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Mulching effects on vegetation recovery following high severity wildfire in north-central Washington State, USA

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ABSTRACT

Straw mulch application after high severity wildfire has gained favor in recent years due to its efficacy in reducing soil erosion hazards. However, possible collateral effects of mulching on post-fire vegetation recovery have received relatively little study. We assessed mulching effects on plant cover and species richness, tree seedling establishment, and exotic species densities in the second year following the 2006 Tripod Wildfire in north-central Washington State, USA, by observing vegetation responses to spatial variability in mulch cover and depth. Mulch cover averaged about 35%, with a median depth of 0.5 cm. Vegetation recovery was generally slow, with median plant cover of only 10%. Tree seedling densities were low and spatially variable. Vegetative cover, species richness, and seedling densities all declined with increasing elevation. Mulch cover was positively associated with plant cover, plant species richness, and conifer seedling densities when second year mulch cover did not exceed 40%. Only when mulch cover exceeded 70% did mulching begin to negatively affect vegetation recovery relative to areas with no mulch. Vegetation responses to mulch depth were minimal at depths under 3 cm, but quite strong when mulch depth exceeded 5 cm. Exotic plant frequency and density were positively associated with mulch cover, but exotic plant cover was low on average (<1%). In this study, mulch added significant cover to sites with slow natural recovery of vegetation, thereby likely reducing erosion hazard. Mulching also appears to have facilitated native plant recovery and conifer seedling establishment except at very high application levels, easing management concerns about longer-term impacts of mulching treatments on post-fire vegetation recovery.

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1. Introduction

Monitoring and assessment of the effectiveness and ecological impacts of post-fire rehabilitation and restoration treatments is a critical step for implementing adaptive management in post-fire forest and range management. High severity wildfires can increase erosion and flooding hazards by reducing ground cover, exposing bare mineral soil, and altering soil physical properties (DeBano et al., 1998; Benavides-Solorio and MacDonald, 2001; Wondzell and King, 2003). Hillslope stabilization treatments such as seeding or mulching are often used following high severity wildfires to reduce these threats and protect important resources and habitats (Robichaud et al., 2000; Beyers, 2004). Hillslope stabilization treatments should ideally act quickly to increase soil cover and reduce erosion and flooding hazards, without impeding natural vegetation recovery (USDA, 1995). Unfortunately, monitoring and reporting of treatment effectiveness and collateral ecological impacts is often lacking (U.S. General Accounting Office, 2003).

Mulching is a hillslope stabilization treatment designed to directly provide organic cover for burned soils, thereby reducing surface erosion caused by raindrop impacts, surface runoff, and transport of eroded sediment (Bautista et al., 1996; Robichaud, 2005; Wagenbrenner et al., 2006; Napper, 2006; Groen and Woods, 2008). As a treatment that is not dependent on biological processes for success, it is most effective in the first year after wildfire, when vegetation recovery is just beginning and soil erosion and runoff hazards are highest (Robichaud, 2005), with diminishing effectiveness in later years as mulch decomposes. The rapid establishment of predictable cover, combined with uncertain efficacy of alternative treatments (Robichaud et al., 2000, 2008) has led to increased mulch usage in recent years, despite high costs, especially in high risk areas (Robichaud, 2005; Robichaud et al., 2009).

The ecological effects of mulch treatments have been variable and may depend on the application rate and site factors. Previous work suggests mulch can inhibit plant establishment, reduce tree regeneration, and introduce exotic species (Beyers, 2004; Kruse et al., 2004; Robichaud, 2005). Mulch may inhibit vegetation establishment and growth by covering bare mineral soil seedbeds, shading emerging plants, reducing nitrogen availability, or producing alleopathic effects (Wu et al., 2001; Kruse et al., 2004). Reductions in plant establishment and growth are more likely when

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deep piles of mulch develop (Robichaud et al., 2000, 2009). Alternatively, mulch may promote plant establishment and growth by retaining soil moisture, which could be particularly beneficial on dry sites (Robichaud et al., 2000). Indeed, previous studies have shown that post-fire mulching was associated with increased plant growth or cover (Bautista et al., 1996, 2009; Badía and Martí, 2000; Peterson et al., 2009).

Introduction of exotic plant species is another potential collateral effect of mulching. Invasion by exotic species is well recognized as a threat to ecosystem recovery following wildfire (Crawford et al., 2001; Freeman et al., 2007). Mulching has the potential to facilitate exotic plant introduction and establishment by transporting seeds on contaminated mulch and improving establishment conditions for locally present exotic plant species (Kruse et al., 2004; Robichaud, 2005). Mulch applied on public lands is frequently required to be "weed-seed free" (Beyers, 2004); however, weed-seed free mulch may still carry seeds of exotic species (Robichaud et al., 2003) or such seeds can be introduced at the staging area (Faust, 2008). Kruse et al. (2004) found increased exotic species abundance following post-fire mulching in California, but overall the potential for mulching to introduce exotic species has received little study.

