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Graduating Essay

A decade of change: Assessing the changes in plant community composition following a post-fire seeding treatment.

Abstract

Seeding treatments are a commonly used restoration strategy to stabilize watershed, prevent erosion, suppress invasive species establishment, and promote native plant recovery after a wildfire. However, there is a lack of knowledge around the ecological consequences that occur over the long-term with this management tool, especially after a high severity fire. This study examines the long-term outcome of aerially broadcast seeding of an agronomic seed mix on a Ponderosa pine-Douglas fir forest that experienced a high severity fire 20 years ago. Plant community composition was assessed annually for the first ten years after the treatment was applied. Following this, a decade long gap occurred after 2014, where no assessments were conducted until 2024. Using the vegetation data collected in 2024, this study assesses the current plant community composition to determine what changes have occurred over the last decade between the seeded and unseeded area, and investigates whether seeding has helped suppress invasive species and assisted in the recovery of native species. The results illustrate that the seeded species have maintained dominance in the seeded plots, and unseeded plots. Two species in particular, Poa *compressa* and *Medicago sativa*, have substantially increased in abundance, amplifying competition across the site. Furthermore, native species richness was lowest in the seeded areas, which contrasted with non-native invasive species, which had a greater presence in the seeded treatments. Additionally, species diversity was found to be greater in the unseeded areas, further highlighting the increased competition brought about by the seeded species. These findings show the potential impacts that can come about from seeding treatments, and emphasize the need for adaptive management approaches, which long-term monitoring can provide.

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Acknowledgments

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Introduction

Disturbances are the fundamental drivers of ecosystem change and hold the potential for a single event to pose long-term alterations to the structure and function of ecosystems (D'Antonio, Yelenik and Mack, 2017). Shifts in disturbance regimes and climate increase the opportunity for species with high colonization potential, rapid growth and high fecundity to establish on site (Kluender and Germino, 2023), which in turn, guides plant communities away from their pre-disturbance conditions (Wion et al. 2024). The resulting impacts can drastically alter biological diversity (Grant, Shaffer and Flanders, 2020), making disturbances primary selective agents for plant community composition (Kluender and Germino, 2023). On a global scale, ecosystems have historically been modeled by various forms of disturbances, including overgrazing and wildfire (Knutson, et al. 2014; Grant, Shaffer and Flanders, 2020).

Impacts of wildfires

Wildfires can rapidly reshape ecosystems, with immense changes occurring based on the scale of frequency and severity. The occurrence, scale and intensity of wildfires continues to rise around the world (Grant-Hoffman and Dollerschell 2019; Davies, Boyd, Martyn and Bates 2024; Kerns and Day 2024) due to decadal-scale megadrought, human caused ignitions and anthropogenic climate warming (Davies, Boyd, Martyn and Bates, 2024, Wion et al. 2024). Burns modify abiotic conditions (e.g., nutrient availability, soil exposure, light levels) and biotic processes (seed germination and competition) (Kerns and Day, 2024). High severity fires negatively effect species richness, particularly for native species (Wion et al. 2024) because of the significant loss of vegetative cover (Hunter, Omi, Martinson and Chong, 2006). Although fire is crucial for maintaining heterogeneity in the plant community (Araujo, et al. 2017), it also increases the risk of producing novel ecosystems, where post-fire plant assemblages differ significantly from pre-disturbance conditions (Knutson, et al. 2014). The loss of important functions through the forest transformation can elevate runoff and soil erosion (Hunter, Omi, Martinson and Chong 2006; Wion et al. 2024), while simultaneously decreasing wildlife habitat and biological diversity (Knutson, et al. 2014). The post-fire “clean slate” also provides favourable conditions for exotic species, which are often better adapted to exploit disturbed environments than native plants (Hunter, Omi, Martinson and Chong, 2006; Uzra, et al. 2019; Kluender and Germino, 2023; Kerns and Day, 2024).

Impacts of invasive species establishment

The establishment of non-native species is frequently facilitated by disturbances, and their ability to persist depends on competitive traits and the post-fire response of native species (Uzra, et al. 2019). The increased availability of resources following a wildfire provide a competitive advantage to exotic species (Uzra, et al. 2019), as they can monopolize soil nutrients that are inaccessible to native species (D'Antonio, Yelenik and Mack, 2017), rapidly extract soil water, and reproduce at a high rate (Knutson, et al. 2014). Many exotic grasses, such as the highly competitive *Bromus inermis,* establish successfully in response to increasing disturbances (Adair, Burke, and Laurenroth 2007), contributing fine-scale fuel homogeneity in the ecosystem (Araujo, et al. 2017). The high flammability of this species (Uzra et al. 2019), can alter fire regimes by increasing both fire frequency and intensity with the provided fuel continuity, creating a positive fire-invasion feedback loop (Knutson, et al. 2014; D'Antonio, Yelenik and Mack, 2017; Kluender and Germino 2023). These positive feedbacks promote permanent ecosystem shifts, as recurring fires further limit native species recovery through the loss of stored nutrients in plant materials (D'Antonio, Yelenik and Mack, 2017) and disrupt ecosystem processes such as nutrient cycling, soil water flux and storage, and favour fire-adapted invasive species (Uzra et al. 2019). The capacity for invasive species to persist through these disturbance-recovery cycles (Kluender and Germino 2023) can lead to permanent changes in structure, function, or a complete vegetative state shift (Araujo, et al., 2017). If they remain dominant, the changes to ecosystem properties could be altered for decades (D'Antonio, Yelenik and Mack, 2017).

Seeding for restoration

Effective post-fire management strategies are critical to facilitate long-term ecosystem recovery and prevent the spread of invasive species. Amplified fire frequency has increased the use management inputs post-disturbance (Grant-Hoffman and Dollerschell, 2019). Seeding is a frequently used tool for post-fire rehabilitation (Grant-Hoffman and Dollerschell, 2019; Davies, Boyd, Martyn and Bates, 2024), in an attempt to rapidly establish vegetation cover and prevent ecosystem degradation (Hunter, Omi, Martinson and Chong, 2006; Ott, Kilkenny, Summers, Thompson and Petersen, 2022). Common applications of native and non-native perennial grasses include aerial broadcast or drill seeding (Knutson, et al. 2014). Aerial broadcasting is a relatively inexpensive method to achieve seed cover across large, burned areas, especially with logistical challenges (Grant-Hoffman and Dollerschell, 2019). Drill seeding is often preferred due to the better seed-to-soil contact, though some studies have reported mechanical soil disturbances (compaction) with this method (Kluender and Germino, 2023; Davies, Boyd, Martyn and Bates, 2024). Seeding objectives are often divided into two categories: (1) protect human health and property (Grant-Hoffman and Dollerschell, 2019), which includes reducing the risk of postfire flooding and debris flows, soil erosion and watershed degradation (Knutson, et al. 2014; Wion et al, 2024), (2) restore native plant communities and biodiversity, prevent establishment of exotic species, increase grazing forage and improve wildlife habitat (Knutson, et al. 2014; Grant-Hoffman and Dollerschell, 2019; Wion et al, 2024).

Although seeding is generally considered effective for resisting ecosystem changes (Wion et al. 2024), there are some concerns about the frequent use of non-native species (Hunter, Omi, Martinson and Chong, 2006; Knutson, et al. 2014; Ott, Kilkenny, Summers, Thompson and Petersen, 2022) and their effectiveness is variable (Kluender and Germino, 2023). The common use of aggressive non-native (Grant-Hoffman and Dollerschell, 2019), mostly Eurasian species (Ott, Kilkenny, Summers, Thompson and Petersen, 2022), can interfere with the regeneration and establishment of native vegetation (Knutson, et al. 2014). Their larger seed production and rapid water extraction (Knutson, et al. 2014) can result in unintended consequences of competition (Grant-Hoffman and Dollerschell, 2019) and undesirable impacts to native plant communities (Davies, Boyd, Martyn and Bates, 2024). Additionally, there is large gap in knowledge around the long-term ecological effects of type-converted plant communities (Wion, et al. 2024).

