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Impacts of erosion control treatments on native vegetation recovery after severe wildfire in the Eastern Cascades, USA

Erich K. Dodson[A,C], *David W. Peterson*[A] and *Richy J. Harrod*[B]

[A]US Forest Service, Pacific Northwest Research Station, 1133 North Western Avenue, Wenatchee, WA 98801, USA.
[B]US Forest Service, Okanogan–Wenatchee National Forest, 215 Melody Lane, Wenatchee, WA 98801, USA.
[C]Corresponding author. Email: edodson@fs.fed.us

Abstract. Slope stabilisation treatments like mulching and seeding are used to increase soil cover and reduce runoff and erosion following severe wildfires, but may also retard native vegetation recovery. We evaluated the effects of seeding and fertilisation on the cover and richness of native and exotic plants and on individual plant species following the 2004 Pot Peak wildfire in Washington State, USA. We applied four seeding and three fertilisation treatments to experimental plots at eight burned sites in spring 2005 and surveyed vegetation during the first two growing seasons after fire. Seeding significantly reduced native non-seeded species richness and cover by the second year. Fertilisation increased native plant cover in both years, but did not affect plant species richness. Seeding and fertilisation significantly increased exotic cover, especially when applied in combination. However, exotic cover and riclmess were low and treatment effects were greatest in the first year. Seeding suppressed several native plant species, especially disturbance-adapted forbs, Fertilisation, in contrast, favoured several native under storey plant species but reduced tree regeneration. Seeding, even with native species, appears to interfere with the natural recovery ofnative vegetation whereas fertilisation increases total plant cover, primarily by facilitating native vegetation recovery.

Additional keywords: BAER, burned area emergency response, diversity, exotic invasion, native species, post-fire rehabilitation, species richness.

Introduction

Post-fire management activities in forest and rangeland ecosystems often focus on urgent, near-term objectives - such as controlling erosion and flooding and protecting human health and property - rather than the long-term recovery of ecosystem structure and function, High-severity fires temporarily increase erosion and runoffpotential by consuming vegetation and organic matter, exposing mineral soil, and increasing soil water repellency (DeBano *et al.* 1998; Benavides-Solorio and MacDonald 200 I; Wondzell and King 2003). Emergency slope stabilisation treatments (e.g. seeding and mulching) are often applied after wildfires to accelerate development of organic soil cover and reduce soil erosion and flooding during high-intensity rainfall events (Robichaud *et al.* 2000; Beyers 2004). However, slope stabilisation treatments can also impede native vegetation recovery (Schoennagel and Waller 1999; Barclay *et al. 2004;* Beyers 2004; Keeley 2004), reduce tree regeneration (Keeley 2004; Kruse *et al.* 2004) and introduce exotic species (Barclay *et al.* 2004; Kruse *et al.* 2004), raising questions about their net benefits. The purpose of the present study was to assess the effects of two slope stabilisation treatments - seeding and fertilisation- on native vegetation recovery and exotic species cover and richness following the Pot Peak Wildfire in the eastern Cascade Mountains of Washington State, USA.

Seeding is a common slope stabilisation treatment intended to rapidly increase plant cover by encouraging the establishment and growth of fast-growing species (Robichaud *et at. 2000).* Seeding can also be used after wildfires to retard the spread of noxious weeds by increasing plant competition for resources (Floyd *et al.* 2006). Seeding is not without its drawbacks, however. Some species that are effective for increasing plant cover after wildfire have been abandoned because they proved too persistent (Beyers 2004, and citations therein). Non-persistent cereal grains such as wheat and barley have proved effective at increasing cover on some sites, but can disrupt native plant community recovery when seeded species achieve high cover (Schoennagel and Waller 1999; Keeley 2004). Seeding with native species has been proposed (Richards *et al.* 1998; Beyers 2004), but more information is needed about the effectiveness and impacts of seeding native species compared with cereal grains.

In some cases, development of vegetative cover following wildfire may be limited by nutrient availability. Volatilisation of soil nitrogen during high-severity wildfires may increase nitrogen limitations on plant growth in ecosystems that are already nitrogen-limited. By increasing nutrient availability, fertilisation may increase plant productivity and cover following severe wildfires (Baird el *al.* 1999), thereby reducing erosion potential, Fertilisation has been most frequently used as a complementary treatment

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with seeding to increase the growth of seeded species (e.g. Tiedemann and Klock 1973; Klock *et al.* 1975; Amaranthus 1989), whereas its use as an independent land-surface treatment has been quite limited (Robichaud *et al.* 2000). Previous studies suggest that fertilisation may help increase plant cover after wildfires (Tiedemann and Klock 1973; Robichaud *et al. 2006),* but potential effects on species richness and community dynamics have not been fully investigated

Exotic invasion is increasingly being recognised as a threat to ecosystem recovery following wildfire (Floyd *et at. 2006;* Freeman *et al.* 2007). Exotic species invasion can reduce the abundance and survival of native species, alter community dynamics, and modify environmental conditions (Vitousek *et al.* 1996; Mack *et al.* 2000; Houlahan and Findlay 2004). Seeding can introduce exotic species if contaminated seed is used (Allen *et al.* 2002; Barclay *et al.* 2004), but can also retard exotic species establishment and spread by capturing growing space and resources (Beyers 2004; Keeley 2004). Fertilisation is unlikely to introduce exotic species, but increased resource availability has been linked to exotic invasion (Huenneke *et al.* 1990; Davis *et al.* 2000), and may increase the ability of exotic species to compete with natives (Kolb *et al.* 2002; Brooks 2003).

