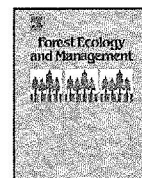




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Seeding and fertilization effects on plant cover and community recovery following wildfire in the Eastern Cascade Mountains, USA

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ABSTRACT

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Slope stabilization treatments are frequently applied following high severity wildfires to reduce erosion, protect water quality, and mitigate threats to human life and property. However, the effectiveness of many treatment options has not been well established. Furthermore, treatments may unintentionally inhibit natural vegetation recovery or facilitate exotic species invasion, compromising long-term ecosystem function. We evaluated the effects of seeding and fertilization treatments on plant cover and vegetation recovery following the Deer Point fire in the Eastern Cascade Mountains of Washington State, surveying vegetation for three consecutive years following fire. We applied a fertilization treatment and two seeding treatments in factorial combination on experimental plots at four sites within the fire. Natural vegetation recovered rapidly on control plots, exceeding 40% average cover the second post-fire year and 53% cover the third year. Seeding and fertilization, applied alone and together, did little to increase total plant cover in any of the three post-fire years. A seed mix containing mostly native species increased seeded species cover, but failed to increase in total plant cover, as reductions in non-seeded species cover largely offset increases in seeded species cover. The seed mix also reduced the cover and frequency of several disturbance-adapted native species and reduced tree seedling abundance by the third year after fire. Exotic species averaged less than 0.5% cover across all treatments, and were not significantly affected by any treatment. Minimal treatment effects on total plant cover suggest that seeding and fertilization did little to reduce erosion hazards. However, seeding with the species mix did interfere with natural vegetation recovery, despite the use of native species and low realized seeded species cover.

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1. Introduction

Forest managers must often balance the potential benefits of active management against anticipated and unexpected impacts of management activities on natural processes. Post-fire forest management provides a good example of such tradeoffs. High severity wildfires consume vegetation and organic matter and may increase soil hydrophobicity, which can greatly increase erosion and compromise watershed functioning and downstream values (Debano et al., 1998; Benavides-Solorio and MacDonald, 2001; Wondzell and King, 2003). Burned area emergency response (BAER) land surface treatments, including seeding, mulching, and log terracing, are often applied after fires to mitigate these hazards (Robichaud et al., 2000). However, treatments may interfere with natural vegetation recovery (Schoennagel and Waller, 1999; Barclay et al., 2004; Beyers, 2004; Keeley, 2004), compromising long-term ecosystem function. Further information about both

treatment efficacy in reducing erosion hazards and ecological consequences is needed to inform management choices among treatment alternatives. Here, we report on a study testing the effects of two post-fire emergency rehabilitation treatments - seeding and fertilization - on the development of organic soil cover and the recovery of native vegetation after high severity wildfires.

Aerial seeding is used to rapidly restore plant cover where severe wildfire has killed vegetation and consumed organic soil cover (Robichaud et al., 2000) and, in some cases, to inhibit the invasion and spread of exotic plant species (Beyers, 2004; Floyd et al., 2006). Seeding is the most widely used post-fire stabilization treatment, but its effectiveness for increasing plant cover has been variable (Robichaud et al., 2000). Furthermore, seeded species can alter community dynamics by displacing native species (Schoennagel and Waller, 1999; Beyers, 2004; Keeley, 2004; Kruse et al., 2004), and several species have been abandoned for seeding because of their persistence in the community (Beyers, 2004). Additionally, seeding may introduce exotic plant species, thereby increasing invasion (Allen et al., 2002; Barclay et al., 2004; Kruse et al., 2004). In recent years, seeding has evolved towards using non-persistent cereal grains or native species to mitigate com-

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munity impacts. However, recent studies have documented negative community effects for even sterile winter wheat (Schoennagel and Waller, 1999; Keeley, 2004) and few studies have documented either the effectiveness or community impacts of seeding native species.

Many essential nutrients for plant growth are primarily contained within the organic layer in and above the soil (Robichaud et al., 2006). Much of this nutrient capital may be lost to volatilization in severe wildfires (DeBano, 1991; Bormann et al., 2008), leading to potential nutrient limitations on vegetation regrowth that could be relieved by fertilization (Baird et al., 1999). Fertilization has also been used in conjunction with seeding to increase the establishment and cover of seeded species (Klock et al., 1975; Robichaud et al., 2000). Effectiveness of fertilization has not been extensively evaluated, but land managers in the northwestern U.S. regard fertilization as effective (Robichaud et al., 2000), which has been corroborated by recent studies in this area (Robichaud et al., 2006; Peterson et al., in press). However, fertilization may also increase competitive exclusion (Wedin and Tilman, 1996; Stevens et al., 2004) or favor exotic species (Kolb et al., 2002; Brooks, 2003). More information is needed on fertilization effects to fully assess its utility as a post-fire erosion control treatment.

