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Effects of postfire seeding and fertilizing on hillslope erosion in north-central Washington, USA

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Abstract

After the 1998 North 25 Fire in the Wenatchee National Forest, eight study sites were established on steep, severely burned hillslopes to examine the effectiveness of postfire seeding and fertilizing treatments in increasing cover to reduce hillslope erosion, and to measure the nutrient content of the eroded sediment. At each site, four 4 by 9 m plots were located with four randomly applied treatments: seed (winter wheat, *Triticum estivum*) at 34 kg ha^{-1}, fertilizer (75% ammonium nitrate and 25% ammonium sulfate) at 31 kg ha^{-1}, seed and fertilizer, and untreated control. Sediment fences were installed at the base of each plot to measure erosion rates and sample the eroded sediments. In addition, precipitation amounts and intensities, surface cover, canopy cover, and nutrient concentrations in the eroded sediments were measured for four years after the fire. Total precipitation was below average during the four-year study period, and most erosion occurred during short duration, moderate intensity summer rainfall events. The overall first year mean erosion rate was 16 Mg ha^{-1} yr^{-1}, and this decreased significantly in the second year to 0.66 Mg ha^{-1} yr^{-1}. There were no significant differences in erosion rates between treatments. In the first year, the seeded winter wheat provided 4.5% canopy cover, about a fourth of the total canopy cover, on the seeded plots; however, the total canopy cover on the seeded plots did not differ from the unseeded plots. The below average precipitation in the spring after seeding may have affected the winter wheat survival rate. In the fourth year of the study, the mean canopy cover in the fertilization treatment plots was 74%, and this was greater than the 55% mean canopy cover in the unfertilized plots ($p = 0.04$); however, there was no accompanying reduction in erosion rate for either the seeding or fertilization treatments. Revegetation by naturally occurring species was apparently not impacted by seeding during the four years of this study. The pH of the sediment as well as the concentrations of NO_3–N, NH_4–N, and K was not affected by seeding or fertilizing. The nutrient loads in the eroded sediment were minimal, with most of the nutrient loss occurring in the first postfire year. These results confirm that seeding success is highly dependent on rainfall intensity, amounts, and timing, and that soil nutrients lost in eroded sediments are unlikely to impair the site productivity.

Keywords: Erosion rate; Wildfire; Grass seeding; Silt fence; Monitoring; Rehabilitation

1. Introduction

Fire is a natural and important part of the disturbance regime for forested ecosystems especially in the western USA (Agee, 1993). Wildfires usually burn in mosaic patterns with portions of the area burned in low, moderate, and high severity conditions as defined by Ryan and Noste (1983) and DeBano et al. (1998). Fire effects on runoff and erosion are related, in part, to the amount of unburned vegetation and organic forest floor materials that are left protecting mineral soil, and the creation and extent of water repellent soil conditions (DeBano et al., 1998). Soil erosion after fires can range from 1.0 Mg ha^{-1} yr^{-1}, (Mg = Megagrams = tons) in flat terrain and in the absence of major rainfall events, to 110 Mg ha^{-1} yr^{-1} in steep terrain affected by high intensity thunderstorms (Robichaud et al., 2000).

In an effort to mitigate these fire effects, land management agencies develop and implement post-wildfire emer-

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gency rehabilitation treatments. The planning teams assess burn severity, climate, and values (life, property, water supplies, etc.) at risk (Robichaud et al., 2000), and recommend actions to reduce the hazards of flooding and erosion. However, these emergency rehabilitation efforts have not been rigorously tested or monitored to determine if plan objectives were met (Robichaud et al., 2000; General Accounting Office, 2003).

The 1998 North 25 Fire in north central Washington provided a unique opportunity to test the effectiveness of seeding and fertilizer treatments for postfire erosion control. Because of the extensive postfire rehabilitation treatment studies done in the nearby Entiat Experimental Forest after the 1970 Entiat Fire (Tiedemann and Klock, 1973, 1976), the use of seeding and fertilization is regarded as highly effective by land managers in this region (Robichaud et al., 2000). Aerial seeding and fertilization were used as emergency rehabilitation on 1160 ha that burned at high severity (about a third of the burned area) of the North 25 Fire to reduce postfire runoff and erosion (USDA Forest Service, 1998). This study is part of the postfire treatment effectiveness monitoring that was done.

1.1. Seed and fertilizer as rehabilitation treatments

During the past 50 years, the most common postfire emergency rehabilitation practice has been broadcast seeding of grasses, usually from aircraft (Robichaud et al., 2000). Rapid vegetation establishment has been regarded as the most cost-effective method to promote rapid infiltration of water, keep soil on hillslopes, and reduce sedimentation in channels and downstream areas (Noble, 1965; Rice et al., 1965; Miles et al., 1989). Grasses are particularly desirable for this purpose because their extensive, fibrous root systems increase water infiltration and hold soil in place. Generally, fast-growing non-native grass species are used because, unlike native species, they are inexpensive and readily available in large quantities when emergencies arise (Barro and Conard, 1987; Miles et al., 1989; Agee, 1993).

Robichaud et al. (2000) provide an extensive review of the reports of postfire seeding effectiveness. Because of the difficulty and expense involved in measuring hillslope erosion directly, most evaluations of seeding effectiveness have been reported in terms of ground cover or canopy cover produced, rather than any direct measurement of erosion reduction (Robichaud et al., 2000; Beyers, 2004). Although there are fewer postfire seeding impact studies from western conifer forests as compared to chaparral and rangeland areas, Robichaud et al. (2000) examined nine seeding studies in conifer forests that provided quantitative ground cover data. In the first growing season after the fire, about half of the studies reported less than 30% ground cover and only 22% reported at least 60% ground cover. In other words, the 60% to 70% ground cover needed for erosion reduction (Robichaud et al., 2000; Pannkuk and Robichaud, 2003), was attained in less than a fourth of the treated areas during the first growing season. After the second growing season, all of the studies reported at least 30% ground cover and a third had at least 60% ground cover. Depending on the timing of the fire, the first erosion-producing rain event can occur before the first growing season is complete. Thus, these studies suggest that seeding does not assure increased ground cover during the first critical rain events after a fire.

