THESIS

PILE BURN SCAR RESTORATION AT LILY LAKE: TRADEOFFS BETWEEN ABUNDANCE OF NON-NATIVE AND NATIVE SPECIES

Submitted by

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ABSTRACT

PILE BURN SCAR RESTORATION AT LILY LAKE: TRADEOFFS BETWEEN ABUNDANCE OF NON-NATIVE AND NATIVE SPECIES

Accumulation of fuels in forests across the western United States is producing larger and more severe wildfires. To decrease wildfire severity and increase forest resilience, foresters regularly remove excess fuel by burning woody material in piles. This common practice can also cause persistent ecosystem changes that include alteration of soil physical and chemical properties due to extreme soil heating, which can favor invasion by non-native plant species. Abundance and species richness of native plant communities may also remain depressed for many years after burning has removed vegetation and diminished propagules in the soil. This adds to the vulnerability of burned areas, which can transition to dominance by invasive species. Research into the use of revegetation techniques following pile burning to suppress invasion is limited. Studies conducted in various woodland types that investigated revegetation of pile burn scars have met with varying success. To assess the effectiveness of restoring pile burn scars in Rocky Mountain National Park, Colorado, we monitored vegetation in 26 scars at Lily Lake the growing season after burning. Later that summer we selected 14 scars for restoration that included soil scarification, seed addition, and pine duff mulch cover. We monitored the scars for 3 years following restoration and found that cover of seeded species exceeded surrounding unburned areas. This suppressed cover of non-native species as well as native species that were not seeded during restoration relative to controls. Productivity of a native forb planted as seed in scars 3 years after restoration was depressed relative to unrestored scars. We conclude that restoration of pile burn scars can be a useful management tool that will likely need to be part of an integrated pest management program addressing preexisting infestations near scars. Monitoring for periods longer than 3 years will help us understand how long suppression of native and non-native species by restoration species may persist.

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Introduction

Abundant fuels in forest landscapes across the American West are a legacy of fire suppression, which in conjunction with climate change can increase the size and severity of wildfires (Adams, 2013). Removal of burnable material has been employed as a restoration method that can decrease fire severity and increase forest resilience (Waltz *et al.*, 2014). A common practice to remove unmarketable wood from the landscape following fuel reduction and other forest management activities is to stack the excess wood or slash and burn the piles when the risk of wildfire is low.

These fires can have dramatic, long lasting impacts (Haskins and Gehring, 2004; Korb *et al.*, 2004; Fornwalt and Rhoades, 2011; Creech *et al.*, 2012; Rhoades and Fornwalt, 2015; Rhoades *et al.*, 2015; DeSandoli *et al.*, 2016). Slash pile burning consumes soil organic material exposing mineral soil (Rhoades *et al.*, 2015) and alters soil chemical characteristics. As soils dry following pile burning, water repellency of soils increases, reducing water infiltration (Hubbert *et al.*, 2015). Soil pH is significantly elevated by burning (Korb *et al.*, 2004) and can remain elevated for at least 6 years afterward (Creech *et al.*, 2012). Significant increases in plant available nutrients such as phosphorus (Creech *et al.*, 2012) as well as mineralized nitrogen (Korb *et al.*, 2004; Fornwalt and Rhoades, 2011; DeSandoli *et al.*, 2016) have also been detected in pile burn scar areas.

Soil nutrient changes associated with pile burning can favor non-native species (Davis *et al.*, 2000), making burn scars more susceptible to invasion (Korb *et al.*, 2004). Exotic plants that invade burned areas can continue to dominate the community after initial establishment.

Exotic plants were still up to four times more abundant in burned areas than adjacent unburned areas 5 years after pile burning in a study by Haskins and Gehring (2004). However, the effects on vegetation are not absolute and may depend on the amount of invasion pressure by exotic plants and the size of the pile as fire intensity increases with pile size. In a chronosequence study, Creech *et al.* (2012) observed very little exotic plant cover (<0.02%) and did not detect a significant difference between 12-15 m wide burn scars and unburned areas. Similarly, in an experiment on restoration of small (2.1-5.4 m diameter) burn piles, Rhoades et al. (2015) observed low exotic plant cover in pile burn scars (1.8%).

Non-native plants also benefit from a lack of native competition as regeneration of plants in pile burn scars is limited by the near elimination of propagules from the soil. Multiple studies that examined the soil seed bank after pile burning showed a significant reduction of viable seed density and species richness (Korb *et al.*, 2004; Creech *et al.*, 2012). Fornwalt and Rhoades (2011) also observed that little or no regeneration occurred from roots or rhizomes at scar interiors. Thus, temperatures near the center of piles were hot enough to penetrate the soil and kill underground plant organs. However, near the edges of the burn pile, where temperatures were lower, some plants emerged from underground structures. These vegetation changes may persist on the landscape. In the chronosequence study, burn scars of varying ages (1-7 years since burning) were assessed to infer how vegetation regenerates over time. A decrease in native species richness was still detectable seven years after pile burning (Creech *et al.*, 2012). The longevity of impacts on native vegetation may also depend on pile size. Unrestored scar interiors recovered native forb and graminoid cover similar to adjacent unburned areas within 3 years after burning in the small scar study (Rhoades *et al.*, 2015). Their findings indicated that impacts of burning smaller piles were short lived and scars can recover without active restoration.

Several methods have been investigated to ameliorate the effects of pile burning. Mulch used as cover in restorations provides a source of carbon, which is likely a limiting resource for microbes once it is consumed during pile burning. Carbon addition promotes growth of microbes that utilize inorganic nitrogen and immobilize it in biomass (Eschen *et al.*, 2007). In the small scar study, a 10 cm deep layer of woodchip mulch rapidly reduced inorganic nitrogen compared to controls, but also suppressed plant establishment even when combined with seeding (Rhoades *et al.*, 2015). Fornwalt et al. (2011) found that a thinner layer of mulch (4-6 cm) still reduced ammonium and nitrate, but also had the benefit of increased native plant cover and richness when combined with seed addition. DeSandoli et al. (2016) found that straw mulch increased the cover of agronomic species and total plant cover but not native species seeded for burn scar restoration. Species richness and diversity were also increased by straw cover.