In this study, we used an experimental approach to evaluate the effects of seeding and fertilization slope stabilization treatments within the Deer Point fire in the eastern Cascade Range of Washington State. We monitored vegetation responses for the first three years after fire to address the following research questions:

- (i) Are seeding and fertilization treatments effective for increasing total plant cover after wildfire?
- (ii) Do seeding and fertilization treatments influence exotic plant cover or spread?
- (iii) Do seeding and fertilization treatments alter the cover, richness, or species composition of recovering native vegetation?

2. Methods

2.1. Study site

The study area is located in north-central Washington State (48.03°N, 120.20°W) within the Lake Chelan drainage. Soils in the study area consist of variable layers of pumice and ash deposited over more coarse textured subsoils. Summers are warm and dry with occasional intense thunderstorms. The majority of the precipitation comes from October to March, much of it falling as snow. Point estimates for our sites indicate that mean annual precipitation ranges between 30 and 43 cm per year (PRISM, 2009).

The nearest climate station (about 20 km to the south of the study area) averages about 28 cm of annual precipitation (Western Regional Climate Center, Lakeside, <http://www.wrcc.dri.edu>). On average about 20% of the total annual precipitation falls during the May–August growing season. Annual precipitation during the study period (2003–2005) was near normal (88–116% of normal), but growing season precipitation varied considerably among years (Fig. 1). The May–August period in the first post-fire year received less than 15% of normal precipitation at Lakeside, while the growing season precipitation in 2005 was more than double the long-term average (Fig. 1). For the state of Washington as a whole, the first post-fire growing season was one of the hottest and driest on record (NCDC, 2007).

The Deer Point fire burned nearly 16,500 ha along the northeast shore of Lake Chelan in the Okanogan-Wenatchee National Forest in the summer of 2002. Much of the area burned by the Deer Point fire had been previously burned by wildfire in 1970. The fire burned through steep terrain across a variety of elevations and

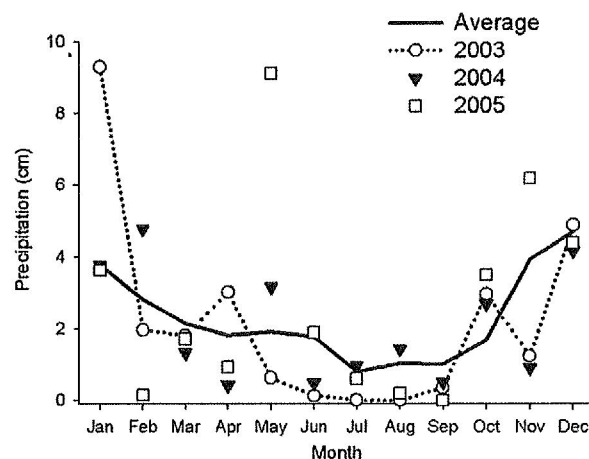


Fig. 1. Monthly precipitation at the nearest weather station for the study period (2003–2005). The solid line is the average monthly precipitation and the dotted line is precipitation for 2003 showing the dry May–September period.

vegetation types ranging from shrub-steppe at low elevations to open dry ponderosa pine (*Pinus ponderosa*) and Douglas-fir (*Pseudotsuga menziesii*) forest at middle elevations and dense young lodgepole pine (*Pinus contorta*) stands at higher elevations. Only 28% of the area burned was classified as moderate to high severity on the burned area reflectance classification (BARC) maps, but 11,000 ha (67%) were classified as being at high risk for post-fire soil erosion due to a combination of site factors, including fire severity, steep slopes, coarse soil textures, soil hydrophobicity, and lack of organic soil cover. BARC maps produced by the U.S. Forest Service were used to identify candidate areas where fire severity and site factors combined to produce high soil erosion potential.

For this study, we selected four study sites in stands that were candidates to receive operational erosion control treatment but which could be excluded from treatment. All sites selected for treatment had complete overstory mortality and near complete consumption of duff and litter as confirmed by site visits shortly after wildfire containment. We chose study sites to represent a variety of environmental conditions and vegetation types (Table 1). Within each site, we established a grid of 48-m² (6 m x 8 m) experimental plots. We left a 1-m untreated buffer between adjacent plots to reduce cross-plot contamination and to minimize trampling within the plots. We established 33 plots at Antilon Lake and 64 plots at each of the three other sites for a total of 255 plots.

