatial arrangement, seed coating, and spring grazing did not alter cheatgrass density (Figure 4b). While all two-way interactions involving high seed rate significantly increased cheatgrass density, the combination of high seed rate and fall grazing had the largest increase on cheatgrass density (3120 m^{-2} , Figure 4c). Monoculture spatial arrangement in conjunction with high seed rate or coated seeds also exhibited increased cheatgrass density (Figure 4c).

Within the monoculture seeded plots, we found that native grass species identity did not significantly alter cheatgrass density, although high seed rates and coated seeds were associated with greater cheatgrass abundance (Figure S3a,b). At high seed rates, cheatgrass density was also significantly higher in plots experiencing fall grazing, planted with coated seeds or planted with POFE (Figure S3c).

3.3 | Cheatgrass, forb and native grass cover

Across all plots, cheatgrass had the highest median cover at 30% while introduced forbs had the second highest median cover at 18%. Median cover of native grasses and forbs were 1% and 0%, respectively, while the median cover of bare ground was 47%. For all surveyed plots, pre-fire herbicide treatments significantly reduced post-fire cheatgrass cover by 23.5% relative to the ungrazed

controls (38.5%, Figure 5a,b), with even larger reductions (Figure 5c) in conjunction with fall (29.9%) and spring grazing (31.4%). Spring grazing marginally decreased cheatgrass cover by 6.7% (Bayesian $p = 0.052$, Figure 5b).

Within the seeded plots, monoculture spatial arrangement increased cheatgrass cover by 9.7%, high seed rates increased cheatgrass cover by 13.5%, and fall grazing increased cheatgrass cover by 16.8%, relative to ungrazed mixture plots planted with uncoated seeds at low seed rates (20.7%, Figure 6a,b). All combined two-way effects significantly increased cheatgrass cover (Figure 6c), confirming that the reference condition experienced the lowest cheatgrass cover among all treatment combinations (Figure 6a).

Among the monoculture seeded plots, species identity did not significantly alter cheatgrass cover (Figure S4), although high seed rates significantly increased cheatgrass cover by 19.8% relative to the ungrazed ELTR planted with uncoated seeds at low rates (27%, Figure S4b). Plots planted with POFE and ELEM at high seed rates and plots planted with coated seeds of VUMI and ELEM also had significantly higher cheatgrass cover (Figure S4c).

Among all plots, pre-fire herbicide did not significantly reduce post-fire forb cover (largely dominated by introduced forbs; Figure S5a,b). Fall grazing significantly increased forb cover by 6.8% relative to the ungrazed control (19.5%), while seeding treatments marginally reduced forb cover by 3.8% (Bayesian $p = 0.027$, Figure S5b). None of the combined two-way interactions were significant (Figure S6c). Within seeded plots, fall and spring grazing

