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ARTICLE

Restoration of slash pile burn scars to prevent establishment and propagation of non-native plants

L. DeSandoli, R. Turkington, and L.H. Fraser

Abstract: Logging and burning of the resultant woody debris is a management tool to reduce fire risk. Burning of the debris as piles affects the underlying soil biota and soil physical and (or) chemical properties. The resulting disturbance created by the burns may create opportunities for the establishment and spread of non-native plant species. Here, we test three restoration treatments on recent, approximately 1-year-old, pile burn scars, including an arbuscular mycorrhizal fungal (AMF) inoculant (present or absent), a ground cover (straw or no straw added), and different seeding types (native seed mix, agronomic seed mix, and no seed). The most effective treatment in reducing undesired non-native species cover was the seeding of agronomic species; here "native" and "non-native" groups exclude sown agronomic species. Undesired non-native cover was 15.1% in plots with no seed, 9.1% in plots with native seed added, and 3.5% in plots with agronomic seed added. Total vegetation cover, mostly through the increase of agronomic species, was increased by seeding and by the application of straw cover. Commercial AMF inoculum was an ineffective treatment, suggesting that a better understanding of host specificity is warranted. Restoration efforts should be directed at burn scar sites after burning to ameliorate the effects of invasive species colonization, and the use of agronomic species may prevent non-native invasive plants from establishing.

Key words: arbuscular mycorrhizal fungi, invasive plants, grassland restoration, pile burn scars, ponderosa pine.

Résumé : La coupe suivie du brûlage des débris ligneux est un outil d'aménagement qui vise à réduire les risques d'incendie. Le brûlage des débris empilés a des impacts sur les biotes du sol sous-jacent ainsi que sur les propriétés physiques et chimiques du sol. La perturbation qui s'ensuit, engendrée par le brûlage, peut favoriser l'établissement et la propagation d'espèces végétales exotiques. Nous avons testé trois traitements de restauration de sites portant les traces laissées récemment, environ un an, par le brûlage d'empilements incluant l'inoculum d'un champignon mycorhizien à arbuscules (CMA) (présent ou absent), un couvre-sol (avec ou sans paille) et l'ensemencement (mélange de semences de plantes indigènes, mélange de semences de plantes agricoles, pas d'ensemencement). Le traitement le plus efficace pour réduire le couvert d'espèces exotiques indésirables était l'ensemencement d'espèces agricoles : ici, les groupes « indigène » et « exotique » excluent les espèces agricoles ensemencées. Le couvert d'espèces exotiques indésirables atteignait 15,1 % dans les parcelles non ensemencées, 9,1 % avec l'ensemencement de semences de plantes indigènes et 3,5 % avec l'ensemencement de semences de plantes agricoles. Le couvert végétal total, surtout dû à l'augmentation des espèces agricoles, était accru par l'ensemencement et l'application du couvre-sol. L'inoculum commercial du CMA n'était par un traitement efficace; ce qui indique qu'une meilleure compréhension de la spécificité de l'hôte est nécessaire. Les efforts de restauration devraient avant tout être dirigés vers les endroits où le brûlage a laissé des traces pour améliorer les effets de la colonisation des espèces invasives, et l'utilisation d'espèces agricoles pourrait prévenir l'établissement d'espèces exotiques invasives. [Traduit par la Rédaction]

Mots-clés : champignon mycorhizien à arbuscules, plantes invasives, restauration des prairies, traces laissées par le brûlage d'empilements, pin ponderosa.

Introduction

Management to reduce the threat of interface fires occurring at the boundaries of urban areas involves the harvesting of brush and trees. Unmarketable harvested woody debris are piled on site and burned. Pile burns cause soil properties to be greatly altered, including increases in soil nutrient availability and hydrophobicity, decreases in soil organic carbon and biotic communities, and a general blackening (Rhoades et al. 2004; Certini 2005; Esquilin et al. 2007; Busse et al. 2013) and reddening (Rhoades and Fornwalt 2015) of the soil. Another undesired consequence of these burns may be the facilitation and spread of non-native species (Korb et al. 2004; Hunter et al. 2006; Keeley 2009; Hebel et al. 2009). Non-native species are of worldwide ecological concern, causing widespread changes to biotic, abiotic, and economic components of ecosystems. For example, non-native plant species can alter soil nutrient cycles (Ehrenfeld 2003), fire cycles (Billings 1994), biodiversity (Fraser and Carlyle 2011), and belowground biotic communities (Richardson et al. 2000). Non-native species can be early colonizers of postburn environments in many ecosystems (e.g., Hebel et al. 2009; Fornwalt et al. 2010). Of particularly concern is if these invaded pile burn scars become the sources of non-native species propagules and affect the surrounding, less disturbed vegetation.

A common restoration technique of pile burn scars aimed at reducing the establishment and spread of non-native species is to promote the growth of non-invasive species (Korb et al. 2004; Hunter et al. 2006; Peppin et al. 2010). Burn scars are generally plant depleted (Korb et al. 2004; Halpern et al. 2014) — little to no seed

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bank or plant remnants remain in the soil to regenerate. In these plant-depleted environments, non-native species may establish more readily because they can have highly efficient seed-dispersal mechanisms (Grime 1977; Davis et al. 2000). Many plant community restoration efforts involve the seeding of domesticated agronomic grasses (Eiswerth et al. 2009). As agronomic species can be of native and non-native origin, we refer to three separate groups throughout, namely "native", "non-native", and "agronomic". Agronomic grasses can establish quickly, provide cover and stability for soils, and prevent the invasion of unwanted non-native species. However, the use of these species in restoration is controversial because agronomic species can establish self-sustaining monocultures (e.g., *Agropyron cristatum* (Bieb.) Tzvelev), creating biologically impoverished areas that may result in the decline of endangered or rare species (Redente et al. 1989) and could facilitate the establishment of non-native species into surrounding areas if they themselves are non-native.

Next to seed limitation, soil mycorrhizal fungi are an important factor that can limit native plant growth on highly disturbed soils (Allen 1989; Korb et al. 2004). Pile burning depletes soil micro-organisms, including arbuscular mycorrhizal fungi (AMF) (Korb et al. 2004; Esquilin et al. 2007). In nutrient-limiting environments, plants may benefit from their association with AMF through improved acquisition of nutrients and water. In exchange, AMF receive carbon from the plants (Malloch et al. 1980). In addition, AMF associations may provide plants with protection from drought, soil-borne pathogens and disease, heavy metal toxicity, and high salinity (Singh et al. 2011). AMF abundance has been shown to promote vegetation diversity (van der Heijden et al. 2008). Thus, the re-establishment of a viable AMF community along with seed treatments may promote vegetation establishment on slash and (or) pile burn scars.