Postfire rehabilitation seed mixes often contained annual grasses to provide quick cover and perennials to establish longer term protection (Klock et al., 1975; Ratliff and McDonald, 1988). However, in a recent review of seeding effectiveness and impacts on native plant communities, Beyers (2004) found that when seeded grass growth generates enough ground cover to effectively control erosion, the grasses appear to displace native or naturalized species, including shrub and tree seedlings. Recent rehabilitation seed applications have used non-reproducing annuals, such as cereal grains or sterile grass hybrids, that provide quick cover and then die to let native vegetation reoccupy the site for long-term protection.

Over the past 30 years, chemical fertilizer has occasionally been applied in an effort to enhance the growth of postfire seeded species as well as regeneration of native species, with mixed results (Robichaud et al., 2000). On disturbed firelines, Klock et al. (1975) seeded various grasses and legumes and found that fertilization greatly increased initial cover of most species tested. On granitic soil in Idaho, fertilization with 50 kg ha^{-1} drilled urea significantly increased native plant regrowth, but not production of seeded species (Cline and Brooks, 1979). In contrast, Amaranthus (1989) found that annual ryegrass seeding and fertilization after an Oregon fire did not significantly increase plant cover or reduce erosion by early December, when the major rain events of winter occurred. He pointed out that timing of rainfall is critical to both grass establishment and erosion, and that different rainfall patterns could have produced different results (Amaranthus, 1989).

The combined treatment of seeding and fertilization has been tested very little (Robichaud et al., 2000), but the most significant work, relative to this study, was done after 1970 Entiat Fire swept through the Entiat Experimental Forest—located only 12 km from the North 25 Fire burned area. Tiedemann and Klock (1973) used four burned watersheds to compare the effects of postfire seeding (perennial grasses and clover) and fertilizer (ammonium sulfate and urea) on plant cover. At the end of the first growing season, plant cover on the three seeded watersheds was 9.3%, with 18% to 32% of the total being the seeded species, compared to 5.6% plant cover on the unseeded watershed. Flooding and debris torrents occurred in the study watersheds at the end of the first postfire year affecting subsequent plant cover amounts (Helvey, 1975; Tiedemann and Klock, 1976). Nonetheless, in terms of seeding effectiveness, these results are consistent with the majority of the studies reviewed by Robichaud et al. (2000) and Beyers (2004) where seeding did not produce sufficient ground cover to effectively reduce erosion.

1.2. Nutrient loss in sediment

The organic material within and above the soil is the primary source of most of the available phosphorus (P) and sulfur (S), and holds virtually all of the available nitrogen (N). In addition, the organic matter in the upper horizons of the soil provides the chemically active cation exchange sites for ammonium (NH_4–N), potassium (K), calcium (Ca), and magnesium (Mg) (DeBano et al., 1998). Several studies have attributed the losses and/or increases of nutrients due to forest fires to a variety of mechanisms including volatilization, leaching, and erosion (DeBell and Ralston, 1970; Grier, 1975; Robichaud et al., 1994; DeBano et al., 1998; Robichaud and Brown, 2000). The volatilization temperatures of N, S, P, and K are lower than the flaming temperatures of woody fuels (1100 °C); consequently, these nutrients are readily volatized from organic material above and within the soil during combustion (DeBano, 1991). N and S, with volatilization temperatures of 200 and 375 °C, respectively, are lost to the atmosphere in larger proportions than K and P, with a volatilization temperature of 774 °C for both (White et al., 1973; Raison et al., 1985; Tiedemann, 1987). During combustion, steep temperature gradients are produced in the upper soil layers, and some of the vaporized compounds are transferred downward and condense in the cooler soil (DeBano et al., 1976; Robichaud and Hungerford, 2000). Although large amounts of total N are lost during the combustion of plants and litter, NH_4–N in the underlying soil is usually higher immediately after a fire because of this downward translocation. P does not translocate downward as readily as NH_4–N and, as a result, increases mainly in the ash and on, or near, the soil surface (DeBano, 1991). The highly available nutrients that are located in the ash and upper soil layers are vulnerable to leaching into and through the soil profile as well as wind and runoff erosion processes (DeBano et al., 1998).

The postfire soil nutrient balance is strongly connected to site productivity, revegetation, and water quality (DeBano et al., 1998). A common rational for postfire seeding is the retention of site nutrients by reducing erosion, while the rational for fertilization is to replace nutrients lost through fire-induced processes. However, few fire effects studies have examined the soil nutrients that are removed from hillslopes in the eroded sediment, as opposed to dissolved in the runoff, from smaller upland watersheds (DeBano et al., 1998). In addition, the effect of fertilizer application on the nutrient content of the sediment has not been systematically measured.

This study was designed to examine the effectiveness of postfire seeding and fertilizer treatments, not only in terms of ground cover produced, but more importantly, through the direct measurement of hillslope erosion. Specifically, the objectives of this study were to compare the effects of two treatments–seeding and fertilizer application–on 1) postfire erosion rates; 2) postfire ground cover and vegetative regrowth; and 3) the amounts of soil nutrients in the eroded sediments.

2. Methods

2.1. Site description and study design

The North 25 Fire in north-central Washington ignited 4 Aug 1998 and burned 3200 ha before it was completely controlled in early November. The burned area was classified as 48% high severity, 12% moderate severity, and 40% low severity (USDA Forest Service, 1998). In late summer of 1998, eight sites (Grouse Mountain-1 and -2, Lone Peak-1 and -2, and View Point-1, -2, -3, and -4) were selected for erosion and vegetation recovery monitoring within high severity burned areas of the North 25 Fire (Fig. 1). The elevation of the eight sites ranged from 1341 to 1780 m with slopes from 28% to 54%. Four of the sites were on south aspects, two sites had east aspects, and two sites had west aspects (Table 1).

