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Post-fire seeding with ryegrass: implications for understorey plant communities and overall effectiveness

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Abstract. Seeding following high-severity wildfires is motivated by the goals of increasing vegetative cover and decreasing bare soil in order to minimise soil erosion and exotic plant invasions. We compared the ground cover and vegetation response of seeded versus non-seeded areas located in the Warm Fire in northern Arizona, where post-fire seeding treatments with Italian ryegrass (*Lolium perenne* spp. *multiflorum* (L.)) were conducted in 4000 ha of high-severity burned areas. Over the course of the study, we observed no significant difference between seeded and non-seeded plots in percentage of bare soil, total vegetative cover or exotic plant cover. However, there were significant differences in plant community composition as revealed by PERMANOVA and Indicator Species Analysis. Two years post-fire there were significantly fewer ponderosa pine seedlings, and the cover of annual and biennial forbs was significantly lower in plots that were seeded with Italian ryegrass. In the third year, the cover of native bunch grasses was significantly lower in seeded plots. The differences we observed may be due to differences in pre-existing vegetation composition because of the geographic separation of the plots across the landscape. Our results illustrate the ineffectiveness of post-fire seeding in achieving the goals of increasing vegetative cover and decreasing the invasion of non-native plants, and we suggest that alternative post-fire remediation should be considered in the future.

Additional keywords: Burned Area Emergency Response, exotics, fire effects, *Lolium perenne* spp. *multiflorum*, non-native plants, northern Arizona, ponderosa pine, Warm Fire.

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Introduction

The ponderosa pine (*Pinus ponderosa var. scopulorum* C. Lawson) forests of the American Southwest are vastly different in structure, function and ecological processes than 150 years ago (Covington and Moore 1994; Allen *et al.* 2002). The interactions of grazing and fire suppression have created unnaturally high levels of fuel accumulation resulting in historically uncommon large and severe crown fires (Covington and Moore 1994; Allen *et al.* 2002). The effects on the forest community from these fires outside the historical range of variability include increased tree mortality (McHugh and Kolb 2003), an initial decrease of understorey plant cover (Springer and Laughlin 2004; Hunter *et al.* 2006) and the subsequent susceptibility of the landscape to invasion of non-natives, soil erosion and flooding (Beyers 2004; Keeley 2004; Hunter *et al.* 2006). Many managers are responding to these fires by emphasising post-fire rehabilitation and restoration. The immediate loss of vegetation leaves the land more susceptible to erosive forces, and managers often employ treatments such as landscape-scale seeding, contour felling of logs, or mulching to mitigate this effect (Robichaud *et al.* 2000). Seeding with non-native grasses is the technique most often used to accomplish the objectives of controlling soil erosion and exotic plant invasions (Robichaud *et al.* 2000; Keeley 2004; Peppin *et al.* 2010). Although this has been common practice for many years, it is controversial because evidence supporting its effectiveness is lacking and conflicts with other management goals (Robichaud *et al.* 2000; Beyers 2004; Keeley *et al.* 2006; Peppin *et al.* 2010).

Post-fire seeding is done on the assumption that once seeds germinate and establish, plant cover will increase, thus decreasing exposed bare soil and preventing soil erosion (Robichaud *et al.* 2000; Beyers 2004). The ratio of plant and ground cover to bare soil is important in determining the amount of post-fire erosion. Increased plant cover intercepts precipitation before it hits the soil and can prevent rain-splash and sheet erosion, and plant roots increase the rate of water infiltration (DeBano *et al.* 1998; Benavides-Solorio and MacDonald 2001).

In field studies, estimates of ground cover are often substituted for actual measurements of erosion because of the difficulty and expense of accurately measuring erosion. A minimum of 60% ground cover (including vegetation, litter, rock, wood, etc.) is normally used as the threshold required to prevent erosion (Orr 1970; Noble and Slatyer 1977; Robichaud *et al.* 2000; Beyers 2004). One study found that when bare soil reaches 60–70% in high-severity burn areas the amount of soil erosion increases (Johansen *et al.* 2001), and other studies reported a significant decrease in erosion with ground cover levels between 50 and 70% (Benavides-Solorio and MacDonald 2001; Wagenbrenner *et al.* 2006). In general, the more ground cover, the less potential for erosion.

Seeding is also done to control the invasion and spread of exotic species (Robichaud *et al.* 2000; Peppin *et al.* 2010). Forested landscapes that have been relatively undisturbed by fire or human influences typically have low cover of exotic species (Keeley 2004; Huisinga *et al.* 2005) but when fire is introduced it opens the canopy, exposes bare mineral soil, increases soil pH and releases phosphorous and nitrogen, creating an improved habitat for plant germination, especially exotics (Keeley 2004; Floyd *et al.* 2006). Successful inhibition of exotic invasions is closely linked to the total cover of the seeded species whether they are native or exotic (Peppin *et al.* 2010). Most studies that report a decline in exotic cover also report a significant difference in cover between seeded and non-seeded sites (Barclay *et al.* 2004; Keeley 2004; Floyd *et al.* 2006). If cover of the seeded species is low, there is usually no difference in total plant cover and no decrease in exotic plant cover (Hunter and Omi 2006*a*; Stella *et al.* 2010). Contamination of post-fire seed sources with other unwanted, weedy, exotic species can lead to large-scale spread of undesirable plants across the landscape (Keeley *et al.* 2006).