Scars denuded of vegetation can be rehabilitated by replenishing propagules destroyed by fire. Seeding can establish native vegetation in a variety of environments. In open ponderosa pine (*Pinus ponderosa* Lawson & C. Lawson) park lands of British Columbia, native, herbaceous seed additions increased native and total vegetative cover (DeSandoli *et al.*, 2016). Korb *et al.* (2004) found that seeding increased native forb and graminoid cover in scars of ponderosa pine forest near Flagstaff, AZ. This effect was increased by addition of soil containing microbial propagules. Rhoades et al. (2015) studied scars across an elevation gradient along the Colorado Front range in forests with dominant overstory species comprised of ponderosa pine, Douglasfir (*Pseudotsuga menziesii* (Mirb.) Franco), and lodgepole pine (*Pinus contorta* Douglas ex Loudon). They found that seeding increased total native species richness, but it did not increase cover of native forbs or graminoids. Seeding native grasses similarly increased species richness without affecting plant cover of burn scars in woodlands of piñon pine (*Pinus edulis* Engelm.) and Utah juniper (*Juniperus osteosperma* (Torr.) Little) (Redmond *et al.*, 2014). Conversely, Havrilla *et al.* (2017) found that seeding a mix of native and non-native herbaceous plants and shrubs increased native cover beyond nearby, undisturbed upland piñon-juniper woodlands of southeast Utah. Seeding of native herbaceous plants also increased native plant cover so that seeded species had greater cover in scars than unburned Juniper woodlands in Central Oregon (Kerns and Day, 2014). Seeding in combination with mulch or soil scarification increased native cover and species richness within a year of treatment in upper montane forest dominated by lodgepole pine and quaking aspen (*Populus tremuloides* Michx.) in Colorado (Fornwalt and Rhoades, 2011).

Seeding to establish native plants has an additional benefit of creating a plant community to compete with non-native plants as they invade. The capacity for a community to decrease the success of an invading species is called biotic resistance (Levine *et al.*, 2004). In a meta-analysis of research on biotic resistance to invasion by non-native plants, Levine *et al.* (2004) found competition to have a strong suppressive effect. Given interest in conservation applications, they invoke a need for restoration ecologists to develop community assemblages that maximize invasion resistance. Fourteen years later Schuster *et al.* (2018) lamented that there is limited literature on the efficacy of revegetation to suppress reinvasion following invasive species removal. They concede that interest and the body of literature on this subject is growing as the majority of studies show revegetation suppresses invasive species, but most studies last 3 years or fewer. They also note that revegetation of woodlands is particularly underrepresented in research, which is dominated by studies in grasslands.

Although revegetation studies are less common in woodlands, some studies have shown that seed additions can be used to establish vegetation that suppresses non-native species in pile burn scars specifically (Korb *et al.*, 2004; Kerns and Day, 2014; Redmond *et al.*, 2014; DeSandoli *et al.*, 2016). Success has been mixed, as some studies found that establishing vegetation in pile burn scars does not always decrease the success of non-native species (Fornwalt and Rhoades, 2011; Kerns and Day, 2014; DeSandoli *et al.*, 2016). Like other revegetation research, studies of seeding on pile burn scars has primarily been of short duration. Schuster *et al.* (2018) observed that revegetation experiments lasting fewer than 3 years were less likely to detect suppression of invasive species. This illustrates a need for longer duration studies that test the efficacy of revegetating pile burn scars to suppress invasion by non-native species.

In addition to suppressing non-native species, existing vegetation can affect the performance of native species that colonize later. Early establishing vegetation can facilitate later arrivals, inhibit them, or have neutral effects (Connell and Slatyer, 1977; Temperton and Zirr, 2004). Because existing literature indicates that revegetation can limit the future composition of communities (Schuster *et al.*, 2018), the inhibition of native species following restoration should be evaluated and considered when making management decisions.

To better understand how plant community development is affected by restoration of pile burn scars in Rocky Mountain National Park, Colorado, we monitored plant cover for four consecutive years after burning and assessed performance of plant species seeded 3.5 years after burning in scars with and without restoration. We tested the following hypotheses in two field studies.

> **Hypothesis 1:** Restoration of pile burn scars can ameliorate impacts of pile burning to establish native vegetation.
>
> > **Prediction 1:** Seeded native species will have greater cover in restored scars.
>
> **Hypothesis 2:** Restoration of pile burn scars suppresses non-native species relative to unrestored scars.
>
> > **Prediction 2:** Non-native species will have less cover in restored scars.
>
> **Hypothesis 3:** Restoration of pile burn scars suppresses native species that were not seeded during restoration relative to unrestored scars.
>
> > **Prediction 3:** Native species that were not seeded during restoration will have less cover in restored scars.
> >
> > **Prediction 4:** Native species seeded 3 years after restoration will have fewer plants establish and they will have lower productivity in restored scars.

Materials and Methods

Site Selection and Plot Establishment

The study site was located at Lily Lake near the eastern border of Rocky Mountain National Park in Northern Colorado's Front Range at an elevation of 2,706 m. Lodgepole pine (*Pinus contorta* Douglas ex Loudon) is the dominant overstory species. Average annual precipitation over the past 30 years was 711 mm at the nearby Copeland Lake SNOTEL station (USDA, 2018).

Park foresters stacked fuel into piles south of the lake in the summer of 2013 and burned them the following winter. Impacts, including vegetation loss and charring, were visible as ~5 m wide scars the next growing season. In 2014, we selected 26 scars on a hill with north- and south-facing slopes. We randomly selected seven scars on the north side of the hill and seven scars on the south side of the hill to be restored. We left six scars on each side of the hill (12 total) untreated as controls.