In warm, dry climates, restoration efforts may be hampered by the lack of soil cover, typically provided by standing vegetation and litter. At mid-day during summer months, dark, exposed soils may become extremely hot, providing an inhospitable environment for seed germination and seedling growth (Farrell et al. 2011). In warm climates, soil cover will decrease soil moisture loss (Shay et al. 2001; Rhoades et al. 2015), reduce soil temperatures (Hogg and Lieffers 1991; Rhoades et al. 2015), and protect seedlings from the desiccating effects of wind (Facelli and Pickett 1991). Soil cover may also prevent soil nutrient loss caused by wind and water erosion (Li et al. 2007). The tempering effect of soil cover is magnified when the litter and vegetation layers, as well as the forest cover, are removed. Invasive species may preferentially establish on these exposed sites because they are often more tolerant of difficult growing conditions (Davis et al. 2000). The re-establishment of cover would remove this tolerance advantage of undesired non-native species.

In this investigation, we tested the following hypotheses:

(*i*) Early establishment of native and agronomic species through seed addition will reduce non-native species establishment and growth;

(*ii*) commercially available AMF inoculant will improve native species growth compared with the growth of non-native and agronomic species;

(*iii*) the presence of soil cover (e.g., as straw) will preferentially improve native species growth compared with the growth of non-native and agronomic species.

Methods

The study was conducted in three open parkland sites near Kamloops, British Columbia: Noble Creek, 50°49′57″N–120°18′56″W, 430 m above sea level (a.s.l.); Heffley Creek, 50°52′6.50″N–120°12′44.70″W, 650 m a.s.l.; and Barnhartvale, 50°38′42.29″N–120°9′38.86″W, 595 m a.s.l. These sites were located at the northern end of the Great Basin ecosystem. All three sites are in the Very Dry Hot Ponderosa Pine (PPxh) biogeoclimatic (BEC) subzone (Meidinger and Pojar 1991). This region has a semi-arid, continental climate with warm, dry summers and cool, dry winters. Mean annual precipitation ranges between 250 and 400 mm, and the mean annual temperature ranges between 5.4 and 9.0 °C (Meidinger and Pojar 1991). Soils of this BEC subzone are typically Chernozemic or Brunisolic. Ecosystems in the PPxh subzone consist of ponderosa pine (*Pinus ponderosa* Douglas ex P. Lawson & C. Lawson), bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) A. Love), rough fescue (*Festuca campestris* Rydb.), and big sagebrush (*Artemisia tridentata* Nutt.). Very dry sites within this subzone often have an association of arrowleaf balsamroot (*Balsamorhiza sagittata* (Pursh) Nutt.) and prickly pear cactus (*Opuntia fragilis* (Nutt.) Haw.) (Meidinger and Pojar 1991). Common non-native species in this BEC zone are cheatgrass (*Bromus tectorum* L.), knapweed species (*Centaurea* spp.), Dalmatian toadflax (*Linaria genistifolia* spp. *dalmatica* (L.) Marie & Petitm.), sulphur cinquefoil (*Potentilla recta* L.), common houndstongue (*Cynoglossum officinale* L.), leafy spurge (*Euphorbia esula* L.), and Kentucky bluegrass (*Poa pratensis* L.).

In November and December 2007, heavy machinery was used to log and assemble woody debris piles at the three sites. The reduction in overstory was relatively minor. The piles were burned in January 2008. Most of the burn scars were a result of piles approximately 2 m in height and 3 m in width. However, there were one to three burn scars at each site that were a result of piles approximately 3 m in height and 5 m in width.

The treatments (factors) investigated in this experiment were: two AMF fungi levels — commercial inoculum added, no commercial inoculum added; two straw levels — present, absent; and three seed addition levels — native seed mix, agronomic seed mix, and no seed added. Treatments were imposed in May 2008, 5 months after burning occurred. At each of the three sites, a fully factorial stratified randomized experiment was established on red soil. Each site had four replications of each treatment combination, for a total of 48 plots·site^{-1} and 144 plots in total. Each plot was 1 m^2, and plots were separated by at least 50 cm. Treatment plots were randomized, except that treatments were distributed so as not to be clustered or duplicated within a single burn scar. We selected 9 burns from the Noble Creek site, 12 burns from the Heffley Creek site, and 6 burns from the Barnhartvale site. Plots per scar ranged from 3 to 14.

Both the native and agronomic species mixes consisted of 1200 seeds·plot^{-1}. The native seed mix was comprised of bluebunch wheatgrass, rough fescue, junegrass (*Koeleria macrantha* (Lebed.) Schult.), stiff needlegrass (*Achnatherum occidentale* (Thurb.) Barkworth), spreading needlegrass (*Achnatherum richardsonii* (Link) Barkworth), foxtail barley (*Hordeum jubatum* L.), yarrow (*Achillea millefolium* L.), brown-eyed susan (*Gaillardia aristata* Pursh), and old man's whiskers (*Geum triflorum* Pursh). These species were chosen because they produce high vegetation cover and represent dominant species in the areas surrounding the burn scars. We were unable to collect enough viable seed of bluebunch wheatgrass, rough fescue, and junegrass in the field, and these species were purchased at Quality Seeds West (Surrey, British Columbia). Seeds of all other species were collected by hand near Kamloops during the summer of 2007 and stored indoors in the laboratory in dry paper bags until needed. The agronomic species mix was also purchased from Quality Seeds West and was composed of creeping red fescue (*Festuca rubra* var. *rubra* L.) native to Eurasia, hard fescue (*Festuca longifolia* auct. non Thuill) native to Europe, slender wheatgrass (*Elymus trachycaulus* (Link) Gould ex Shinners) native to North America, tall wheatgrass (*Agropyron elongatum* (Host). Beauv) native to Eurasia, and annual ryegrass (*Lolium multiflorum* Lam.) native to Europe. These species were chosen because they all grow well on disturbed soils and, typically, do not invade native plant communities (Newman 2007). All species underwent germination tests, and percent germination ranged from 60% to 95%. Before being sown in the field treatment plots, native seeds were stratified by mixing them with 500 mL of wet sand and storing them at 4 °C for 10 days.