The purpose of this study was to assess the effects of straw mulch cover and depth on live plant cover, plant species richness, tree seedling densities, and exotic plant abundance (presence and density) following the 2006 Tripod Fire. We employed an observational approach, using variability in mulch cover and depth following operational mulching treatments to evaluate mulching intensity effects on vegetation recovery. We sought to answer three questions:

(1) Does mulching intensity (cover or depth) significantly influence live plant cover, plant species richness, tree regeneration densities, or exotic species densities after wildfire?
(2) Are there thresholds of mulching intensity, below which treatment effects on vegetation recovery are neutral to positive?
(3) How does topographic position (elevation, slope, aspect) affect native vegetation recovery rates and the need for land surface treatments to reduce erosion?

2. Methods

2.1. Study site

The Tripod Wildfire began on July 24, 2006 and burned about 80,000 ha in north-central Washington State, USA. About 24% of the burned area was classified as high burn severity and 27% at moderate burn severity (USDA Forest Service, 2006). The fire mostly burned coniferous forest, but forest types varied with elevation. Ponderosa pine (*Pinus ponderosa* C. Lawson) and Douglas-fir (*Pseudotsuga menzesii* (Mirb.) Franco) dominated low elevation forests, while higher elevation forests were dominated by lodgepole pine (*Pinus contorta* Douglas ex Louden) and subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.).

Precipitation at the nearest weather station (Winthrop, about 6 km west of fire boundary at an elevation of 539 m) averages about 35 cm/year, with about 23% of precipitation falling during the May–August growing season (Western Regional Climatic Center, Winthrop, http://www.wrcc.dri.edu). Climate within the study area is cooler and wetter, with estimated mean annual precipitation of 60–78 cm and mean annual temperatures of 1.4–5.6 °C (PRISM, 2009). Annual precipitation during the study year (2008) was about 83% of normal at Winthrop, and growing season precipitation was about 65% of normal (Western Regional Climatic Center, Winthrop, http://www.wrcc.dri.edu). The first post-fire year (2007) also had below average annual (80% of normal) and growing season (52% of normal) precipitation.

The terrain is mountainous with high relief and steep slopes. Soils are mostly well drained sandy loams derived from volcanic ash deposited over glacial till and glacial outwash (USDA Forest Service, 2006).

2.2. Mulch application

After the wildfire, BAER teams prescribed mulching to control soil erosion on 37 management units (treatment polygons) where high severity fire consumed most of the vegetation and soil organic matter, slopes were less than 60%, surface rock was not abundant, and elevated soil erosion and runoff posed threats to downstream resources. Helicopters applied straw mulch at a prescribed intensity of 2.24 Mg/ha, treating about 3480 ha in the autumn of 2006, immediately following the fire, and about 2226 ha shortly after snowmelt in the spring of 2007. Mulch consisted of certified weed-seed free wheat straw from eastern Washington, Idaho, and Montana.

2.3. Vegetation sampling

We assessed mulching effects on vegetation recovery by examining correlations between spatial variability in mulch cover and vegetation cover (or density) within treated management units in 2008, the second growing season following fire. Mulching resulted in an uneven distribution of mulch within treatment units, including areas with no mulch that served as baseline levels for vegetation responses in this study. We sampled vegetation along 18 transects in 14 management units (Table 1). Sampled mulch units were chosen subjectively from the 37 total units based on size (large enough to accommodate sampling) and proximity to roads (accessibility). Transects were 100 m long and were oriented along slope contours. We selected the starting point for each transect by first subjectively selecting an initial point that was at least 50 m from the road and any residual live trees, and then applying a random offset of 0–25 m in a random direction. We placed a 0.25 m^2 sampling quadrat at the beginning of each transect and placed subsequent quadrats at random intervals of 1–5 m along the transect, resulting in totals of 29–36 quadrats per transect.

At each sampling location, we identified and recorded all vascular plant species found within a square 0.25 m^2 quadrat. Difficulties in identification led to some species being identified only to the genus level. We assigned a unique identifier to 3 species that could not be identified to genus. We estimated relative cover of vegetation (including non-vascular species), litter (including mulch), rock, wood, and soil based on what a vertically falling raindrop would hit first so that the total cover at each sampling point summed to 100%. We estimated absolute mulch cover separately by disregarding overtopping vegetation and litter. Cover values were estimated to the nearest percent. We also measured mulch depth at 4 systematically placed points in each quadrat.