In this region, over 80% of the mean annual precipitation of 1000 mm occurs between October and March, with three fourths falling as snow. Most of the remaining precipitation falls during high intensity summer thunderstorms. July is usually the driest month with a mean precipitation of less than 20 mm (Western Regional Climate Center, 2004). The dominant geologic type is colluvium from grandiorite or rhyolite, and the soils are ashy, sandy loams (Typic Vitrixiands) (Palmich soil series) (USDA Forest Service, 1998). Earlier research has shown that these soils are often sulfur deficient (Klock et al., 1971; Tiedemann and Klock, 1973). Prior to the fire, subalpine fir/pinegrass (*Abies lasiocarpa*/*Calamagrostis rubescens*) and subalpine fir/grouse huckleberry (*Abies lasiocarpa*/*Vaccinium scoparium*) plant associations dominated the study area (Lillybridge et al., 1995).

At each site, four 4 by 9 m plots were established in October 1998, immediately following wildfire suppression efforts. Seed, fertilizer, and seed plus fertilizer were applied by hand to three of the plots in a randomized complete block design. The fourth plot at each site was left untreated as a control. White winter wheat (*Triticum estivum*) was applied at a rate of 34 kg ha^{-1} (~90 seeds m^{-2}), and the fertilizer, a blend of 75% ammonium nitrate and 25% ammonium sulfate, was applied at a rate of 31 kg ha^{-1}. These application rates matched, as closely as possible, the aerial application rates for the emergency rehabilitation treatments throughout the burned area (USDA Forest Service, 1998).

2.2. Precipitation

Precipitation was measured using a recording tipping bucket rain gauge (Fig. 1). Rain events with at least

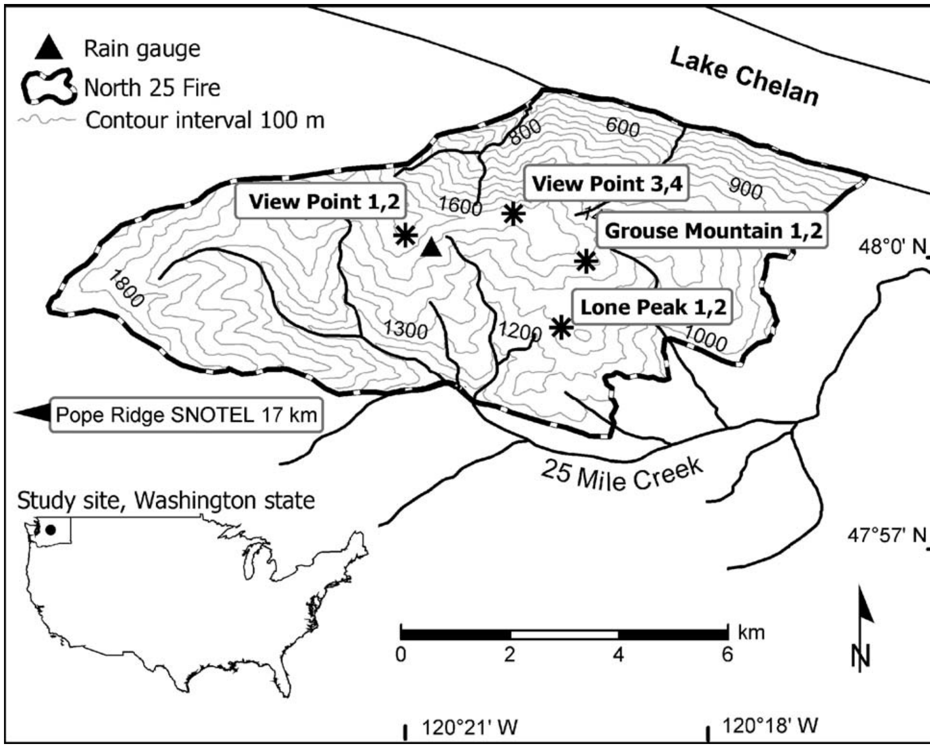

Fig. 1. Location of study plots and rain gauge within the North 25 Fire burn perimeter.

6 h between tips were considered separate storms. Since the sites were not visited after each rain event, the total rainfall, the average rain event duration, and the maximum 10-min intensity (I_{10}) that occurred between site visits were associated with each sediment clean-out. Precipitation between 1 Nov and 31 Mar was assumed to be snowfall. The Pope Ridge Snowfall Telemetry (SNOTEL) weather station, located 17 km west of the study area, was used to compare the study period to the climatic trends discerned from 21 years of records.

2.3. Surface cover and canopy cover

Surface cover (non-living material covering the mineral soil) and canopy cover (layers of live vegetation above the soil) were characterized separately. In many erosion studies, these two cover types are combined into a single ground cover measurement; however, because these data were to be used for several types of analysis, they were estimated separately using the mean of 10 subplot samples, taken within a 0.6 m square frame, in each plot (Daubenmire, 1959). The square subplots were first used for cover estimates in July 1999, and the frame locations were marked with pins. Cover estimates were repeated at the same locations in late July–early August 2000, late July 2001, and early–mid-August 2002.

The surface cover was estimated by summing the percent of area within the square sample frame covered by non-living material, specifically cobble (>75 mm), litter, and wood. Gravel (>2 mm) and charcoal were also estimated as components of surface cover; however, both of these materials moved downslope and were found in the collected sediments. Hence, these two categories were removed from surface cover for analysis. The percent canopy cover was estimated for each species within the subplots and summed to obtain the total canopy cover for each measurement period. Because species occurred at different heights above the soil surface, the sum of these cover amounts can exceed 100%.

Table 1
Elevation, slope, and aspect for the eight study sites

Site	Elevation (m)	Slope (%)	Aspect (degrees)
Grouse-1	1548	42	160
Grouse-2	1524	42	180
Lone Peak-1	1378	28	280
Lone Peak-2	1341	45	270
View Point-1	1780	53	110
View Point-2	1756	54	95
View Point-3	1670	30	155
View Point-4	1640	52	180

2.4. Sediment

In October 1998, sediment fences were installed along the bottom of each plot to collect eroded sediment, and trenches were dug along the top of each plot to limit the contributing area (Dissmeyer, 1982; Robichaud and Brown, 2002). Sediment collected in the silt fences was removed and weighed during 20 site visits between June 1999 and October 2002. Several grab samples from the collected sediment in each fence were used to measure the soil water content. The field-measured weights were then corrected by the soil water content to obtain dry sediment weights. The sediment removed from each plot was summed for each calendar year and divided by the plot area to obtain the annual erosion rate.