In the present study, we examined the effects of seeding and fertilisation treatments the first 2 years following wildfire. We asked three basic questions:

1. Did treatments alter the cover or species richness of native vegetation?
2. Did treatments introduce exotic species to plant communities or alter exotic species cover?
3. Did treatments alter the relative cover and frequency of individual plant species in post-fire plant communities?

Fertilisation and seeding resulted in small to moderate increases in ground cover (the primary objectives for the treatments), but we limit our discussion to treatment effects on ecological variables as effects on ground cover are discussed elsewhere (Peterson *et al. 2009).*

Methods

Site description

The study area was near the south-western shore of Lake Chelan in north central Washington State, USA, east of the crest of the Cascade Mountains (4r55'N, 120°18'W). The Pot Peak-Sisi Ridge wildfire complex burned 19000ha (47000 acres) of coniferous forest in this area in the summer of 2004, with the Pot Peak fire burning 7000 ha. Post-fire assessments concluded that 90% of the burned area was at high risk for soil erosion owing to fire effects, soil properties, and steep topography, even though only 45% of the area burned at moderate to high severity with regard to soil impacts. Much of the area burned by the Pot Peak fire consisted of young forest stands that developed after a large wildfire in 1970. Forests within the study area were dominated by ponderosa pine *(Pinus ponderosa* C. Lawson) and Douglas-fir *(Pseudotsuga menziesii* (Mirbel) Franco) at lower elevations and by lodgepole pine *(Pinus contorta* Dougl. ex Loud.), Douglas-fir, and subalpine fir *(Abies lasiocarpa* (Hook). Nutt.) at higher elevations. Soils in the area are primarily gravelly sandy loams of the Palmich series (NRCS 2007) with parent material of volcanic ash and pumice deposited over colluvium or glacial till derived from granodiorite or rhyolite.

The regional climate has warm, dry summers and cold, relatively wet winters. Summer thunderstorms occur periodically and can produce intense rainfall. Mean annual precipitation estimates for the eight sites range from 40 to 57 ern, with an overall mean of 48 em (PRISM Group, Oregon State University, see http://www.prismclimate.org accessed 11 May 2010). Only 22% of annual precipitation falls during the May-August period (PRISM Group). Precipitation at the nearest weather station (Lakeside, ~ 1000m lower and 22 km to the south-east; Western Regional Climate Center, see http://www.wrcc.dri.edulclimsum.htrnl, accessed 5 May 2009) was above average during the study period, with 116 and 146% of historical values in 2005 and 2006 respectively. Growing season (May-August) precipitation was above average at Lakeside during the study period, with 213 and 119% of historical values in 2005 and 2006 respectively (Western Regional Climate Center).

We used fire severity maps (based on soil effects) to identify potential study sites within the perimeter of the Pot Peak fire. Selection criteria for potential study sites included areas that were burned at moderate or high severity and had reasonable access from existing road systems. From the pool of potential sites, we chose eight study sites that encompassed a wide range of environmental conditions, including elevation, aspect, pre fire vegetation, and fire severity (Table 1). Each site had complete overstorey tree mortality. At each site, we established a grid of

Table 1. Site characteristics for Pot Peak Study, including topographic setting (slope, aspect, and elevation), fire severity (based on soil effects), mulching status, and soil properties

Site	Slope (%)	Aspect (°)	Elevation (m)	Fire severity	Mulch[A]	Soil bulk density (g cm^{-3})	Soil pH	Soil organic matter (%)	Soil fraction <2 mm (%)
Hug me	35	280	1196	Moderate	No	0.87	6.8	3.5	67.7
Mouse	47	305	1221	High	Yes	0.80	6.8	3.8	79.4
Rainbow	57	360	1297	High	Yes	0.80	6.6	4.4	83.8
Stairway	45	325	1313	Moderate	No	0.84	7.1	3.5	91.0
Beast	68	90	1321	High	Yes	0.87	6.9	4.0	79.9
Big tree	45	320	1380	High	Yes	0.81	6.8	3.4	45.6
Nice view	12	20	1393	Moderate	No	0.85	7.1	3.4	88.6
Squirrelly	43	345	1507	High	No	0.80	6.7	3.8	87.2

[A]Mulch was operationally applied to the site.

Table 2. Scientific and common names, lifeform and origin of seeded species used in the Pot Peak study

Scientific name	Common name	Lifeform	Origin
Warm mix			
Achillea millefolium	Common yarrow	Forb	Native
Elymus wawawaiensis	Snake River wheatgrass	Grass	Native
Festuca ovina[A]	Sheep fescue	Grass	Exotic
Poa secunda[A]	Sandberg bluegrass	Grass	Native
Cool mix			
Chamerion angustifolium	Fireweed	Forb	Native
Elymus lanceolatus	Thick-spike wheatgrass	Grass	Native
Festuca idahoensis[A]	Idaho fescue	Grass	Native
Wheat			
Triticum aestivum	Winter wheat (Eltan)	Grass	Exotic

[A]Owing to difficulties in identification of young plants, *Festuca* and *Poa* species were identified to genera only.

96 study plots, attempting to minimise environmental variability among plots. Study plots were 40 m^2 each (4 x 10 m, with the short axis oriented along slope contours). We left an untreated 2-m buffer between study plots to reduce treatment contamination among plots.