2.2. Treatments

We tested the effects of three potential erosion control treatments - seeding with wheat, seeding with a perennial seed mix, and fertilization - on vegetation cover and plant community recovery. For the wheat seeding treatment (prescribed operationally for middle to high elevation sites), we applied soft white winter wheat (Eltan) at a target rate of 97 seeds/m² (34 kg/ha). For the seed mix treatment (prescribed operationally for some low

Table 1
Site characteristics for the Deer Point study.

Site	Elevation (m)	Aspect	Slope (°)	Forest type
Antilon	701	N	5–15	Open ponderosa pine
Mitchell	701	W	25	Open ponderosa pine
Grade	1036	SE	0–10	Mixed conifer
Oss Peak	1463	SE	10–20	Mixed conifer

Table 2
Species in the seed mix at the Deer Point Study and application rates.

Species	Common name and cultivar	Origin	Seeds (m ²)
<i>Achillea millefolium</i>	Western yarrow	Native	75
<i>Elymus lanceolatus</i>	Thickspike wheatgrass	Native	269
	'Schwendimar'		
<i>Festuca ovina</i>	Sheep fescue 'Covar'	Exotic	172
<i>Poa secunda</i>	Big bluegrass 'Sherman'	Native	226

elevation areas), we applied a mixture of three perennial graminoids and one perennial forb that were expected to do well on warm, dry sites and compete strongly with noxious weeds at a target rate of 742 seeds/m² (See Table 2 for species composition). The seeding rates for both the wheat and seed mix followed operational protocols for the Deer Point fire, and we used the same seed source as the operational treatments. For the fertilization treatment, we applied a mixture of ammonium nitrate and ammonium sulfate (30-0-0-6) at a rate of 56 kg N/ha. Combined factorially, the three treatments produced eight unique treatment combinations which we assigned randomly to experimental plots at each site. We applied seed treatments in fall 2002, and fertilizer treatments in the spring 2003 following operational timetables for treatment application in this region.

2.3. Sampling

We assessed plant cover in the summer (June–August) when plant biomass was near its peak at each site. We sampled total plant cover in the first two years after fire using the line-intercept method (Kent and Coker, 1992) with a 9-m diagonal transect on each plot extending to within 0.5 m of the corner posts. We also measured seeded species cover in the first year after fire, but not the second year. In the third year, we adjusted the sampling protocols to better evaluate treatment effects on community composition and exotic species. On each 48-m² plot, we visually estimated the cover for each non-plant cover type (bare ground, rock, wood, litter). We then identified all vascular plant species and visually estimated their cover to the nearest percent, assigning 0.2% over to species present only in trace amounts. We estimated cover for each plant species as if other species were not present, so plot totals could exceed 100%. We then adjusted total plant cover and the cover of each plant group (e.g., seeded species, exotic species) proportionally so that the total cover on a plot summed to exactly 100% when plant cover was combined with non-plant cover. We defined exotic species as species introduced to the contiguous United States as noted in the United States Department of Agriculture PLANTS database (USDA NRCS, 2007).

2.4. Statistical methods

This study employed a generalized randomized block experimental design (Hinkelmann and Kempthorne, 1994), which is similar to a common randomized block design, but with treatments replicated within blocks (sites). Prior to analysis, we chose a Type I error rate of 5% ($P < 0.05$) for assessing the statistical significance of treatment effects. We used a mixed effects model analysis (PROC MIXED, SAS Institute, 2004, version 9.1) to assess seeding and fertilization effects. We evaluated treatment effects on total plant cover separately for each of the first three post-fire years and on seeded species cover in the first and third post-fire years. We also evaluated treatment effects on exotic and native non-seeded species (native species not included in the seed mix) cover and richness only in the third post-fire year. We analyzed the effects of wheat seed, seed mix, and fertilization treatments as a complete factorial experiment considering all possible interac-

tions. We included treatments in the models as categorical fixed factors with two levels each (applied or not). We included a random site variable to account for site differences in mean cover and richness across all treatments. The site variable was never significant (all P -values > 0.1) but was retained in the models to maintain the experimental design and account for autocorrelation among plots on the same site. We also tested for random site differences in treatment effects (site by treatment interactions), but none were significant and therefore were not included in the final analyses. Where significant interactions were found we performed post-hoc tests among significant treatment combinations with a Tukey adjustment for multiple comparisons. We visually examined model residuals to assess assumptions about normality and independence. Model residuals for seeded species cover and exotic species richness and cover showed some deviation that was not corrected with transformations due to a number of plots with no exotic or seeded species. We present the results on the untransformed data.