The organic matter content of each sediment sample was determined by the colorimetric method (Sims and Haby, 1971), and sample pH was measured using the saturation paste method (Rhoades, 1982; McLean, 1982). In addition, each sediment sample was analyzed for nutrient concentrations: sulfate (SO_4–S) by the calcium phosphate extractable method (Gavlak et al., 1997a); nitrate–nitrogen (NO_3–N) and ammonium–nitrogen (NH_4–N) by the potassium chloride extractable method (Keeney and Nelson, 1982); and potassium (K) and phosphorous (P) by the acetate extractable method (Gavlak et al., 1997b). The nutrient loads in the sediment were determined by the measured nutrient concentrations and the dry sediment weight for each plot. The nutrient loads in each sediment collection were calculated and summed for each year.

2.5. Statistical analysis

Since the cover and sediment yield data were not normally distributed, non-parametric techniques were chosen for analysis (Ott, 1993). Multi-response permutation procedures were used to test if differences existed in cover and annual erosion rates between treatments and between years (Mielke and Berry, 2001; King, 2003). Multiple comparisons were tested using a Peritz closure method (Petrondas and Gabriel, 1983).

To examine the effects of the treatments on nutrient data, repeated measures analyses (Littell et al., 1996) were conducted. With the exception of pH, these data were log transformed to increase the normality of the sample distributions and the uniformity of variance of the errors of the models (Ott, 1993). Differences between treatments and between years were tested using least squares means (Littell et al., 1996). For all statistical tests, $\alpha = 0.05$.

3. Results and analysis

3.1. Precipitation

Based on field observations, there were no sediment producing rain events between the installation of the sediment fences in October 1998 and April 1999, and snowmelt did not produce measurable sediment yields during any year of the study. There were a total of 299 rainfall events between 3 Apr 1999 and 24 Oct 2002, which resulted in twenty periods of sediment accumulation (Table 2). The majority (127) of spring/summer rain events were long duration, low intensity rain storms resulting from low pressure frontal systems that caused little or no measured erosion. Nine short duration rain events with a maximum 10 min rainfall intensity (I_{10}) of at least 13 mm h^{-1} caused most of the measured erosion (Table 2). The rain event on 25 Aug 2002 was a 10-yr, 30-min storm (Miller et al.,

Table 2
Sediment clean-out dates with number of rain events, mean duration, total rainfall, and date and amount of the maximum 10 min rainfall intensity (I_{10}) associated with each clean-out

Year	Sediment clean-out date(s)	Rain events between clean-outs (number)	Mean rain event duration (min)	Total rainfall (mm)	I_{10} (mm h^{-1})	Date of maximum I_{10}
1999	25 Jun–12 Jul	25	93	36.6	10.7	15 May
	26 Jul	6	58	8.9	7.6	18 Jul
	9–12 Aug	5	68	21.1	29.0	6 Aug
	16–18 Aug	2	325	12.7	13.7	14 Aug
	31 Aug–9 Sep	1	956	15.0	9.1	30 Aug
	2–9 Nov	9	186	26.7	7.6	28 Oct
2000	5–6 Jun	18	167	32.0	9.1	30 May
	13–14 Jun	4	343	17.5	3.0	11 Jun
	10–17 Jul	4	485	28.7	16.8	9 Jul
	31 Oct–7 Nov	16	265	45.5	16.8	4 Aug
2001	30 May–4 Jun	8	134	23.1	22.9	4 Apr
	2–3 Jul	6	547	38.9	9.1	12 Jun
	21 Aug[a]	5	340	19.8	9.1	21 Jul
	22–27 Aug	6	553	37.6	13.7	21 Aug
	25–26 Oct[a]	8	252	39.4	16.8	23 Oct
	8 Nov	9	437	53.8	16.8	23 Oct
2002	18–27 Jun[a]	24	128	45.0	10.7	17 Apr
	1–2 Jul	25	157	53.1	10.7	17 Apr
	5–9 Sep	7	73	40.1	38.1	25 Aug
	22–24 Oct	3	162	7.9	3.0	16 Sep

Snowmelt did not produce any measurable sediment yields during the study.
[a] An additional storm occurred before all sites were cleaned out during these periods.

Table 3
Fall/winter, spring, summer, and total annual precipitation during the study period compared to the 21-year mean from the Pope Ridge SNOTEL weather station

Water year	Precipitation				Annual precipitation compared to the 21-year mean (%)
	Fall/winter Oct–Mar (mm)	Spring Apr–Jun (mm)	Summer Jul–Sep (mm)	Annual (mm)	
1999	1016	74	33	1123	122
2000	775	56	25	856	93
2001	386	69	51	506	55
2002	673	76	28	777	84
21-yr mean	738	128	57	923	

1973). All other rain events had return periods of less than 1 year.

The data from the Pope Ridge SNOTEL weather station indicate that the August, 1998 North 25 Fire occurred during a drier than average year where no rainfall was recorded during August and September. The winter immediately following the fire was wetter than average (mostly due to heavy winter snowfall) followed by a drier than average spring and summer in 1999 (Table 3). The drier than average trend continued throughout the study years (1999–2002) with total precipitation in 2001, the driest year, of only 55% of the 21-year mean (Table 3).

3.2. Surface cover and canopy cover

In 1999, the mean surface cover (percent of soil covered by non-living material) for all treatments was 11%, and this value increased each year of the study to 31% in 2002 (Table 4). Most of this increase is due to the increase in mean litter cover, which went from 6% in 1999 to 30% in 2002. Neither the seed nor fertilizer treatments significantly increased the surface cover as compared to the untreated areas.