Treatments

At each site, we applied four seeding treatments and three levels of fertilisation to study plots using a completely randomised factorial design (12 treatment combinations). Seeding treatments included a 'warm' species mix with three graminoids and one forb that were expected to do well on wanner and chief sites; a 'cool' seed mix with two graminoids and one forb species expected to perform better on more mesic sites; a monoculture of soft white winter wheat (Eltan; *Triticum aestivum* L.), the standard operational seeding treatment for this area; and a control with no seeding. We used mostly native species (Table 2), but we did not require seeds to be from local seed sources owing to time and availability constraints. We designed the 'warm' and 'cool' seeding treatments to provide an average of 646 seeds m^{-2} (60 seeds *fooC*^{2}) and the winter wheat treatment to provide an average of 162 seeds m^{-2} (15 seeds *fooC*^{2}), with roughly equal proportions of all species in the mixes (based on estimated seed counts). This was more than the local standard for operationally applying winter wheat, which called for only 65 live seeds m^{-2}.

The fertilisation treatments consisted ofa mixture of ammonium nitrate and ammonium sulfate (30-{)-0-6) at quantities calculated to provide 0 kg, 56 kg (low dosage) or 112 kg (high dosage) ofN ha^{-1} (0, 50 or 100 Ib $acre^{-1}$). The local operational treatment called for 56 kg N ha^{-1} We applied fertiliser and seed with a hand-held spreader, attempting to achieve a relatively uniform application rate, but did not quantify seed densities in the field. We applied treatments in the spring following the wildfire, shortly after snowmelt. Spring application is typical for fertilisation treatments, but seeding treatments are often applied in the fall (autumn), after wildfires are extinguished but before snowfall.

We randomly assigned treatments to plots so that there would be eight replicates of each of the 12 possible seeding and fertilisation combinations at each site (Fig. I). Implementation errors at two sites led to as many as 10 replicates or as few as seven replicates for some treatment combinations. However, each fertilisation level was replicated 32 times and each seeding treatment was replicated 24 times at each site.

Four of the study sites also received an aerial application of wheat straw mulch as part of operational erosion control efforts. Helicopters dropped loose bales of straw from elevations of 200-300 m with the goals of achieving good dispersal and even cover of straw (actual cover averaged <15%). Mulching operations were started in the fall of2004 and completed shortly after snowmelt in spring 2005; in most cases, mulching preceded our treatment applications.

Field sampling

We measured plant cover and species richness near the peak of the growing season in July and August of2005 and 2006. At each plot (40 m^2), we first estimated percentage cover for bare ground (exposed soil), straw mulch, other plant litter, coarse woody debris, cryptogams and rock. We then identified and recorded all vascular plant species present and estimated relative cover for each species so that total plot cover (plants plus other variables) summed to 100%. We based all cover estimates on visual assessments of what a vertically falling raindrop would hit first, Where plant species overlapped, we attributed cover to the taller species. Similarly, plant cover took precedence over other cover.

Statistical analyses

Prior to analysis, we chose an ıx value of 0.05 as statistically significant. Our experimental design was a generalised randomised block design (Hinkelmann and Kempthome 1994) with replication of treatment combinations within blocks. We analysed treatment effects on native vegetation cover and richness and exotic species cover and richness using mixed statistical models (*SAS PROC MIXED;* Littell *et al.* 1996). Seeding, fertilisation, year, and their interactions were included as categorical fixed effects in the model. The study unit (site) and site-treatment interactions were included as random factors in the model. We also included the individual plot (within a site) as a random effect to account for positive correlations among repeated measurements on the same plot.

We evaluated the potential influence of mulch cover on vegetation responses by including mulch cover (linear and quadratic terms) and mulch cover interactions with seeding and fertilisation treatments as potential covariate terms in our statistical models. For each response variable, we initially fitted a full model with all potential fixed effects, covariates, and random site effects included. We then eliminated non-significant mulch covariate and random effects ($P > 0.05$) using a backward elimination process to arrive at the final model. We retained the fixed treatment effects (seeding, fertilisation, and their interaction) in all models to match the experimental design. Where significant fixed effects were found, we performed post-hoc pairwise comparisons (with a Tukey adjustment for multiple comparisons) to assess differences among seeding and fertilisation treatment main effects and their interactions.

Exotic species were infrequent in this study (several plots with none), leading to moderate violations of statistical assumptions that were not correctable with common transformations.