The AMF inoculum used was purchased from BioOrganics™ (Santa Maria, California) and consisted of a mix of *Glomus aggregatum*, *Glomus clarum*, *Glomus deserticola*, *Glomus intraradices*, *Glomus monosporus*, *Glomus mosseae*, *Gigaspora margarita*, and *Paraglomus brasilianum* at a minimum of 50 spores·cm^{-3}. Each treatment consisted of 2 mL (~100 spores) of the inoculum or approximately twice the rate of recommended application. Although the efficacy of commercially available inoculum can be low, a similar product from BioOrganics™ was shown to be viable in a greenhouse study (Cavender and Knee 2006).

The wheat straw cover treatment was applied evenly to each plot so that it covered about 75% of the ground area to a depth of approximately 0.5 cm. The straw was held in place using 5 cm mesh wire, which was cut into 1 m² sections and stapled to the ground outside of the plot corners. Any seedlings arising from seeds mixed in with the straw were removed immediately (<5 such seedlings were identified during the course of the study).

The AMF and seed addition treatments were mixed with the aforementioned 500 mL of slightly moistened sand. The mixture was spread evenly on plots and gently worked into the top surface of the soil in May 2008. Control plots also had 500 mL of sand applied, and this likewise was gently worked into the top surface of the soil. No seed germination was observed in 2008, most likely due to unseasonably hot and dry weather.

In July 2009, response variables were measured. Plant cover was estimated by species in the centre 50 cm × 50 cm of each plot to minimize edge effects. A 1 m × 1 m square grid was placed over the plot, with strings stretched across the grid every 10 cm in both directions. Plant cover was estimated to the closest 10% when cover was over 25% and 0–1%, >1%–5%, >5%–10%, >10%–15%, >15%–20%, and >20%–25% otherwise. Multiple canopy layers were considered, and therefore, percent cover can potentially exceed 100%. Species nomenclature and origins followed Klinkenberg (2012) and Parish et al. (1999). Identification to genus was done when identification to species was not possible.

Soil nutrient availability was measured in the Noble Creek site using Plant Root Simulator PRS™ probes (Western Ag Innovations, Saskatoon, Saskatchewan) during the 2009 growing season. Four sets of probes were placed each in the red-burned and unburned soils. These probes were used in pairs: one probe measured plant-available cations ($NH4^+$-N, K^+, Mg^{2+}, Ca^{2+}, Cu^{2+}, Mn^{2+}, Zn^{2+}, Fe^{3+}, and Al^{3+}) and the other measured plant-available anions ($NO3^-$-N, Cl^-, $SO4^-$-S, $H_2PO_4^-$-P, and $B(OH)_4^-$-B). Pb^{2+}, Cu^{2+}, and Cd^{2+} were also measured but excluded from the analysis because the majority of measurements were below the detection limits of 0.2 mg·(10 cm²)$^{-1}$.

For analyses, species were grouped into native, non-native (excluding agronomic), and agronomic species. Total species richness, non-native (excluding agronomic) species richness, and Shannon–Weiner diversity indices (H') were also calculated to compare community indices. Data were analyzed using general linear models (GLMs), with site and burn nested within site as blocking variables and seed, straw, and AMF as factor variables. A Bonferroni correction was applied post-hoc for differences among treatments if there were more than two treatments. Paired t tests were performed on soil nutrient data within years. If necessary, data were log(n+1) transformed and tested for normality and equivalence of variance. Significance level was set at α = 0.05. All statistics were done using Systat 13.0 (Systat 2009).

Results

Red-burned soil had higher plant-available nutrients than the surrounding unburned soil two growing seasons after burning (Table 1). Ammonium was approximately three times higher in the red-burned soil than in the unburned soil. Nitrate was almost 19 times higher in red-burned soil than in unburned soil.

Table 1. Mean (±1 standard error) nutrient supply values for the Noble Creek site in 2009.

Nutrient	Unburned soil (mg·(10 cm²)$^{-1}$)	Red-burned soil (mg·(10 cm²)$^{-1}$)	Multiplication factor
$NH4^+$-N	1.67 (0.24)	4.97 (1.12)	3.0
$NO3^-$-N	18.53 (3.22)	348.80 (61.10)	18.8
$B(OH)_4^-$-B	1.29 (0.10)	1.88 (0.14)	0.7
Mg^{2+}	327.37 (24.08)	580.29 (40.03)	1.8
Ca^{2+}	1389.97 (62.80)	1790.91 (129.64)	1.3
K^+	224.91 (16.62)	494.94 (50.79)	2.2
$H_2PO_4^-$-P	8.40 (1.28)	25.74 (2.81)	3.1
Fe^{3+}	5.34 (1.13)	19.13 (3.20)	3.6
Cu^{2+}	0.29 (0.09)	1.16 (0.15)	4.0
Mn^2	4.50 (0.83)	39.57 (4.63)	8.8
SO_4^--S	37.58 (26.55)	296.70 (46.15)	7.9
Al^{3+}	32.60 (0.81)	40.63 (1.77)	1.2

Note: All values are significantly different between treatments, p < 0.05. Multiplication factor indicates the difference (ratio) in nutrient supply values (red-burned soil:unburned soil).

Seed treatments significantly affected all of the vegetation response variables (total, non-native, native, and agronomic vegetation covers; Table 2), where total vegetation cover was greatest for the agronomic seed addition, less for the native seed addition, and least for the control, no seed added treatment (Fig. 1). The proportion of agronomic vegetation was highest when agronomic seed was added, the proportion of native vegetation was highest when native seed was added, and the proportion of non-native vegetation was highest when no seed was added (Fig. 1).

The total and agronomic vegetation covers were increased by the straw treatment when straw was added (Table 2; Fig. 2). There was no main effect of the straw treatment on non-native and native vegetation covers (Table 2).

The only interacting effects detected by the GLM was between the straw and the AMF treatments for non-native and native vegetation covers (Table 2). For native vegetation cover, plots receiving AMF but no straw had less cover than plots receiving AMF with straw and all plots without the addition of AMF (Fig. 3). For non-native vegetation cover, plots receiving AMF plus straw had greater cover than the other treatment combinations (Fig. 3).