In 1999, the mean canopy cover (percent cover provided by live plants) for all plots was 18%, and this increased significantly ($p<0.001$) to 53% in 2000 (Table 4). In the first postfire year, the winter wheat provided 4.5% canopy cover on the seeded plots and only 0.3% on the unseeded plots (Table 4). The first year mean canopy cover included one plot (Grouse Mountain-2, no treatment) with a much greater than average canopy cover of 72%, of which 37% was fireweed (*Epilobium angustifolium*). If this plot is removed, the first year mean canopy cover on all plots ($n=31$) decreases from 18% to 16%, and on unseeded plots ($n=15$) decreases from 18% to 14%. By the second postfire year (2000), the mean winter wheat canopy cover had decreased to 0.5% in the seeded plots and 0.0% in the unseeded plots; and in 2001 and 2002 no winter wheat was recorded on any plots (Tables 4 and 5).

Given that the first year canopy cover amounts were the same for all treatments, the seeded plots, with a fourth of the canopy cover being seeded winter wheat, had a lower proportion of naturally occurring plant species than the unseeded plots (Table 4). Seeding did not affect the mean canopy cover from naturally occurring vegetation after the first growing season (Tables 4 and 5). The fertilized plots did have a greater mean canopy cover by the fourth year after treatment (Table 4), and because seeded winter wheat did not persist beyond the first year, this greater mean canopy cover was from revegetation by naturally occurring species (Table 5).

3.3. Erosion rates

The mean erosion rate for all plots in 1999 was 16 Mg ha^{-1} yr^{-1}. In 2000, the mean erosion rate was 0.66 Mg ha^{-1} yr^{-1}, which was significantly less than in 1999 ($p<0.0001$). In 2001, the erosion rate decreased again to 0.39 Mg ha^{-1} yr^{-1}. The lower erosion rates in 2000 and

Table 4
The mean surface cover and canopy cover by treatment for each year of the study

Year	Seeded S and SF plots	Unseeded NT and F plots	Fertilized F and SF plots	Unfertilized NT and S plots	All plots NT, S, F, and SF
Surface cover (%)					
1999	12 (14)[a]	9 (5.7)	10 (8.0)	12 (13)	11a[b] (10)
2000	19 (15)	15 (6.8)	16 (8.0)	18 (15)	17b (12)
2001	24 (18)	17 (6.6)	23 (14)	19 (14)	21b (14)
2002	33 (19)	29 (10)	35 (17)	27 (13)	31c (15)
Canopy cover (%)					
1999	17 [4.5][c] (12)	18 [0.3] (17)	18 (11)	17 (18)	18a (15)
2000	49 [0.5] (30)	56 [0.0] (20)	57 (24)	49 (26)	53b (25)
2001	57 [0.0] (22)	62 [0.0] (17)	66 (19)	53 (18)	59b (20)
2002	59 [0.0] (24)	70 [0.0] (19)	**74**[d] (20)	**55** (20)	64b (22)

In order to analyze the effects of seeding separately from the effects of fertilization, the treatment plots were combined ($n=16$) as seeded and unseeded, then recombined ($n=16$) as fertilized and unfertilized. Treatment designations: NT = no treatment, control; S = seed only; F = fertilizer only; and SF = seed and fertilizer ($n=8$ for each treatment).

[a] Standard deviation is in parenthesis following the mean cover value.

[b] Different letters within a set of 4 years indicate a significant difference at $\alpha=0.05$.

[c] Mean seeded winter wheat canopy cover (%) is in brackets.

[d] Bold type indicates a significant difference between treatments at $\alpha=0.05$.

Table 5
The mean canopy cover provided by the most abundant species is listed by year and by treatment

Plant species	Mean canopy cover (%)																
	Year	1999				2000				2001				2002			
	Treatment	NT	S	F	SF	NT	S	F	SF	NT	S	F	SF	NT	S	F	SF
Shrubs	Snowbrush (*Ceanothus velutinus*)	2.6	0.6	4.6	5.7	16	6.6	19	20	26	9.0	32	28	25	14	42	26
	Pachistima (*Pachistima myrsinities*)	0.8	0.4	0.7	0.5	2.0	1.4	2.3	1.5	3.2	1.1	3.0	2.8	4.2	2.3	4.2	3.2
	Currant species (*Ribes* spp.)	0.1	0.1	0.8	0.3	1.3	1.3	4.5	2.6	1.5	2.2	4.4	1.3	2.6	2.8	5.8	2.4
	Willow species (*Salix* spp.)	0.1	0.9	0.4	0.1	1.7	2.7	1.7	2.0	2.4	3.7	1.4	2.2	3.4	3.6	2.4	1.8
Herbs	Fireweed (*Epilobium angustifolium*)	4.8	0.8	2.0	0.7	7.6	7.1	7.9	10	10	9.0	7.6	15	4.2	4.9	4.1	7.0
	Lupine species (*Lupinus* spp.)	2.2	5.1	4.6	1.1	5.2	12	11	3.0	4.4	12	8.1	3.8	3.6	11	9.7	5.1
	Sedge species (*Carex* spp.)	0.8	0.5	1.1	1.2	2.9	2.4	3.8	1.9	3.6	3.7	3.8	3.0	3.4	2.7	3.4	2.7
	Pinegrass (*Calamagrostis rubescens*)	1.2	2.4	3.3	1.2	2.0	1.8	3.6	1.8	1.2	1.6	2.8	2.4	1.8	1.4	2.4	1.7
	Spreading groundsmoke (*Gayophytom diffusm*)	1.4	1.3	0.7	0.9	5.0	6.4	3.0	2.9	0.5	1.7	0.5	0.4	0.3	1.0	0.1	0.2
	Winter wheat (seeded)(*Triticum estivum*)	0.0	4.4	0.3	4.6	0.1	0.4	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Other species	2.8	1.2	1.0	0.6	5.3	5.8	6.4	4.3	3.1	5.2	4.8	5.2	10	6.8	7.5	17
	Mean canopy cover from all species combined	17	18	19	17	49	48	63	50	56	49	68	64	59	50	81	68

Mean canopy cover from all non-listed species is combined under 'other species'. Treatments designations: NT = no treatment, control; S = seed only; F = fertilizer only; and SF = seed and fertilizer ($n=8$ for each treatment).