To test treatment effects on individual species, we used indicator species analysis (ISA) in PC-ORO version 5.0 (McCune and Mefford, 1999) with third-year plant species cover data to identify individual species that increased or decreased cover and frequency in response to treatments. ISA combines the relative cover and frequency of a species to determine an indicator value (IV) for a group or treatment (Dufrene and Legendre, 1997; McCune and Grace, 2002). We performed comparisons of plots where each treatment was applied vs. plots where the treatment was not applied. We performed three separate analyses for the wheat seed, seed mix, and fertilization treatments. This approach allowed us to identify both species that were favored by specific treatments and those that were reduced. We used a Monte Carlo test with 5000 runs with individual species observations randomized among plots to determine the probability (P -value) of obtaining an equally large IV for a treatment by chance. We designated species as significant indicators if the IV exceeded 15 and $P < 0.05$.

3. Results

Total plant cover on control plots (no seeding or fertilization) averaged 23% in the first year after fire (2003) and 42% in the second year (2004, Fig. 2). By the third year after fire (2005), total plant cover averaged 53% on control plots with bare ground averaging 41%. Native non-seeded species (native species not included in the seeding treatments) provided most of the cover in

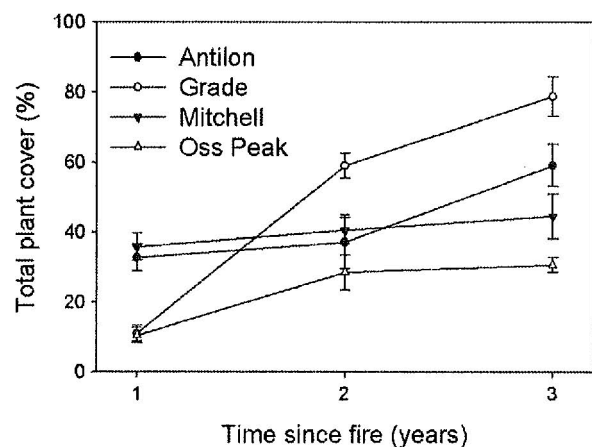


Fig. 2. Mean plant cover on control plots at each of the four sites during the first three years following wildfire. Error bars indicate standard errors.

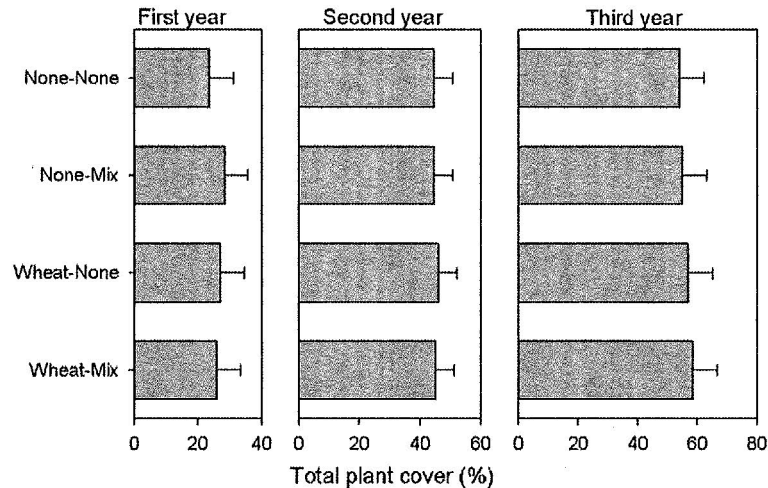


Fig. 3. Least square means of total plant cover for the wheat and species mix seeding treatments and their interaction, for each of the first three years following wildfire at Deer Point. Error bars indicate standard errors. Pair-wise comparisons indicated no significant differences among treatments.

each year. Plant recovery rates and total plant cover varied considerably among the four sites (Fig. 2). The low elevation sites (Antilon and Mitchell) recovered most rapidly, surpassing 30% average plant cover on control plots the first year after fire. The middle elevation site (Grade) had little plant cover the first year, but recovered rapidly thereafter, surpassing the cover on the low elevation sites by the second year. Plant cover remained low through the third year after fire on Oss Peak, the highest elevation site, due to low densities of surviving plants (Fig. 2).