The seed and straw treatments significantly affected total species richness and non-native species richness (Table 3), where total species richness was higher in plots with native and agronomic seed additions compared with the no-seed control (Fig. 3*a*). Non-native species richness, however, was lower in the agronomic seed addition treatment compared with the native seed addition and no seed added treatments (Fig. 3*a*). When straw was added, species richness of the total vegetation was higher (Fig. 3*b*). There was a similar trend (p < 0.10) with the straw treatment for the species richness of non-native vegetation (Fig. 3*b*).

The Shannon–Weiner diversity (H') was affected by seed and straw treatments (Table 3; Fig. 4), where H' followed the same pattern as total species richness, being higher when native and agronomic seed was added compared with the no-seed control (Fig. 5*a*) and being higher when straw was added than when it was not (Fig. 5*b*).

Forty-four species were identified in the unburned areas, and 36 species, including those which were seeded, were identified in treatment plots (Appendix A, Table A1). Twenty-two species were common between unburned areas and treatment plots. Many of these species were non-native, including cheatgrass, Japan brome (*B. japonicus* Thunb.), and corn brome (*B. squarrosus* L.). Although Japan brome was relatively uncommon in the study area (2.5% of unburned plots), both corn brome and cheatgrass were widespread (46.3% and 35.0% of unburned plots, respectively, compared with 7.2% and 11.6% of all treatment plots, respectively).

Table 2. Summary of the nested generalized linear model (GLM) for total vegetation cover, non-native vegetation cover, native vegetation cover, and agronomic vegetation cover.

		Total vegetation cover		Non-native vegetation cover		Native vegetation cover		Agronomic vegetation cover	
	Df.	*F* value	Pr(>*F*)	*F* value	Pr(>*F*)	*F* value	Pr(>*F*)	*F* value	Pr(>*F*)
Site	2	1.613	0.205	*2.983*	*0.055*	0.494	0.612	0.409	0.665
Pile burn (site)	27	1.084	0.375	**1.681**	**0.035**	0.729	0.824	0.908	0.600
Cover	1	**6.973**	**0.010**	1.532	0.219	0.844	0.361	**4.366**	**0.039**
Seed	2	**15.134**	**<0.001**	**3.093**	**0.050**	**107.570**	**<0.001**	**127.382**	**<0.001**
AMF	1	1.443	0.356	0.052	0.820	1.483	0.226	0.400	0.529
Cover × seed	2	1.374	0.258	2.250	0.111	0.000	1.000	1.088	0.341
Cover × AMF	1	2.495	0.117	**7.190**	**0.009**	*3.758*	*0.055*	0.432	0.513
Seed × AMF	2	0.871	0.422	0.221	0.802	1.454	0.239	0.135	0.874
Cover × AMF × seed	2	1.368	0.260	0.848	0.432	0.587	0.558	0.825	0.441

Note: Native and non-native groups exclude sown agronomic species. Data were blocked by site (site) and burn nested by site (pile burn (site)). Bold numbers indicate $p < 0.05$; italicized numbers indicate $p < 0.10$. Residuals = 97 degrees of freedom. Df., degrees of freedom; AMF, arbuscular mycorrhizal fungi.

Fig. 1. The effect of seed treatment on mean percent total vegetation cover. Bars with the same letters indicate no significant difference among treatments for total vegetation cover ($p < 0.05$, Bonferroni posthoc test). *Agronomic vegetation cover is higher for agronomic seed added (Agronomic+) than the no seed (Control) and native seed added (Native+) ($p < 0.05$, Bonferroni posthoc test). **Native vegetation cover is higher for the Native+ than the Control and Agronomic+ seed treatment ($p < 0.05$, Bonferroni posthoc test). ***Non-native vegetation cover is significantly different between Control and Agronomic+ seed treatment ($p < 0.05$, Tukey's posthoc test). Native and non-native groups exclude sown agronomic species.

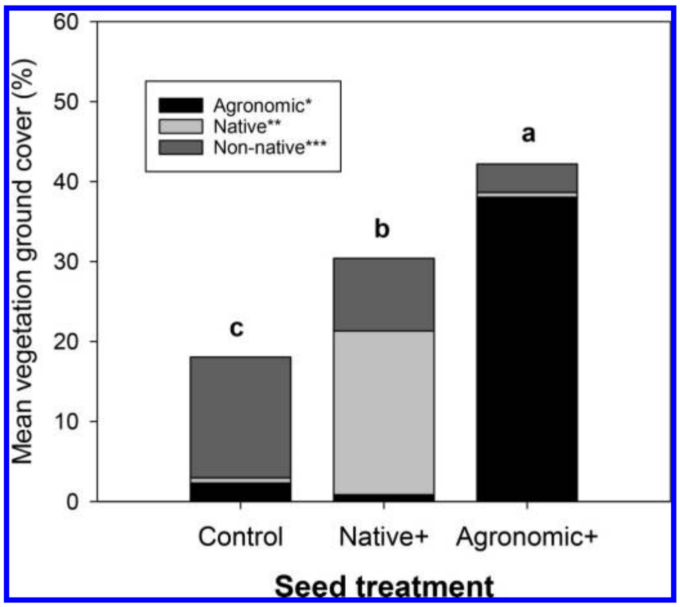
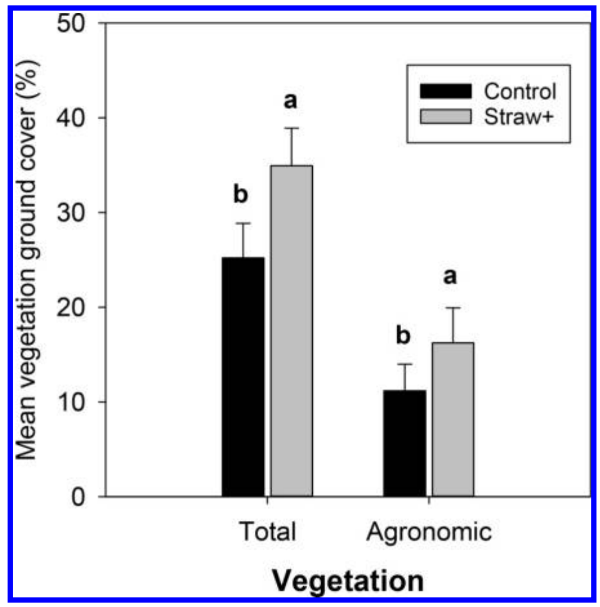


Fig. 2. The effect of straw treatment (Control and Straw+) on total and agronomic vegetation covers. Error bars represent ± 1 standard error. Bars with the same letters indicate no significant difference among treatments for total vegetation cover ($p < 0.05$, Bonferroni posthoc test).