For exotic plants and conifer seedlings, we counted individuals by species within the sampling quadrat and also estimated absolute cover. Exotic species were defined as non-native to the contiguous U.S. (USDA NRCS, 2007, PLANTS database; Table 2).

2.4. Analyses

We calculated species richness from species lists at each sampling point for exotic plants and all vascular plants combined. We also calculated mean and maximum mulch depth at each sampling point from the 4 depth measurements. We calculated heat load index for each transect (McCune and Keon, 2002; Eq. (1)), giving minimum values to northeasterly aspects. To assess envi-

Table 1
Site characteristics for the 18 transects in the Tripod fire mulch units.

Transect	Elevation (m)	Aspect (°)	Slope (%)	Heat load index[a]	Mulch cover (%)	Plant cover (%)	Species richness	Exotic (density/m^2)
1	1750	162	18	−0.29	57.36	8.13	2.28	11.03
2	1786	279	28	−0.20	21.57	5.30	1.74	0.06
3	1596	349	35	−0.59	19.31	16.18	2.53	2.13
4	1627	255	42	−0.08	30.63	17.72	2.69	8.69
5	1492	338	14	−0.44	21.38	33.28	3.59	0.66
6	1546	195	10	−0.32	51.15	8.68	1.42	0.09
7	1604	253	21	−0.22	32.76	19.00	2.58	0.76
8	1362	331	42	−0.47	32.03	37.59	4.50	1.91
9	1297	53	40	−0.95	41.60	20.93	3.70	1.03
10	1281	36	57	−1.16	29.70	24.30	4.30	3.21
11	1331	72	55	−1.00	40.47	27.00	5.18	3.94
12	1440	354	50	−0.73	28.13	28.67	3.27	0.87
13	1868	207	56	−0.04	33.33	12.06	1.59	0.28
14	1529	309	35	−0.31	45.39	5.38	2.36	2.55
15	1637	286	33	−0.20	38.42	18.72	3.03	4.82
16	1699	328	35	−0.43	36.60	13.39	3.26	0.40
17	1716	220	25	−0.22	12.24	8.06	1.32	0.18
18	1546	89	25	−0.60	58.23	30.46	2.49	2.86

[a] From McCune and Keon (2002), Eq. (1).

ronmental effects on mulch and vegetation recovery, we calculated Pearson correlation coefficients between transect-level environmental variables (elevation, heat load index, percent slope) and transect averages for mulch cover and depth, vegetation cover, and species richness.

Prior to analyses, we chose a type I error rate of 5% ($P < 0.05$) for evaluating statistical significance of mulching effects. We partitioned variability in mulch cover and vegetation response variables into within- and between-transect components using unconditional means mixed-effects models (Singer, 1998; Raudenbush and Bryk, 2002) with only a random term for each transect.

We evaluated mulch cover effects on plant cover and species richness by developing a series of random-intercept-and-slopes mixed-effects models using SAS PROC MIXED (Littell et al., 2006; SAS Institute, 2008, version 9.2). We included the transect identifier as a random term to account for probable non-independence of quadrats grouped within transects. We assessed mulch unit as another potential random term (4 units had more than one transect), but it was not significant and was dropped from all final models. We included a random transect–mulch interaction term to assess potential variability in mulching effects among transects (random slopes model; Singer, 1998). We included mulch cover (linear and quadratic terms) as a fixed effect at the sample point level and heat load index, slope angle, and elevation as fixed effects at the transect-level. For each response, we first fit the full model with all fixed and random effects (including interactions) and then used a backward elimination procedure to remove non-significant effects. We evaluated model residuals for adherence to assumptions of independence, equal variance, and normality. Plant cover required a square root transformation in order to meet these assumptions. For total plant cover and species richness, we added random variables to the models to model error variances individually for each transect (Littell et al., 2006) after preliminary analyses suggested potential violation of the equal variance assumption.

Mulch depth appeared to have a non-linear effect on plant cover that transformations could not convert into a linear relationship. To evaluate the effects of mulch depth on plant cover, therefore, we applied a sigmoidal three-parameter Gompertz equation (SigmaPlot10 for Windows, Systat Software Inc., San Jose, CA, USA).