While fire recovery strategies are more commonly used today, many ecosystems have evolved through periodic fire disturbance prior to their use (Knutson, et al. 2014). The reoccurrence of low-severity fires has helped dry, coniferous forests build resiliency (Wion et al. 2024), with some species acquiring traits to respond positively to fire (Kerns and Day, 2024), such as *Pinus ponderosa*, with serotinous cones that require heat to release seeds (Nolan et al. 2021). However, changes in land use over the last century, including intense grazing practices, fire suppression, increased logging of fire-resistant trees (Wilon et al. 2024) and the invasion of non-native annual plants (Ott, Kilkenny, Summers, Thompson and Petersen, 2022) has reduced the resiliency of these uncharacteristically dense forests. This is particularly prominent in

Ponderosa pine and dry mixed conifer stands in semiarid ecosystems, where irreversible outcomes of stand-replacing fires and invasive plant dominance (Underwood, Klinger, and Brooks, 2019) are propelled by drought, historical land uses, and fire (Wion, et al. 2014).

Study Objectives

Given the complex ecological interactions between fire, native species recovery, and invasions of exotic species, it is important to evaluate the long-term effectiveness of post-fire seeding treatments, to provide informed adaptive management strategies (Knutson, et al. 2014). This study assesses the success of aerial broadcast seeding as a restoration tool by examining the plant community composition of a Ponderosa pine-Douglas fir stand that experienced a high severity wildfire 20 years ago and has since transformed into a grassland ecosystem. Uniquely, this research compares both seeded and unseeded areas, offering valuable insight into the long-term ecological outcomes of post-fire seeding treatments.

Following the initial seeding treatment, annual vegetation assessments were conducted over the first 10 years (2004-2014) to monitor species establishment. However, no further assessments occurred for another 10 years until 2024, leaving an important gap in understanding long-term plant community dynamics. This focus of this study is changes to the plant community that have taken place since the last monitoring a decade ago. This study aims to analyze changes in plant community composition since 2014, assess whether seeding has successfully suppressed invasive species establishment and evaluate whether the seeding treatment facilitated the recovery of native plant species.

Methods

Site Description

The study site is approximately 10 km north of Kamloops, BC, in the Rayleigh community (50º47’40” N, 120º,17’44” W). The site is situated in the Thompson Very Dry Hot Interior Douglas Fir (IDFxh2) biogeoclimatic zone (Meidinger and Pojar, 1991). The slope aspect of the site is west – south-west, with elevation ranging between 650 – 750 m. The hillside has a slope average of 31%.

Original Plant Community

In the mid-1960s, a large portion of the area was logged. Prior to the fire, the site was dominated by coniferous trees, which totalled 70% of the landscape, and 30% was comprised of grasslands. Within that 70%, 115-year-old Douglas fir (*Pseudotsuga menziesii*) made up 65%, while 175-year-old ponderosa pines (*Pinus ponderosa*) supplemented the remaining 35% of the forested portion. The main species in the grassland openings included bluebunch wheatgrass (*Pseudoroegneria spicata*), needle-and-thread grass (*Festuca campestris*) and a mixture of grassland forb species. Pinegrass (*Calamagrostis rubescens*) was rarely found, only occurring in small draws on shadier portions of the site. Similarly, rough fescue (*Festuca campestris*) was uncommonly found, despite its dominance anticipated on cool, open slopes. Invasive species in the area included Canada thistle (*Cirsium arvense*) and spotted knapweed (*Centaurea biebersteinii*). Plumeless thistle (*Carduus acanthoides*) was found within 1 km of the site, and along the roadsides, dalmatian toadflax (*Linaria genistifolia*) developed on disturbed ground.

Grazing

The study site occurs in the 650-ha cattle pasture of the Seven-O Ranch grazing lease. Grazing was halted on the study site for the first two year after the fire. Following this period, a rest rotation grazing system was used, where grazing occurred within the study site in June of alternating years. Between 2012 and 2015, the grazing lease management plan transitioned to a spring and fall rotation of 3-week periods, for two bulls and 50 cow-calve pairs. Moderately averaged levels of grazing were observed between 3 to 11 years post-fire – though observations were limited to the annual site visit for plant assessments. After the 11th year, there was a reduced level of grazing observation, however, it is assumed that grazing rates remained consistent. The current grazing lease management plan held by the Seven-O Ranch is set for 1500 AUMs (Valentine, pers. comm).

Strawberry Hill Fire and Treatment

On August 1, 2003, the Strawberry Hill fire (K20298) began in the Rayleigh suburb of Kamloops, with windy, dry conditions producing a high rate of spread. Overall, 5,731 ha were burnt on the east side of Highway 5. The high severity fire burnt the understory of the study site and the canopy fire scorched all the tree needles, which resulted in complete mortality for all trees on site. Additionally, soil burn severity was moderate-to-high, greatly impacting propagules and plant growth structures. No salvage logging occurred on the study site, and no trees were planted following the fire. An assessment for plant survival was done 11 months after the fire and 25 plant species were documented, mostly found in rocky areas. The maximum cover found was 0.4% and only eight species had cover greater than 0.1%.

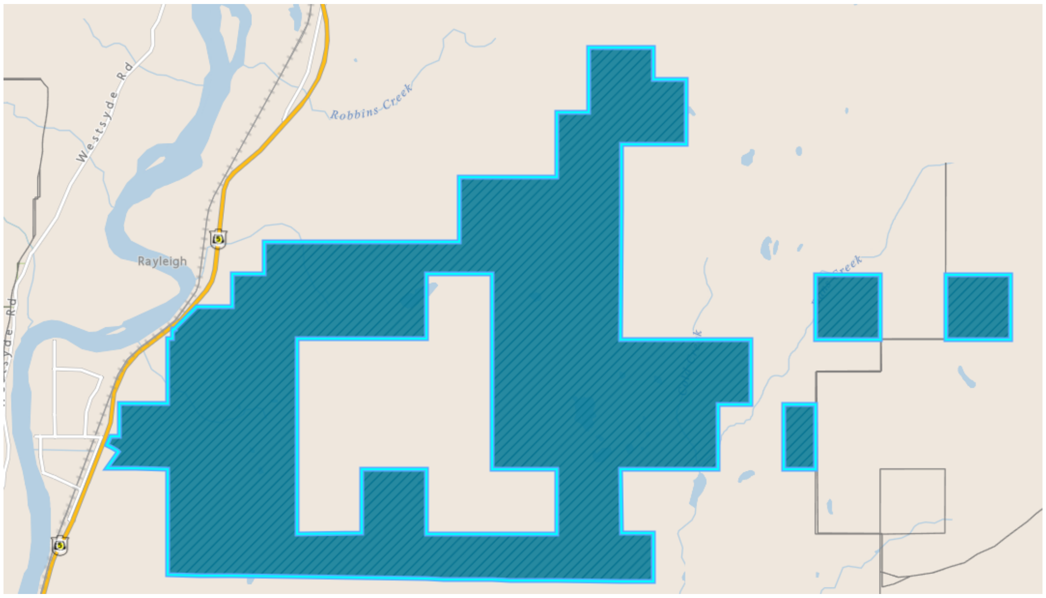

Figure 1. Basic outline of the Seven-O ranch grazing pasture held in the grazing management lease plan where the study site is located.

With the goal of hindering invasive species from establishing, while also re-establishing forage availability, the decision was made to seed some portions on land within the grazing lease. Fortunately, the May rainfall doubled the anticipated amount in 2004, providing ideal conditions for germination and establishment. Nearly 10 months after the destructive fire started, a standard Kamloops District weed and erosion control mix was aerially dispersed by helicopter on May 26, 2004. The agronomic seed mix was spread over four 250 m X 50 m plots, with equally sized unseeded plots rotated in-between them. The aerial application resulted in eighty-seven percent seed cover across the four treatment blocks. Visible seed density was estimated to be 289,000 seeds/ha (28.9 seed/m^2). After the treatment was placed on site, precipitation was 37% higher than average for the rest of the growing season. Soil analysis performed one year after seeding determined the soil to be moderately hydrophobic and precipitation was double the predicted amount in June 2005.