Exotic grasses and cultivars are commonly used in post-fire seeding because they are readily available and inexpensive, and germinate quickly and establish in a variety of conditions providing quick vegetative cover (Robichaud *et al.* 2000). Managers seed with the goal of promoting the long-term health and sustainability of an ecosystem but may produce a perverse result by intensifying competition with non-native seeded species (Beschta *et al.* 2004). However, some studies show that when seeding with native species, the results can be beneficial in that exotic species can be controlled and plant community composition can be diversified (Floyd *et al.* 2006).

Italian ryegrass (*Lolium perenne* ssp. *multiflorum* [(Lam.) Husnot]) is a commonly used exotic grass for post-fire seeding even though it can have deleterious effects on native vegetation regeneration (Robichaud *et al.* 2000; Beyers 2004). Exotic grass species often capture resources more aggressively than do native bunch grasses, thus giving non-natives a competitive advantage in a post-fire environment (Nadkarni and Odion 1986; Conard *et al.* 1991; Barclay *et al.* 2004; Keeley 2004; Hunter and Omi 2006*a*). Ryegrass in particular can compete with native plant species (Taskey *et al.* 1989) and decrease the biomass, species richness, and diversity of native species (Nadkarni and Odion 1986; Conard *et al.* 1991). Further, ryegrass can spread after a fire and may become invasive and displace more desirable native species (USDA NRCS 2002).

Overall, post-fire seeding with exotic grasses tends towards the suppression of native species, but results can vary depending on the seeded species, method of seed application, initial cover and post-fire precipitation (Barclay *et al.* 2004; Keeley 2004; Hunter and Omi 2006*a*; Peppin *et al.* 2010). The increase in propagule pressure as a result of seeding can inhibit native seed establishment and lead to lower native grass density (Hunter and Omi 2006*b*). It can also reduce the abundance of fire-following annuals and disrupt root establishment of soil-stabilising perennials (Griffin 1982; Taskey *et al.* 1989; Keeley 2004). Ponderosa pine seedling regeneration and survivorship can also be inhibited when the cover of the seeded species exceeds 40% (Griffin 1982; Barclay *et al.* 2004). Unfortunately, much of the information available on the effects of seeding is the result of short-term investigations (less than 5 years post-fire), and the long-term significance of seeding remains unknown (Robichaud *et al.* 2000; Beyers 2004; Peppin *et al.* 2010).

In this study we examined the differences between seeded and non-seeded sections of the Warm Fire on the North Kaibab Plateau in northern Arizona in a predominately ponderosa pine forest (Fig. 1). The Warm Fire was the first large wildland fire use (WFU) fire on the North Kaibab Ranger District, and burned a total of ~23 500 ha with 7700 ha of mixed-severity fire in the WFU section and another 15 800 ha outside the management area (classified as a wildfire). Almost 4000 ha of high-severity burned area in the wildfire section were seeded with ryegrass. We aimed to determine if there were any differences between the vegetation communities in seeded versus non-seeded areas 1 to 3 years post-fire. We hypothesised that seeding would (1) increase vegetative cover and decrease bare soil, (2) decrease the abundance of other exotics and (3) alter native plant community composition and decrease species richness.

Methods

Study area

The North Kaibab District of the Kaibab National Forest is located just north of Grand Canyon National Park in northern Arizona on the Colorado Plateau. The Kaibab Plateau ranges in elevation from 2000 to 2800 m. The lower elevations consist of pinyon (*Pinus edulis* (Engelm.))–juniper (*Juniperus* ssp.) forests, the mid-elevations are predominately ponderosa pine forests, and higher elevations are mixed conifer forests. The average annual precipitation is 61 cm (Western Regional Climate Center see www.wrcc.dri.edu, accessed January 2010). Half of the precipitation comes during winter months in the form of snow and the remainder falls during summer (July–September) monsoon rainstorms. These monsoon rainstorms are usually localised, intense and of short duration.

The Warm Fire (8 June–4 July 2006) was a WFU fire that burned in mixed severities across three vegetation types on the Kaibab Plateau in the Kaibab National Forest in northern Arizona, USA (Fig. 1). Plot locations for this study range in elevation from 2300 to 2590 m and at the landscape scale have a similar disturbance history in terms of grazing and logging (Trudeau 2006). Pre-fire vegetation in the study area consisted of a ponderosa pine-dominated overstorey and an understorey composed of muttongrass (*Poa fendleriana* (Steud.) Vasey), squirreltail (*Elymus elymoides* (Raf.) Swezey), Junegrass (*Koeleria macrantha* (Ledeb.) Schult), small leaf pussytoes (*Antennaria parvifolia* Nutt.), Fendler's sandwort (*Arenaria fendleri* A. Gray) and woolly cinquefoil (*Potentilla hippiana* Lehm.). Soils are derived from Kaibab Limestone parent material (Brewer *et al.* 1991).