Tree seedlings and exotic species occurred on less than half of the sampling quadrats. Therefore, we used mixed logistic regression models (PROC GLIMMIX; SAS Institute, 2008, version 9.2) to assess mulch cover effects on the probability of at least one tree seedling or exotic plant species being present in a sample. We developed separate models for lodgepole pine and trembling aspen (*Populus tremuloides* Michx.), the only 2 tree species encountered. A square root transformation of mulch cover was necessary to meet model assumptions for lodgepole pine. We also used mixed logistic regression to assess mulch effects on the probability of an exotic species being present in a sampling quadrat. Because wheat (*Triticum aestivum* L.) is an exotic species, but also expected to be encountered regularly due to residual seed heads in the straw mulch, we analyzed exotic species presence both with and without wheat. For each seedling species and exotic species group, we initially fit a full model with mulch cover (linear and quadratic terms) and transect-level environmental variables as fixed effects and transect as a random effect. We eliminated non-significant variables using a backward elimination process to arrive at the final models.

Finally, exotic species density varied considerably among the sampling quadrats where exotic species were present. Therefore, we used a conditional linear mixed model to assess mulching effects on exotic plant density, given that exotic plants are present (a subset of the sampling points), using the same modeling process as above. Exotic plant density required a logarithmic transformation to meet model assumptions of normality and equal variance.

3. Results

3.1. Mulch cover and depth

Mulch cover in the second growing season following fire averaged 35.3% across the 18 sampling areas. Point estimates of mulch cover ranged from 0% to 100%, with a median value of 25%. Mulch cover was less than 5% on 80 quadrats, which provided a baseline for vegetation responses. At the transect-level, mean mulch cover ranged from 3.1% to 59.7%. However, variance partitioning with the unconditional means model indicated that over 85% of the variability in mulch cover was due to variability within sampling transects, suggesting a high potential for distinguishing mulch cover effects from transect-level environmental effects. Mulch cover did not vary significantly with aspect (heat load index) or elevation (both P-values > 0.30), but did vary with slope angle ($P = 0.05$), with mulch cover being somewhat lower on steep slopes than on gentle slopes.

Mulch depth averaged 1.6 cm overall, with a range of 0–24 cm and a median depth of 0.5 cm. Spotty cover produced 172 quadrats with no measurable mulch depth (at any of the 4 predetermined

Table 2
Exotic species present in the mulch units, with number of occurrences and status as a noxious weed (in Washington State) for each species.

Species	Occurrences	Noxious weed (WA)
Anthemis cotula	3	No
Bromus tectorum	17	No
Chenopodium album	35	No
Cirsium vulgare	1	Yes
Lactuca serriola	349	No
Logfia arvensis	60	No
Lolium perenne	2	No
Polygonum aviculare	22	No
Sisymbrium altissimum	252	No
Sonchus spp	115	Yes[a]
Spergularia rubra	4	No
Taraxacum officinale	3	No
Triticum aestivum	656	No
Tragopogon dubius	3	No

[a] Only one species in the genera *Sonchus* is designated as a noxious weed in Washington State.

measurement points), which was much greater than the number of quadrats with no mulch cover. Mulch depth exceeded 5 cm on only 42 of the 591 sampling quadrats. Over 90% of the variability in mulch depth was due to variability within transects. Mulch depth was not significantly related to slope, elevation, or heat load index ($P > 0.2$ for all transect-level predictors).

3.2. Plant cover and species richness

Plant cover was strongly correlated with vascular plant species richness ($r = 0.74$). Plant cover (vascular and non-vascular species) averaged 18% overall, with a median cover of 10%. Mean plant cover for individual transects ranged from 5% to 38%, but variance partitioning indicated that 79% of the variability in plant cover occurred among samples within transects, suggesting a high potential for observing mulching effects within and among sampling transects. Vascular plant species richness averaged 3 species per sample (0.25 m^2). Mean species richness at the transect-level ranged from 1 to 5 species per sample, but almost 80% of the variability in plant species richness occurred among samples within transects.

Elevation was negatively correlated with mean plant cover and species richness at the transect-level. Mean plant cover declined by an average of almost 4% for each 100 m of increasing elevation ($P < 0.01$), with elevation explaining 44% of the variation in mean plant cover among transects (Fig. 1a). Mean species richness also declined by about one species for every 200 m of increasing elevation, with elevation explaining 63% of the variability in species richness among transects (Fig. 1b). None of the other transect-level variables were significantly correlated with plant cover or species richness, however.

Plant cover generally declined with increasing mulch depth, but in a non-linear way (Fig. 2). Mulch depths of less than 3 cm produced little or no effect on plant cover. However, plant cover was negatively correlated with mulch depth at higher levels, and predicted plant cover declined to near 0% when mulch depth exceeded 7–8 cm (Fig. 2).

Mulch cover produced only weak effects on plant cover and species richness (Table 3 and Fig. 3). There was some evidence that mulching reduced mean plant cover when second year mulch cover exceeded 75%. At lower levels, however, mulching effects on plant cover appeared to be neutral to positive relative to no mulching (Fig. 3a). There was no evidence that mulching reduced total plant species richness at any level, and species richness was maximized at intermediate levels of mulch cover (Fig. 3b).