Discussion

Pile burning created soil with elevated nutrient levels and areas initially devoid of vegetation. Both factors have implications for the restoration of vegetation communities and the susceptibility of these communities to colonization by non-native species. Immediate increases to soil nutrients are commonly observed after burning (Certini 2005), caused largely by the combustion of organic matter. Although soil nutrients may eventually reach or fall below pre-burn levels due to nutrient loss primarily through leaching and erosion (Creech et al. 2012), this temporary nutrient pulse may make systems more susceptible to invasion (Davis et al. 2000). However, nutrients are increasingly volatized with increasing burn severity, and red-burned soils may not have sufficient nutrients to make them susceptible to invasion.

Red-burned soil had higher plant-available nutrients than the surrounding unburned soil after burning. In particular, nitrate was almost 19 times higher in red-burned soil than in unburned soil. Soil nitrate typically increases after burning but can lag for weeks or months as nitrifying bacteria respond to the increase in ammonium and soil pH (Certini 2005). The relative difference in soil potassium was also high between burned and unburned soils, possibly resulting from potassium becoming more available as sodium leaches from the soil.

The immediate increases of soil nutrients and the resultant susceptibility to invasion indicate that restoration efforts are warranted. Our treatments had mixed success in reducing the colonization of burn scars by unwanted non-native species. Our findings supported the hypothesis that the early establishment of agronomic species, but not the early establishment of native species, limited non-native species establishment and vegetative cover, although there was no significant difference between native and agronomic vegetation covers. It should be noted that agronomic seed had higher germination rates than native seed, and it is also possible that 10 days of cold stratification may not have been

Fig. 3. The interaction effect of straw (Control and Straw+) and arbuscular mycorrhizal fungi (Control and AMF+) treatments on (*a*) native and (*b*) non-native vegetation covers. Native and non-native groups exclude sown agronomic species. Error bars represent ± 1 standard error. Bars sharing the same letter are not significantly different (p < 0.05, Bonferroni posthoc test).

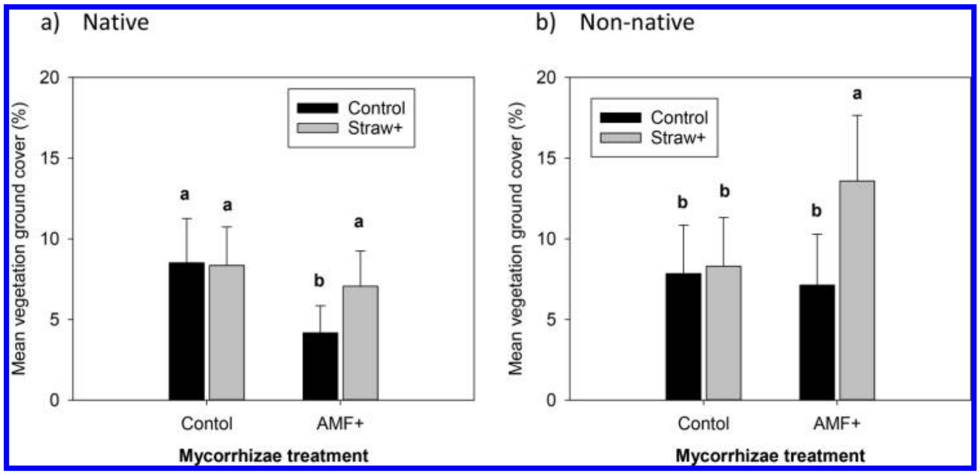


Table 3. Summary of nested GLM for total species richness, non-native species richness, and Shannon–Weiner diversity (H'). Native and non-native groups exclude sown agronomic species.

		Total species richness		Non-native species richness		Shannon–Weiner diversity (H')	
	Df.	*F* value	Pr(>*F*)	*F* value	Pr(>*F*)	*F* value	Pr(>*F*)
Site	2	1.134	0.326	0.037	0.964	0.720	0.489
Pile burn (site)	27	1.136	0.317	**1.880**	**0.010**	1.006	0.470
Cover	1	**5.365**	**0.023**	*3.588*	*0.061*	**8.491**	**0.004**
Seed	2	**23.761**	**<0.001**	**5.317**	**0.007**	**27.175**	**<0.001**
AMF	1	0.275	0.601	0.102	0.750	0.341	0.560
Cover × seed	2	1.316	0.273	2.036	0.137	1.660	0.195
Cover × AMF	1	2.060	0.154	3.624	0.060	0.380	0.539
Seed × AMF	2	0.268	0.765	0.338	0.714	0.177	0.838
Cover × AMF × seed	2	2.352	0.101	1.073	0.346	2.145	0.123

Note: Data were blocked by site (site) and burn nested by site (pile burn (site)). Values are significant at p < 0.05 (bold). Bold numbers indicate p < 0.05; italicized numbers indicate p < 0.10. Residuals = 97 degrees of freedom. Df., degrees of freedom; AMF, arbuscular mycorrhizal fungi.

sufficient to break dormancy in some of the native species. The broadscale seeding of agronomic species after wildfire is a common restoration or rehabilitation technique, although these programs have had mixed success in suppressing invasive species (Peppin et al. 2010).

Our findings tentatively support the hypothesis that the presence of a soil cover such as straw will improve plant establishment and growth. The presence of soil cover in our study not only increased plant cover for agronomic species and total cover, but also increased species richness and diversity. Soil cover seemed to create conditions that were unfavourable to native species but favorable to agronomic species. Rhoades et al. (2015) added a thick layer of woodchips (~10 cm deep) as cover to pile burn scars in Colorado and found that the treatment suppressed plant cover, including seeded species, some of which would be considered agronomic (e.g., *Elymus trachycaulus* and *Pascopyrum smithii*). Conversely, Fornwalt and Rhoades (2011) found that a thin layer of woodchips (~5 cm deep) enhanced the cover of *E. trachycaulus*, which is more in line with our findings. Our results suggest that the cover aspect of litter is a significant contributor to non-native plant community composition and aboveground cover, at least on soils affected by intense heat due to pile burning.