Table 1. Composition of the agronomic seed mixture used on the Strawberry Hill fire

Common Name	Scientific Name	% by weight	% by density	Seeds/kg	Seeds/m^2
Italian ryegrass	*Lolium multiflorum*	32	22	501,000	80
Creeping red fescue	*Festuca rubra*	5	8	1,068,380	27
Canada bluegrass	*Poa compressa*	4	31	5,511,500	110
Timothy	*Phleum pratense*	4	15	2,712,000	54
Western wheatgrass	*Pascopyrum smithii*	35	12	243,000	43
Rambler alfalfa	*Medicago sativa*	20	12	441,000	44

Data Collection

The 2024 vegetation sampling began on May 15th and took two days to complete. The first day eight individuals split into four groups of two to increase efficiency. Four members finished the sampling on the second day and continued monitoring in groups of two. Metal stakes were permanently placed in the ground in 2003, to identify the start and finish of 50 m transect lines. Each treatment plot has five 50 m transect lines, creating a total of 200 quadrants for both the seeded and unseeded plots. They were identified using a metal detector and flagged for easy detection. Vascular plant species were assessed using the canopy cover method, with 20 cm x 50 cm quadrats (Daubenmire, 1959). Bare soil, litter, wood, ash and cryptogrammic crust cover were also estimated within the quadrants. Using randomly generated numbers from excel, 10 quadrants were assessment on each line, resulting in 50 quadrants per treatment block (250 m X 50 m).

2004-2014 Sampling

The first quadrant sampling occurred in 2004. Sampling continued annually through to 2014 (Year 11), taking place during late June or early July. The only year sampling did not occur was 2011 (Year 8). To exclude wild ungulate and domestic cattle grazing, three portable cages made of meshed wire panels (1 m X 1 m X 1 m) were placed in each treatment in 2014. This is the only year the biomass was assessed. In August 2014, the three cages in treatment plot had three 0.5 m^2 areas clipped to ground level. Peak annual standing crop of grasses and forbs was used to estimate the living ground biomass. The living plant materials was grouped into functional groups (all grasses, all forbs) except for seeded species, which were identified by individual species. To minimize decomposition, all plant material was stored in paper bags to air dry before being oven-dried to a constant weight at 70°C and weighed to the nearest 0.1 g.

Figure 2. The first plant community assessment in the study site on May 15, 2024

Data Analysis

All vegetation data from 2014 and 2024 was provided in an excel spreadsheet and had been organized by year, functional group, treatment, species, and average cover for each of the four blocks in both treatments from the study. The data was uploaded into the R version 4.2.0 (R Core Team, 2022), where all data analysis and visualizations were performed. First, the total number of species in each functional group (forbs and graminoids) where assess to determine the level of change that has occurred. Afterwards, the vegan package was used to calculate Shannon's diversity index to identify species richness between native and exotic species. Simpson's diversity index was also calculated to assess dominance occurring within the treatments. A Two-way ANOVA was used to test for mean cover differences between native and exotic species in the seeded and unseeded plots.

Next, the mean cover was calculated for each all species and filtered into a new dataset to only include those who had cover greater than 1%. A second filter was applied to group the vegetation data by year and treatment. The ggplot2 package was used to generate histograms to portray variations in dominance species about treatments and changes that have occurred over the last decade. Species origin was used to colour the bars to illustrate native and invasive presence. Following this, 2024 datasets were combined and filtered to identify the dominant species in both treatment blocks to display dominance between functional groups. Again, origin was displayed as bar colours, though seeded species were removed from the exotic group and placed in their own origin (seeded).

Results

Initial Plant Community Establishment After Seed Application

Seeded Species Establishment

Even with optimal weather conditions and above-normal precipitation, only two seeded species established 30 days after the seed was applied (June 2004). The remaining four seeded species did not establish until the second growing season. Italian ryegrass (*Lolium multiflorum*) had 0.4% cover across 10% of the plots in 2004 and grew to reached 17% cover by 2005 and a small portion of 0.8% cover was found on the control plots. After this, it quickly decreased and essentially disappeared from site by year 4. Rambler alfalfa (*Medicago sativa*) had 0.1% cover in the first year but persisted to reach 14.5% cover by the 4th year.

The most abundant specie in the seed mixture, Canada bluegrass (*Poa compressa*) was also the most productive of the seeded species (Table 1), however, it appeared at a similar rate in the unseeded plots, presumably from drift during application due to its light seed weight. Western wheatgrass (*Pascopyrum smithii*) was the least abundant species in the mix, establishing slowly at 0.3% per year until year 7, where annual growth increased to reach 5.4% by 2014. Only 0.8% cover was recorded in unseeded plots. Creeping red fescue (*Festuca rubra*) gradually established, reaching a maximum cover of 10% in year 6. It is assumed that the prescribed rate of

Timothy (*Phleum pratense*) was excluded from the seed mixture since it is typically a successful specie and only reached at maximum cover of 0.19%.

Despite not being listed in the seed mixture, Smooth brome (*Bromus inermis*) was found in the treatment plots in the second growing season and obtained 3.8% cover by 2014. Similarly, Kentucky bluegrass (*Poa pratensis*) was identified in the third year and reached 4.1% by 2013. It is assumed that these species were contaminants in the seed mixture, since their cover on the treatment plots was considerably greater than the control plots.

Invasive Establishment

By the second growing season, 14 invasive species were recorded, and grew to 18 species by the fifth year. The species with the highest maximum cover included cheatgrass (*Bromus tectorum*) (18.1%); prickly lettuce (*Lactuca serriola*) (14.9%); common dandelion (*Taraxacum officinale*) (10.7%); horseweed (*Conyza canadensis*) (8.2%); yellow salsify (*Tragopogon dubius*) (2.9%) and corn brome (*Bromus squarrosus*) (2.4%). Provincially listed species found in low cover (<1.5%) were plumeless thistle (*Carduus acanthoides*), Canada thistle (*Cirsium arvense*), bull thistle (*Cirsium vulgare*), spotted knapweed (*Centaurea biebersteinii*) and Dalmation toadflax (*Linaria genistifolia*). Common dandelion appeared to be unaffected by the seeded species and was dominant in both treatments. In year 7, cheatgrass had reduced to 8.3% cover in the treatment plots and declined to 2.7% by 2014. Additionally, the seeded species were able to supress yellow salsify, horseweed and prickly lettuce to less than 1.5% cover.

Native Establishment

The dominant specie pre-fire, Bluebunch wheatgrass (*Pseudoroegnaria spicata*) was found in both treatments after a very slow establishment, reaching only 1.6% cover by year 10. Tall annual willowherb (*Epilobium brachycarpum*) reached 3.4% cover in the control area by year 3 but was supressed to 1.5% in the treatment. Similarly, yarrow (*Achillea millefolium*) had a slow establishment, eventually reached 4.3% in unseeded and 1.5% in seeded areas by 2014. Lastly, Columbia needlegrass (*Achnatherum nelsonii*) cover stayed consistently low, reaching a maximum of 1% in the control and remained absent in the seeded plots.

2024 Sampling Results

Through the vegetation sampling, 59 plant species were counted, which included 41 forbs and 18 graminoids. For the seeded agronomic species, 4 were found on the site (1 forb and 3 grasses), and will be identified separately from other exotic species that were not purposefully placed on site. The forbs identified consisted of 9 exotic species and 31 native species. The total graminoids count contained 3 exotic [*Bromus inermis, Bromus tectorum and Poa pratensis*] and 12 native species. In 2014, graminoids made up over half of the treatment plots, while forbs were more dominant in the control plots. However, in 2024, graminoids are leading both the treatment and control plots (2).

Table 2. Changes in function group abundance on seeded and unseeded plots in 2014 and 2024.

Functional Group	2014		2024	
	Seeded	Unseeded	Seeded	Unseeded
Forbs	41.09 %	52.94 %	38.66 %	39.34 %
Graminoids	58.91 %	47.06 %	61.34 %	60.66%

ANOVA and Diversity Indices

The comparison of plant communities is considerably different between the natural post-fire recovery and seeded treatment. Though they had the lowest cover amongst treatments in the previous assessment, the decline that has occurred over time is substantial, suggesting long-term recovery limitations across the site. This is particularly noteworthy in the control areas. This trend could have been influenced by the expansion of exotic species, creating higher competition in the control. However, the exotic growth in the treatment plots is exponentially higher (Table 3).

Table 3. Two-way ANOVA significance test on mean cover changes of native and exotic species (excluding seeded species) across treatments between 2014 and 2024.