Annual precipitation measured from October to September (2007–2009) was 48 cm in 2007, 33 cm in 2008 and 30 cm in 2009 (Western Regional Climate Center see www.wrcc.dri.edu,

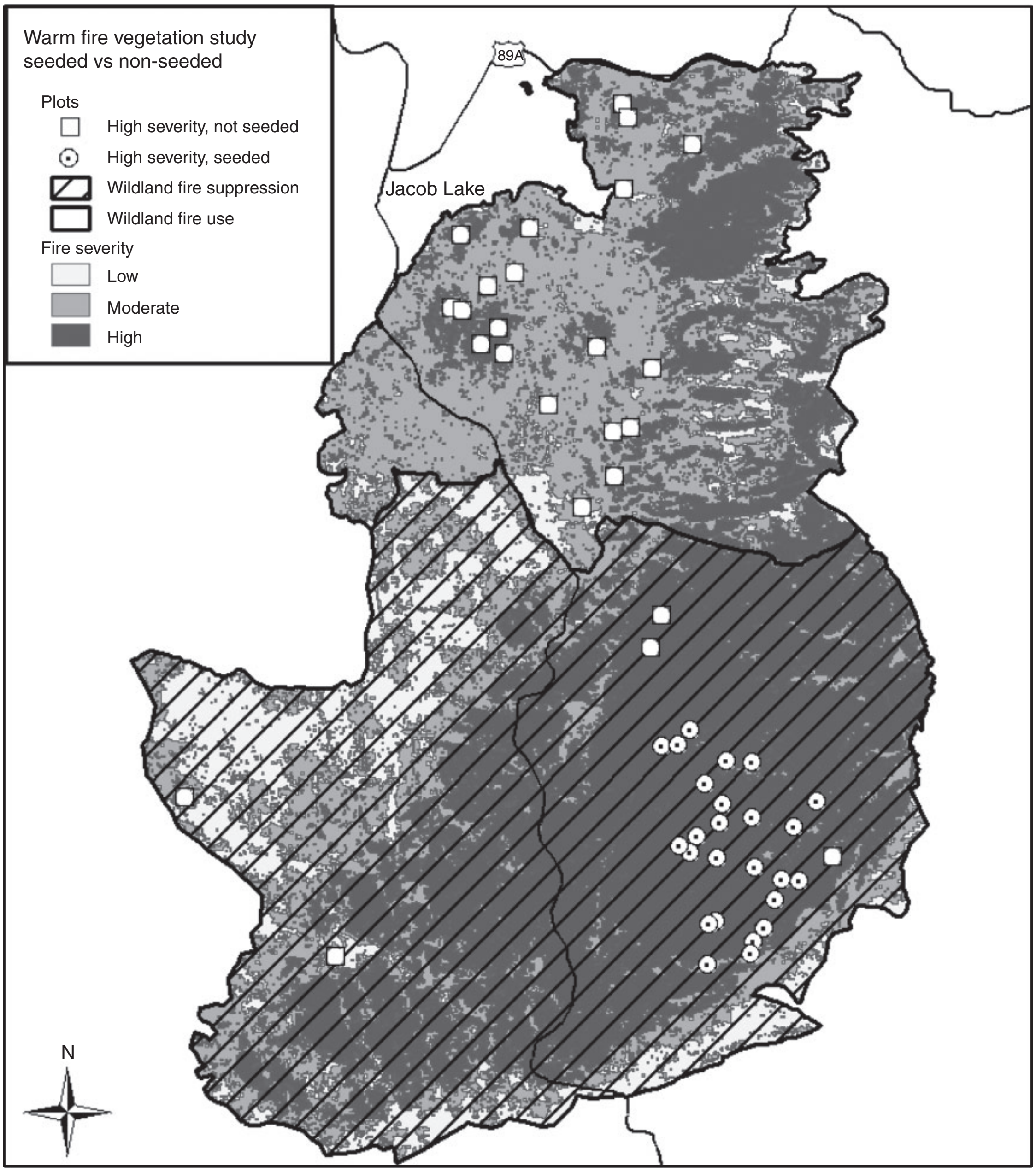

Fig. 1. Location of study site. Perimeter of the Warm Fire; the northern portion of the fire (no crosshatch) was managed as a wildland fire use, and the southern portion (crosshatched) was managed as a wildfire and was subjected to post-fire mitigation. The fire encompassed 24 000 ha across three vegetation types. Burn severity is indicated by varying shades of grey. Seeded plots are indicated by circles and non-seeded plots by squares.

accessed January 2010). During our study, precipitation in May of 2009 was higher than the previous 10 years (www.wrcc.dri.edu).