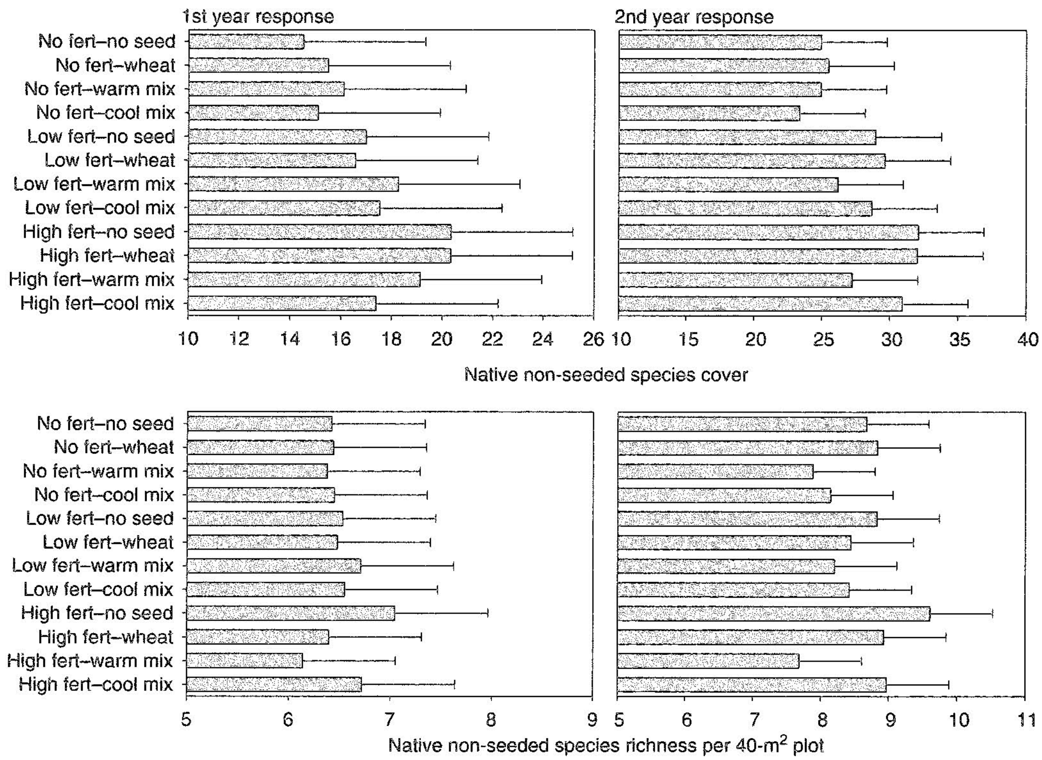

Fig. 1. Least square means (plus one standard error) for seeding and fertilisation (fert) treatment effects on native non-seeded species cover and richness in the first and second post-fire years.

We present results based on analysis of untransformed data, relying on the robustness of analysis of variance procedures to moderate deviations from normality in data, especially given our large sample sizes ($n = 768$ plots).

We employed indicator species analysis (lSA; Dufrene and Legendre 1997), as implemented in *PC-ORD,* version 5 (McCune and Mefford 1999), to identify species that responded positively or negatively to seeding and fertilisation treatments. We performed separate pairwise analyses with each seeding treatment compared with the non-seeded control. We ran the same analysis comparing each of the two levels of fertilisation with the unfertilised control. ISA produces an indicator value (IV) that ranges from 0 to 100 based on the relative cover and frequency of a species in one group (treatment; Dufrene and Legendre 1997; McCune and Grace 2002). We ran Monte Carlo tests with 5000 randomisations to calculate the probability *(P)* of obtaining an equal or larger indicator value with data randomised among treatments. Species were considered significant indicators of a treatment if the indicator value was at least 15 and the probability of obtaining the indicator value with randomised data was low ($P < 0.05$).

Results

We identified a total of III species across all study sites during the first 2 post-fire years. Non-seeded species comprised the majority of plant cover during the first 2 years following the wildfire (Table 3). Shrub species provided the majority of native non-seeded species cover.

Native non-seeded species

Native non-seeded species cover increased from an average of 17% in the first post-fire year to 28% the second year, averaged across all treatments. Cover of native non-seeded species varied significantly *(P=0.031)* among the eight sites, with average cover between sites differing by as much a841% (11.5 to 52.3%) by the second year.

Seeding and fertilisation effects on native non-seeded species cover varied significantly among the two years of the study (Table 4). Seeding did not significantly alter native non-seeded species cover the first year (Fig. I). In the second year, however, the warm mix seeding treatment significantly reduced native non-seeded species cover relative to the control Cno seed') and wheat seeding treatments (both $P < 0.03$). The high dosage of fertilisation increased native non-seeded species cover by an estimated 6% ($P < 0.001$) in the second year when averaged across all seeding treatments and by more than 7% ($P <$ 0.00 I) in the absence of seeding (Table 3). Post-hoc tests revealed that both levels of fertilisation significantly increased native non-seeded species cover relative to the unfertilised plots in both years (all $P < 0.03$) and that the high dosage of fertilisation increased cover significantly more than the low dosage in both years (both $P < 0.04$).

Table 3. Percentage cover of native non-seeded species, seeded species, exotic species and the most abundant individual species in each treatment in each year

Fertilisation categories are: N, no fertilisation; L, low fertilisation; H, high fertilisation

Seeding	None			Wheat			Warm			Cool		
Fertilisation	N	L	H	N	L	H	N	L	H	N	L	H
Year 1												
Native non-seeded species												
Total cover	14.5	17.0	20.4	15.1	16.6	20.7	15.7	18.8	19.0	15.9	16.9	17.1
Salix scouleriana	5.4	5.8	6.7	5.3	5.6	7.5	5.2	6.7	6.9	5.8	6.7	6.8
Spiraea betulifolia	4.1	6.1	7.0	3.9	6.6	7.3	4.5	6.2	5.8	4.3	5.2	5.3
Ceanothus velutinus	1.8	1.4	2.0	2.1	0.9	1.5	2.6	1.7	2.3	1.8	1.8	1.4
Alnus viridis	1.2	1.2	0.9	1.1	1.1	1.2	1.1	1.1	1.1	1.2	1.0	1.2
Seeded species												
Total cover	2.8	4.5	5.8	2.9	5.6	6.2	3.5	8.6	8.7	2.8	5.7	5.2
Achillea millefolium	0.1	0.1	0.3	0.1	0.1	0.1	1.7	4.4	4.6	0.1	0.2	0.2
Chamerion angustifolium	2.2	3.7	4.8	2.2	4.2	4.7	1.1	3.1	3.1	2.0	4.1	3.6
Triticum aestivum	0.5	0.7	0.6	0.6	1.1	1.4	0.3	0.6	0.6	0.5	0.9	0.7
Exotic species												
Total cover	0.1	0.3	0.2	0.1	0.2	0.3	0.2	0.4	0.5	0.2	0.4	0.4
Year 2												
Native non-seeded species												
Total cover	25.0	29.0	32.1	25.1	29.7	32.5	24.5	26.7	27.1	24.2	28.1	30.7
Salix scouleriana	9.2	10.9	12.4	8.7	11.6	14.0	9.3	10.1	10.1	8.8	11.6	12.6
Spiraea betulifolia	5.9	8.5	8.9	6.3	9.5	9.5	6.1	7.3	7.3	5.7	7.0	9.0
Ceanothus velutinus	4.5	3.7	4.5	4.2	3.0	3.6	5.1	3.7	4.5	4.5	4.2	3.7
Alnus viridis	2.3	2.7	2.2	2.7	2.2	2.2	1.8	2.2	2.2	2.4	2.4	2.2
Seeded species												
Total cover	4.3	7.1	8.1	5.1	7.1	6.5	7.7	17.0	20.2	4.8	7.7	8.5
Achillea millefolium	0.1	0.4	0.7	0.3	0.4	0.4	4.7	10.9	15.0	0.3	0.6	1.0
Chamerion angustifolium	4.1	6.6	7.2	4.6	6.3	5.8	2.2	4.5	3.7	3.6	5.6	5.5
Triticum aestivum	0.1	0.1	0.1	0.2	0.2	0.2	0.1	0.1	0.1	0.1	0.2	0.1
Exotic species												
Total cover	0.1	0.2	0.1	0.1	0.1	0.2	0.0	0.1	0.1	0.2	0.3	0.3