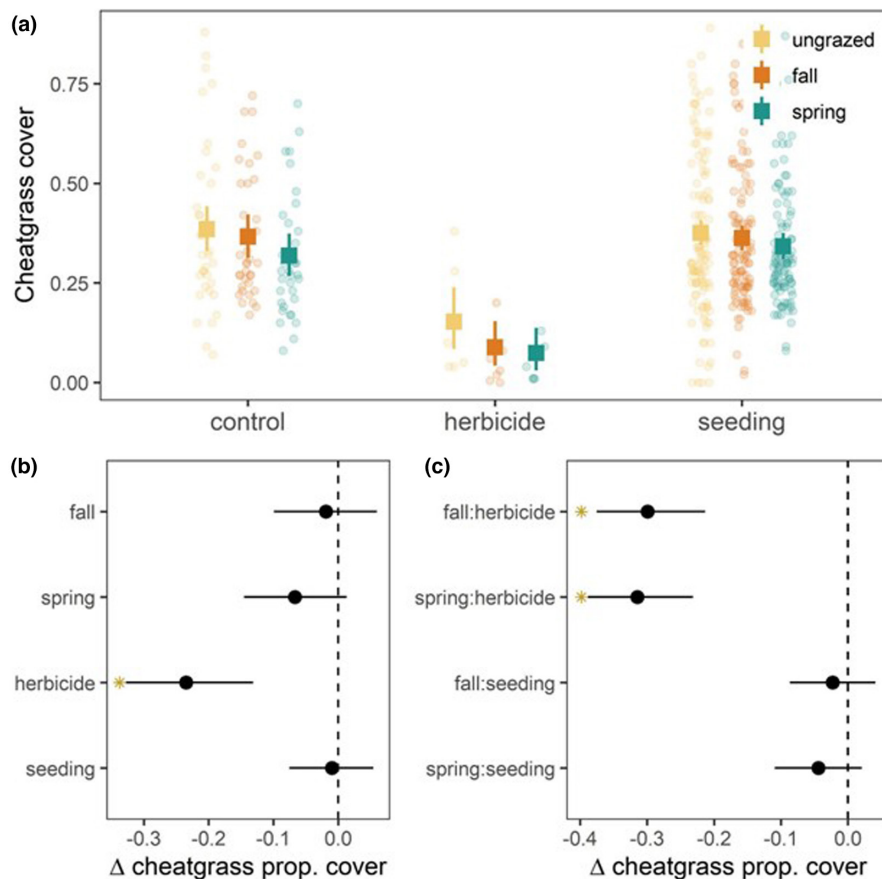


FIGURE 5 (a) Proportion of cheatgrass cover as raw data (background) and modelled means (foreground) for all plots across grazing and plot type treatments (Model I, $n = 453$). Main effect (b) and two-way interaction (c) offsets relative to the ungrazed and untreated control (39.0% cheatgrass cover [33.0%, 44.2%]). Foreground points and bars represent the posterior mean and central 95% credible interval. Significant negative effects are denoted by a gold asterisk.

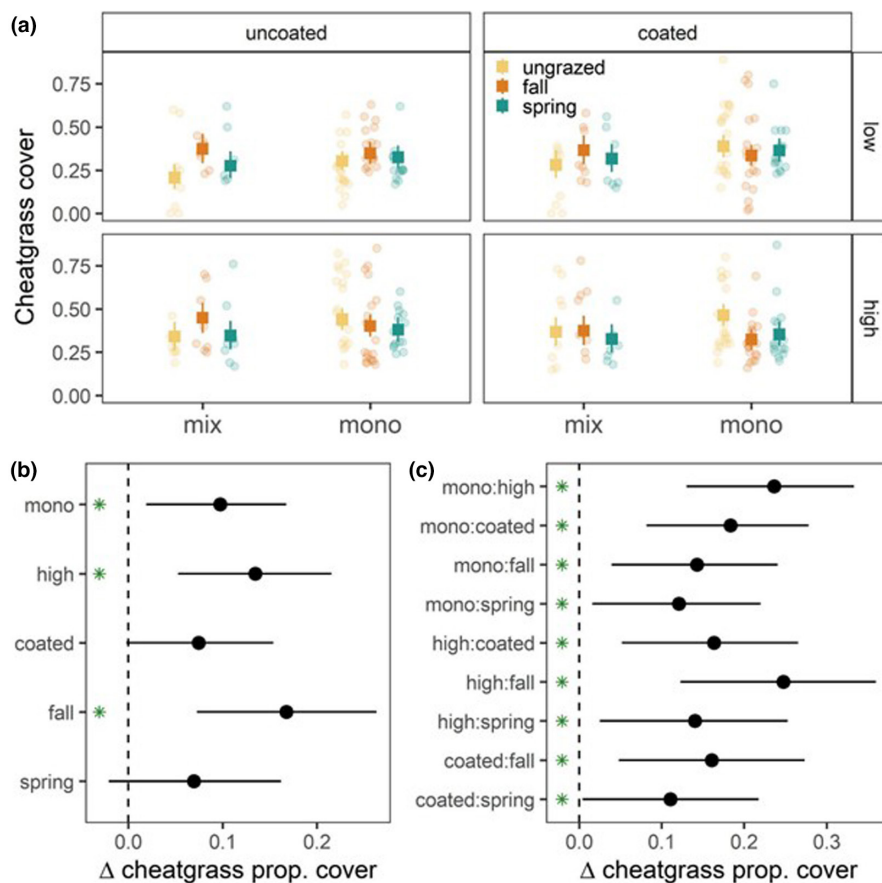


FIGURE 6 (a) Proportion of cheatgrass cover as raw data (background) and modelled means (foreground) for seeded plots across spatial arrangement, seed rate, seed coat, and grazing treatments (Model II, $n = 310$). Main effect (b) and two-way interaction (c) offsets relative to the mixture, low seed rate, uncoated, and ungrazed control (20.7% cheatgrass cover [14.2%, 28.4%]). Foreground points and bars represent the posterior mean and central 95% credible interval. Significant positive effects are denoted by a green asterisk.