Moderate- and high-severity burn areas of the wildfire section were seeded with Italian ryegrass by a federal Burned Area Emergency Rehabilitation (BAER) team during the first week of July 2006 (USDA Forest Service 2006). Across the landscape, certified weed-free seed was applied by fixed wing aircraft to 4063 ha of ponderosa pine and mixed conifer forests at a rate of 3.92 kg ha^{-1}. The WFU portion of the fire was not seeded and served as our control. Few data are available for the first growing season after the fire, but reports indicate that summer monsoonal rains began within a week of seeding and washed the majority of the seed off the steepest slopes before germination could occur (USDA Forest Service 2007; www.wrcc.dri.edu, accessed January 2010). Establishment of ryegrass was not consistent across the landscape with more seed and plants in drainages and areas of low slope than on the steeper slopes (USDA Forest Service 2007).

Sampling design

Our study took advantage of a natural, unplanned disturbance (wildfire) and therefore we were unable to employ a more robust sampling design including replication and randomisation (van Mantgem *et al.* 2001). Potential plot locations were restricted to high-severity ponderosa pine forests and points were randomly selected within those areas. High-severity areas were delineated using the BAER burn severity map derived from Landsat satellite imagery, and ground truthed using visual assessments in accordance with the Composite Burn Index (Key and Benson 2006). Site elevation was restricted to within the ponderosa pine vegetation type on slopes less than 28° (53%). Soil types were restricted to Mollic Eutroboralfs and were determined using Terrestrial Ecosystem Survey of the North Kaibab National Forest (Brewer *et al.* 1991). We attempted to control for environmental variation as much as possible, but seeded plots in the wildfire portion of the burn are closer to the eastern end of the plateau and are often located along exposed ridges. Non-seeded plots were scattered across the WFU portion of the fire (Fig. 1).

As a pilot study 1 year post-fire, we established seven plots in each treatment area. Plot design was based on previous vegetation sampling done by Grand Canyon Trust on the Kaibab Plateau in 2005. The following year, we added 15 plots to each treatment and sampled a total of 44 plots in 2008 and 2009. Plots were circular with a 7.32-m radius and contained six transect lines that were permanently marked with nails (Fig. 2). There were two 1 × 1-m subplots (quadrats) placed along the right side of the 30°, 150° and 270° transects at 2.43 m and 4.57 m; and one quadrat at 6.1 m on the 90°, 210° and 310° transects. A 3.6-m radius circular subplot was marked in the centre of the circle for recording tree seedling recruitment.

We recorded estimates of the following variables in each of the nine quadrats: total plant canopy cover (foliar), plant canopy cover by species, bare soil, rock, wood, litter, duff, lichen, moss and scat. Plant canopy cover was further categorised and recorded by life cycle (annual, biennial, perennial) and growth form (graminoids, forbs, shrubs and trees). Estimates were made to the nearest percentage (0.5–100%) and mean plant canopy cover was calculated per plot. In the smaller, 3.6-m radius subplot we recorded conifer seedling density.

All sampling occurred during August and early September to capture the greatest plant production as a result of the monsoon rains. Species were classified on the generic or family level when vegetative characteristics were insufficient to identify at the species level. Plant nomenclature and nativity are based on the Plants Database (http:plants.usda.gov) and voucher specimens are located in the Deaver Herbarium, at Northern Arizona University in Flagstaff, Arizona.

Statistical analyses

We analysed all vegetation data using a permutational multivariate analysis of variance (PERMANOVA) (McCune and Mefford 1999; Anderson 2001; PC-ORD V 5.1); a non-parametric test that can be used with non-normal univariate or multivariate datasets. We conducted one-way analysis using Bray–Curtis distance measures for multivariate data and Euclidean distance for univariate data using 9999 permutations, with significance at $\alpha = 0.05$. Differences in species richness

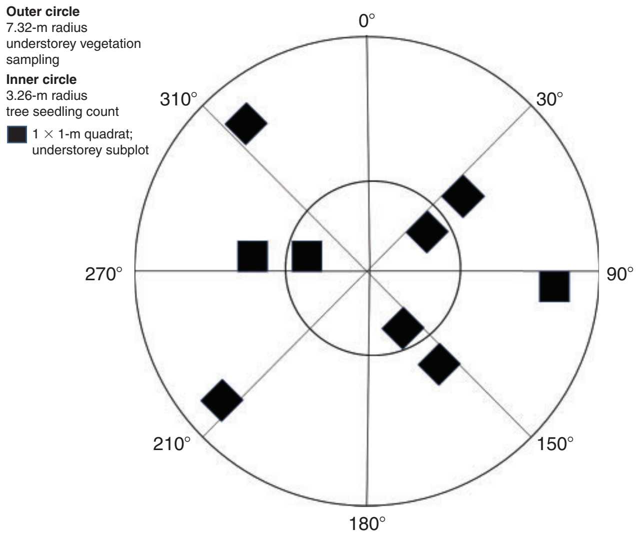

Fig. 2. Plot design for vegetation sampling. Understorey vegetation was collected in the full plot (7.32-m radius) and tree seedlings were sampled in the circular subplot (3.26-m radius). Cover estimates by species and substrates (rock, bare soil, litter, wood litter, duff, lichen and scat) were estimated in the 1 × 1-m plots.