Origin	Mean Cover Control		Mean Cover Treatment		P value
	2014	2024	2014	2024	
Native	3.05	1.88	1.28	0.57	0.0119
Exotic	5.41	6.54	5.95	9.53	0.554

Despite the shrinkage of native cover, Shannon's index indicates that species diversity is greater in the control plots, likely reflecting a more even distribution of plant cover (Table 4). Although there was a noticeable difference in mean cover between native and exotic species (Table 3), the results of the Simpson's diversity index did not show a significant difference in dominance occurring between the treatment and control plots.

Table 4. Results and significance of Shannon's and Simpson's diversity indices for the control and treatment areas

Diversity Index	Control Mean	Treatment Mean	P value
Shannon's	3.23	2.82	0.05929
Simpson's	0.935	0.921	0.2422

Comparison of Grass Community Composition between treatments

Grass Community in Control

A visible shift in the composition occurred in the grass community of the control plot over the last decade (Figure 3). Although the relative abundance of most native species has remained low, the number of native species increased from six to nine [JUBA (*Juncus balticus)*; VUOC (*Vulpia octoflora)*, POSE (*Poa secunda*)]. The opposite occurred for exotic species,

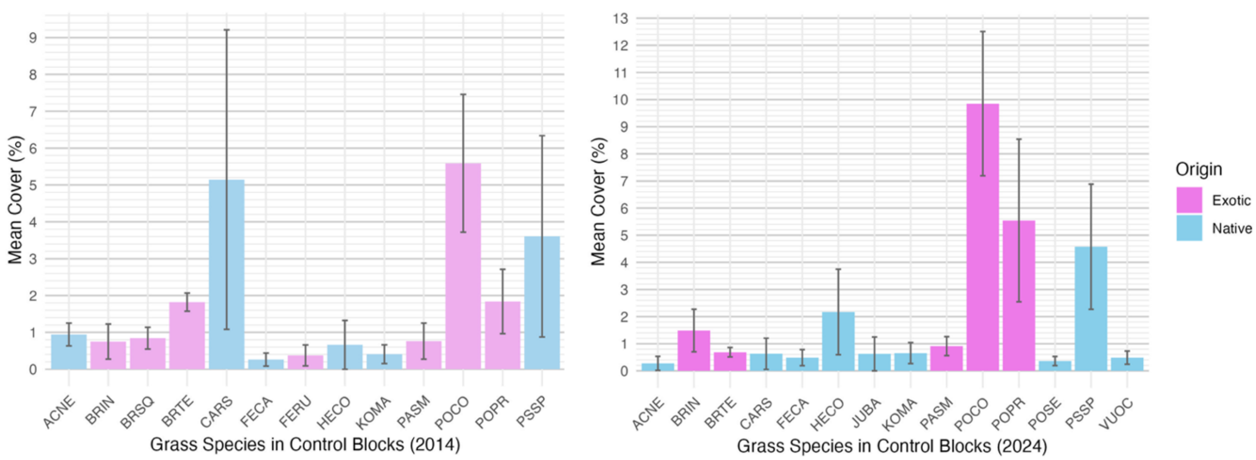

Figure 3. Grass community changes that have occurred to species with more than 1% cover in the unseeded plots over the last decade. Seeded and invasive species are grouped as exotics.

which decreased to five species present [BRSQ (*Bromus squarrosus*), FERU (*Festuca rubra*)]. Despite the decrease in numbers, seeded and invasive species exhibit the greatest dominance in the control plot, particularly POCO (*Poa compressa*) and POPR (*Poa pratensis*), who have both drastically grown in abundance.

A dramatic increase occurred with the pre-fire dominant understory species PSSP (*Pseudoroegneria spicata*) and HECO (*Hesperostipa comata*), who now represent the most abundant native species. However, CARS (*Carex rossii*), the previously dominant native species, experienced a decline. ACNE (*Achnatherum nelsonii*) also had a noticeable reduction, while other native species maintained relative low cover, similar to 2014 (Figure 3).

Grass Community in Treatment

Across both years of study, exotic species (primarily seeded species), dominated the seeded plots. Over the past 10 years, significant shifts in competition for dominance occurred, along with a loss of species richness in the grass community. The highly competitive exotic species significantly reduced the ability for native species to establish. CARS completely disappeared from the area, leaving only two native species in 2024 (Figure 4). CARU (*Calamagrostis rubescens*), was rarely found on the study site before the fire, but did have a slight increase in abundance, and PSSP slightly declined.

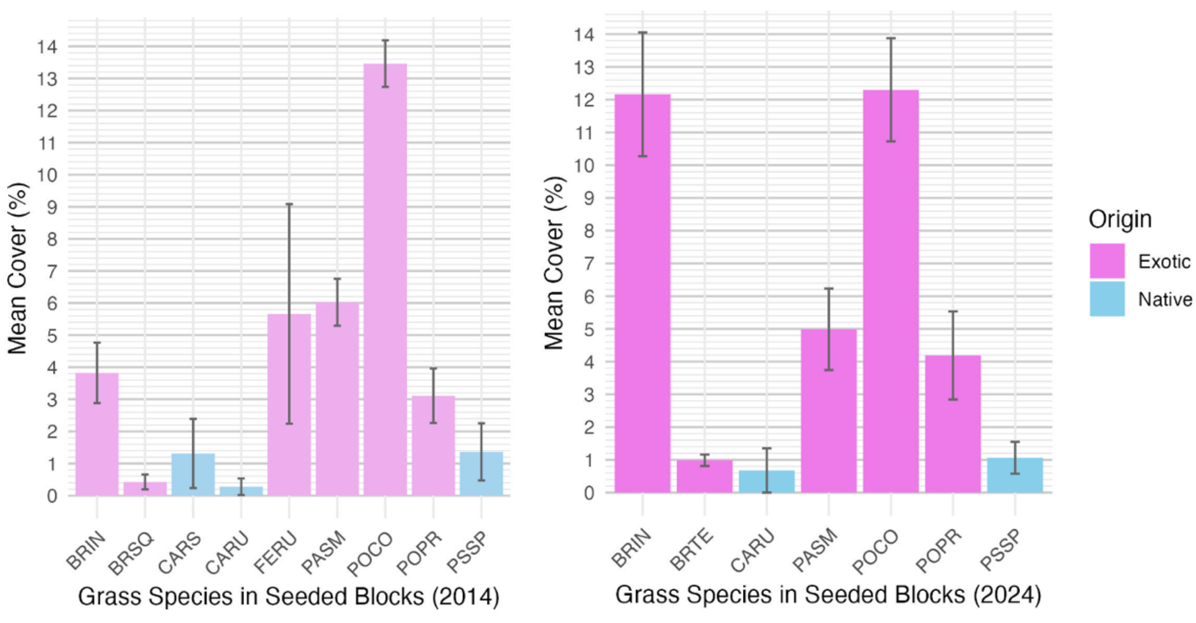

Figure 4. Grass community changes that have occurred to species with more than 1% mean cover in the seeded plots over the last decade. Seeded and invasive species are grouped as exotics.

Between 2014 and 2024, POCO had just over a 1% decline in mean cover, but maintained dominance as the most abundant species - though not by much, due to the exponential growth seen by BRIN (*Bromus inermis*). Despite having the third highest cover in 2014, the seeded species FERU (*Festuca rubra)* has disappeared from the area, which was also seen in the control plots. Similarly, BRTE (*Bromus tectorum*) abundance declined in the unseeded area, as seen in Figure 4; however, the opposite effect is seen in the seeded plots, where had not been previously found in 2014. Furthermore, the seeded species, PASM (*Pascopyrum smithii*) has sustain relatively high cover even with the small decrease. In contrast, mean cover growth is found with the invasive POPR.