significantly increased forb cover by 10% and 7.8%, respectively, relative to ungrazed mixture plots planted with uncoated seeds at low seed rates (11.0%, Figure S6a,b), while seed monocultures and high seed rates marginally increased forb cover by 3.9% and 3.8%, respectively (Bayesian $p = 0.032$ for both). Most of the combined two-way interactions also significantly increased forb cover (Figure S6c). Among the monoculture seeded plots, none of the main effects of species identity, seed rate, grazing nor seed coating was significant.

Of the 453 plots across all treatments, native grass was absent in 192 plots and median cover in the remaining plots was only 3%. The models of cover yielded no significant differences and we focus on describing how probability of absence (ρ) varied among the ANOVA factors. For all plots, spring grazing alone and in conjunction with seeding significantly reduced native grass probability of absence relative to the ungrazed controls, while the combination of fall grazing and herbicide treatment made absence more likely (Figure S7a–c). Among seeded plots, monoculture spatial arrangement increased the probability of absence for native grass (Figure S8a,b), while many combined two-way interactions also increased the probability of absence (Figure S8c). Among monoculture seeded plots, *Poa secunda* had significantly reduced probability of absence of native grass, both alone and in conjunction with all other treatments (Figure S9a–c).

4 | DISCUSSION

In the Western US, there is a critical need to manage wildfire associated with the invasion of cheatgrass (Fenesis et al., 2011). We investigated the utility of coupling common rangeland management strategies pre-fire to modify plant community outcomes 2 years post-fire. Because environmental effects from a previous year can significantly alter plant community response in subsequent years through legacy effects (Stampfli & Zeiter, 2020), we expected that pre-fire grazing and seeding would modify post-fire cheatgrass dynamics. Even after 5 years and a fire, we found that herbicide, alone or in conjunction with grazing, was most effective at reducing cheatgrass in an arid grassland. Spring grazing did reduce cheatgrass density and cover 2 years post-fire, but this treatment was less effective than herbicide individually. Among seeding treatments, high seed rate and coated seeds increased cheatgrass, suggesting that treatments meant to promote native grasses may have inadvertently benefitted cheatgrass.

Three years of herbicide application (1 year of glyphosate followed by 2 years of imazapic) were the most effective treatments to maintain a long-term reduction in cheatgrass density and cover, compared to control treatments. Although herbicide has been shown to be effective in controlling cheatgrass (Monaco et al., 2017), the response of native plant communities to herbicide treated areas can be unpredictable (Elseroad & Rudd, 2011). Broad-scale herbicide application can also be accompanied by significant expense, logistical challenges, and need for repeated application, which can make it a difficult strategy to apply at scale (Merriam et al., 2006; Pellent, 1990). The pairing of spring

grazing with herbicide did significantly enhance control of cheatgrass, reducing cover beyond herbicide alone, which is consistent with findings that integrated weed control strategies often produce greater weed control than singular approaches (Gornish et al., 2018; Lehnhoff et al., 2019). Deploying only spring targeted grazing for 2 years pre-fire decreased cheatgrass biomass, cover and density measured 2 years post-fire. Interestingly, spring grazing effects were generally dampened by the presence of seeding treatments. Spring grazing treatment effects were in stark contrast to fall grazing treatment effects, which generally resulted in enhanced cheatgrass dominance. This was unexpected as fall grazing typically reduces cheatgrass via a reduction in the seed-bank (Perryman et al., 2020). Clearly, grazing may not produce consistent results, particularly when paired with a fire, and it is therefore critical to consider timing, duration, and intensity of use when designing targeted grazing treatments.

Among seeding treatments, seed mixtures proved more effective than monocultures for reducing both cheatgrass count and cover. High diversity plantings are known to modify a variety of habitat characteristics, including those that might affect cheatgrass dominance, such as soil microbial communities and soil nutrients (e.g. Jones et al., 2015), although this is not always the case (Yang et al., 2022). Clearly, cheatgrass control techniques that include a seeding component should consider a high diversity seed mixture. This might be particularly effective when coupled with multiple years of herbicide application. However, the application of this approach should be considered on a case by case basis as the outcomes of native species seeding are often driven by site specific conditions (Stroot et al., 2021).