2001 were due to a combination of vegetative recovery (Table 4), and all rain events having return periods much lower than one year. The mean erosion rate increased to 2.1 Mg ha^{-1} yr^{-1} in 2002 due to the 10-yr, 30-min rainfall event on 25 Aug (Table 2); however, this increase was not statistically significant.

Given the high variability of the erosion rate data (Fig. 2), it is not possible to show any significant differences in mean annual erosion rates between treatments for any year of the study. Even though there was a significant increase in 2002 fertilized plot canopy cover (Table 4), there was no accompanying reduction in erosion rate as compared to the unfertilized plots (Fig. 2).

When considered separately, the seeded plots at the Grouse Mountain-1 and Lone Peak-1 plots did show a significant reduction in erosion rates as compared to all other seeded plots ($p=0.02$ and $p=0.04$, respectively) (Table 6). In 1999, the seeded Grouse Mountain-1 plots had 44% surface cover, mostly from the litter within these plots. This value for surface cover was higher than all other

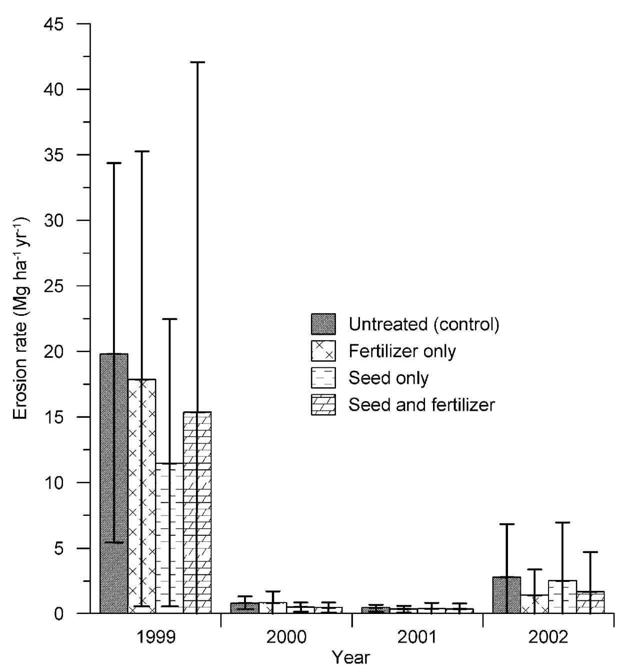

Fig. 2. Mean annual erosion rates by treatment ($n=8$) for 1999 to 2002. Error bars represent one standard deviation above and below the mean, except for values less than zero.

Table 6
Mean annual erosion rate, surface cover, and canopy cover, for Grouse Mountain-1, Lone Peak-1, and all other sites

Year	Grouse Mountain-1		Lone Peak-1		All other sites	
	Seeded	Unseeded	Seeded	Unseeded	Seeded	Unseeded
Erosion rate (Mg ha^{-1} yr^{-1})						
1999	0.0	31	2.0	16	17	17
2000	0.0	0.4	0.04	0.6	0.6	0.9
2001	0.0	0.3	0.05	0.3	0.5	0.4
2002	0.01	0.5	0.02	0.2	2.8	2.7
Surface cover (%)						
1999	44	16	5	4	8	9
2000	47	20	22	12	14	14
2001	55	22	18	19	20	16
2002	46	39	17	21	34	29
Canopy cover (%)						
1999	7	11	30	17	17	19
2000	30	45	60	46	50	60
2001	47	82	64	64	57	58
2002	53	94	75	61	57	67

Mean erosion rates for Grouse Mountain-1 seeded ($n=2$) and Lone Peak-1 seeded ($n=2$) plots were significantly less than from all other sites ($n=12$) at $p=0.02$ and $p=0.04$ respectively. Grouse Mountain-1 seeded plots had more surface cover than all other sites ($p=0.01$).

Table 7
Mean annual nutrient loads in the eroded sediments from all sites are listed by year as well as the four year totals

Year	Nutrients					Erosion rate (Mg ha^{-1})	Organic material (%)
	NO_3–N (kg ha^{-1})	NH_4–N (kg ha^{-1})	SO_4 (kg ha^{-1})	P (kg ha^{-1})	K (kg ha^{-1})		
1999	0.81 (95%)[a]	0.14 (93%)	0.17 (100%)	0.64 (63%)	2.9 (100%)	16 (84%)	5.2
2000	0.00	0.00	0.00	0.01	0.00	0.7	6.8
2001	0.00	0.00	0.00	0.01	0.00	0.4	5.9
2002	0.04	0.01	0.00	0.36	0.01	2.1	5.9
4 yr total	0.85	0.15	0.17	1.0	2.9	19	

Mean annual erosion rates and percent organic material in the sediment are also shown.
[a] The proportion of the four year nutrient load and erosion rate collected in the first year is shown in parentheses.

plots for that year—significantly higher than the 8% average for all other seeded sites ($p=0.01$), and higher than the average for all other sites for the subsequent three years of the study. Although the Lone Peak-1 seeded plots had a lower than average amount of surface cover in 1999 (5%), and near or below the overall average for 2000–2002, the canopy cover for these two plots was greater than the average for all other seeded plots during all four years of the study (Table 6). These differences in canopy cover were not significant, but in 1999, the Lone Peak-1 seeded plots had 30% canopy cover, which was nearly double the 17% mean canopy cover found on all other seeded sites (Table 6).