We established transects starting two meters outside of each scar by marking the ends with wooden stakes for consistent plot measurements in subsequent years. Transects ran generally downhill to scar centers. Outside stakes were north of the scar on the south side of the hill and south of the scar on the north side of the hill.

Restoration Methods

Students from the Public Lands History Center at Colorado State University conducted restoration of more than 300 scars near Lily Lake during summer 2014 because it is an area of high visitor use with known populations of Canada thistle (*Cirsium arvense* (L.) Scop.) and cheatgrass (*Bromus tectorum* L.). Students scarified the soil to a depth of ~5-8 cm using hand tools to disrupt the hydrophobic layer of soil. Once the seed bed was prepared, they hand broadcast a seed mix of 24% squirreltail (*Elymus elymoides* (Raf.) Swezey, 80-90% PLS), 64% Canada wildrye (*Elymus canadensis* L., 63-87% PLS), and 12 % prairie sagewort (*Artemisia frigida* Willd., 85-90% PLS) at a rate of 1,086 seeds m^{-2}. Species were selected from readily available seeds that were surplus from another restoration project. Seeds were collected within the park and increased through a contract with the Natural Resources Conservation Service (NRCS) Upper Colorado Environmental Plant Center (Meeker, CO). Students then raked pine duff from the surrounding area onto seeded scars to create approximately 50% litter cover.

National Park Service employees applied herbicide to Canada thistle at the study area in 2014 and 2016 because it is a List B noxious weed in Colorado and is targeted for eradication by Rocky Mountain National Park (NPS, 2003). They spot sprayed plants until evenly wet with Milestone (aminopyralid, Dow AgroSciences LLC, Indianapolis, IN) at a concentration of 0.07 % product (0.03 % active ingredient) mixed with a nonionic surfactant at a concentration of 0.5 %. Control and restored scars were equally likely to receive herbicide if they contained Canada thistle.

Vegetation Sampling

We made ocular estimates of pre-restoration cover in July 2014 and post-restoration cover during summer 2015, 2016, and 2017 at three positions for each scar. To estimate cover in the same location each year, we placed 0.5 m^2 (0.5 x 1 m) quadrats (1) outside of the burn scar at 1.5-2 m from the edge (0-0.5 m on the transect), (2) just inside the edge of the scar (2-2.5 m), and (3) at the scar center (3-3.5m) with the transect on the left side of the short edge when facing the outside stake. Before estimating cover, we identified vascular plants to species. When we could not identify plants to species, we recorded their genus or growth form (i.e. graminoid, forb, tree, or shrub). We described cover as ranges of 0-1, 1-3, 3-5, 5-10, 10-25, 25-50, 50-75, 75-90, 90-95, 95-99, or 99-100% according to modified Daubenmire cover classes (Daubenmire, 1959).

Seedling Establishment Study

After observing lower establishment in restored scars of native species that were not seeded, we conducted a second experiment to examine effects of restoration on plants seeded 3 years after burning/restoration. We established three 25 cm by 25 cm subplots within 1 m of the center stake for each previously monitored scar. Over 4 days, June 5-8, 2017, we added 40 prairie junegrass (*Koeleria macrantha* (Ledeb.) Schult.) seeds and 40 common yarrow (*Achillea millefolium* L.) seeds to each new subplot and raked them into the ground with fingertips to increase contact with soil while minimizing disturbance. Seed was collected from within Rocky Mountain National Park. On September 15, 2017, we counted, collected, dried, and weighed aboveground biomass of yarrow seedlings from each new subplot. Because seedlings were so small, we weighed all seedlings from a subplot together. Very few (~5 total) grass seedlings were observed and could not be positively identified as junegrass, thus, they were not counted or collected.

Data Analysis

We used the midpoint of the cover class range as the response variable for each observation. To analyze cover data, we pooled species into three response variables (seeded species, native species not seeded, and non-native species) following native status reported by the USDA-NRCS PLANTS Database (2016). We analyzed each cover response variable with the MIXED procedure in SAS software, Version 9.4 of the SAS System for Windows (SAS Institute Inc., Cary, NC, USA). To correct *p*-values of fixed effects for multiple testing, we performed the Holm's Step-Down-Procedure in R, Version 3.5.1 (R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/). We performed the GLIMMIX procedure in SAS software on cover response variables to obtain covariance parameter estimates with 95% confidence intervals and pairwise comparisons for fixed factors that had significant effects. We created models for our cover response variables with quadrat and pile burn scars as random effects, where quadrats were nested within pile burn scars. To test for fixed effects, we used treatment, position, year, and aspect as main effects with aspect as a fixed blocking factor with two levels (north or south), then tested for all two-way interactions and main effects. We also verified no spatial autocorrelation by testing for a fixed effect of X and Y coordinates (UTM NAD 83 Zone 13). To meet assumptions of normality and homogeneity of variance, cover data were log transformed before analysis.

We used the total number of seedlings alive at the end of the growing season as a response variable for seedling establishment. To analyze seedling productivity, we summed the seedling biomass for each subplot within a pile burn scar and divided the mass by the number of seedlings weighed. This gave us the average aboveground biomass per plant for each scar. We tested the effect of restoration on seedling establishment and productivity with the GLIMMIX procedure in SAS software and adjusted *p*-values with the Holm's Step-Down-Procedure in R to correct for multiple tests. We considered treatment, aspect, and the interaction between treatment and aspect as fixed effects and verified no spatial autocorrelation by testing for a fixed effect of X and Y coordinates. To meet assumptions of normality and homogeneity of variance, we log transformed the average mass per seedling for each scar.