Table 4. Mixed model Type III tests of fixed effects for each response variable

Num d.f., numerator degrees of freedom; Den d.f., denominator degrees of freedom. Significant effects are in bold

Effect	Num d.f.	Den d.f.	*F* value	*P*	Effect	Num d.f.	Den d.f.	*F* value	*P*
Native non-seeded species cover					Native non-seeded species richness				
seed	3	749	1.3	0.2749	seed	3	749	5.44	**0.0010**
fertilisation (fert)	2	749	27.3	**<0.0001**	fert	2	749	1.87	0.1555
seed × fert	6	749	0.9	0.4729	seed × fert	6	749	1.96	0.0693
year	1	756	1131.9	**<0.0001**	year	1	7	59.13	**0.0001**
seed × year	3	756	6	**0.0005**	seed × year	3	749	6.63	**0.0002**
fert × year	2	756	3.6	**0.0289**	fert × year	2	749	1.64	0.1952
seed × fert × year	6	756	1.7	0.1191	seed × fert × year	6	749	0.39	0.8867
Exotic species cover					Exotic species richness				
seed	3	21	1.49	0.2472	seed	3	28.6	23.54	**<0.0001**
fert	2	14	4.96	**0.0235**	fert	2	1395	25.39	**<0.0001**
seed × fert	6	714	0.65	0.6914	seed × fert	6	1394	3.73	**0.0011**
year	1	756	39.2	**<0.0001**	year	1	7	0.01	0.9363
seed × year	3	756	6.03	**0.0005**	seed × year	3	29	16.12	**<0.0001**
fert × year	2	756	3.68	**0.0256**	fert × year	2	709	1.65	0.1931
seed × fert × year	6	756	0.74	0.6215	seed × fert × year	6	708	2.96	**0.0073**
					mulch	1	1063	9.60	**0.0020**

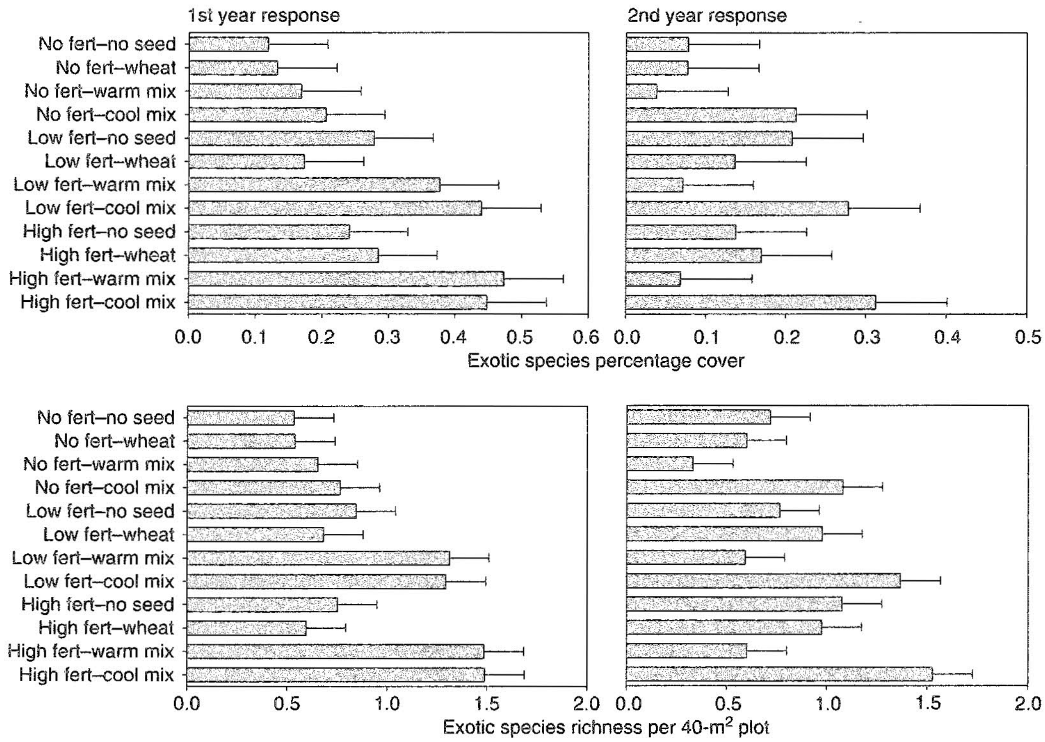

Fig. 2. Least square means (plus one standard error) for seeding and fertilisation (fert) effects on exotic species cover and richness in the first and second post-fire years.