Our findings do not support the hypothesis that commercial AMF inoculum would improve the establishment and growth of native species. In fact, native species were reduced in the presence of AMF when there was also no straw cover added. More worrisome is the fact that the cover of non-native species was increased with the addition of AMF in plots where straw was added. The addition of commercial AMF did not have any significant effect on species richness or diversity. Several mechanisms may explain this lack of significant response: (*i*) AMF inoculum did not contain active spores; (*ii*) spores migrated off plot during the winter season; (*iii*) AMF spores did not survive the winter season; and (*iv*) the AMF inoculum used was inappropriate for the soil conditions. Results from another related study (DeSandoli 2013) suggest that the AMF inoculum did contain active spores, and the secure cover would have prevented some of the spores from migrating off plot. Of the remaining possible explanations, the last is most likely. Although the death or grazing of spores cannot be ruled out, other studies have also reported that commercial AMF inoculum is not always effective when used in a field setting (Rowe et al. 2007; White et al. 2008), possibly because AMF is less of a generalist in nature than previously assumed.

The use of commercial AMF inoculum is controversial (Schwartz et al. 2006). This technology generally involves the application of, at most, a few AMF species with little or no intraspecific genetic variability. This technology also assumes a highly generalist mutualism between AMF and their host plant species, when in reality the relationship between AMF and their host plant species varies from highly mutualistic to neutral to highly parasitic (Klironomos 2003). The broadscale application of limited AMF species and genomes may therefore have positive, neutral, or negative effects on any species. Individual plant responses to the inoculum are difficult to predict prior to investigation. Controlled experiments have shed doubt on the efficacy of commercially available AMF inoculum (e.g., Rowe et al. 2007; White et al. 2008). Considering that the only AMF effect we found was an interaction effect with cover such that non-native invasive species increased when AMF was added in combination with a soil cover, we do not recommend the use of AMF for restoring pile burn scars.

The plant species that naturally regenerated on burns were mostly non-native, primarily composed of non-mycorrhizal (Brassicaceaa and Chenopodiaceae) and facultative mycorrhizal (*Bromus* spp.) species (Wang and Qiu 2006). Those from non-mycorrhizal families were primarily found only on treatment plots. Multiple non-native species, primarily from the Poaceae and Asteraceae families, were found only in the unburned surrounding area, suggesting that the burns are host to some, but not all, non-native species.

Fig. 4. The effect of (*a*) seed treatment (Control, Native+, and Agronomic+) and (*b*) straw treatment (Control and Straw+) on species richness of total vegetation and non-native vegetation. Native and non-native groups exclude sown agronomic species. Error bars represent ± 1 standard error. Bars sharing the same letter are not significantly different (p < 0.05, Bonferroni posthoc test).

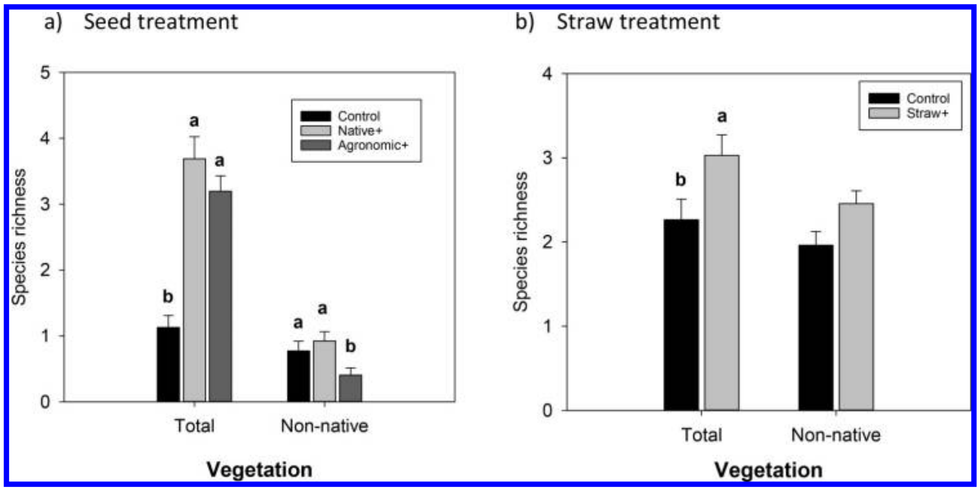
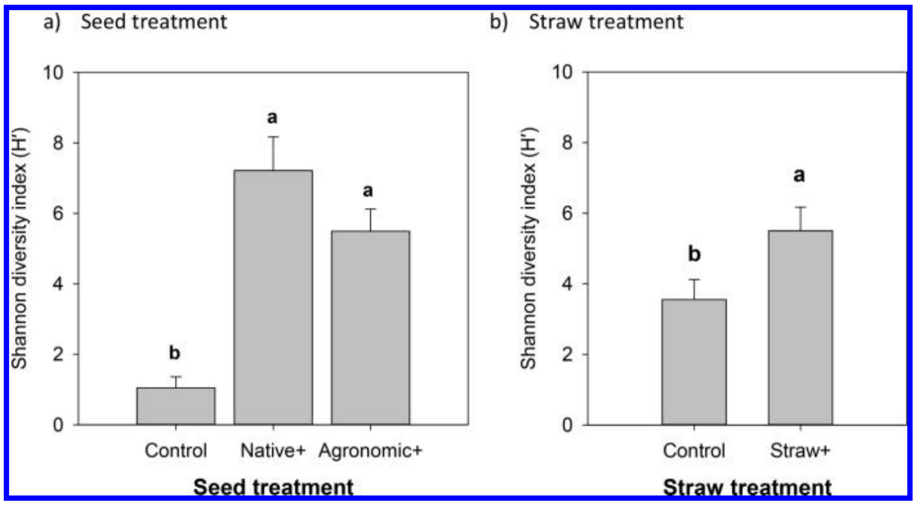


Fig. 5. The effect of (*a*) seed treatment (Control, Native+, and Agronomic+) and (*b*) straw treatment (Control and Straw+) on the Shannon–Weiner diversity index of total vegetation and non-native vegetation. Native and non-native groups exclude sown agronomic species. Error bars represent ± 1 standard error. Bars sharing the same letter are not significantly different (p < 0.05, Bonferroni posthoc test).