were determined by the total number of species per plot per year in each treatment. Species that occurred in less than 5% of the plots were omitted from species composition analysis but included in species richness and univariate analysis (McCune and Grace 2002). To specifically test the rate of plant community change in the seeded and non-seeded plots, we used PC-ORD to calculate the Bray–Curtis distance for each plot. This method is similar to calculating the difference between two values in a paired *t*-test. The observed dissimilarity was then analysed with a PERMANOVA using Euclidean distance.

When differences in plant composition were identified in PERMANOVA analyses for 2008 and 2009, we ran an indicator species analysis (ISA) to determine which species were driving those differences (McCune and Mefford 1999; PC-ORD V 5.1). ISA takes into account both relative abundance and relative frequency. Species with an indicator value >30 and $P < 0.05$ were identified as indicator species (Dufrêne and Legendre 1997).

Results

Ryegrass and total vegetation cover; exposed bare soil and total ground cover

Results from our pilot study in 2007 revealed a mean canopy cover of ryegrass at 1.86%, but individual plots ranged in cover from 0.05–7.00%. By 2008 mean ryegrass cover per plot had decreased slightly to 1.35% with individual plots ranging from 0.05–13.00%, and in 2009 cover dropped to 0.5% with individual plots ranging from 0.05–3.00% (Table 1). The amount of ryegrass cover in 2009 was not significantly different from previous years ($P = 0.074$). Mean total plant canopy cover of all vegetation was not significantly different between seeded and non-seeded plots in any year and was less than 25% cover in both seeded and non-seeded plots in all years (Fig. 3).

Table 1. A summary of mean canopy cover values (standard error) and species richness for all plots within a treatment
Asterisks (*) indicate a significant difference ($P < 0.05$) between non-seeded and seeded plots within a year; t, <0.5% cover

Category	2007		2008		2009	
	Non-seeded	Seeded	Non-seeded	Seeded	Non-seeded	Seeded
Vegetative cover (%)						
exotic (non-native)	0.2(0.1)	1.9(0.8)	1.6(0.5)	0.2(0.1)	1.5(0.4)	0.5(0.1)
native	5.9(1.8)	3.6(1.0)	20.1(2.3)	15.9(1.8)	21.6(2.0)	19.0(1.5)
Community composition						
richness (avg)	28	27	29	29	33	32
Species (%)						
Bromus tectorum	t	0	0.6(0.3)*	0.1(0.0)*	t	t
Carex spp.	1.3(0.6)	0.2(0.0)	0.6(0.1)	0.6(0.2)	1.0(0.6)	0.9(0.2)
Elymus elymoides	t	t	t	t	t	t
Koeleria macrantha	t	t	t	t	t	t
Lolium perenne	–	1.9(0.1)*	–	1.4(0.6)*	–	0.5(0.1)*
Muhlenbergia montana	t	0	0.1(0.0)*	0*	0.1(0.1)*	0*
Poa fendleriana	1.1(0.5)	0.2(0.1)	0.7(0.2)	0.4(0.1)	1.6(0.3)*	0.8(0.2)*
Pinus ponderosa avg. per plot	n/a	n/a	1.1(0.4)*	0.1(0.1)*	1.1(0.0)	0.4(0.3)
Pinus ponderosa (# plots)	n/a	n/a	14.0*	2*	14	4
Abiotic variables (%)						
litter	31.2(9.7)	6.5(1.8)	33.0(3.6)*	18.6(3.0)*	46.1(2.8)*	28.8(3.5)*
rock	12.3(2.8)	19.8(5.7)	16.5(3.2)*	33.3(4.5)*	13.4(2.4)*	25.5(3.6)*
bare soil	23.5(7.1)	34.2(4.7)	20.4(3.1)	29.4(5.3)	9.1(1.7)*	19.1(3.0)*