Seeding and fertilization treatments failed to significantly increase total plant cover in any of the first three years after wildfire (Fig. 3). The only significant effect was an interaction of seeding wheat and the species mix the first year (Table 3). When seeded separately the wheat and seed mix increased plant cover slightly, but when seeded together cover was lower than when each was seeded alone (Fig. 3). However, none of the seeding treatments were significantly different from the no-seeding control in post-hoc comparisons. No treatment or treatment interaction significantly affected total plant cover in either the second or third years after fire (Table 3).

Although treatments failed to significantly increase total plant cover, the perennial seed mix and fertilization treatments did significantly alter seeded species cover (Table 3, Fig. 4). Seeded species cover was low on control plots, averaging less than 0.5% the first year and increasing to only about 2% cover by the third year after fire (Fig. 4). Seeding with the perennial seed mix significantly increased seeded species cover in the first year after fire (Fig. 4). In the third year, there was a significant interaction between the seed mix and fertilization treatments (Table 3). The seed mix treatment significantly increased seeded species cover, while the combination of the seed mix and fertilization treatments produced significantly higher seeded species cover than the seed mix treatment without fertilization (Fig. 4). Seeding winter wheat failed to significantly increase seeded species cover in either year, alone or in combination with fertilization.

Although the perennial seed mix treatment increased seeded species cover, it produced a corresponding reduction in non-seeded native species cover (Table 4, Fig. 5). Application of the perennial seed mix reduced native non-seeded species cover by an average of almost 7.5%, an effect that was apparent whether or not

Table 3
Type III tests of fixed treatment effects on total plant cover and seeded species cover in each year from the mixed models.

Effect	Plant cover 2003		Plant cover 2004		Plant cover 2005	
	F-value	P	F-value	P	F-value	P
Wheat	0.16	0.6920	0.46	0.4691	2.89	0.0904
Mix	1.66	0.1995	0.16	0.6853	0.37	0.5411
Wheat × mix	5.00	0.0264	0.12	0.7262	0.01	0.9194
Fert	1.64	0.2018	3.24	0.0734	2.67	0.1038
Wheat × fert	0.31	0.5779	1.36	0.2444	3.44	0.0649
Mix × fert	0.00	0.9730	0.38	0.5407	0.93	0.3356
Wheat × mix × fert	0.91	0.3417	0.00	0.9845	1.89	0.1708
Effect	Seeded species cover 2003		Seeded species cover 2005			
	F-value	P	F-value	P		
Wheat	1.57	0.2119	2.07	0.1517		
Mix	29.44	<0.0001	193.58	<0.0001		
Wheat × mix	0.00	0.9819	2.14	0.1447		
Fert	1.07	0.3014	7.47	0.0068		
Wheat × fert	1.28	0.2589	0.01	0.9290		
Mix × fert	0.97	0.3268	6.31	0.0127		
Wheat × mix × fert	0.48	0.4880	0.42	0.5199		

Significant effects ($P < 0.05$) are bolded.

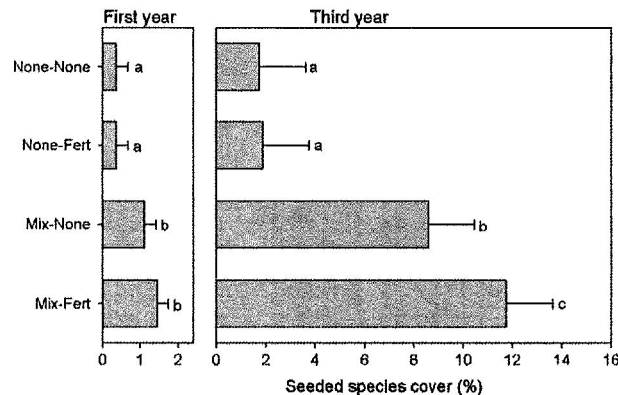


Fig. 4. Least square means of seeded species cover for the seed mix and fertilization treatments and their interaction, for the first and third years after wildfire at Deer Point. Error bars indicate standard errors. Different letters signify significant treatment differences in pair-wise comparisons.

wheat and fertilizer were also applied. As a result, reductions in non-seeded native species cover largely offset increases in seeded species cover (Fig. 5). Unlike plant cover, treatments did not alter species richness for native non-seeded species (Table 4).