Native non-seeded species richness varied significantly in response to seeding treatments (Table 4) and among sites ($P = 0.034$). The first year after fire, post-hoc comparisons revealed no significant differences among seeding treatments (all $P > 0.5$). In the second year, the warm seed mix treatment significantly reduced species richness by 1.1 species ($P < 0.05$) and the cool seed mix treatment reduced species richness by 0.5 species ($P < 0.05$) relative to the 'no seed' treatment (Fig. I). The warm seed mix treatment also produced significantly lower native non-seeded species richness relative to the wheat treatment ($P = 0.0005$). Fertilisation did not significantly influence native non-seeded species richness.

Exotic species

Exotic plant species cover was generally low, with total exotic plant cover averaging less than 0.5% across all treatments in both years (Fig. 2). Seeding and fertilisation both significantly influenced exotic plant cover, with effects varying by year (Table 4). In the first year after fire, seeding with the warm ($P = 0.032$) and cool ($P = 0.006$) mixes significantly increased exotic cover relative to no seeding in pairwise comparisons, but seeding wheat did not ($P = 0.98$). In the second year, seeding with the cool mix significantly increased exotic plant cover relative to no seeding ($P = 0.032$), but seeding with the warm mix no longer had any effect ($P = 0.29$). Both levels of fertilisation significantly increased exotic plant cover relative to no fertilisation in the first year (both $P < 0.00$ I), but not the second year (both $P > 0.15$). The effects of seeding ($P < 0.01$) and fertilisation ($P = 0.05$) both varied significantly among the eight sites, but site differences in mean exotic plant cover were generally small.

Exotic plant species richness varied by seeding and fertilisation treatments, with treatment effects also varying among years (Table 4; Fig. 2). In the first year after fire, fertilisation (low or high) combined with the cool or warm mix seeding treatments significantly increased exotic species richness relative to the control treatment (all $P < 0.00$ I). In the second year, the cool mix seeding treatment combined with high ($P < 0.001$) or low ($P = 0.014$) levels offertilisation significantly increased exotic plant species richness relative to the control, but the warm seed mix treatment had no significant effect (all $P > 0.50$). Fertilisation alone did not significantly influence exotic species richness in either year (all $P > 0.30$). Mulching increased exotic species richness (Table 4), with predicted exotic species richness increasing by 0.1 species per plot for each additional 10% mulch cover.

Individual species

In the pairwise indicator species analyses, comparing the warm seeded treatment with no seeding (control) revealed the greatest number of indicator species. Three of the four species seeded in the warm mix were significantly indicative of plots where they

Table 5. Indicator species analysis for each seeded treatment compared with the plots without seeding
IV is the indicator value and *P* is the probability of obtaining as high an IV with data randomised among treatments 5000 times in a Monte Carlo procedure. Only significant indicator species are shown. NS, not significant

Species	Nativity	Life-form	2005			2006		
			Seed	IV	*P*	Seed	IV	*P*
Cool *v.* none								
Bromus tectorum	Exotic	Graminoid	Cool	52.9	<0.001	Cool	43.1	<0.001
Elymus lanceolatus[A]	Native	Graminoid	Cool	51.1	<0.001	Cool	66.7	<0.001
Elymus wawawaiensis	Native	Graminoid			NS	Cool	20.0	<0.001
Festuca spp.[A]	Both	Graminoid	Cool	54.3	<0.001	Cool	78.7	<0.001
Warm *v.* none								
Achillea millefolium[A]	Native	Forb	Warm	88	<0.001	Warm	94.9	<0.001
Chamerion angustifolium	Native	Forb	None	54.2	0.009	None	58.7	<0.001
Bromus tectorum	Exotic	Graminoid	Warm	18.7	<0.001			NS
Elymus wawawaiensis[A]	Native	Graminoid	Warm	55.2	<0.001	Warm	79.9	<0.001
Epilobium brachycarpum	Native	Forb			NS	None	17.6	<0.001
Festuca spp.[A]	Both	Graminoid	Warm	61.7	<0.001	Warm	86.8	<0.001
Sisymbrium altissimum	Exotic	Forb	Warm	33.3	<0.001			NS
Unknown grass		Graminoid	Warm	19.3	<0.001			NS
Lactuca serriola	Exotic	Forb			NS	None	25	<0.001
Pseudognaphalium canescens	Native	Forb			NS	None	54.3	<0.001
Anaphalis margaritacea	Native	Forb			NS	None	15.2	0.011
Pinus contorta	Native	Tree	None	43.8	0.013			NS
Wheat *v.* none								
Triticum aestivum[A]	Exotic	Graminoid	Wheat	57.3	<0.001	Wheat	54.3	<0.001

[A]Species seeded in the seed mix being compared with the unseeded control.

were seeded in both 2005 and 2006 (*Achillea millefolium* L., *Elymus wawawaiensis* J. Carlson & Barkworth., *Festuca* spp. L.; Table 5). Also, two exotic annuals were indicative of the warm seeding treatment in 2005 (*Bramus tee/arum* L. and *Sisymbrium altissimum* L.), but not in 2006 (Table 5). In contrast, several species were significant indicators of the non-seeded plots in the first year, including *Chamerion angustifolium* (L.) Holub and *Pinus contorta* (Table 5), suggesting that seeding the warm mix reduced their abundance (cover, frequency or both). By the second post-fire year, seeding the warm mix reduced the abundance of several forb species, including *Chamerion angustifolium, Epilobium brachycarpum* K. Presl, *Lactuca serriola* L., *Pseudognaphalium canescens* (DC.) W.A. Weber, and *Anaphalis margaritacea* (L.) Benth.