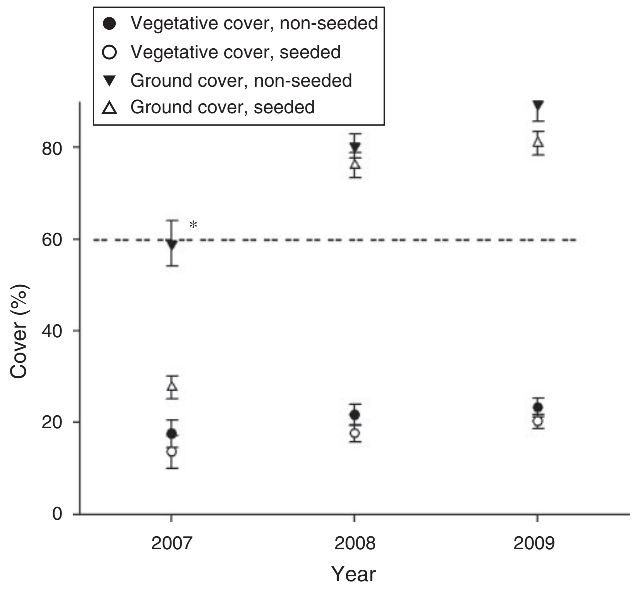

Fig. 3. Mean vegetative canopy cover (%) of all species by treatment and year, and total ground cover including litter, duff, rock, wood and vegetation in all 3 years in both treatments; the dashed line represents the target vegetative or ground cover of 60% that is associated with a lower risk of erosion runoff (Robichaud *et al.* 2000). Total ground cover was significantly different ($P < 0.05$) between seeded and non-seeded plots in 2007 (denoted with an asterisk (*)). Vertical bars represent +/− 1 standard error.

In 2007, the mean exposed bare soil was 23% in non-seeded plots and 34% in seeded plots and continued to decline in both treatments in 2008 and 2009 (Table 1). There was a significant difference between treatments only in 2009 with less exposed bare soil in the non-seeded plots. In 2007 total ground cover (including live vegetation, litter, duff, wood and rock) was significantly less in seeded (27%) than non-seeded (59%) plots (Fig. 3). By 2008 total ground cover for all plots was well above 60% (the threshold for decreased erosion potential), and the trend continued and was not significantly different between treatments after 2007 (Fig. 3).

Exotic plant cover

The mean plant canopy cover of exotic species other than Italian ryegrass was not significantly different between treatments in any year ($P = 0.89$, 0.89, 0.32). Overall, exotic cover was low; less than 2.0% on non-seeded plots, less than 1.0% on seeded plots (excluding Italian ryegrass) (Table 1) and less than 10.0% for all individual plots. Cheatgrass (*Bromus tectorum* L.) and musk thistle (*Carduus nutans* L.) were two exotic species of concern immediately following the fire. Cheatgrass was present in trace amounts on two plots in 2007, but in 2008 its presence and cover was significantly more in non-seeded plots ($P = 0.02$) (Table 1). In 2009 it was an indicator species for non-seeded plots, but canopy cover was not significantly different from seeded plots (Tables 1 and 2). Musk thistle was found on one non-seeded plot, and two other populations were observed but not on a plot. Other exotic plants that were present in more than 5% of the plots were orchardgrass (*Dactylis glomerata* L.), intermediate wheatgrass (*Thinopyrum intermedium* [(Host) Barkworth and D.R. Dewey] and prickly lettuce (*Lactuca serriola* L.).

Plant community composition and native plant cover

The plant community composition was significantly different between treatments in all years ($P = 0.003$, 0.0002, 0.0014).

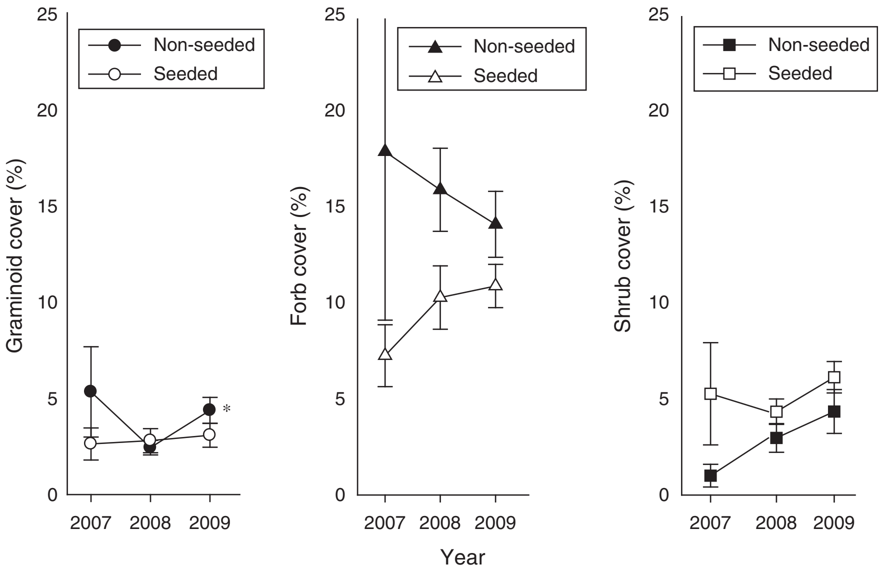

Fig. 4. Mean canopy cover (%) of life forms (graminoids, forbs and shrubs) by treatment and year. Graminoid cover was significantly different ($P < 0.05$) between seeded and non-seeded plots in 2009 (denoted with an asterisk (*)). Vertical bars represent +/− 1 standard error.