Non-seeded exotic species cover and richness were very low in the third year after fire, averaging less than one species per 48 m² plot, and less than 0.5% cover across all treatments. There were no significant effects of either seeding or fertilization on exotic species cover or richness in the third post-fire year (Table 4).

The perennial seed mix had the strongest effects on community composition. The seed mix increased the abundance of four seeded species, including *Achillea millefolium*, *Festuca ovina*, *Poa secunda* and *Elymus lanceolatus* (Table 5). The seed mix reduced the abundance of several other species, including two disturbance-adapted forbs (*Chamerion angustifolium* and *Pseudognaphalium canescens*), a graminoid (*Carex rossii*) and an important tree species (*P. contorta*). Fertilization affected few species, apparently increasing the abundance of *Calamagrostis rubescens* and decreasing the abundance of *Purshia tridentata* (Table 5). The indicator species analysis revealed no species that were significantly affected by seeding wheat (Table 5).

Table 4
Type III tests of fixed treatment effects on native non-seeded and exotic species cover and richness in the third post-fire year.

Effect	Native non-seeded cover		Native non-seeded richness	
	F-value	P	F-value	P
Wheat	1.77	0.1851	0.93	0.3361
Mix	16.16	<0.0001	1.61	0.2066
Wheat × mix	0.09	0.7687	0.44	0.5091
Fert	0.59	0.4445	0.66	0.4173
Wheat × fert	3.64	0.0577	0.04	0.8444
Mix × fert	0.02	0.8799	2.01	0.1575
Wheat × mix × fert	1.54	0.2167	0.16	0.6907
Effect	Exotic cover		Exotic richness	
	F-value	P	F-value	P
Wheat	0.63	0.4301	0.27	0.6049
Mix	1.42	0.2345	0.70	0.4022
Wheat × mix	1.33	0.2496	2.99	0.0851
Fert	0.53	0.4679	0.00	0.9710
Wheat × fert	0.35	0.5575	0.46	0.4981
Mix × fert	0.48	0.4878	0.13	0.7210
Wheat × mix × fert	0.04	0.8499	0.00	0.9710

Significant effects ($P < 0.05$) are bolded.

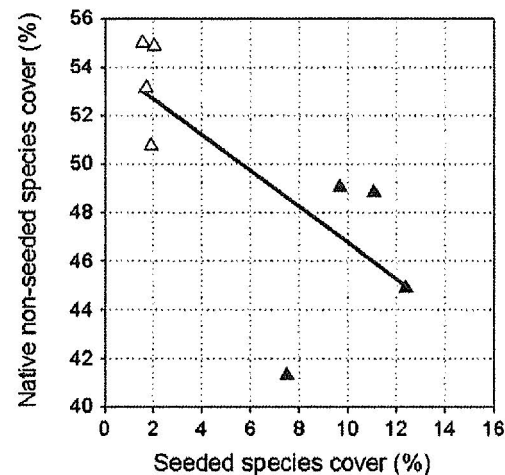


Fig. 5. Trade-off between seeded species cover and native non-seeded species cover in the third year following fire at Deer Point. Symbols represent the least square means for the eight treatment combinations with closed symbols indicating the application of the seed mix. The regression line is a least squares fit. Wheat seed and fertilization did not significantly affect native non-seeded species cover.

4. Discussion

4.1. Treatment effects on plant cover

The rising costs of post-fire stabilization treatments (Government Accountability Office, 2003; Groen and Woods, 2008) and the increasing frequency of large wildfires (Westerling et al., 2006) underscores the need for better monitoring and critical evaluations of treatment efficacy (Government Accountability Office, 2003). In this study, seedling and fertilization treatments, applied alone or together, failed to significantly increase total plant cover. Our experimental results were consistent with results from the operational seeding treatments following this fire, which also produced little additional soil cover (D.W. Peterson, personal observation). Given that previous studies have found that soil erosion rates are strongly negatively correlated with ground cover (Benavides-Solorio and MacDonald, 2001; Johansen et al., 2001), it is unlikely that treatments significantly reduced soil erosion and flooding hazards.