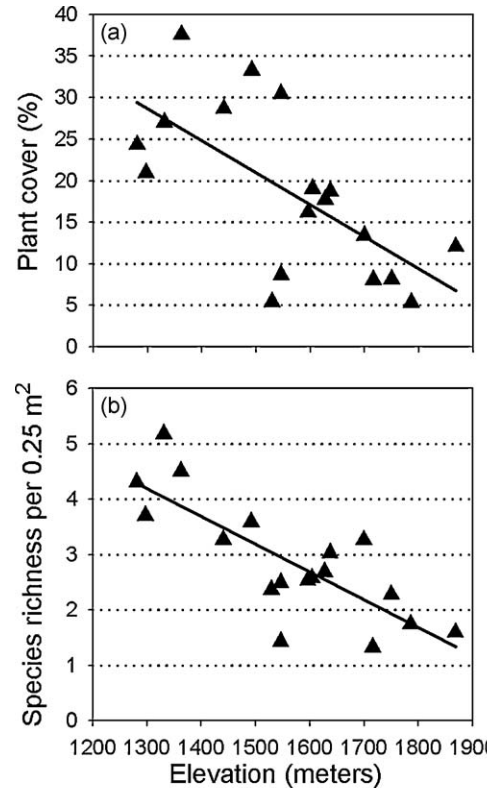

Fig. 1. Effects of elevation on (a) total plant cover and (b) plant species richness. Lines represent linear regression model predictions.

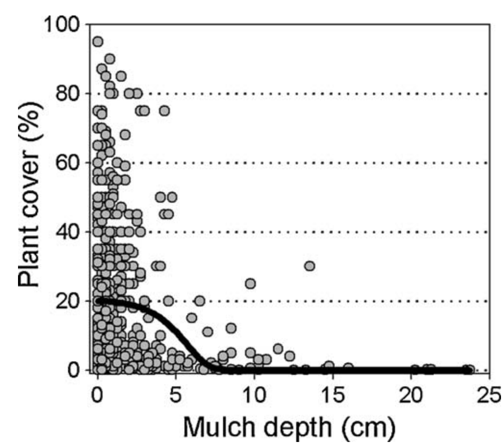

Fig. 2. Mulch depth effects on plant cover. Scatter plot points are values from individual sampling quadrats. The fitted line displays predicted values from a Gompertz 3-paramater non-linear regression model.

Table 3
Type III tests of mulch and elevation effects on plant cover and species richness from the mixed model analysis. Plant cover was square root transformed.

Effect	Degrees of freedom[a]	*F*-value	*P*
Plant cover			
Mulch cover	219	10.34	0.002
Mulch cover quadratic	419	16.74	<0.001
Elevation	16	16.86	<0.001
Plant species richness			
Mulch cover	492	20.20	<0.001
Mulch cover quadratic	484	21.91	<0.001
Elevation	17	27.68	<0.001

[a] Denominator degrees of freedom, numerator degrees of freedom are 1 for each variable.

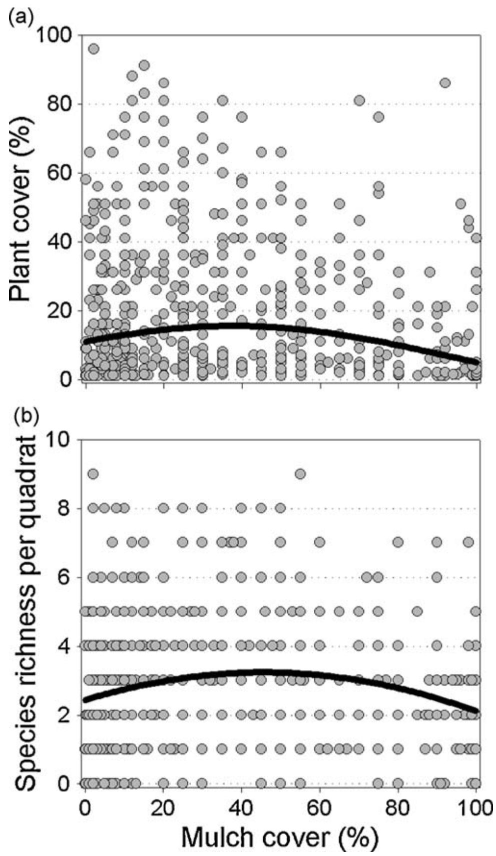

Fig. 3. Mulch cover effects on (a) total plant cover and (b) vascular plant species richness. Scatter plot points are values from the individual sampling quadrats. The fitted lines are predicted values from the mixed model analysis with elevation held constant at the median value for the study site.