The rate of species composition change between seeded and non-seeded plots was not significantly different between 2008 and 2009; thus, the floristic composition was changing at a relatively consistent rate in both treatments. Species richness did not differ between treatments in any year (Table 1).

We tested for differences in mean canopy cover of native plants based on growth form, life cycles and specific family characteristics. In 2009 there was significantly more graminoid cover in non-seeded than in seeded areas ($P = 0.03$) (Fig. 4). A more detailed examination showed non-seeded plots had significantly greater cover of mountain muhly (*Muhlenbergia montana)* [(Nutt.) Hitchc] in 2008 and 2009; and of muttongrass and squirreltail in 2009 (Table 1). We also detected significantly higher cover of annual and biennial forbs in non-seeded plots in 2008 but not in any other year (Fig. 5). There were significantly more ponderosa pine seedlings in non-seeded plots in 2008 ($P = 0.002$) and a significant difference in the number of plots with tree seedlings: 14 in non-seeded plots and 2 in seeded plots (Table 1). We observed no significant difference in cover of forbs, shrubs, perennials, total native plant canopy cover or leguminous plants between treatments in any year (Table 1, Figs 3 and 4).

The ISA identified several species strongly associated with treatments in both years. Twelve species were listed as indicators for non-seeded plots in 2008 with the notable presence of two tree species, ponderosa pine and quaking aspen; two common exotics, dandelion (*Taraxacum officinale* F.H.Wigg) and mullein (*Verbascum thapsus* Bertol.); and one invasive exotic, cheatgrass. Significant species in the seeded plots were two legumes, Wright's deervetch (*Lotus wrightii* (A. Gray) Greene) and crescent milkvetch (*Astragalus amphioxys* A. Gray); one shrub, New Mexico locust (*Robinia neomexicana* A.Gray) and the seeded species, ryegrass (Table 2). In 2009 fourteen species were closely associated with non-seeded plots with the same species listed above and the addition of three

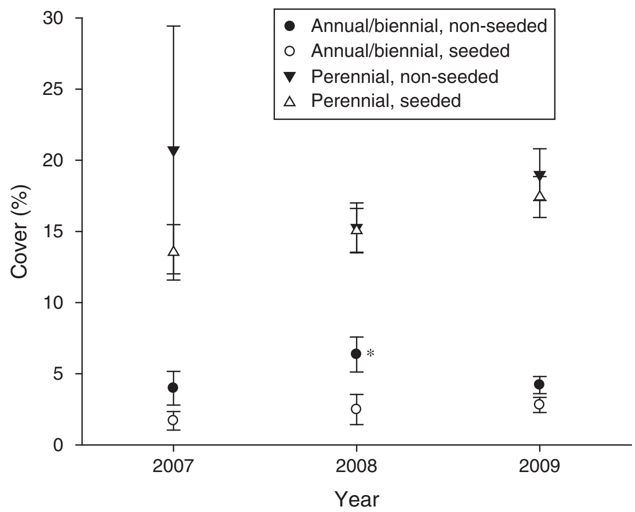

Fig. 5. Mean canopy cover (%) of species by life history: annual and/or biennial *v.* perennial. Annual/biennial cover was significantly different ($P < 0.05$) between seeded and non-seeded plots in 2008 (denoted with an asterisk (*)). Vertical bars represent +/− 1 standard error.

native bunchgrasses: muttongrass, squirreltail and mountain muhly. Seeded plots maintained the same species and added two more weedy species: horseweed (*Conyza canadensis* (L.) Cronquist) and the exotic invasive, yellow salsify (*Tragopogon dubius* Scop.) (Table 2).

Discussion

There are limits to the inferences that can be drawn from this study on post-fire seeding with Italian ryegrass in the Warm Fire, due to the inherent nature of studies on wildfires.

Table 2. Indicator species for 2008 and 2009

Only species that were significant indicators (IV >30, $P < 0.05$) are listed in this table (IV is the indicator value for each indicator species); P, perennial; A, annual; B, biennial; G, grass; F, forb; S, shrub, T, tree