Results

Seeded Species Cover

Restoration increased cover of species seeded (SD) in restored scars depending on year and position (center, edge, and outside) (Table 1). Restored scars had higher SD cover in all years of the study. The difference between treatments was small, but statistically significant in 2014 (pre-treatment). A single scar, with 3-5% and 0-1% cover of Canada wildrye at the center and edge, respectively, along with 1-3% cover of squirreltail at the center and edge, was responsible for the difference between treatments before restoration. At the end of the study, SD cover in restored scars was 8.6 times higher than controls. For both restored and control scars, SD cover increased in 2015, but no other years differed significantly (Figure 1). Restored scars had nearly 13 times greater cover of SD at their centers than controls when averaged over time from pre-restoration to 3 years after restoration (Figure 2). At scar edges SD cover was 10.77 ± 1.99% (mean ± standard error of the mean) in restored scars, but SD cover was essentially zero in controls. There was no difference between treatments outside of the scars.

Table 1. Linear mixed model analysis of variance of log transformed plant cover at Lily Lake in Rocky Mountain National Park. *P*-values were adjusted using Holm step-down procedure. Bold *p*-values are statistically significant ($\alpha = 0.05$).

	Seeded Species				Non-native Species				Native Species Not Seeded			
	Num df	Den df	*F*	*p*	Num df	Den df	*F*	*p*	Num df	Den df	*F*	*p*
Treatment	1	52.8	126.11	**0.0012**	1	33.4	7.80	0.0860	1	34.2	14.44	**0.01**
Year	3	118.0	45.65	**0.0012**	3	204.0	13.59	**0.0012**	3	190.0	54.05	**0.00**
Position	2	36.5	94.43	**0.0012**	2	39.8	4.37	0.1728	2	41.3	14.51	**0.00**
Treatment*Year	3	49.2	5.98	**0.0105**	3	196.0	5.19	**0.0198**	3	188.0	8.73	**0.00**
Treatment*Position	2	37.9	63.52	**0.0012**	2	40.4	0.06	1.0000	2	41.9	2.60	0.34
Position*Year	6	113.0	20.26	**0.0012**	6	169.0	0.67	1.0000	6	156.0	2.83	0.10
Block	1	39.7	3.01	0.5442	1	20.7	0.27	1.0000	1	22.7	5.69	0.15
Block*Treatment	1	15.5	0.51	1.0000	1	18.6	0.22	1.0000	1	19.4	0.80	0.76
Block*Year	3	49.2	0.97	1.0000	3	196.0	1.65	1.0000	3	188.0	0.50	0.76
Block*Position	2	37.9	1.96	0.7745	2	40.4	0.39	1.0000	2	41.9	4.19	0.15
UTMX	1	15.5	0.49	1.0000	1	18.6	0.24	1.0000	1	19.4	3.85	0.32
UTMY	1	15.5	1.25	1.0000	1	18.6	0.43	1.0000	1	19.4	2.06	0.50

There was no difference in SD cover among positions for control scars, but it increased moving from outside to the center of restored scars (Figure 2).

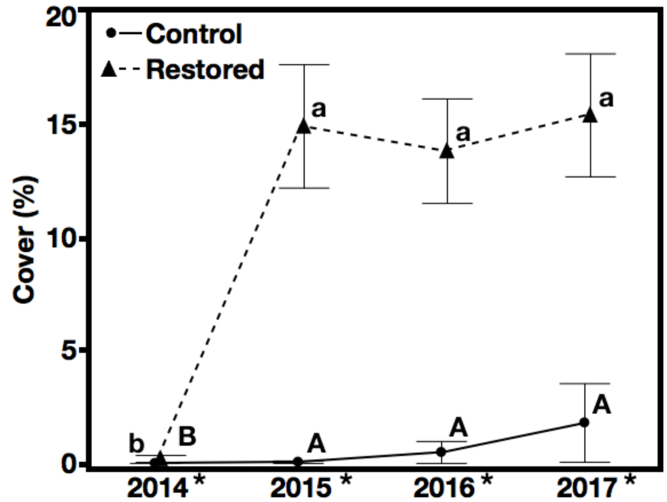

Figure 1. Cover of seeded species [squirreltail (*Elymus elymoides* (Raf.) Swezey), Canada wildrye (*Elymus canadensis* L.), and prairie sagewort (*Artemisia frigida* Willd.)] in pile burn scars before (2014) and after restoration at Lily Lake in Rocky Mountain National Park. Points are means ± standard error of the mean. Means labeled with different letters within a treatment level indicate that log transformed means differed between years and asterisks indicate years when means differed between treatment levels (Holm adjusted $p < 0.05$).

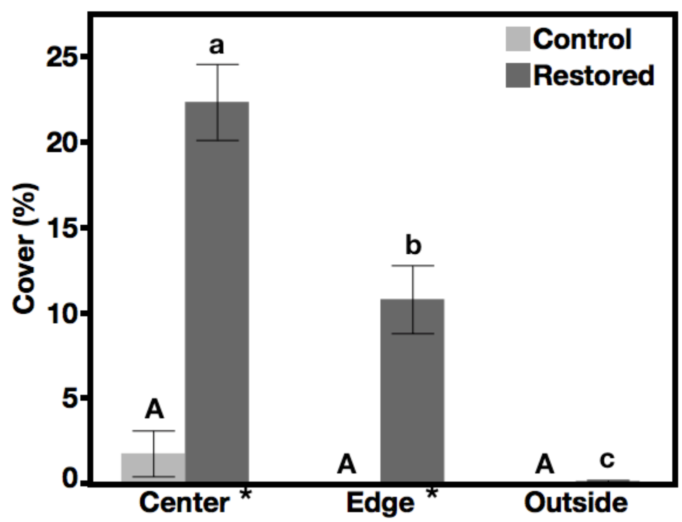

Figure 2. Cover of seeded species [squirreltail (*Elymus elymoides* (Raf.) Swezey), Canada wildrye (*Elymus canadensis* L.), and prairie sagewort (*Artemisia frigida* Willd.)] in pile burn scars at Lily Lake in Rocky Mountain National Park averaged over time (pre-restoration to 3 years after restoration). Bars are means ± standard error of the mean. Means labeled with different letters within a treatment level (restored or control) indicate that log transformed means differed between positions and asterisks indicate positions where means differed between treatment levels (Holm adjusted $p < 0.05$).