When compared to mixture seedings of uncoated seeds at low rates (a “business as usual” seeding approach), the majority of alternative seed treatments were either benign or actually enhanced cheatgrass dominance 5 years after seeding (and 2 years post-fire). For example, the combination of high seed rate and coated seeds significantly increased cheatgrass count and cover, which suggests that more intensive interventions may not only be more expensive in the short-term, but also have a short-term benefit and a long-term cost. While high seed rates and coated seeds enhanced the density of the planted native grasses 2 years after seeding (Porensky et al., 2018), these treatments increased cheatgrass count and cover by year five. These seed treatments might have generally enhanced facilitation by shading the soil surface and increasing local water availability (e.g. Abella & Smith, 2013). Alternatively, high seed rates and coated seeds may have increased intraspecific competition among seeded plants to the ultimate benefit of cheatgrass, for example by delaying the development of the seeded species such that smaller individuals were unable to survive the wildfire or some other disturbance. Finally, Porensky et al. (2018) reported that increases in cheatgrass may have been associated with the mechanical disturbance of the drill used to plant seeded plots. It could be that initial increases in cheatgrass associated with drill seeding persisted and were amplified post-fire. Regardless of mechanism, we can conclude that although

certain seeding treatments led to high native grass establishment in the first 2 years after seeding (Porensky et al., 2018), these treatments were not associated with persistent cheatgrass reductions in the post-fire context.

Five years after seeding in our experiment, native grass cover was extremely low across the seeded treatments. The failure of our seeding treatments after 5 years is concerning, but perhaps not surprising, given the generally low success rates of active restoration in arid, invaded ecosystems (Shackelford et al., 2021). Such failures severely limit the ability of restoration seedings to reduce invasion and fire over longer timescales in this system. In the face of low success rates overall, we found that spring grazing did enhance the probability that native grasses would be present. There were also some indications that plots seeded with *Poa secunda* performed better than other plots, while plots seeded with *Poa fendleriana* performed worse.

Unintended wildfire is a hugely problematic ecological and social disturbance that is becoming more common across rangelands worldwide (Balch et al., 2013). Because post-fire management can be extremely difficult, logistically challenging and expensive, pre-fire management might become a more common component of arid land management. Our study shows how typical management practices that are expected to resist the dominance of a flammable invasive grass might not always operate as intended. Although herbicide use is sometimes associated with ecological and human health risks (e.g. Norgaard, 2007), its disproportionate ability to control cheatgrass on its own or in combination with other approaches highlights its value as a pre-fire management tool. While spring grazing was also able to reduce the dominance of cheatgrass post-fire, these changes were modest. Clearly, additional research is needed to identify unintended negative consequences of common restoration techniques (such as drill seeding desired native species) and foster the development of approaches that ultimately lead to more effective and fire-resilient rangeland vegetation management strategies.

AUTHOR CONTRIBUTIONS

Lauren M. Porensky, Elizabeth A. Leger, and Barry L. Perryman installed the experiment and collected data; Jessica S. Guo analysed data; Elise S. Gornish and Jessica S. Guo wrote the manuscript; Lauren M. Porensky and Elizabeth A. Leger edited the manuscript.

ACKNOWLEDGEMENTS

We thank Albert Kline for field help. We thank the Elko Land and Livestock Company and Newmont Mining Corporation for providing facilities, labor, and permission for this research. For assistance setting up the experiment, we are grateful to Owen Baughman, Jason Sprott, Scot Ferguson, Marenna Disbro, Quinn Campbell, Eleni Nicholson, Sarah Barga, Bryce Wehan, Sage Ellis, Erick Larsen, Jen Peterson, and Brian Von Seggern. Funding was from a National Institute of Food and Agriculture, U.S. Department of Agriculture award within the Western Integrated Pest Management Program (2013-34103-21325).

CONFLICT OF INTEREST STATEMENT

The authors declare no conflict of interest.

DATA AVAILABILITY STATEMENT

All data and scripts were versioned on GitHub and archived on Zenodo <https://doi.org/10.5281/zenodo.7272546> (Guo et al., 2022).