Treatment	Species	IV 2008	IV 2009	Life cycle	Growth form
Non-seeded	*Antennaria parvifolia*	45.4	66.1	P	F
	Androsace septentrionalis	41.4	–	A	F
	Arenaria lanuginosa ssp. *saxosa*	40.8	48.5	P	F
	*Bromus tectorum**	56.9	61.4	A	G
	Chenopodium fremontii	79.1	67.6	A	F
	Elymus elymoides	–	76.5	P	G
	Hieracium fendleri	33.7	38.9	B	F
	Lupinus argenteus	–	57.0	P	F
	Muhlenbergia montana	–	31.8	P	G
	Packera multilobata	63.8	71.7	A/P	F
	Poa fendleriana	–	66.8	P	G
	Pinus ponderosa	56.0	53.5	P	T
	Populus tremuloides	52.6	72.0	P	T
	Pseudognaphalium macounii	59.9	–	B	F
	*Taraxacum officinale**	61.6	70.9	A/B	F
	*Verbascum thapsus**	36.4	40.9	B	F
Seeded	*Astragalus amphioxys*	61.3	–	P	F
	Conyza canadensis	–	58.7	A	F
	Erigeron divergens	–	86.0	B	F
	Hymenopappus filifolius	–	39.9	P	F
	Lolium perenne ssp. *multiflorum**	98.5	99.5	A/B/P	G
	Lotus wrightii	75.0	58.0	P	F
	Robinia neomexicana	57.0	59.1	P	S
	*Tragopogon dubius**	–	46.9	A/B	F

*exotic.

Specifically, the sizes of the high-severity burn patches were different in WFU and wildfire areas, and there were much larger patches of high severity in the wildfire section, which could influence vegetation recovery. Ideally we would have begun collecting data in all plots during the first post-fire season to capture quantifiable data regarding the actual germination and establishment of ryegrass. There is also a lack of independence as the application of seed was determined by the management status of the fire, and we regret the inability to more precisely control for environmental factors. Ours is a short-term study, and ideally studies of this nature should be continued over decades to determine if the differences in plant community composition persist or gradually converge over time.

Many studies have failed to produce evidence demonstrating that seeding with non-natives is effective (Robichaud *et al.* 2000; Barclay *et al.* 2004; Peppin *et al.* 2010; Stella *et al.* 2010). Successful post-fire seeding is particularly difficult in the Southwest where high-intensity summer monsoon rainstorms directly follow the fire season, and much of the seed washes off the steepest slopes before it is able to establish (Robichaud *et al.* 2000). On the Warm Fire the ryegrass did not have time to establish before monsoon rains and much of the seed washed away.

In our study, seeding was not successful at effectively increasing vegetation cover. There was no significant difference in the total vegetative cover between seeded and non-seeded sites in any year. There was however a significant difference in total ground cover 1 year post-fire which includes vegetation, and the non-seeded sites had higher total cover. Vegetation cover of 30%, has been shown to be partially effective at reducing erosion in other studies (Beyers, 2004) but cover in our plots 1 year post-fire was well below that threshold.

We found no significant difference between treatments in exotic plant cover and it appears that seeding had little to no effect on non-native plants. In an evidence-based review of post-fire seeding studies, Peppin *et al.* 2010 reported that 60% of the reviewed studies indicated seeding was ineffective at controlling exotics, mostly due to inadequate cover of the seeded species. On the Warm Fire, ryegrass cover was inconsistent and relatively low in all treated plots. Seeding with native species may produce a working alternative to using exotics (Floyd *et al.* 2006).

The floristic composition of seeded and non-seeded plots was significantly different but the source of these differences is unclear. Pre-fire conditions can carry through and dilute the effects on seeding when cover of seeded species is below 10% (Conard *et al.* 1991; Barclay *et al.* 2004; Stella *et al.* 2010). Many of the post-fire colonisers found in ponderosa pine forests are adapted to conditions of high disturbance but the added stress of the presence of ryegrass could prevent those species from establishing (Springer *et al.* 2001; Laughlin *et al.* 2004).

Ponderosa pine seedlings were more abundant in non-seeded plots, and seedlings were present in more non-seeded plots. However, it is challenging to make strong assertions due to the variability of the fire mosaic across the landscape. At the plot level, there was no relationship between the presence of nearby seed trees and seedling establishment, but we did observe rodent activity on one seeded plot that had more than 10 ponderosa seedlings. We looked for relationships at the landscape scale between cover of ryegrass, environmental variables (slope, aspect and litter cover), and seedlings and found no strong associations, in part because of the high variability in the data.

Conclusion

Our data suggest that post-fire seeding with a non-native grass was ineffective at providing adequate ground cover to mitigate erosion or control exotic invasion. The maintenance and rehabilitation of damaged watersheds is an important component of post-fire treatments and alternative actions warrant more research and implementation (Robichaud *et al.* 2000; Beyers 2004; Peppin *et al.* 2010). Innovative solutions using wood shred mulching with ground-up hazard trees, aerially applied to high-burn severity areas, have shown great potential (Robichaud *et al.* 2013). Seeding with native and local genotypes may provide a more ecologically appropriate solution to post-fire rehabilitation, but studies continue to report varying results (Floyd *et al.* 2006; Stella *et al.* 2010). Seeding itself may be entirely unwarranted because the positive effects of fire such as higher nutrient levels and increased light exposure often lead to rapid growth of vegetative cover (Keeley *et al.* 2006; Huisinga *et al.* 2005; Stella *et al.* 2010). This study adds to the growing body of evidence that seeding with non-native grasses is often ineffective, in that there was no positive effect on total plant cover.

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