We found no indicator species for un seeded plots when compared with either the plots that received the cool or wheat seed treatments (Table 5). Not surprisingly, the wheat seed treatment increased the abundance of wheat *(Triticum aestivum),* and the cool seed mix increased the abundance of cool mix species *(Festuca* spp., *Elymus lanceolatus* (Scribn. & J.G. Sm.) Gould). The exotic annual, *Bromus tectorum,* was also a significant indicator of the cool seed treatment, suggesting potential seed contamination.

Fertilisation increased the abundance of numerous species in each of the first 2 post-fire years when compared with the unfertilised plots (Table 6). This included both seeded and non-seeded species, native and exotic species, and all understorey Iifeforms. The only species for which abundance was significantly reduced by fertilisation was *Pinus contorta* (Table 6).

Discussion

Treatment effects on native vegetation

Recently, there has been an emphasis on using native species in seeding treatments to increase post-fire plant cover while avoiding the negative impacts on native vegetation recovery and biodiversity associated with seeding exotic species (Richards *et al.* 1998; Beyers 2004). However, seeding with native species has seldom been applied in practice, so ecosystem effects are not well understood. In this study, impacts of seeding on native vegetation recovery were more closely related to cover produced than to the origin of the species. Three of the four species in our warm species mixture were natives (with the one non-native species averaging less than 0.5% cover), and yet the warm seed mix treatment significantly reduced the species richness and cover of native non-seeded species and reduced the abundance of several early successional native species. Species that are likely to provide sufficient cover with seeding following wildfire likely will grow early and vigorously (Beyers 2004); therefore, they may reduce water availability or other limiting nutrients later in the growing season whether the seeded species are native or exotic. The key to maintaining biodiversity may be preventing dominance by one or a few species, regardless of their origin (Houlahan and Findlay 2004), which is contrary to the objectives of post-fire seeding treatments that attempt to establish rapid cover with one or a few species.

The negative relationship between seeding and native 110n-seeded species cover found in this study may have important ecological implications as well as implications for monitoring treatment efficacy. Previous work has demonstrated that the

Table 6. Indicator species analysis for each level of fertilisation compared with the plots without fertilisation
IV is the indicator value and *P* is the probability of obtaining as high an IV with data randomised among treatments 5000 times in a Monte Carlo procedure. Only significant indicator species are shown. NS, not significant

Species	Nativity	Life-form	2005			2006		
			Fert	IV	*P*	Fert	IV	*P*
Low fert *v.* none								
Achillea millefolium[A]	Native	Forb	Low	36.3	<0.001	Low	48.0	<0.001
Bromus tectorum	Exotic	Graminoid	Low	15.8	0.001			NS
Chamerion angustifolium[A]	Native	Forb	Low	63.1	<0.001	Low	57.1	<0.001
Elymus lanceolatus[A]	Native	Graminoid			NS	Low	20.6	0.020
Elymus wawawaiensis[A]	Native	Graminoid	Low	15.8	0.003			NS
Festuca spp.[A]	Both	Graminoid	Low	28.0	<0.001	Low	33.1	0.030
Lactuca serriola	Exotic	Forb	Low	21.8	0.003			NS
Pinus contorta	Native	Tree			NS	None	52	0.003
Pseudognaphalium canescens	Native	Forb	Low	21.2	0.033	Low	42.9	0.001
Salix scouleriana	Native	Shrub	Low	49.1	0.053	Low	53.0	0.004
Sisymbrium altissimum	Exotic	Forb	Low	19.0	0.001			NS
Spiraea betulifolia	Native	Shrub	Low	52.7	0.001	Low	49.9	0.007
Triticum aestivum[A]	Exotic	Graminoid	Low	46.4	0.001	Low	35.9	0.046
High fert *v.* none								
Achillea millefolium[A]	Native	Forb	High	36.5	0.001	100N	50.0	<0.001
Anaphalis margaritacea	Native	Forb			NS	High	15.4	0.013
Bromus tectorum	Exotic	Graminoid	High	20.4	<0.001			NS
Chamerion angustifolium[A]	Native	Forb	High	62.6	<0.001	High	55.8	<0.001
Elymus lanceolatus[A]	Native	Graminoid	High	15.1	0.002	High	21.6	0.005
Epilobium brachycarpum	Native	Forb			NS	High	15.5	0.004
Festuca spp.[A]	Both	Graminoid	High	23.0	<0.001	High	34.6	0.007
Gayophytum diffusum	Native	Forb			NS	High	37.3	0.006
Lactuca serriola	Exotic	Forb	High	18.9	0.033			NS
Logfia arvensis	Exotic	Forb			NS	High	16.2	0.002
Pinus contorta	Native	Tree	None	43.1	0.013	None	56.4	<0.001
Pseudognaphalium canescens	Native	Forb	High	22.0	0.020	High	43.0	<0.001
Salix scouleriana	Native	Shrub	High	50.8	0.005	High	56.2	<0.001
Sisymbrium altissimum	Exotic	Forb	High	24.8	<0.001			NS
Spiraea betulifolia	Native	Shrub	High	52	<0.001	High	50.8	0.001
Triticum aestivum[A]	Exotic	Graminoid	High	45.9	<0.001			NS

[A]Seeded species.

ecological effects of seeding are likely dependent on the cover and density of seeded species (Schultz *et al.* 1955; Beyers 2004). For example, large reductions in native cover and richness following post-fire seeding in the Eastern Cascades (Schoennagel and Waller 1999) and California (Keeley 2004) were associated with high seeded-species cover. The smaller impacts of seeding on native vegetation in the present study were likely due to lower seeded-species cover. However, even low seeded-species cover interfered with native vegetation recovery, suggesting that competition for limiting resources plays an important role in post-fire vegetation recovery in these dry forest types. It also suggests that the common practice of monitoring only seeded species cover to assess treatment effectiveness (Robichaud *et al.* 2000) is likely to overestimate seeding treatment effectiveness, as such an approach cannot quantify associated reductions in native non-seeded vegetation cover.