3.3. Tree seedlings

Tree regeneration was roughly equally divided between lodgepole pine and aspen seedlings. Lodgepole pine seedlings were found on 60 of the 591 sample points (83 total individuals), while aspen seedlings were found on 48 sample points (78 total individuals). Seedling densities varied among transects; two transects contained no lodgepole pine seedlings while 5 transects contained no aspen seedlings. Almost one-third of the lodgepole pine seedlings (26) were found on 1 transect. The probability of finding a tree seedling – either lodgepole pine or aspen – at a sample point declined significantly with increasing elevation (Table 4 and Fig. 4b and d).

Mulch effects on seedling presence varied between lodgepole pine and aspen. The probability of finding a lodgepole pine seedling at a sample point increased with mulch cover when mulch cover was low (0–25%), but was largely unresponsive to mulch cover at higher levels (Fig. 4a). The probability of finding an aspen seedling was not significantly influenced by mulch cover (Table 4 and Fig. 4c).

3.4. Exotic plant species

Exotic plant species (including wheat) were found on 249 of the 591 sampling points (172 sample points excluding wheat). A total of 14 exotic plant species were found, of which wheat, prickly lettuce (*Lactuca serriola* L.), and tall tumblemustard (*Sisymbrium altissimum* L.) were the most common (Table 2). For the 249 sampling quadrats with at least 1 exotic plant, exotic plant density averaged 6 plants per quadrat (24 plants/m^2), exotic plant cover averaged 1.4%, and exotic plant species richness averaged 1.4 species.

Table 4
Type III tests of mulch effects on seedling and exotic species occurrences from logistic mixed models.

Effect	Degrees of freedom[a]	*F*-Value	*P*
Conifer seedlings			
Mulch cover[b]	587	5.22	0.023
Mulch cover quadratic	587	3.86	0.050
Elevation	18	7.76	0.012
Aspen seedlings			
Mulch cover	588	0.01	0.918
Elevation	16	13.36	0.002
Exotic species with wheat			
Mulch cover	589	22.48	<0.001
Exotic species without wheat			
Mulch cover	587	3.65	0.057
Mulch cover quadratic	587	2.93	0.088
Elevation	14	12.29	0.004

[a] Denominator degrees of freedom, numerator degrees of freedom are 1 for each variable.
[b] Mulch cover (linear and quadratic) was square root transformed for the logistic regression of conifer seedlings.

Table 5
Type III tests of mulch effects on exotic plant species density given that exotic plant species are present. Exotic plant density was log-transformed.

Effect	Degrees of freedom[a]	*F*-Value	*P*
With wheat			
Mulch cover	212	10.32	0.002
Mulch cover quadratic	235	6.26	0.013
Without wheat			
Mulch cover	164	9.03	0.003
Mulch cover quadratic	165	5.86	0.017

[a] Denominator degrees of freedom, numerator degrees of freedom are 1 for each variable.

Exotic plant presence was positively associated with mulch cover, but not influenced by elevation (Table 4). The probability of an exotic species occurring on a quadrat increased from about 25% on quadrats with no mulch to over 60% on quadrats with mulch cover over 90% (Fig. 5a). Among sampling points where exotic plant species were found, exotic plant densities were greatest at 50–80% mulch cover (Fig. 5c). The relationship between mulch cover and exotic plant presence was considerably weaker when wheat was excluded from the analysis. Without wheat, exotic plant frequency was greatest at about 60% mulch cover (Fig. 5b), and declined with increasing elevation. Given exotic plant presence, exotic plant density excluding wheat was also maximized at about 60% mulch cover (Fig. 5d), but did not vary significantly with elevation (Table 5).

4. Discussion

We found little evidence that mulching inhibited vegetation recovery following wildfire. Plant cover and species richness were largely unaffected by mulching even with relatively high second year mulch cover. Furthermore, mulching appeared to provide a small benefit to recovering plants at moderate mulch cover and low mulch depths. Previous studies of mulch effects on vegetation recovery have been variable, with different studies suggesting that mulch facilitated (Bautista et al., 1996; Badía and Martí, 2000) or interfered with (Kruse et al., 2004) recovering vegetation. Some of this variability may be due to differences in dominant environmental factors limiting plant establishment and growth. Mulch can help retain soil moisture, which could benefit plant establishment and growth on water-stressed sites more than on mesic sites (Robichaud et al., 2000). Indeed, below-average precipitation during the first two years following the Tripod Fire may have increased