Despite the widespread use of seeding as a post-fire rehabilitation treatment, few studies have documented decreased erosion with seeding (Robichaud et al., 2000). Wheat seed, one of the primary operational treatments in this region of the United States (Robichaud et al., 2000), was particularly ineffective at producing cover in this study, with wheat never exceeding 2% on any plot. This contrasts with another recent study of post-fire wheat application in California where much higher rates of wheat application resulted in very high wheat cover the first post-fire year (Keeley, 2004). Similar applications rates of wheat as used in this study have also produced much higher cover following wildfire in the Eastern Cascades (Schoennagel and Waller, 1999). However, other regional studies have also found that wheat seed failed to produce much cover with the same application rate as used in this study (Robichaud et al., 2006) and even with higher application rates (Peterson et al., in press). The germination and establishment of wheat may be strongly influenced by precipitation patterns the first year following the fire (Robichaud et al., 2006). The dry growing season the first post-fire year in this study may have contributed to the failure of wheat to establish (Fig. 1), but large fires often occur during multi-year dry periods (Robichaud et al., 2006) suggesting this may be an on-going issue.

Table 5
Species significantly affected by treatments as determined by Indicator Species Analysis.

Wheat seeded vs. no wheat seeded						
No significant indicators						
Mix seeded vs. no mix seeded						
Species	Seeded	Lifeform	Origin	Group	IV	P
<i>Achillea millefolium</i>	Yes	Forb	Native	Mix	87.5	0.0002
<i>Chamerion angustifolium</i>	No	Forb	Native	None	45.5	0.0002
<i>Festuca ovina</i>	Yes	Graminoid	Exotic	Mix	88.7	0.0002
<i>Poa secunda</i>	Yes	Graminoid	Native	Mix	80.1	0.0002
<i>Pseudoroegneria spicata</i>	No	Graminoid	Native	Mix	26.6	0.0002
<i>Agropyron</i> spp.	No	Graminoid	Native	Mix	19.6	0.0004
<i>Elymus lanceolatus</i>	Yes	Graminoid	Native	Mix	28.6	0.0016
<i>Pseudognaphalium canescens</i>	No	Forb	Native	None	31.1	0.0054
<i>Carex rossii</i>	No	Graminoid	Native	None	15.2	0.0200
<i>Pinus contorta</i>	No	Tree	Native	None	29.1	0.0354
Fertilization vs. no fertilization						
<i>Purshia tridentata</i>	No	Shrub	Native	None	15.7	0.0094
<i>Calamagrostis rubescens</i>	No	Graminoid	Native	Fert	49.2	0.0562

IV is the indicator value and P is the probability of obtaining as large an IV with data randomized 5000 times in a Monte Carlo test.

The perennial seed mix, prescribed operationally to reduce erosion and combat the spread of exotic plant species in low-elevation dry forest areas, provided low to moderate seeded species cover, but failed to increase total plant cover and provided very little cover in the first post-fire year when erosion hazard is the highest. Previous studies have found seeded perennial species may take several years to establish significant cover (Leege and Godbolt, 1985). This study adds to the growing body of evidence that suggests seeding often has little to no impact on total plant cover (Schoennagel and Waller, 1999; Robichaud et al., 2000; Beyers, 2004; Wagenbrenner et al., 2006; Robichaud et al., 2006; Groen and Woods, 2008; Peterson et al., in press). This lack of consistent efficacy in reducing erosion hazard raises questions about the continued wide-spread use of seeding to reduce post-fire erosion hazard.

Fertilization was also ineffective for increasing plant cover, suggesting that vegetation recovery on these sites was not limited by soil nutrient availability. This contrasts with previous studies in the eastern Cascades where fertilizer increased plant cover or vigor (Klock et al., 1975; Robichaud et al., 2006; Peterson et al., in press). Soil water is by definition a major limiting resource in dry coniferous forests and regional climatic models predicted very low average annual precipitation levels at each of our sites (< 44 cm; PRISM, 2009). Furthermore, the first summer after the fire (2003) was very dry (NCDC, 2007), which may have further reduced fertilization efficacy. Fertilization has received relatively little study as a post-fire slope stabilization treatment (Robichaud et al., 2000) and further study may be needed to determine what site environmental and other factors influence fertilization effectiveness.

4.2. Treatment effects on exotic species

Non-seeded exotic species cover and richness were very low at all four sites evaluated in this study and did not respond to either seeding or fertilization. This lack of post-fire invasion contrasts with other recent studies where exotic species had high abundance following wildfire (e.g., Crawford et al., 2001; Griffiths et al., 2001; Floyd et al., 2006). Had the treatments increased plant cover significantly, the lack of response in this study would have been encouraging, suggesting that exotic plant introductions through seeding were minimal and that fertilization did not significantly facilitate the spread of existing exotic plant populations. As it is, the lack of overall plant cover response to treatment prevents us

from drawing any strong inferences about potential seeding and fertilization effects on exotic plants. Low background levels of exotic plant cover and species richness, combined with unusually dry weather conditions, likely minimized any stimulatory effects of seeding and fertilization and made any real effects difficult to detect, even at high levels of treatment replication.