The cover of SD in different positions depended on year (Table 1). In 2014, before restoration, all positions had the same cover of SD, but it varied by position in the following years. Cover of SD increased in 2015 at the center and edge of scars and then remained constant so that SD cover was greater at the center of scars than the edges for all years after restoration. We detected no changes outside of scars (Figure 3).

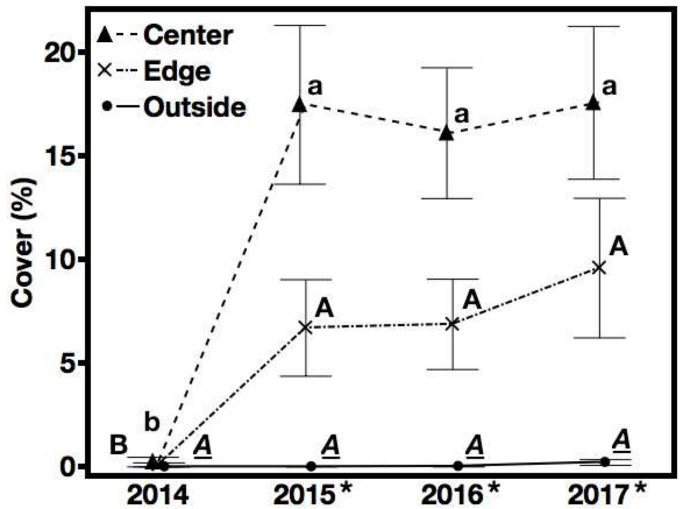

Figure 3. Cover of seeded species [squirreltail (*Elymus elymoides* (Raf.) Swezey), Canada wildrye (*Elymus canadensis* L.), and prairie sagewort (*Artemisia frigida* Willd.)] in pile burn scars before (2014) and after restoration at Lily Lake in Rocky Mountain National Park. Points are means ± standard error of the mean. Means labeled with different letters within a position indicate that log transformed means differed between years and asterisks indicate years when means differed between positions (Holm adjusted $p < 0.05$).

Non-native Species Cover

Non-native species (NN) cover increased over time depending on treatment (Table 1). In scars without restoration, NN cover increased in 2015 and 2016, but not 2017. Conversely, it remained constant in restored scars. We did not detect a difference between treatments in 2014 and 2015, but NN cover was higher in controls than restored scars in 2016 and 2017. By the end of the study, restored scars had less than one third of the non-native cover found in unrestored scars (Figure 4).

Position did not have an effect on non-native cover, and this did not depend on year or treatment (Table 1).

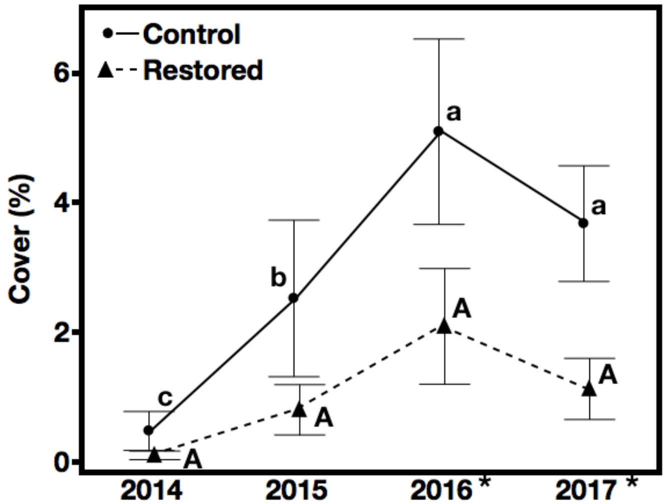

Figure 4. Cover of non-native species in pile burn scars before (2014) and after restoration at Lily Lake in Rocky Mountain National Park. Points are means ± standard error of the mean. Means labeled with different letters within a treatment level indicate that log transformed means differed between years and asterisks indicate years when means differed between treatment levels (Holm adjusted $p < 0.05$).

We found 10 non-native species throughout the study (Supplemental Tables, Table 3). The most abundant non-native species in control scars was Canada thistle, which comprised 40% of the total non-native cover. It had a mean cover of 1.49 ± 0.46% in control scars, when averaged over all years. Its cover in restored scars averaged 0.13 ± 0.06%. Canada bluegrass (*Poa compressa* (L.) Scop.) was the most abundant non-native species in restored scars, where it was less abundant than control scars (0.42 ± 0.17% and 0.53 ± 0.22%, mean cover across years in restored and control scars respectively).

Native Species, Not Seeded Cover

Restoration treatment decreased cover of native species that were not seeded (NSD) depending on year (Table 1). Cover of NSD increased in both restored and control scars, but increased more in controls. Controls had greater NSD cover than restored scars in 2016 and 2017 (Figure 5). In 2015, log transformed NSD cover was slightly higher in restored scars than controls, but this was marginally significant (p = 0.0497) and the raw data did not have the same relationship. By the end of the study, controls had 65% more NSD cover than restored scars.

NSD differed among positions independent of treatment and year (Table 1). Cover of NSD was lower at the center than the edge and outside, where it did not differ (Figure 6).