Comparison of Forb Community Composition between treatments

Forb Community in Control

Important changes in relative cover and composition occurred in the forb community since 2014, most notably, the shift away from native dominance due to the influx of exotic species (Figure 5). Although native richness increased to ten species, the abundance of native species declined. Additionally, a pronounced change in the native forb community occurred, with [EPAN (*Epilobium angustifolium*), EPBR (*Epilobium brachycarpum*), LATA (*Lactuca tatarica*)] disappearing from the area, while [ARHO (*Arabis holboellii*), CARO (*Campanula rotundifolia*), COPA (*Collinsia parviflora*) MYST (*Myosotis stricta*)] emerging over the last decade. Although the overall abundance of native cover is relatively low, the dominant native species ACMI (*Achillea millefolium*) and ASMI (*Astragalus miser*) have experienced a significant decline. In

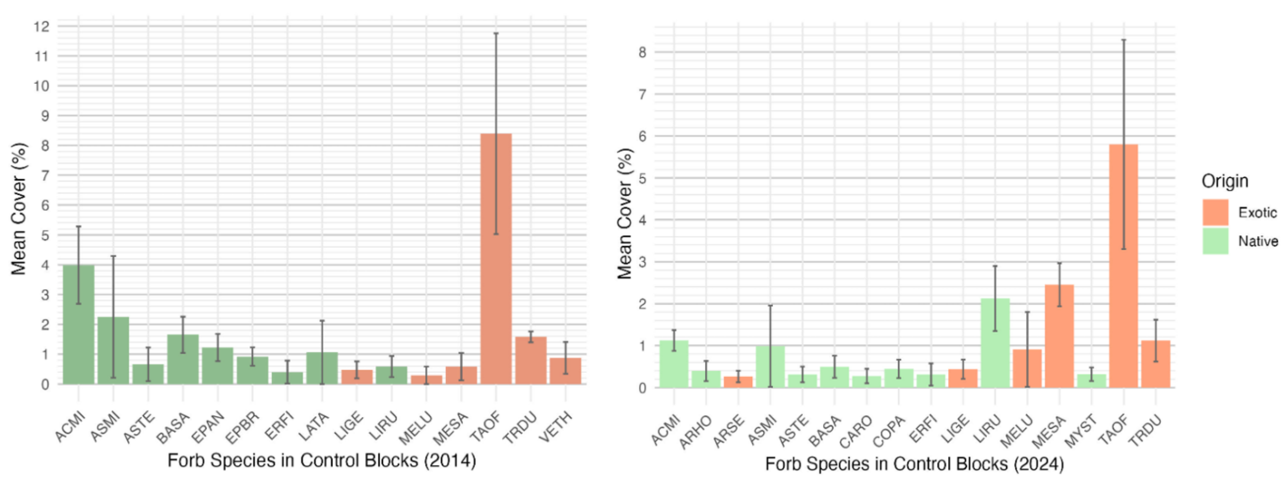

Figure 5. Forb community changes that have occurred to species with more than 1% mean cover in the unseeded plots over the last decade. Seeded and invasive species are groups as exotic.

contrast, LIRU has climbed and now places as the most dominant native forb. Exotic forbs cover increased, with TAOF (*Taraxacum officinale*) maintaining dominance on the control plot over the last decade, despite some decline. The seeded species MESA (*Medicago sativa*) had a significant increase in cover, which was also seen for MELU (*Medicago lupulina*).

Forb Community in Treatment

Over last 10 years, the composition of the forb community remained relatively stable in the seeded treatment, with exotic species dominating the community (Figure 6). The seeded species MESA remains the dominant component of the forb community, even with a slight decrease in cover. Meanwhile, the invasive species TAOF and TRDU, have continued to increase in cover since 2014.

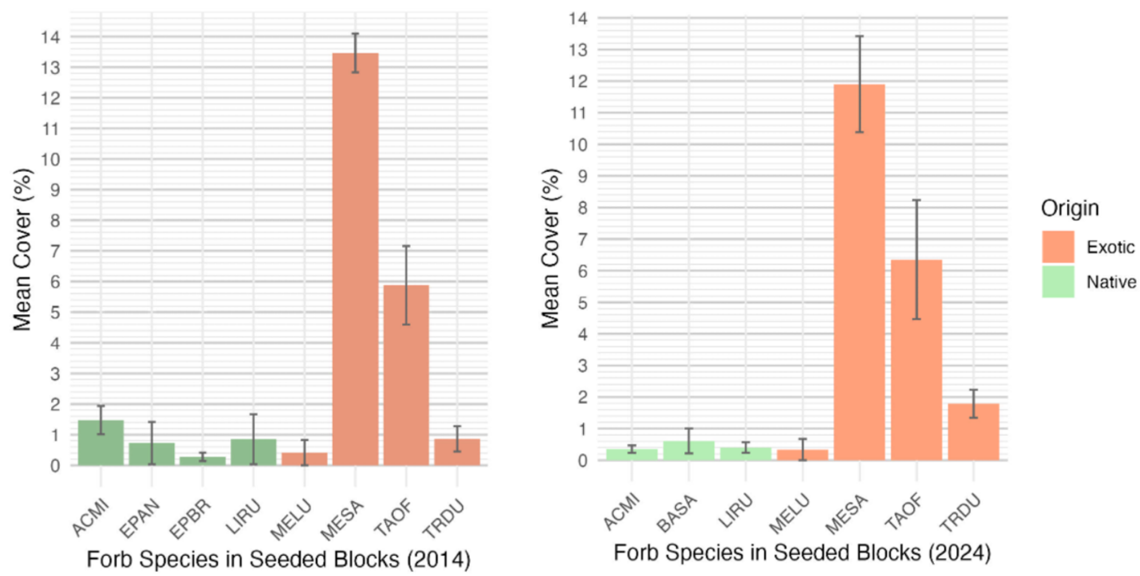

Figure 6. Forb community changes that have occurred to species with more than 1% mean cover in the seeded plots over the last decade. Seeded and exotic species are grouped as exotic.

Additionally, a similar shift in native forb species has occurred, mirroring that of the unseeded plot (Figure 5), with EPAN and EPBR dropping from the community, while BASA has established over time and emerged as the most dominant native forb. Both ACMI and LIRU showed a notable decline over time. As seen with the grass community in the seeded plots (Figure 4), these results suggest that the seeding treatment did not suppress invasive species. Instead, it seems to have suppressed the recovery of native forb species.

Comparison of current dominant plant species between treatments

After 20 years since the seeded species were applied to the study site, POCO and MESA remain dominant in the treatment plots; however, both show a notable presence in the control plots. POCO in particular, is the most dominant species overall. POPR exhibits a slightly lower presence in the in the treatment plot compared to the control, suggesting that the seeded species may have somewhat hindered it growth. However, this effect has not observed with BRIN, which has a significantly greater presence in the treatment plot and remains the second most dominant species (Figure 7). Additionally, TAOF shows a slightly higher presence in the treatment than the control, suggesting that the seeded species provided more favourable conditions for its establishment.

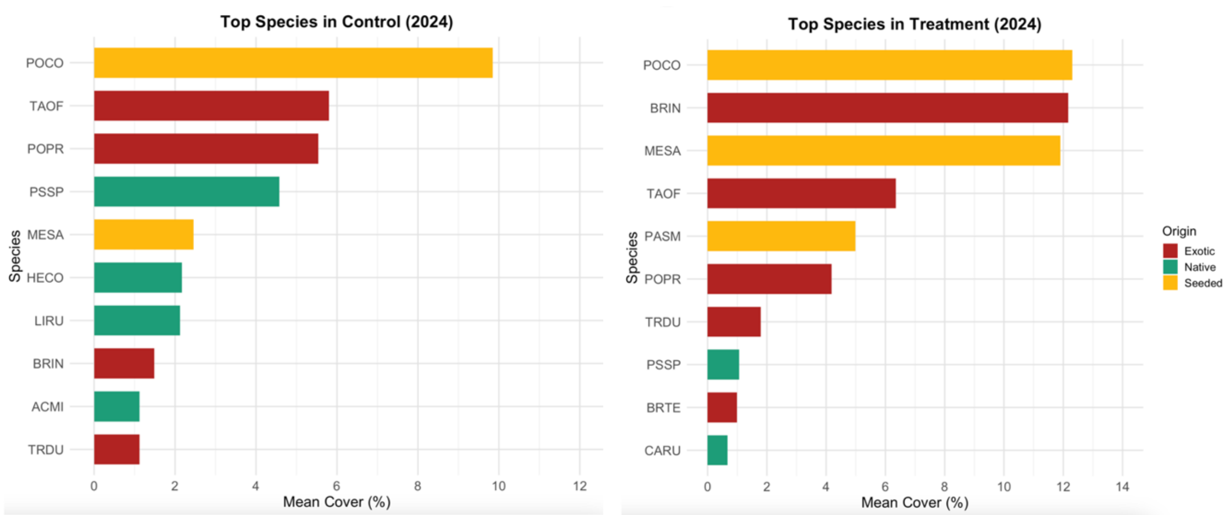

Figure 7. Comparison of the 10 most dominant species with the largest mean cover in the control and treatment areas in 2024.