Other research has reported that mycorrhizal colonization of roots can be low in post-slash burn environments (Korb et al. 2004; Esquilin et al. 2007) and, combined with our findings, suggest that colonization after a pile burn would favour non- and facultative-mycorrhizal species.

Percent vegetation cover of treatment plots was generally higher than what was observed in similar studies (Haskins and Gehring 2004; Korb et al. 2004; Creech et al. 2012). Although differences in climatic regimes and site factors such as soils and slope among the study sites may contribute to some of this difference, the most likely explanation is the presence of an “intact” vegetation cover in the surrounding areas, possibly combined with the sharp increases to soil nutrients created when burning on frozen soils. Disturbed areas are often seed limited (Korb et al. 2004; White et al. 2008), and the ready supply of propagules from surrounding vegetation contributes to rapid recolonization. The surrounding vegetation in similar studies was burned along with the slash piles; hence, there were fewer recruitment opportunities.

The remaining non-seeded species identified only in the treatment plots were non-native. These included species from the Brassicaceae or Chenopodiaceae family, including Russian thistle (*Kochia scoparia* (L.) Schrad.), tall tumble-mustard (*Sisymbrium altierrssimum* L.), and Loesel's tumble-mustard (*Sisymbrium loeselli* L.). Multiple non-native species were identified in the unburned areas but not in treatment plots, including species from the Poaceae and Asteraceae family such as smooth brome (*Bromus inermis* Leyss.), diffuse knapweed (*Centaurea diffusa* Lam.), oxeye daisy (*Leucanthemum vulgare* Lam.), and Dalmation toadflax (*L. genistifolia* subs. *dalmatica*).

We found that the seeding of agronomics in recent pile burn scars was associated with a reduction in non-native plant species, suggesting a beneficial use of agronomics within this context. However, it should be noted that many agronomics are also non-native and pose the risk of establishing persistent monocultures that may have long-term effects on species diversity (Redente et al. 1989; Isselstein et al. 2005) and thus a reduction in native species diversity. To prevent the prevalence of non-natives in general, whether they are invasive (e.g., *B. tectorum*) or agronomic (e.g., *Lolium perenne*), we suggest the use of native agronomics in the restoration of pile burn scars.

Conclusion

The findings here suggests that pile burning, along with the nonburning of surrounding vegetation, creates areas that are susceptible to high cover of non-native species. Restoration efforts should be directed at these sites after burning to ameliorate the effects of invasive species colonization. The amelioration treatments tested in this research had mixed success in reducing non-native species cover on the red soil of pile burn scars. The most effective treatment in reducing non-native species cover was the seeding of agronomic species, and the additional straw tended to increase plant cover, specifically agronomic species. Commercial AMF inoculum was an ineffective treatment, suggesting that a better understanding of host specificity is warranted.

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Appendix A

Table A1 appears on the following pages.

Table A1. Frequency (%) of identified species in burn treatments and in unburned surrounding plots.