PEER REVIEW

The peer review history for this article is available at <https://www.webofscience.com/api/gateway/wos/peer-review/10.1002/2688-8319.12215>.

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SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.

Supplement Method S1. Modeling background.

Figure S1. (a). Timeline of treatments; (b). Treatment orientation (figure adapted from Porensky et al., 2018). Black and red squares represent 1 × 1 m sampling quadrats. In monoculture plots, multicoloured strips represent planting zones for different species; (c). Spatial arrangement of experimental blocks and grazing treatments.

Figure S2. Diagram of the three kinds of models performed on cheatgrass density, cheatgrass cover, forb cover, and native grass cover. Bolded levels of each factor were established as the reference for the offset ANOVA models.

Figure S3. (a) Cheatgrass density as raw data (background) and modelled means (foreground) for monoculture seeded plots across species, seed rate, seed coat, and grazing treatments (Model III, $n = 220$). Main effect (b) and two-way interaction (c) offsets relative to the ELTR, low seed rate, uncoated, and ungrazed control (2380 cheatgrass m^{-2} [1440, 4040]). Foreground points and bars represent the posterior mean and central 95% credible interval. Significant positive effects are denoted by a green asterisk.

Figure S4. (a) Proportion of cheatgrass cover as raw data (background) and modelled means (foreground) for monoculture seeded plots

across species, seed rate, seed coat, and grazing treatments (Model III, $n = 214$). Main effect (b) and two-way interaction (c) offsets relative to the ELTR, low seed rate, uncoated and ungrazed control (27.0% cheatgrass cover [18.0%, 38.4%]). Foreground points and bars represent the posterior mean and central 95% credible interval. Significant positive effects are denoted by a green asterisk.

Figure S5. (a) Proportion of forb cover as raw data (background) and modelled means (foreground) for all plots across grazing and plot type treatments (Model I, $n = 453$). Main effect (b) and two-way interaction (c) offsets relative to the ungrazed and untreated control (19.3% forb cover [15.8%, 22.8%]). Foreground points and bars represent the posterior mean and central 95% credible interval. Significant positive effects are denoted by a green asterisk.

Figure S6. (a) Proportion of forb cover as raw data (background) and modelled means (foreground) for seeded plots across spatial arrangement, seed rate, seed coat, and grazing treatments (Model II, $n = 310$). Main effect (b) and two-way interaction (c) offsets relative to the mixture, low seed rate, uncoated, and ungrazed control (11% forb cover [7.7%, 15%]). Foreground points and bars represent the posterior mean and central 95% credible interval. Significant positive effects are denoted by a green asterisk.

Figure S7. (a) Proportion of native grass absence as raw data (bars) and modelled means (foreground) for all plots across grazing and plot type treatments (Model I, $n = 453$). Main effect (b) and two-way interaction (c) offsets relative to the ungrazed and untreated control (49.9% absent [33.9%, 67.9%]). Foreground points and bars represent the posterior mean and central 95% credible interval. Significant positive (negative) effects are denoted by a green (gold) asterisk.

Figure S8. (a) Proportion of native grass absence as raw data (bars) and modelled means (foreground) for seeded plots across spatial arrangement, seed rate, seed coat, and grazing treatments (Model II, $n = 310$). Main effect (b) and two-way interaction (c) offsets relative to the mixture, low seed rate, uncoated, and ungrazed control (8% absence [0.6%, 22.5%]). Foreground points and bars represent the posterior mean and central 95% credible interval. Significant positive effects are denoted by a green asterisk.

Figure S9. (a) Proportion of native grass absence as raw data (bars) and modelled means (foreground) for monoculture seeded plots across species, seed rate, seed coat, and grazing treatments (Model III, $n = 214$). Main effect (b) and two-way interaction (c) offsets relative to the ELTR, low seed rate, uncoated and ungrazed control (65.7% absence [31.4%, 94%]). Foreground points and bars represent the posterior mean and central 95% credible interval. Significant negative effects are denoted by a gold asterisk.

How to cite this article: Gornish, E. S., Guo, J. S., Porensky, L. M., Perryman, B. L., & Leger, E. A. (2023). Pre-fire grazing and herbicide treatments can affect post-fire vegetation in a Great Basin rangeland. *Ecological Solutions and Evidence*, 4, e12215. <https://doi.org/10.1002/2688-8319.12215>