Three rain events in 1999 caused 10 of 32 sediment fences to be overtopped once, and two other sediment fences to be overtopped on two separate occasions. The amount of sediment lost from these fences was estimated based on visual observations of downstream deposition and judged to be negligible for 11 of the 12 sediment fences. The sediment lost from the remaining single fence, View Point-2 seed and fertilizer plot, was estimated and added to

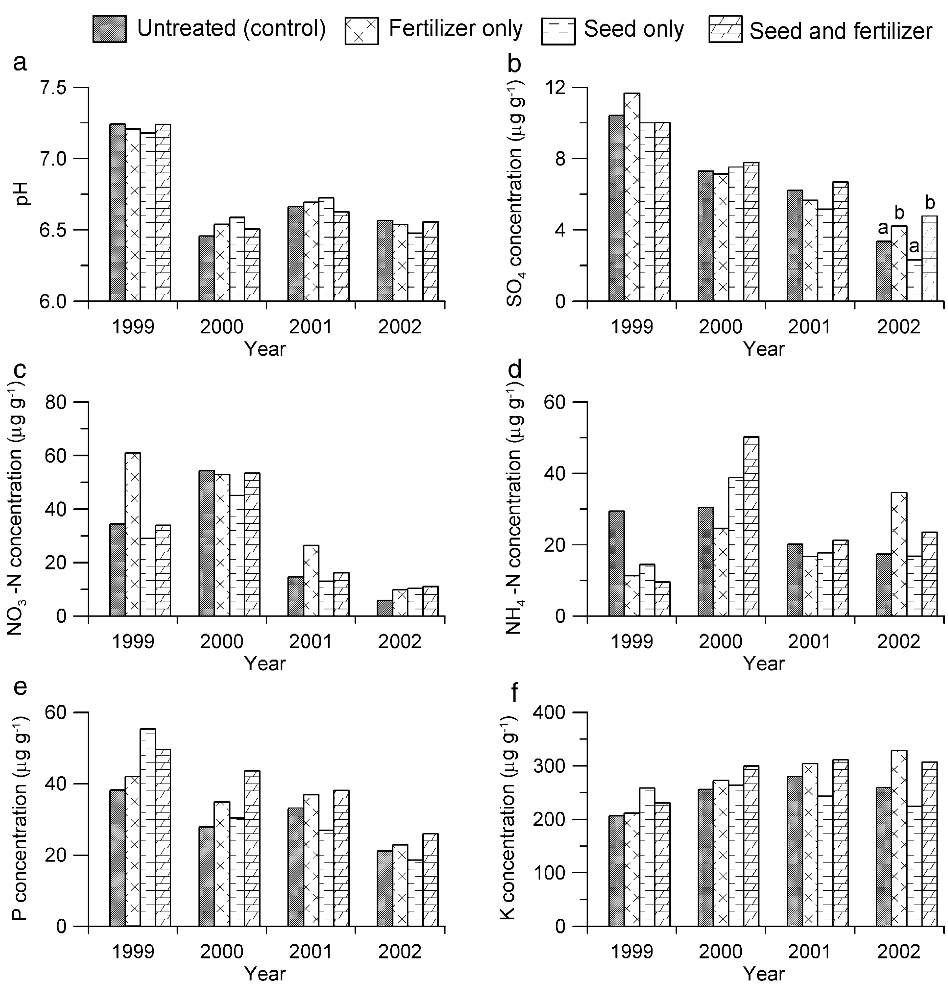

Fig. 3. a–f. The mean sediment pH (a) and mean sediment concentrations of SO_4 (b), NO_3–N (c), NH_4–N (d), P (e), and K (f) by year and by treatment. Different letters on the top of a set of bars indicate a significant difference between treatments at $\alpha=0.05$.

the measured value for two rain events. These estimates of lost sediment did not affect the non-significance of the treatment effect ($p=0.50$ with estimates included, $p=0.57$ with estimates excluded) or the significance of the reduction in erosion from 1999 to 2000 ($p<0.0001$ for both cases).

3.4. Organic matter, pH, and nutrients

The mean organic matter content in the eroded sediment was 5.2% in 1999, 6.8% in 2000, and 5.9% in 2001 and 2002 (Table 7). In 1999, the average pH of the eroded sediment was 7.2 and decreased to an average of 6.5 to 6.6 for 2000 through 2002 (Fig. 3a). The mean concentrations of some nutrients in the eroded sediments changed over time–SO_4 and P decreased each year and K generally increased–but no discernable trend was observed for NO_3–N or NH_4–N (Fig. 3b–f). The pH of the sediment as well as the concentrations of organic matter, NO_3–N, NH_4–N, and K in the collected sediments were not affected by seeding or fertilization. A fertilizer treatment effect was observed in the 2002 sediment concentrations of SO_4 (Fig. 3b).

Over 90% of the four year total sediment nutrient load was collected in 1999, except for P with 63% of the total load in 1999 and 35% in the fourth year, 2002 (Table 7). These proportions of total sediment nutrient loads generally reflect the proportion of total sediment loss, where 84% of the four year total sediment was collected in 1999, the first postfire year, and 11% was collected in 2002 (Table 7). Although differences in the sediment nutrient loads between treatments were sometimes statistically significant, the magnitudes were small.

4. Discussion and conclusions

The success of seeding as a postfire emergency rehabilitation treatment is largely dependent on the timing and amount of spring and summer rainfall. Although the winter (1998–1999) preceding this study was wetter than average, the spring (April through June) of 1999 was dry (Table 3). The total rainfall for this three month period was 74 mm, just 58% of the 21-yr mean. Field observations noted that seeded winter wheat which had germinated died due to lack of spring moisture (USDA Forest Service, 2001). However, fire cycles often follow drought cycles, and it is not surprising to have several drier than normal years before and after a fire. The lower than average rainfall in the years following the North 25 Fire (Table 3), is a common postfire condition, and should be considered when evaluating the efficacy of postfire seeding.

The first postfire year (1999) erosion rate, 16 Mg ha^{-1} yr^{-1}, was consistent with results from previous studies on postfire erosion (Robichaud and Brown, 2000; Dean, 2001; Spigel, 2002; Wagenbrenner, 2003). Although postfire erosion rates are highly variable, they generally decrease by an order of magnitude with each year of recovery (Robichaud et al., 2000). In this study, general postfire recovery processes were augmented by the low number of high intensity rainstorms in 2000 and 2001 and the drier than average summers in years 2000 to 2002 resulting in lower second and third year erosion rates—0.66 Mg ha^{-1} yr^{-1} in 2000 and 0.39 Mg ha^{-1} yr^{-1} in 2001. Then, in 2002, a 10-year, 30-min rain event increased the erosion rate to 2.2 Mg ha^{-1} yr^{-1}. Had this event occurred earlier in the study, prior to three years of vegetative recovery, the measured erosion rate would likely have been much higher.