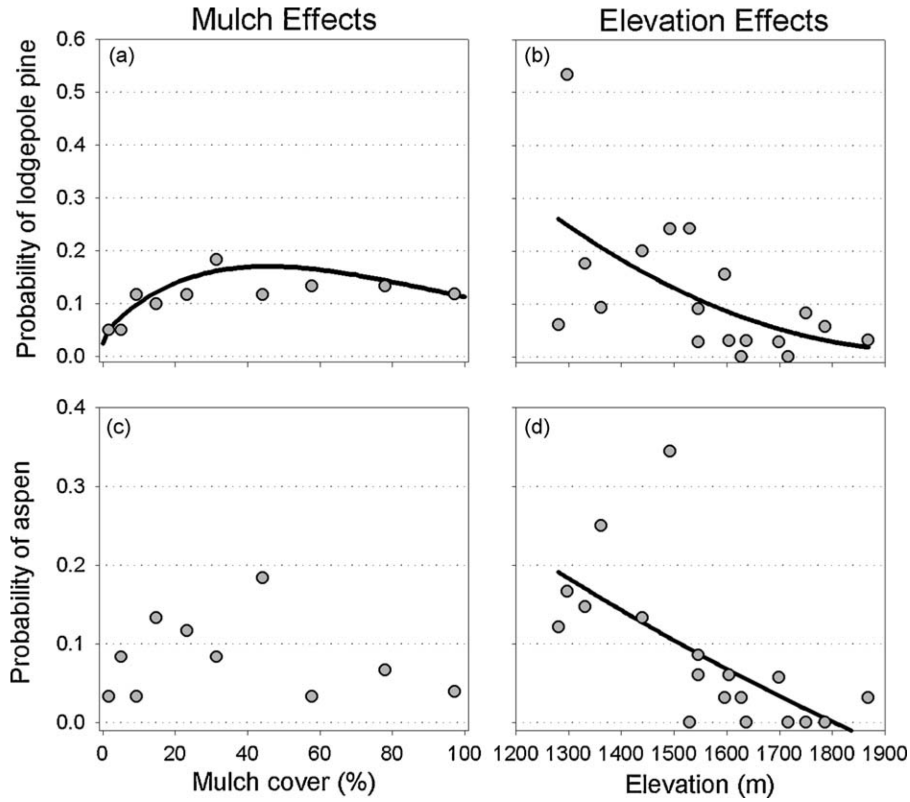

Fig. 4. Mulch cover effects on (a) conifer seedling frequency and (c) aspen seedlings. Elevation effects on (b) conifer seedlings and (d) aspen seedlings. Scatter plot points for mulch cover effects were calculated by dividing sample quadrats into groups of 60 based on ranked order of mulch cover, then calculating the frequency of seedlings within that group. Scatter plot points for (b) and (d) are the proportions of sample points with seedlings for each of the 18 transects. The line in (a) shows predicted seedling frequency response to varying mulch cover with elevation held constant at the median value where conifers were found (mulch cover was square root transformed for the analysis). The other fitted lines (b and d) represent second order linear regressions models.

the benefits of mulch in this study by limiting establishment in areas without mulch.

Only at very high cover or depth did mulching reduce plant cover compared to areas with no mulch, and these conditions were relatively infrequent in our study (and inconsistent with management prescriptions). A study in southwestern Washington also found better post-fire vegetation recovery where straw mulch was evenly distributed and did not form deep piles (Robichaud et al., 2009). Clearly, excessive application of mulch or aggregation of mulch redistributed by wind could negatively affect vegetation recovery, either through excessive shading or by creating a physical barrier to plant growth.

Mulch application has the potential to impede conifer regeneration (Kruse et al., 2004), but we found no such evidence in this study. If anything, mulching may have facilitated lodgepole pine seedling establishment. Seedlings in dry coniferous forests can be killed by high ground surface temperatures and drought stress during the summer (Kolb and Robberecht, 1996). Mulch may facilitate tree seedling establishment by reducing soil surface temperatures through shading and by increasing soil water availability (Kolb and Robberecht, 1996; Robichaud, 2005). Such benefits appear to have compensated for any negative effects of mulching in our study.

Mulching has increased exotic species abundance modestly following wildfire in California (Kruse et al., 2004) and the Eastern Cascades (Dodson et al., 2010). We also found that mulching increased the frequency and density of exotic plants. Wheat cover increased with mulching cover, probably because the straw retained significant amounts of wheat seed. Wheat persistence is not expected to be a problem, however, as post-fire seeding studies with wheat have shown that wheat cover declines rapidly after the initial growing season (Dodson and Peterson, 2009; Peterson et al., 2009). At intermediate cover, mulching could have increased exotic plant frequency by introducing exotic plant seeds (hitchhiking agricultural weeds), promoting invasion and establishment by seed dispersing from neighboring populations, increasing growth and reproductive output of locally established plants, or some combination of the above. Exotic plant frequency and density (excluding wheat) declined with high mulch cover, presumably due to shading and reduced access to bare mineral soil.