4.3. Treatment effects on native vegetation recovery

There is growing evidence that seeding after fire interferes with natural vegetation recovery (Schoennagel and Waller, 1999; Barclay et al., 2004; Beyers, 2004; Keeley, 2004; Hunter and Omi, 2006). In this study, the perennial seed mix significantly reduced the average cover of native non-seeded species and reduced the cover and frequency of several individual native species. The decrease in native non-seeded species cover was almost identical to the additional cover of seeded species on plots that received the seed mix, implying a trade-off between seeded species and non-seeded species. Previous studies have suggested that when seeded species cover is high it may interfere with native vegetation recovery (Beyers, 2004; Keeley, 2004); however, in this study trade-offs between seeded and non-seeded species cover were evident even at relatively low seeded species cover (<13% average for all treatments).

This evidence for suppression of native vegetation cover by seeded species has important ramifications for forest managers developing protocols for monitoring seeding treatment effectiveness. Where seeded species are exotics that can be clearly distinguished from native species (e.g., wheat seeding), it is tempting to simply assess cover of the seeded species and assume that represents a net increase in total plant (and soil) cover. Our study shows that comparisons with untreated control areas are needed to accurately quantify net increases in total plant cover and assess likely treatment effects on soil erosion. Without such comparisons, seeded species cover can only provide an upper bound on net cover increases. Unfortunately, effectiveness monitoring of operational treatments rarely includes untreated control sites (Robichaud et al., 2000).

In recent years, some have suggested using native plant species in seeding treatments to reduce seeding impacts on native vegetation recovery and to prevent introduction of exotic plant species. Most of the seeded species cover produced by the seed mix treatment was from native species (>88%), and yet interference with non-seeded vegetation recovery was still evident. This

suggests that efforts to alleviate ecological impacts by seeding native species (Richards et al., 1998; Beyers, 2004) may not be effective. Species that produce sufficient plant and soil cover to significantly reduce soil erosion and runoff following wildfire will tend to grow early and vigorously (Beyers, 2004; Hunter and Omi, 2006), whether they are native or exotic. In doing so, they may reduce soil water or nutrient availability to other species later in the growing season. Likewise, seeding treatments designed to compete strongly with exotic invaders (e.g. the seed mix in this study) are also likely to compete strongly with recovering native vegetation.

Understory vegetation may compete strongly with tree seedlings (Davis et al., 2004; Rose and Ketchum, 2002). The seed mix treatment significantly reduced the relative abundance (cover) of one conifer species (lodgepole pine) in this study. Previous studies have also documented that post-fire seeding can reduce conifer regeneration (Keeley, 2004). However, we did not quantify seedling densities in this study, and conifers may continue to establish for several years after fire (Turner et al., 1997) so the long-term consequences for forest re-establishment remain uncertain.

Many understory plant species in dry coniferous forests are adapted to resprout and colonize vigorously, even after intense wildfires (Lyon and Stickney, 1976; Turner et al., 1997; Schoennagel and Waller, 1999). Previous studies of soil cover relationships with soil erosion have found that returning soil erosion rates to background levels required organic soil cover in the ranges of 30–40% (Johansen et al., 2001) to 60–70% (Pannkuk and Robichaud, 2003). In this study, vegetative cover on control plots averaged 40% in the second year after fire and 53% the third year after fire, suggesting that soil erosion hazards on many sites is reduced in a relatively short time through natural vegetation recovery alone, although erosion hazards are often greatest in the first post-fire year (Robichaud et al., 2000). Recent studies have shown that thinning and burning treatments designed to reduce fuel loads can reduce wildfire severity (Pollet and Omi, 2002; Strom and Fule, 2007). While the short-term effects of such treatment on understory vegetation have been variable (e.g., Schwilk et al., 2009), several studies that have monitored vegetation several years after thinning and/or burning treatments in dry coniferous forests have found increased understory abundance (McConnell and Smith, 1970; Moore et al., 2006). Therefore, restoration treatments may also reduce the need for post-fire slope stabilization treatments, both by reducing fire severity and increasing density and cover of understory vegetation that is likely to survive and quickly recover following a wildfire.

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