There is a striking imbalance in species origins between the two treatments. In the unseeded plots, native species account for half of the top species; however, their richness is reduced to only two in the seeded plot, where they also exhibit substantially lower mean cover than in the control. In contrast, invasive species are more abundant in the treatment plots, despite the greater presence of seeded species relative to the control plots.

Discussion

These results show a decline in native species cover, while exotic species increased over the last decade, particularly seeded POCO and invasive PORP. The higher availability of resources in burned areas has the potential to facilitate success of non-native species (Hunter, Omi, Martinson and Chong, 2006; Shive, Sieg and Fule, 2013; Woongsoon, et al. 2021), which underscores the need for strategic post-fire management. Post-fire seeding is a common approach that aims to promote succession towards a preferred ecological condition (Ott, et al. 2022), while preventing the establishment of non-native species (Peppin, et al. 2010). Although, these results suggest the seed application caused more harm on native species recovery, through the established dominance of seeded species in areas they were not intentionally placed, which increased competition across the site.

The effect of seeded species on invasive cover

Since the 2014 plant community assessment, the seeded plots have experienced a substantial increase in invasive species abundance. While seeded species remain dominant, their decline has coincided with the rise of invasives species. Additionally, native species richness is dramatically lower than both invasive and seeded species treatment (Figure 3-6). This reinforces the idea that the seeded species where not effective in reducing invasive species, rather, they primarily contributed to the suppression of native species.

Both *Bromus inermis* and *Poa pratensis* had exponential growth in the last decade, and achieved widespread dominance across the site, with *B. inermis* having greater success in seeded areas (Figure 4), while *P. pratensis* thrived more in the control areas (Figure 3). This raises the question of whether the lower presence of *P. pratensis* in the treatment plot was due to competition from the seeded species, or the high presence of *B. inermis*. These highly competitive, cool-season invaders are often found in co-occurrence (Otfinowski, Kenkel and Catling, 2006) and have been known to dominant over 60% of exotic species cover in some grasslands (Rakhi and DeKeyser, 2022). *B. inermis* is known spread faster where soil disturbance is high, especially in rangelands (Rakhi and DeKeyser, 2022), therefore cattle disturbance could account for its rapid increase in cover. On the other hand, both *B. inermis* and *P. pratensis* have been managed through grazing (Rakhi and DeKeyser, 2022), although their abundance indicates they aren't actively being selected by grazers in the study site. Though they were not detected on site prior to the fire, it is likely these species were unintentionally introduced during the seeding application as contaminants, which is not an uncommon occurrence (Peppin, et al. 2004; Hunter, Omi, Martinson and Chong, 2006).

Another successful invasive that was accounted for prior to the fire, *Taraxacum officinale*, had a high presence in both treatments, though its abundance is greater in the seeded plots. Although the rapid growth and resource utilization of seeded species is believed to help mitigate the establishment and spread of invasive species (Peppin, et al. 2010), the findings of this study do not align with this expectation, where seeded species have not been successful at managing invasive species. Instead, they have been found to increase competition and reduce both species richness and diversity in the treatment area.

Impacts of seeding on native plant recovery

Native species, which already had low cover in 2014, experienced further declines and reduced species richness in the seeded plots. Across both seeded and unseeded plots, native plant cover had a significant decline in mean cover (Table 2). Though native species richness in the control plots increased slightly over time (Figure 3, Figure 5) the seeded plots exhibited a loss of native species by 2024. The results indicate that seeding did not facilitate native plant recovery and may have contributed to its suppression. This suggests that seeding did not promote natural regeneration but rather intensified competition for native plants, particularly given the dominance of agronomic grasses in treatment plots. This finding aligns with the literature review by Peppin et al. (2010), which reported that 62% of studies found lower native species cover on seeded plots compared to unseeded ones. Further supporting this, the higher Shannon's and Simpson's diversity indices indicate that unseeded areas had greater species diversity compared to the seeded plots (Table 3). This suggests that seeded plots were dominated by fewer species, leading to lower species evenness and reduced diversity, which is seen in Figures 4 and 6. In contrast, the decreased diversity of the treatment plots signifies that dominance of fewer species is driving the plant community composition.

The continued increase in invasive species cover in treatment plots, despite the presence of seeded species, further indicated that seeding did not provide a competitive advantage for native plants. Additionally, the decline of previously dominant native species in the control plots, like Carex rossii (Figure 3), highlight the negative effects of seeded species on native vegetation, with their spread into unseeded areas intensified competition. Two native species, *Pseudoroegneria spicata* and *Hesperostipa comata,* both of which were dominant species before the fire (Figure 3, Figure 7), did see a significant growth in the control plots. However, their potential for greater success may be constrained by the dominance of seeded species in the control area, as these species have been shown suppress native plant regeneration (Beyers, 2004; Grant-Hoffman and Dollerschell, 2019).

While adaptive traits such as seedbanks and dispersal from nearby unburnt stands can help facilitate native recovery (Araujo, et al. 2017), high fire severity and intensity can diminish these strategies (Nolan, et al. 2021). This can lead to delayed native species recovery compared invasives species, who often excel on post-fire conditions (Woongsoon, et al. 2021), as observed in this study. Furthermore, severe disturbances that promote invasive species establishment (Woongsoon, et al. 2021), also create intense competition for limited resources (Nolan, et al. 2021). This competition is further exacerbated by the dominance of seeded species in the initial years following the fire, ultimately contributing to reduced native species richness (Peppin, et al. 2010). Natural vegetation recovery and seedbanks can experience long term effects when using aggressive species in postfire seeding (Grant-Hoffman and Dollerschell, 2019). This study provides insight into the extent and persistence of these impacts over time; and underscores the need for land managers to carefully consider the ecological cost associated with seeding decisions (Beyers, 2004).

Study Limitations

One limitation to this study is the continued persistence and expansion of seeded species beyond the treatment plots. Canada bluegrass (*Poa compressa*) and alfalfa (*Medicago sativa*), which were introduced as part of the post-fire seeding treatment, have not only maintained high dominance in the treatment plots but have also successfully established in the unseeded areas, with *P. compressa* in particular, exhibiting the highest cover on the study site since 2014. The unintended spread of these species raises concerns about the long-term effectiveness of seeding an agronomic mix as a restoration strategy, as their presence in the control area appears to have impeded the recovery of native forbs and graminods. Their high presence could be linked to drift during the broadcast application, or through increased spread over time, since the seeded and unseeded plots are directly adjacent to one another. Future research should address using other applications, such as hand seeded or drones, to reduce the risk of wind drift. Additionally, the placement of plots adjacent to one another increases the risk of ground spread. Placing plots further apart could assist in reducing the rate of spread and allow a better understanding of natural vegetation without the unintentional dominance of seeded species. Lastly, cattle access to the study site could unequally impact species through selective grazing.

Conclusion

This study examined the long-term effects of post-fire seeding on plant community composition after a high severity fire 20 years ago, with a focus on changes over the last decade. The results found a shift in community dynamics, which indicate that seeding did not successfully suppress invasive species nor facilitate the recovery of native plant species. Over the last decade, invasive species had a significant increase in both treatments, however, their success was greater within the seeded treatment. In contrast, native species showed further decline in the seeded area, demonstrating that seeding did not assist in their recovery. Instead, seeded agronomic species persisted and, in some cases, expanded beyond the treatment areas, contributing to increased competition with both native and invasive species, and a reduction in native species richness. These findings challenge the assumption that seeding mitigates invasive establishment, illustrating the risk of heighten invasive competition with the use of agronomic species. Additionally, the potential ecological consequences that can occur over time because of seeding, such as invasive establishment, persistence of unintended species (seeded species or contaminate in the seed mix) or increase competition for native species; underscores the need for adaptive management strategies and further research into the use of native seed mixes.