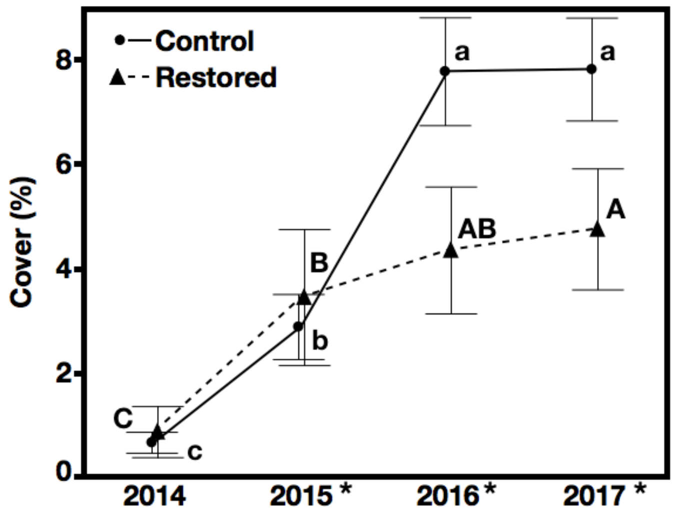

Figure 5. Cover of native species that were not seeded in pile burn scars before (2014) and after restoration at Lily Lake in Rocky Mountain National Park. Points are means ± standard error of the mean. Means labeled with different letters within a treatment level indicate that log transformed means differed between years and asterisks indicate years when means differed between treatment levels (Holm adjusted $p < 0.05$).

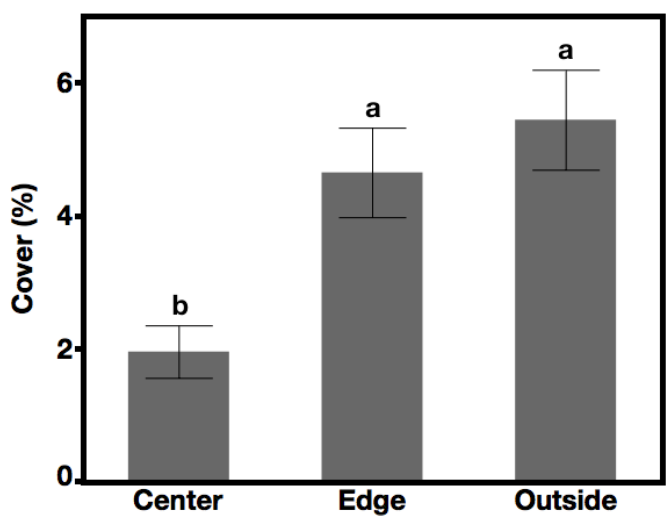

Figure 6. Cover of native species that were not seeded in pile burn scars at Lily Lake in Rocky Mountain National Park averaged over time and treatment. Bars are means ± standard error of the mean. Means labeled with different letters indicate that log transformed means differed between position (Holm adjusted $p < 0.05$).

We found 45 native species other than those seeded in the initial restoration (Supplemental Tables, Table 4). American red raspberry (*Rubus idaeus* L.) was the most abundant NSD species in control scars with an average cover of 0.47 $\pm$ 0.19%. Silverleaf phacelia (*Phacelia hastata* Douglas ex Lehm.) was the most abundant native species not seeded in restored scars, with a cover of 0.26 $\pm$ 0.15%.

Recruitment and productivity of Species Seeded Three Years After Restoration

Seedlings established equally regardless of restoration (Table 2), but seedlings in control scars accrued more than 15 times more aboveground biomass than in restored scars (Figure 7). We did not find a significant correlation between the number of established seedlings and their aboveground biomass (r = 0.19, p = 0.35).

Table 2. Linear mixed model analysis of variance of establishment and logit transformed productivity (mean aboveground biomass per plant) of common yarrow (*Achillea millefolium* L.) seeded into scars 3.5 years after pile burning at Lily Lake in Rocky Mountain National Park. Treatment levels were restored or unrestored control. *p*-values adjusted using Holm step-down procedure. Bold *p*-values are statistically significant (α = 0.05).

	Survival				Productivity			
	Num df	Den df	*F*	*p*	Num df	Den df	*F*	*p*
Treatment	1	20	0.75	0.3980	1	20	30.88	**<0.00050**
Block	1	20	5.74	0.1320	1	20	0.91	0.14108
Treatment*Block	1	20	1.80	0.3886	1	20	1.97	0.46530
UTMX	1	20	2.54	0.3801	1	20	2.18	0.46530
UTMY	1	20	5.00	0.1476	1	20	2.18	0.46530

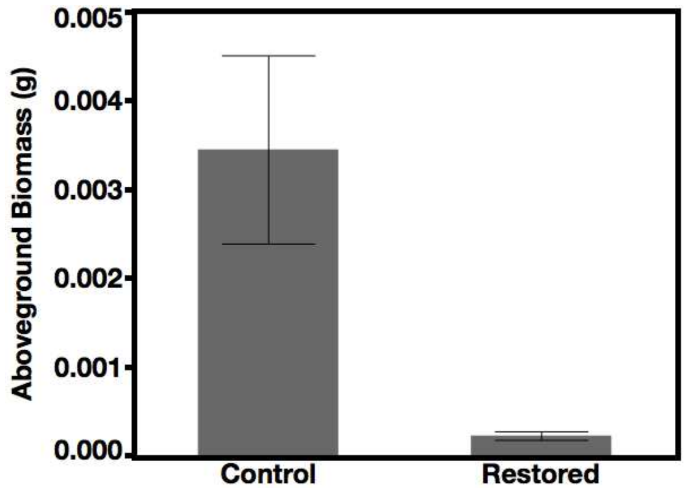

Figure 7. Productivity of common yarrow (*Achillea millefolium* L.) seeded into scars 3.5 years after pile burning at Lily Lake in Rocky Mountain National Park. Bars are means ± standard error of the mean of aboveground biomass per plant. Means labeled with different letters indicate that logit transformed means differed between treatment levels (Holm adjusted $p < 0.05$).

Discussion

Burn scar restoration may not be necessary in all situations (Rhoades *et al.*, 2015), but it can be a useful tool to establish native vegetation quickly (Korb *et al.*, 2004; Fornwalt and Rhoades, 2011; Kerns and Day, 2014; DeSandoli *et al.*, 2016) and reduce risks of invasion (Korb *et al.*, 2004; Kerns and Day, 2014; Redmond *et al.*, 2014; DeSandoli *et al.*, 2016).