4.1. Effects of seeding and fertilization on erosion rates

There was no reduction in erosion rates for either the seeding or fertilization treatments. To effectively reduce erosion during the first postfire year, an effective ground cover of 60% to 70% is needed (Robichaud et al., 2000; Pannkuk and Robichaud, 2003), and this level of ground cover was never attained on either the treated or the untreated sites. The two exceptions, Grouse Mountain-1 with higher surface cover and Lone Peak-1 with higher canopy cover, demonstrate the effect of soil-protecting cover on erosion. The greater canopy cover in Lone Peak-1 seeded plots, like the greater surface cover in the Grouse Mountain-1 seeded plots, probably reduced erosion rates at these sites.

Surface cover and canopy cover were evaluated separately in this study, but, in terms of erosion, both types of cover can intercept rainfall to reduce raindrop impact, increase infiltration, and thereby reduce overland flow. The 20% increase in surface cover between 1999 and 2002 was due to the increase in natural vegetation, such as snowbrush (*Ceanothus velutinus*), fireweed (*E. angustifolium*), and lupine (*Lupinus* spp.) (Table 5), which subsequently increased the litter component of surface cover. Seeding did not affect canopy cover amounts and the seeded winter wheat did not persist beyond the first growing season. In year two there was a 36–38% increase in canopy cover that occurred fairly evenly across all treatments (Table 4). There were more modest increases in canopy cover in 2001 and 2002, and a significant fertilization treatment effect on canopy cover in 2002 (Table 4). Although this study did not evaluate the treatment effects on individual species, Table 5 suggests there was no obvious species-specific response to seeding or fertilizing by the plants which revegetated the area during the four years of the study.

The first year canopy cover measured in this study compares reasonably well with the first year plant cover measured by Tiedemann and Klock (1973). In this study, if the 1999 high canopy cover (72%) on the Grouse Mountain-2, no treatment plot is removed, the first year mean canopy cover on the remaining no treatment plots ($n=7$) decreases from 17% (Table 5) to 9%. A 9% canopy cover value is similar to the plant cover value of 5.6% reported by

Tiedemann and Klock (1973) for the untreated watershed in their study. Tiedemann and Klock (1973) did not measure erosion, but it is unlikely that the plant cover reported would have had any effect on erosion rates.

The postfire seeding and fertilization treatments did not reduce postfire erosion at any time during this study. In order for annual or sterile grasses to establish sufficient cover to effectively reduce postfire erosion, periodic low intensity rainfall is needed for germination and to sustain growth. In addition, this germination and growth needs to occur before the first erosion-producing precipitation event occurs. Consequently, the effectiveness of seeding sterile or annual grasses as a postfire rehabilitation treatment is highly dependent on rainfall intensity, amount, and timing. If the purpose of postfire treatment is erosion control, the decision to use grass seeding should be based on the probability of a cover-producing rainfall pattern occurring before a major erosion-producing rainfall.

4.2. Effects of seeding on native plant revegetation

Beyers (2004) reported that when postfire seed growth provides enough cover to substantially reduce erosion (60% to 70%), it generally suppresses revegetation by naturally occurring species. This effect was verified by Keeley (2004) in a study of a burned site in California that had been seeded with winter wheat at a rate of 157 kg ha^{-1}, nearly 5 times greater than the application rate of 34 kg ha^{-1} used in this study. The postfire seeding treatment resulted in 95% first year ground cover, with 67% being seeded winter wheat. Total species richness was 152 species on unseeded sites compared to 104 species on seeded sites (Keeley, 2004). On the other hand, given the lack of effective canopy cover provided by the seeded winter wheat in our study, it is not surprising that seeding had little impact on the reestablishment of naturally occurring vegetation even during the first postfire year. The increasing canopy cover measured annually in 2000–2002 provides a measurement of postfire vegetative recovery for this area, and suggests that the effect of seeding with annual or sterile grasses may be inconsequential by the second year of recovery.

4.3. Nutrient losses through erosion

Except for SO_4 in the fourth year of the study, nutrient concentrations in the eroded sediments were not significantly impacted by either the seeding or fertilization. Based on earlier studies (Tiedemann and Klock, 1973), there was a concern that SO_4 may be a limiting nutrient in this area; therefore, the fertilizer mix that was applied included 25% ammonium sulfate. In the fourth year of the study, the SO_4 concentration in the sediments from the fertilized plots was small (less than 6 μg g^{-1}) but significantly greater than the unfertilized plots (Fig. 3). Given that this difference was not observed in years one through three, no conclusive connection between fertilizer application and SO_4 concentration in the sediments can be made.

Fire effects certainly include changes in the soil nutrients; however, the sampled eroded sediments contained negligible amounts of these nutrients and are unlikely to affect long-term site productivity (Johnson and Curtis, 2001). The magnitudes of the nutrient loads in the eroded sediment (Table 7) were minimal when compared to the characteristic total nutrient pools in the mineral soil (Means et al., 1992; Busse, 1994; Baird et al., 1999), nutrient losses from burning (Feller, 1988), or amounts of nutrients in the surface ash deposited after a conifer forest wildfire (Grier, 1975). Helvey et al. (1985), in a study of nutrient losses after the 1970 wildfire in the Entiat Experimental Forest, stated that most nutrients in sediment are bound into stable inorganic compounds and are not readily available to plants, whereas the nutrients in solution are often in more available (exchangeable) chemical forms. They concluded that, in terms of site productivity, 'solution losses are probably more important than the larger nutrient losses from soil erosion and debris torrent processes' (Helvey et al., 1985, p. 172). In this study, the measured nutrient losses due to erosion did follow the trend of erosion rates with more than 90% (except for P at 65%) of the losses occurring in the first year, and near zero losses in subsequent years. However, the magnitude of these losses was minimal and likely had little to no affect on site productivity. Although postfire erosion may impact downstream water quality, sedimentation, and property, it is unlikely to affect hillslope productivity.

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