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Appendices

GRASS	COMMON NAME	LATIN
ACNE	Columbia needlegrass	Achnatherum nelsonii
BRIN	smooth brome	Bromus inermis
BRSQ	corn brome	Bromus squarrosus
BRTE	cheatgrass	Bromus tectorum
CARS	Ross' sedge	Carex rossii
CARU	pinegrass	Calamagrostis rubescens
FECA	rough fescue	Festuca campestris
FERU	red fescue	Festuca rubra
HECO	needle-and-thread grass	Hesperostipa comata
JUBA	Baltic rush	Juncus balticus
KOMA	junegrass	Koeleria macrantha
PASM	western wheatgrass	Pascopyrum smithii
POCO	Canada bluegrass	Poa compressa
POPR	Kentucky bluegrass	Poa pratensis
POSE	Sandberg's bluegrass	Poa secunda
PSSP	bluebunch wheatgrass	Pseudoroegneria spicata
VUCO	six-weeks grass	Vulpia octoflora

FORBS	COMMON NAME	Latin
ACMI	Yarrow	Achillea millefolium
ARHO	Holboell's rockcress	Arabis holboellii
ARSE	thyme-leaved sandwort	Arenaria serpyllifolia
ASMI	timber milk-vetch	Astragalus miser
ASTE	pulse milk-vetch	Astragalus tenellus
BASA	arrowleaf balsamroot	Balsamorhiza sagittata
CARO	common harebell	Campanula rotundifolia

COPA	small-flowered blue-eyed Mary	Collinsia parviflora
EPAN	Fireweed	Epilobium angustifolium
ERFI	thread-leaved fleabane	Erigeron filifolius
LATA	blue lettuce	Lactuca tatarica
LIGE	Dalmatian toadflax	Linaria genistifolia
LIRU	lemonweed	Lithospermum ruderale
MELU	black medic	Medicago lupulina
MESA	alfalfa	Medicago sativa
MYST	blue forget-me-not	Myosotis stricta
TAOF	common dandelion	Taraxacum officinale
TROU	yellow salsify	Tragopogon dubius
VETH	great mullein	Verbascum thapsus

Year	Functional	Treat	Species	1	2	3	4
2014	Forbs	Treat	ACMI	1.40	2.70	1.35	0.45
2014	Forbs	Control	ACMI	1.58	7.65	3.25	3.45
2014	Forbs	Control	ANMI	0.00	0.30	0.00	0.00
2014	Forbs	Treat	ANUM	0.00	0.00	0.00	0.05
2014	Forbs	Control	ARCA	0.00	0.65	0.00	0.00
2014	Forbs	Control	ARHO	0.00	0.25	0.15	0.10
2014	Forbs	Treat	ARHO	0.05	0.00	0.05	0.00
2014	Forbs	Control	ARSE	0.00	0.05	0.45	0.45
2014	Forbs	Treat	ARSE	0.00	0.15	0.15	0.00
2014	Forbs	Control	ASCO	0.00	0.00	0.05	0.00
2014	Forbs	Treat	ASMI	0.00	0.00	0.00	0.60
2014	Forbs	Control	ASMI	0.00	0.65	0.00	8.35
2014	Forbs	Control	ASTE	2.33	0.00	0.30	0.00
2014	Forbs	Treat	ASTE	0.60	0.00	0.00	0.00
2014	Forbs	Control	BASA	0.00	2.90	2.05	1.65
2014	Forbs	Treat	BASA	0.30	0.05	0.00	0.00
2014	Forbs	Control	CAMA	0.00	0.00	0.05	0.05
2014	Forbs	Control	CARO	0.00	0.00	0.00	0.05
2014	Forbs	Control	CIAR	0.08	0.00	0.90	0.00
2014	Forbs	Treat	CIUN	0.75	0.00	0.00	0.00
2014	Forbs	Control	CRAT	0.00	0.00	0.05	0.05
2014	Forbs	Treat	CRAT	0.00	0.05	0.00	0.00
2014	Forbs	Control	EPAN	1.58	0.05	1.05	2.20
2014	Forbs	Treat	EPAN	0.05	0.00	0.05	2.80
2014	Forbs	Control	EPBR	0.67	1.75	0.95	0.30
2014	Forbs	Treat	EPBR	0.10	0.60	0.40	0.00
2014	Forbs	Treat	ERCM	0.00	0.00	0.00	0.15
2014	Forbs	Control	ERCM	0.00	0.00	0.00	0.50
2014	Forbs	Control	ERFI	0.00	1.55	0.05	0.00
2014	Forbs	Treat	ERFI	0.00	0.05	0.00	0.00
2014	Forbs	Control	ERSP	0.00	0.05	0.00	0.00
2014	Forbs	Control	FIAR	0.00	0.15	0.05	0.05
2014	Forbs	Treat	FIAR	0.00	0.15	0.00	0.00
2014	Forbs	Control	GAAR	0.00	0.30	0.10	0.10
2014	Forbs	Treat	GAAR	0.05	0.20	0.00	0.00
2014	Forbs	Treat	LATA	0.65	0.00	0.00	0.00
2014	Forbs	Control	LATA	4.25	0.00	0.00	0.00
2014	Forbs	Control	LIGE	0.00	0.65	1.20	0.05
2014	Forbs	Treat	LIGE	0.00	0.05	0.00	0.00
2014	Forbs	Treat	LIRU	3.30	0.00	0.05	0.05
2014	Forbs	Control	LIRU	0.00	1.50	0.80	0.05
2014	Forbs	Treat	LODE	0.00	0.00	0.15	0.00
2014	Forbs	Control	LODE	0.00	0.00	0.30	0.00
2014	Forbs	Control	LOMA	0.00	0.05	0.00	0.00
2014	Forbs	Control	MELU	1.17	0.00	0.00	0.00

2024	Forbs	Treat	ANMI	0.00	0.00	0.00	0.05
2024	Forbs	Control	ANMI	0.08	0.00	0.00	0.00
2024	Forbs	Control	ARCA	0.00	0.40	0.00	0.00
2024	Forbs	Treat	ARFR	0.00	0.05	0.00	0.00
2024	Forbs	Treat	ARHO	0.10	0.20	0.10	0.05
2024	Forbs	Control	ARHO	0.08	1.10	0.30	0.10
2024	Forbs	Control	ARSE	0.00	0.50	0.50	0.05
2024	Forbs	Treat	ARSE	0.00	0.15	0.35	0.00
2024	Forbs	Treat	ASCO	0.00	0.05	0.00	0.00
2024	Forbs	Control	ASCO	0.00	0.10	0.00	0.00
2024	Forbs	Treat	ASMI	0.05	0.00	0.00	0.05
2024	Forbs	Control	ASMI	0.00	0.05	0.00	3.90
2024	Forbs	Control	ASTE	0.50	0.00	0.75	0.00
2024	Forbs	Control	BASA	0.08	0.45	1.25	0.20
2024	Forbs	Treat	BASA	0.80	1.65	0.00	0.00
2024	Forbs	Treat	CAMA	0.00	0.10	0.05	0.15
2024	Forbs	Control	CAMA	0.00	0.00	0.00	0.20
2024	Forbs	Treat	CAMI	0.00	0.05	0.05	0.00
2024	Forbs	Control	CARO	0.00	0.75	0.30	0.05
2024	Forbs	Treat	CARO	0.00	0.00	0.00	0.05
2024	Forbs	Treat	CIAR	0.05	0.00	0.00	0.00
2024	Forbs	Control	COPA	0.08	0.05	0.85	0.80
2024	Forbs	Treat	COPA	0.00	0.05	0.65	0.00
2024	Forbs	Control	CRAT	0.00	0.00	0.00	0.05
2024	Forbs	Treat	CRAT	0.00	0.00	0.00	0.05
2024	Forbs	Treat	DEPI	0.05	0.00	0.00	0.00
2024	Forbs	Control	EPAN	0.00	0.00	0.00	0.15
2024	Forbs	Treat	EPAN	0.10	0.00	0.00	0.00
2024	Forbs	Control	EPBR	0.00	0.00	0.25	0.10
2024	Forbs	Treat	ERCO	0.00	0.00	0.00	0.05
2024	Forbs	Control	ERCO	0.00	0.00	0.00	0.10
2024	Forbs	Treat	ERFI	0.00	0.00	0.00	0.35
2024	Forbs	Control	ERFI	0.00	0.00	0.15	1.10
2024	Forbs	Treat	ERHE	0.00	0.30	0.00	0.00
2024	Forbs	Control	GAAR	0.00	0.05	0.00	0.00
2024	Forbs	Treat	GAAR	0.00	0.05	0.00	0.00
2024	Forbs	Control	LATA	0.67	0.00	0.00	0.05
2024	Forbs	Treat	LATA	0.05	0.00	0.00	0.00
2024	Forbs	Control	LIGE	0.00	0.95	0.70	0.10
2024	Forbs	Treat	LIGE	0.00	0.20	0.00	0.00
2024	Forbs	Treat	LIRU	0.35	0.85	0.05	0.35
2024	Forbs	Control	LIRU	0.00	3.70	2.50	2.30
2024	Forbs	Control	LODE	0.00	0.00	0.00	0.10
2024	Forbs	Treat	LOMA	0.00	0.00	0.00	0.30
2024	Forbs	Control	LOMA	0.00	0.25	0.15	0.00
2024	Forbs	Control	MELU	3.58	0.05	0.00	0.00