Family	Species	Common name	Status	Control (no treatment)	Agronomic seed	Native seed	No seed	Cover	No cover	Mycorrhizae	No mycorrhizae	All treatment plots	Percentage of all plots unburned
Asteraceae	*Achillea millefolium**	yarrow*	Na	7.7	—	22.2	2.1	5.7	10.3	7.5	8.5	8.0	16.3
Poaceae	*Achnatherum occidentale**	stiff needlegrass*	Na	—	2.2	42.2	6.4	22.9	10.3	14.9	18.3	16.7	18.8
Poaceae	*Achnatherum richardsonii**	spreading needlegrass*	Na	—	—	22.2	—	8.6	5.9	6.0	8.5	7.2	11.3
Rosaceae	*Amelanchier alnifolia*	saskatoon	Na	—	—	—	—	—	—	—	—	—	2.5
Asteraceae	*Antennaria* spp.	pussytoes	Na	—	—	—	—	—	—	—	—	—	13.8
Brassicaceae	*Arabis holboellii*	holboell's rockcress	Na	—	—	—	—	—	—	—	—	—	2.5
Asteraceae	*Artemisia frigida*	pasture sage	Na	—	—	—	—	—	—	—	—	—	5.0
Fabaceae	*Astragalus collinus*	hillside milkvetch	Na	—	—	—	—	—	—	—	—	—	3.8
Fabaceae	*Astragalus miser*	timber milkvetch	Na	—	2.2	4.4	4.3	2.9	4.4	3.0	4.2	3.6	12.5
Asteraceae	*Balsamorhiza sagittata*	arrow-leaf balsamroot	Na	—	—	—	—	—	—	—	—	—	8.8
Poaceae	*Bromus inermis*	smooth brome	NN	—	—	—	—	—	—	—	—	—	1.3
Poaceae	*Bromus japonicus*	Japanese brome	NN	—	—	—	2.1	1.4	—	—	1.4	0.7	2.5
Poaceae	*Bromus squarrosus*	corn brome	NN	—	4.3	8.9	8.5	7.1	7.4	6.0	8.5	7.2	46.3
Poaceae	*Bromus tectorum*	cheatgrass	NN	15.4	8.7	8.9	17.0	14.3	8.8	13.4	9.9	11.6	35.0
Asteraceae	*Centaurea diffusa*	diffuse knapweed	NN	—	—	—	—	—	—	—	—	—	2.5
Chenopodiaceae	*Chenopodium album*	lamb's quarters	NN	—	2.2	—	—	—	1.5	—	1.4	0.7	1.3
Asteraceae	*Cirsium arvense*	Canada thistle	NN	7.7	—	—	2.1	—	1.5	—	1.4	0.7	—
Asteraceae	*Crepis* spp.	hawksbeard	Unk	—	—	4.4	—	2.9	—	—	2.8	1.4	—
Poaceae	*Elymus trachycaulus**	Slender wheatgrass*	Ag	—	82.6	2.2	6.4	34.3	26.5	31.3	29.6	30.4	—
Asteraceae	*Erigeron corymbosus*	long-leaved fleabane	Na	—	—	—	—	—	—	—	—	—	1.3
Asteraceae	*Erigeron filifolius*	thread-leaved fleabane	Na	—	5.0	—	—	—	—	—	—	—	1.3
Poaceae	*Festuca campestris**	rough fescue*	Na	—	—	20.0	—	8.6	4.4	9.0	4.2	6.5	35.0
Poaceae	*Festuca ovina**	sheep fescue*	Ag	—	13.0	—	—	5.7	2.9	4.5	4.2	4.3	—
Poaceae	*Festuca rubra**	red fescue*	Ag	—	50.0	—	2.1	15.7	19.1	14.9	19.7	17.4	—
Liliaceae	*Fritillaria pudica*	yellow bell	Na	—	—	—	—	—	—	—	—	—	3.8
Asteraceae	*Gaillardia aristata**	brown-eyed susan*	Na	—	2.2	13.3	—	7.1	2.9	6.0	4.2	5.1	—
Asteraceae		unkown daisy	Unk	—	—	2.2	—	1.4	—	1.5	—	0.7	3.8
Poaceae	*Hesperostipa comata**	needle-and-thread grass*	Na	—	—	6.7	—	2.9	1.5	4.5	—	2.2	10.0
Poaceae	*Hordeum jubatum**	foxtail barley*	Na	—	—	15.6	—	5.7	4.4	4.5	5.6	5.1	—
Poaceae	*Hordeum vulgare*	common barley	Ag	—	6.5	4.4	2.1	5.7	2.9	7.5	1.4	4.3	—
Chenopodiaceae	*Kochia scoparia*	summer-cypress	NN	—	2.2	4.4	—	2.9	1.5	3.0	1.4	2.2	—
Poaceae	*Koeleria macrantha**	junegrass*	Na	—	—	—	—	—	—	—	—	—	8.8
Asteraceae	*Lactuca serriola*	prickly lettuce	NN	—	4.3	4.4	10.6	1	2.9	6.0	7.0	6.5	6.3
Asteraceae	*Leucanthemum vulgare*	oxeye daisy	NN	—	—	—	—	—	—	—	—	—	2.5
Scrophulariaceae	*Linnaria genistifolia* subs. *damlatica*	Dalmation toadflax	NN	—	—	—	—	—	—	—	—	—	1.3
Boraginaceae	*Lithospermum ruderale*	lemonweed	Na	—	—	—	—	—	—	—	—	—	5.0
Poaceae	*Lolium multiflorum**	Italian ryegrass*	Ag	—	52.2	—	2.1	18.6	17.6	16.4	19.7	18.1	—
Poaceae	*Lolium perenne**	perennial ryegrass*	Ag	—	8.7	—	—	4.3	1.5	3.0	2.8	2.9	—
Berberidaceae	*Mahonia aquifolium*	tall Oregon-grape	Na	—	—	—	—	—	—	—	—	—	1.3
Asteraceae	*Matricaria discoidea*	pineapple weed	Unk	—	—	—	—	—	—	—	—	—	1.3
Fabaceae	*Medicago lupulina*	black medic	NN	—	—	2.2	—	1.4	—	1.5	—	0.7	6.3
Fabaceae	*Medicago sativa*	alfalfa	Ag	—	2.2	—	—	1.4	—	—	1.4	0.7	6.3
Fabaceae	*Melilotus alba*	white sweet-clover	NN	—	2.2	2.2	2.1	2.9	1.5	1.5	2.8	2.2	1.3
Pinaceae	*Pinus ponderosa*	ponderosa pine	NA	—	—	—	—	—	—	—	—	—	1.3
Plantaginaceae	*Plantago patagonica*	woolly plantain	Na	—	—	—	—	—	—	—	—	—	1.3
Poaceae	*Poa secunda*	Nevada bluegrass	Unk	—	2.2	53.3	4.3	22.9	16.2	16.4	22.5	19.6	—

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Table A1 (*concluded*).

Family	Species	Common name	Status	Control (no treatment)	Agronomic seed	Native seed	No seed	Cover	No cover	Mycorrhizae	No mycorrhizae	All treatment plots	Percentage of all plots unburned
Poaceae	*Poa compressa*	Canada bluegrass	Ag	—	—	—	—	—	—	—	—	—	8.8
Poaceae	Poa pratensis	Kentucky bluegrass	NN	—	—	4.4	6.4	4.3	2.9	3.0	4.2	3.6	27.5
Polygonaceae	*Polygonum* sp.		Unk	—	2.2	—	—	1.4	—	—	1.4	0.7	1.3
Poaceae	*Pseudoroegneria spicata**	bluebunch wheatgrass	Na	—	—	53.3	2.1	17.1	19.1	17.9	18.3	18.1	33.8
Rosaceae	*Rosa nutkana*	Nootka rose	Na	—	—	—	—	—	—	—	—	—	2.5
Brassicaceae	*Sisymbrium altissimum*	tall tumble-mustard	NN	—	2.2	4.4	2.1	4.3	1.5	4.5	1.4	2.9	—
Brassicaceae	*Sisymbrium loeselli*	loesel's tumble-mustard	NN	—	2.2	6.7	2.1	2.9	4.4	4.5	2.8	3.6	—
Asteraceae	*Taraxacum officinale*	common dandelion	NN	7.7	8.7	22.2	17.0	17.1	14.7	17.9	14.1	15.9	52.5
Asteraceae	*Tragopogon dubius*	yellow salsify	NN	7.7	—	17.8	4.3	10.0	4.4	3.0	11.3	7.2	51.3
Poaceae	unknown *Elymus**		Ag	—	56.5	—	4.3	22.9	17.6	22.4	18.3	20.3	2.5
Fabaceae	*Vicia americana*	American vetch	Na	—	—	2.2	—	—	1.5	1.5	—	0.7	5.0
Poaceae	*Vulpia octoflora*	six—week fescue	Na	—	—	—	—	—	—	—	—	—	11.3

Note: For species and common names, an asterisk (*) indicates inclusion in seed addition treatment. For status, Na, native; NN, non-native; Ag, agronomic; Unk, unknown.

This article has been cited by:

1. Michael J. Schuster, Peter D. Wragg, Peter B. Reich. 2018. Using revegetation to suppress invasive plants in grasslands and forests. *Journal of Applied Ecology* **54**. . [Crossref]
2. Caroline A. Havrilla, Akasha M. Faist, Nichole N. Barger. 2017. Understory Plant Community Responses to Fuel-Reduction Treatments and Seeding in an Upland Piñon-Juniper Woodland. *Rangeland Ecology & Management* **70**:5, 609-620. [Crossref]