Seeding winter wheat, which was the operational seeding treatment for the Pot Peak Fire, produced very little wheat cover (average ofless than 2% cover where seeded), but also had little effect on the recovery of the native community. Wheat sprouting and survival is strongly correlated with the amount and timing of precipitation (Robichaud *et at.* 2006), so it should not be surprising that wheat seeding efficacy has been variable (Robichaud *et at.* 2000). However, wheat failed to provide much cover in the present study despite above-average precipitation in the 2 years following the fire, suggesting other factors also limit wheat establishment and growth in this area. Stronger community effects than observed in this study could likely be expected where wheat cover is higher. For example, wheat seeding has impaired native community recovery in California (Keeley 2004) and the Eastern Cascades (Schoennagel and Waller \999).

Slope stabilisation treatments may also affect vegetation recovery and successional patterns by altering community composition through differential impacts on constitutive native species. For example, Schoennagel and Waller (I 999) found that seeding exotic grasses decreased early successional wind-dispersed species in the Eastern Cascades and Anderson and Brooks (1975) found that several forb species were decreased by seeding grass in Oregon. In the present study, seeding the warm mix also led to reduced cover and frequency of several early successional forb species. Furthermore, seeded species cover increased from the first to second year in this study and the effects of seeded species can extend beyond their lifecycle (Schoennagel and Waller 1999), suggesting that impacts may be greater than measured in this short-term study.

Fertilisation in undisturbed ecosystems can favour a few dominant species, leading to competitive exclusion and decreased diversity (Wedin and Tilman 1996; Thomas *et al.* 1999; Clark *et al.* 2007). However, the post-fire fertilisation in the present study increased native non-seeded plant cover without any significant effect on species richness, Furthermore, there was no evidence that fertilisation reduced richness in a non-significant fashion, as the effects on richness were small but positive in the absence of seeding. Finally, fertilisation favoured species of all understorey lifeforms (graminoids, shrubs and forbs) and both off-site colonisers and resprouters, suggesting that many species in this ecosystem can simultaneously benefit from fertilisation after wildfire. However, fertilisation can impact communities long after its application (Olsson and Kellner 2006), emphasising the need for longer-term monitoring.

Although fertilisation produced few negative effects, it did reduce tree regeneration. Reduction in tree regeneration is a common problem for BAER treatments, and has been documented for treatments including seeding (Keeley 2004) and mulching (Kruse *et al.* 2004). It is unlikely that the fertilisation decreased tree seedlings directly, but herbaceous vegetation and shrubs can compete strongly with tree regeneration (Davis *et al.* 1998), so fertilisation likely reduced tree seedling growth, density or both by favouring their competitors. The long-term impacts of fertilisation treatments on forest structure remain unclear, however. Where tree regeneration is abundant, self-thinning would be expected to reduce future tree seedling or sapling densities anyway (Kruse *et al.* 2004). In addition, short-term reductions in seedling growth may be offset over time if increased site nutrient capital produces higher growth rates after trees begin to dominate competitive interactions.

Treatment effects on exotic plants

Potential treatment benefits must be weighed against their potential for negative ecosystem consequences such as the introduction of exotic species, which is currently one of the largest problems facing land managers (Mack *et al.* 2000; Brooks *et al.* 2004). The cool and warm seed mixes increased exotic cover and richness in the first post-fire year, suggesting seed contamination, an ongoing concern in seeding treatments (Robichaud *et al.* 2000; Allen *et al.* 2002; Barclay *et al.* 2004). Fertilisation also resulted in increased exotic cover and richness the first post-fire year, especially when combined with seeding. This supports previous research that suggests nutrient addition can favour exotic species (Kolb *et al.* 2002; Brooks 2003; Hunter and Omi 2006). Mulch also resulted in increased exotic species richness in the current study, which is consistent with a study of post-fire mulching in California (Kruse *et al.* 2004). However, long-term treatment effects on exotic plant cover and richness are less clear. Exotic cover and richness declined on seeded and fertilised piots in the second year after fire, leaving treated plots more similar to control plots. This suggests that increases in exotic plant cover may be transient effects that decline as native vegetation recovers.

Conclusions

Post-fire slope stabilisation treatments should be cost-effective, effective at reducing erosion and beneficial to native vegetation recovery, or should at worst have minimal negative ecosystem consequences (USDA 1995). We found evidence that seeding reduced native non-seeded species cover and richness, introduced exotic species, and reduced the cover or frequency of several native forbs despite only very modest increases in total plant cover (Peterson *et at.* 2009). This adds to a growing body of work that suggests seeding is likely to have negative ecological impacts (Schoennagel and Waller 1999; Beyers 2004; Keeley 2004) and suggests that cover added by seeded species comes at the partial expense of native species cover. Although seeding with native species has gained momentum in recent years (Richards *et al.* 1998; Beyers 2004), we found evidence that seeding with native species has similar ecosystem effects as previously documented for exotic species.

In contrast, fertilisation led to increased cover of native non-seeded species, while having no effect on richness and simultaneously favouring several individual native species. Further research and monitoring will be needed to establish the general effectiveness of fertilisation in reducing erosion under varying environmental conditions. Also, the potential for fertilisation to reduce conifer regeneration and for mulch and fertilisation to favour exotic species deserves further study, although the effects were small. Overall fertilisation showed promise as a BAER treatment that can increase plant covel' by facilitating natural vegetation recovery.

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