2014	Forbs	Treat	MELU	1.65	0.00	0.00	0.00
2014	Forbs	Treat	MESA	13.35	14.05	11.75	14.70
2014	Forbs	Control	MESA	0.08	0.30	1.95	0.00
2014	Forbs	Control	SILO	0.67	0.00	0.00	0.00
2014	Forbs	Treat	TAOF	8.20	2.40	5.55	7.35
2014	Forbs	Control	TAOF	17.42	2.15	4.60	9.40
2014	Forbs	Control	TRDU	1.08	1.80	1.55	1.90
2014	Forbs	Treat	TRDU	0.45	0.75	0.20	2.05
2014	Forbs	Control	VETH	0.00	1.35	0.00	2.15
2014	Forbs	Control	VIAM	0.17	0.00	0.00	0.00
2014	Forbs	Control	ZIVE	0.00	0.00	0.00	0.05
2014	Grass and g	Control	ACNE	0.67	1.75	0.30	1.05
2014	Grass and g	Treat	ACNE	0.05	0.00	0.00	0.00
2014	Grass and g	Control	BRIN	0.50	0.30	0.05	2.15
2014	Grass and g	Treat	BRIN	6.40	2.00	3.00	3.90
2014	Grass and g	Control	BRSQ	0.42	0.25	1.30	1.40
2014	Grass and g	Treat	BRSQ	0.05	0.90	0.75	0.00
2014	Grass and g	Control	BRTE	2.33	1.95	1.85	1.15
2014	Grass and g	Treat	BRTE	0.00	0.45	0.50	0.00
2014	Grass and g	Treat	CARS	0.00	4.50	0.75	0.00
2014	Grass and g	Control	CARS	0.08	3.40	17.10	0.00
2014	Grass and g	Treat	CARU	0.05	0.00	0.00	1.05
2014	Grass and g	Control	CARU	0.67	0.00	0.00	0.00
2014	Grass and g	Control	FECA	0.00	0.75	0.00	0.30
2014	Grass and g	Treat	FECA	0.00	0.00	0.00	0.30
2014	Grass and g	Control	FERU	0.00	0.30	0.00	1.20
2014	Grass and g	Treat	FERU	6.40	0.05	1.15	15.05
2014	Grass and g	Control	HECO	0.00	2.65	0.00	0.00
2014	Grass and g	Treat	HECO	0.15	0.00	0.00	0.00
2014	Grass and g	Control	KOMA	0.58	0.00	1.05	0.00
2014	Grass and g	Control	PASM	0.00	0.00	1.00	2.05
2014	Grass and g	Treat	PASM	5.20	5.55	8.20	5.15
2014	Grass and g	Control	POCO	10.50	2.65	2.70	6.50
2014	Grass and g	Treat	POCO	14.80	13.90	13.75	11.40
2014	Grass and g	Control	POPR	4.25	0.10	1.70	1.30
2014	Grass and g	Treat	POPR	2.55	1.00	4.05	4.85
2014	Grass and g	Control	POSE	0.00	0.10	0.10	0.30
2014	Grass and g	Control	PSSP	0.08	11.65	2.40	0.30
2014	Grass and g	Treat	PSSP	0.00	3.75	0.00	1.70
2014	Grass and g	Control	STRI	0.00	0.00	0.05	0.00
2014	Grass and g	Control	VUOC	0.08	0.25	0.00	0.60
2014	Grass and g	Treat	VUOC	0.00	0.00	0.05	0.00
2024	Forbs	Treat	ACMI	0.20	0.70	0.30	0.20
2024	Forbs	Control	ACMI	1.50	1.05	1.50	0.45
2024	Forbs	Control	ALCE	0.00	0.00	0.05	0.05
2024	Forbs	Treat	ALCE	0.30	0.05	0.00	0.00

2024	Forbs	Treat	MELU	1.35	0.00	0.00	0.00
2024	Forbs	Control	MESA	1.92	3.55	3.05	1.30
2024	Forbs	Treat	MESA	9.10	10.55	16.15	11.80
2024	Forbs	Treat	MIGR	0.00	0.05	0.00	0.00
2024	Forbs	Treat	MYST	0.00	0.30	0.20	0.05
2024	Forbs	Control	MYST	0.17	0.15	0.80	0.15
2024	Forbs	Control	PEPR	0.00	0.00	0.05	0.00
2024	Forbs	Control	SILO	0.08	0.00	0.00	0.00
2024	Forbs	Control	SOSP	0.00	0.00	0.35	0.00
2024	Forbs	Control	TAOF	13.25	3.85	3.35	2.75
2024	Forbs	Treat	TAOF	10.50	2.80	3.55	8.55
2024	Forbs	Control	TRDU	2.58	0.55	0.40	0.95
2024	Forbs	Treat	TRDU	3.05	1.70	1.05	1.35
2024	Forbs	Control	VETH	0.00	0.15	0.10	0.05
2024	Forbs	Treat	VETH	0.75	0.00	0.00	0.00
2024	Forbs	Control	VIAM	0.17	0.00	0.00	0.00
2024	Forbs	Treat	ZIVE	0.00	0.00	0.00	0.05
2024	Forbs	Control	ZIVE	0.00	0.00	0.05	0.50
2024	Grass and g	Control	ACNE	0.00	0.00	0.05	1.05
2024	Grass and g	Control	ACRI	0.00	0.00	0.30	0.00
2024	Grass and g	Control	BRIN	0.00	1.50	0.80	3.65
2024	Grass and g	Treat	BRIN	12.75	7.25	12.20	16.45
2024	Grass and g	Control	BRTE	0.50	1.05	0.90	0.30
2024	Grass and g	Treat	BRTE	0.55	0.95	1.05	1.40
2024	Grass and g	Treat	CARS	0.00	0.00	0.00	0.05
2024	Grass and g	Control	CARS	0.17	0.00	2.35	0.00
2024	Grass and g	Control	CARU	0.00	0.00	0.00	0.05
2024	Grass and g	Treat	CARU	0.00	0.00	0.00	2.70
2024	Grass and g	Control	ELTR	0.75	0.00	0.00	0.00
2024	Grass and g	Treat	FECA	0.00	0.00	0.00	0.30
2024	Grass and g	Control	FECA	0.00	1.20	0.00	0.75
2024	Grass and g	Treat	FERU	0.70	0.00	0.00	0.10
2024	Grass and g	Control	FERU	0.00	0.10	0.05	0.00
2024	Grass and g	Control	HECO	1.08	6.85	0.75	0.00
2024	Grass and g	Treat	HECO	0.05	0.00	0.00	0.00
2024	Grass and g	Control	JUBA	2.50	0.00	0.00	0.00
2024	Grass and g	Control	KOMA	0.17	0.05	0.65	1.75
2024	Grass and g	Control	PASM	0.00	1.60	1.30	0.75
2024	Grass and g	Treat	PASM	6.75	6.95	4.65	1.60
2024	Grass and g	Treat	POCO	9.30	16.70	12.05	11.15
2024	Grass and g	Control	POCO	15.75	7.60	3.65	12.40
2024	Grass and g	Control	POPR	14.17	0.80	2.30	4.90
2024	Grass and g	Treat	POPR	3.85	2.90	1.95	8.05
2024	Grass and g	Control	POSE	0.00	0.25	0.40	0.80
2024	Grass and g	Treat	PSSP	1.10	2.35	0.00	0.80
2024	Grass and g	Control	PSSP	0.00	9.85	6.95	1.50

2024	Grass and g	Control	VUOC	0.00	0.30	1.15	0.50
2024	Grass and g	Treat	VUOC	0.15	0.10	0.00	0.00