Establishment of Seeded Species

Establishment of seeded species following restoration increased native plant cover, which supports Hypothesis 1. Restoration species established quickly to dramatically increase the difference between treatments after the first year. Large increases in SD cover at restored scar centers and edges created this difference while SD cover remained low for all positions at control scars. In a similar study along the Front Range of Colorado, native forbs and graminoids recovered to abundance found in unburned areas outside of scars regardless of seeding (Rhoades *et al.*, 2015). This differs from our results where restoration treatments created greater understory plant cover than that found in unburned areas. Similarly, Havrilla *et al.* (2014) found seeding increased native cover beyond areas without fuel reduction and the majority of other studies also found increases in native cover following seeding (Korb *et al.*, 2004; Fornwalt and Rhoades, 2011; Kerns and Day, 2014; DeSandoli *et al.*, 2016; Havrilla *et al.*, 2017).

Suppression of Non-native Species

Restoration suppressed abundance of non-native species relative to unrestored scars, similar to previous studies and in support of Hypothesis 2. Because restoration treatments were successful in this way, we were able to demonstrate that local, native seed can effectively compete with non-natives. Desandoli *et al.* (2016) found that agronomic (primarily non-native) species, seeded into pile burn scars reduced cover of non-native species (excluding seeded species), but native seed did not. This prompted them to suggest testing the effectiveness of native agronomic species to avoid negative consequences of using non-native agronomic species. In a seeding experiment on pile burn scars in juniper woodlands of central Oregon, seeded cultivars decreased cover of tall, non-native annuals, but local seed had no effect on non-native vegetation (Kerns and Day, 2014). Rhoades et al. (2015) achieved a marginal reduction in non-native cover using a mix of local seed and seed purchased from regional suppliers while addition of native seed from regional suppliers decreased cover of non-native species after scars were amended with living soil in Arizona (Korb *et al.*, 2004). These mixed results indicate that seed origin is unlikely to be the determining factor of non-native suppression. Success of our restoration was likely because we used seeds of grasses competitive with invasive species (Arredondo *et al.*, 1998; Ulrich and Perkins, 2014). We also monitored scars long enough for non-native species to increase and differentiate between treatments, while others expected increases after termination of their experiment (Fornwalt and Rhoades, 2011). Establishment of seeded species to increase native cover was also likely important as Redmond *et al.* (2014) attributed their lack of suppression to low seedling establishment during a drought.

Although park employees sprayed Canada thistle in the study area with herbicide, we attribute the difference between treatments to restoration. Cover of non-native species did not differ between treatments and plants had an equal chance of receiving herbicide whether they occurred in a restored or unrestored scar.

Suppression of Unseeded Native Species

While suppression of non-native plants is a goal of restoration, suppression of native plants may be an unintended consequence. Restoration reduced cover and productivity of native species that we did not seed during restoration, as predicted by Hypothesis 3. While cover of NSD was greater in controls than restored scars for the final 2 years of our study, it was continuing to increase in restored scars. If this trend continues, NSD cover in restored scars may eventually equal that in controls, especially if seeded species decline and are replaced by later seral species as expected. However, we still detected a net competitive effect reducing productivity of common yarrow seeded 3 years after restoration. Because seedlings established equally in controls and restored scars, the difference in biomass was not caused by variation in germination or seedling establishment leading to variation in intraspecific competition. Furthermore, there was no correlation between mean aboveground biomass and establishment. Yarrow plants accumulated mass at the same rate regardless of how many conspecifics were around, indicating no net intraspecific competition or facilitation.

Native plants established quickly at scar edges regardless of treatment, but scar centers did not recover to the level of NSD cover found in unburned areas at our study site. In a study of smaller burn scars (3.5 m mean diameter), vegetation near scar centers recovered to equal total plant cover found in unburned areas outside scars (Rhoades *et al.*, 2015). This may be a result of smaller burn piles producing less severe fires that decrease the effects of burning.

Conclusions

Restoring pile burn scars can quickly establish native cover to slow the march of invasion, but it does not prevent it entirely. Because restoration does not address preexisting infestations in unburned surroundings, it will likely need to be part of an integrated pest management program if management objectives are focused on control of invasive species. Even when invasive species control is the primary goal of management, seeding may not be necessary if invasion pressure is low and there are sources of native seed that can disperse into scars. This is particularly true at scar edges where plant cover regenerates quickly or throughout smaller scars. The ability to detect an effect of restoration may also change over longer time periods. Thus, restoration experiments in pile burn scars should be followed for longer periods in the future to see if restored scars are eventually swamped with non-native species and seeded species are replaced. The studies we found that examined seeding into pile burn scars only sampled vegetation for one to two years (Korb *et al.*, 2004; Fornwalt and Rhoades, 2011; Kerns and Day, 2014; Redmond *et al.*, 2014; Rhoades *et al.*, 2015; DeSandoli *et al.*, 2016). One study sampled vegetation six growing seasons after pile burning, but all pile burn plots were seeded and could not be compared to unseeded controls (Havrilla *et al.*, 2017). It will also be important to follow pile burn scars over longer time periods to examine how long restoration suppresses native vegetation. Managers will need to consider objectives of management actions when weighing the benefits of restoration against the consequences for native vegetation. We did not see unseeded native species recover in scar centers, leaving unrestored scars vulnerable to invasion.

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Appendix

Supplemental Tables

Table 3. Cover of non-native plants by species in restored and unrestored pile burn scars at Lily Lake in Rocky Mountain National Park. Means and standard errors of the mean are averaged across years and position.

	Control		Restored	
	Mean	SE	Mean	SE
Agrostis stolonifera L.	0.1111	0.0499	0.0863	0.0426
Bromus inermis Leyss.	0.0486	0.0313	0	0
Bromus tectorum L.	0.0313	0.0198	0	0
Cirsium arvense (L.) Scop.	1.4861	0.4556	0.1310	0.0590
Phleum pratense L.	0.0799	0.0589	0.0744	0.0411
Poa compressa L.	0.5313	0.2156	0.4196	0.1706
Rumex acetosella L.	0.4306	0.2150	0	0
Rumex crispus L.	0	0	0.0685	0.0505
Salsola collina Pall.	0.0035	0.0035	0	0
Taraxacum officinale F. H. Wigg.	0.2118	0.0603	0.2440	0.1226

Table 4. Cover of native plants that were not seeded by species in restored and unrestored pile burn scars at Lily Lake in Rocky Mountain National Park. Means and standard errors of the mean are averaged across years and position.

	Control		Restored	
	Mean	SE	Mean	SE
Achillea millefolium L.	0	0	0.0030	0.0030
Agrostis scabra Willd.	0.1840	0.0737	0.0149	0.0123
Agrostis sp.	0.2986	0.1071	0.0952	0.0531
Androsace septentrionalis L.	0.0139	0.0139	0.0506	0.0448
Antennaria parvifolia Nutt.	0.0035	0.0035	0	0
Antennaria sp.	0.0278	0.0278	0	0
Arabis fendleri (S. Watson) Greene	0	0	0.0030	0.0030
Arctostaphylos uva-ursi (L.) Spreng	0	0	0.0119	0.0119
Arnica cordifolia Hook.	0.0694	0.0539	0.0238	0.0238
Arnica latifolia Bong.	0	0	0.0149	0.0149
Arnica sp.	0.0660	0.0538	0	0
Artemisia ludoviciana (Nutt).	0.0278	0.0278	0.0119	0.0119
Artemisia sp.	0.0278	0.0278	0	0
Astragalus sp.	0.0035	0.0035	0	0
Carex sp.	1.0278	0.2183	0.8542	0.2923
Chamerion angustifolium (L.) Holub	0.0139	0.0139	0.0595	0.0313
Chenopodium fremontii S. Watson	0.1424	0.0758	0.0030	0.0030
Chenopodium sp.	0.0035	0.0035	0	0
Corydalis aurea Willd.	0.0035	0.0035	0.0506	0.0448
Descurainia incana (Bernh. ex Fisch. & C.A. Mey.) Dorn	0.0035	0.0035	0	0
Potentilla fissa Nutt.	0.0035	0.0035	0.0030	0.0030
Epilobium ciliatum Raf.	0.0243	0.0151	0	0
Epilobium lactiflorum Hausskn.	0.0174	0.0143	0.0030	0.0030
Epilobium palustre L.	0.0035	0.0035	0	0
Epilobium sp.	0.0243	0.0151	0.0060	0.0042
Equisetum arvense L.	0	0	0.0030	0.0030
Erysimum capitatum (Douglas ex Hook.) Greene	0.0486	0.0313	0	0
Festuca saximontana Rydb.	0.0139	0.0139	0.0476	0.0290
Festuca sp.	0	0	0.0030	0.0030
Fragaria sp.	0	0	0.0030	0.0030
Fragaria virginiana Duchesne	0	0	0.0060	0.0042
Gayophytum diffusum Torr. & A. Gray	0.0035	0.0035	0.0298	0.0138
Grindelia subalpina Greene	0.0035	0.0035	0.0030	0.0030

	Control		Restored	
	Mean	SE	Mean	SE
Harbouria trachypleura (A. Gray) J.M. Coult. & Rose	0	0	0.0298	0.0173
Heterotheca villosa	0.0278	0.0278	0	0
Koeleria macrantha (Ledeb.) Schult.	0.0278	0.0196	0	0
Leymus ambiguus	0.0174	0.0143	0	0
Lupinus bakeri	0	0	0.0030	0.0030
Monolepis nuttalliana (Schult.) Greene	0.0208	0.0147	0	0
Oreochrysum parryi (A. Gray) Rydb.	0.0278	0.0278	0.0238	0.0238
Oxytropis sp.	0.0660	0.0538	0.0030	0.0030
Packera fendleri (A. Gray) W.A. Weber & Á. Löve	0.0556	0.0391	0	0
Penstemon sp.	0.0035	0.0035	0	0
Penstemon virens Pennell ex Rydb.	0.1736	0.0660	0.0030	0.0030
Phacelia alba Rydb.	0.0035	0.0035	0.0685	0.0505
Phacelia hastata Douglas ex Lehm.	0	0	0.2649	0.1535
Phacelia sericea (Graham) A. Gray	0.1007	0.0605	0.0595	0.0462
Phacelia sp.	0	0	0.0030	0.0030
Pinus contorta Douglas ex Loudon	0.0486	0.0243	0.0446	0.0210
Pinus sp.	0.1111	0.0272	0.0655	0.0279
Potentilla fissa Nutt.	0.0208	0.0147	0.0238	0.0168
Potentilla norvegica L.	0	0	0.0238	0.0238
Pseudognaphalium macounii (Greene) Kartesz	0.0417	0.0310	0	0
Pseudotsuga menziesii (Mirb.) Franco	0.0035	0.0035	0.0060	0.0042
Ribes sp.	0.0035	0.0035	0.0238	0.0168
Rosa woodsii Lindl.	0.0451	0.0241	0	0
Rubus idaeus L.	0.4722	0.1943	0.1220	0.0651
Sedum lanceolatum Torr.	0.0521	0.0245	0.0863	0.0426
Senecio eremophilus Richardson	0.0035	0.0035	0.1280	0.1067
Senecio sp.	0.0035	0.0035	0	0
Solidago simplex Kunth	0.0660	0.0538	0	0
Solidago sp.	0	0	0.0714	0.0505
Unk. Forb	0.5208	0.1295	0.2619	0.1479
Unk. Grass	0.0625	0.0220	0.0476	0.0271
Unk. Sedge	0.7431	0.2190	0.6815	0.2218