Although average second year mulch cover in this study was only about 35%, first year mulch cover was likely much higher and therefore more effective for erosion control. Robichaud (2005) reported that the mulch application rate used on the Tripod Fire (2.24 Mg/ha) typically produces about 70% mean mulch cover. Compaction, decomposition, and redistribution of mulch during the first 12–18 months probably reduced mean mulch cover and increased spatial heterogeneity. We assume that mulch redistribution occurred primarily before the first winter and that most compaction occurred during the first winter after treatment.

Mulch redistribution following application, if it occurred, could have obscured mulching impacts in this observational study. Although we found significant responses of tree seedlings and exotic plants to mulching, some mulching effects could be associated with the treatment application process or initial mulch distribution, rather than the distribution at the time of sampling. For example, exotic plant seeds could detach from contaminated straw mulch as it fell to the ground during application or during an early rainfall event, prior to redistribution of mulch by wind or water. In either case, the resulting plants would contribute to background exotic species frequencies and not be correlated with final mulch cover. It is also possible that resprouting shrubs or

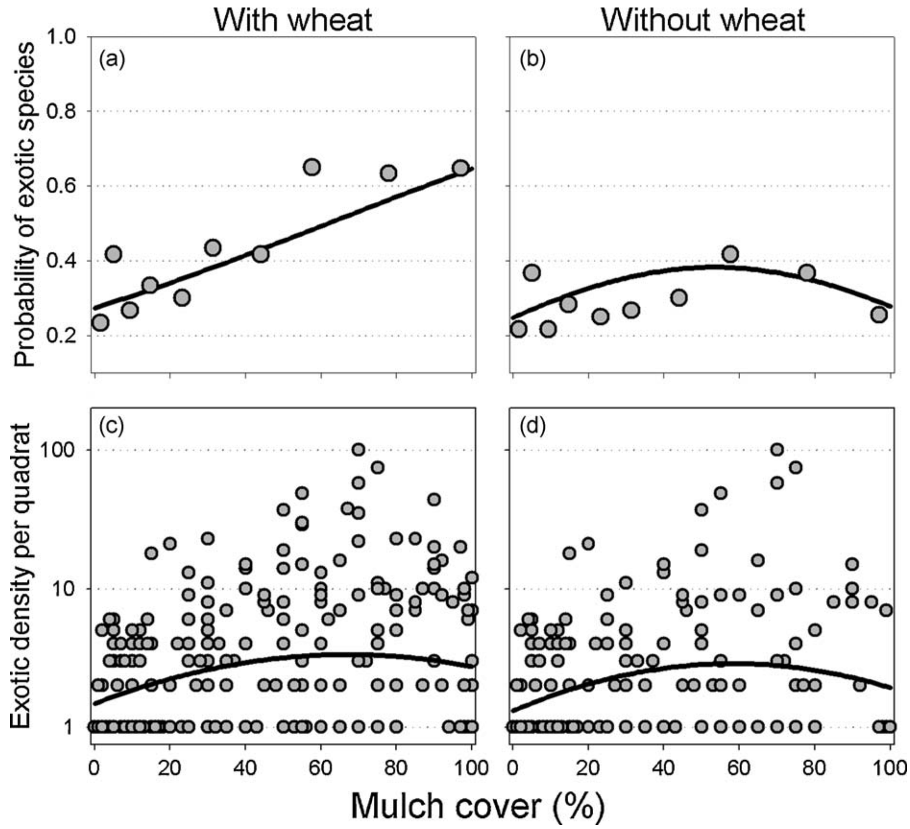

Fig. 5. Effects of mulch cover on (a) exotic species frequency including wheat, (b) exotic species frequency excluding wheat, (c) exotic species density including wheat given that exotic species are present, and (d) exotic species density excluding wheat given that exotic species are present. Lines are modeled values with elevation held constant at the median value where exotic species were found. Scatter plot points are proportions of sampling quadrats (divided into groups of 60 by rank order of mulch cover) with exotic species.

herbaceous vegetation could "trap" mulch during redistribution, creating a positive association between mulch and vegetation without the implied facilitative process. While we do not believe that these problems were dominant in this study, we acknowledge their potential influence and the need for future experimental studies or effectiveness monitoring projects that incorporate untreated control areas.

The mulching prescription used in this study appears to provide adequate protection from soil erosion hazards while producing mostly neutral to positive effects on vegetation recovery. Second year mulch cover averaged 35% which, combined with recovering vegetation, litter, wood, and rock, limited exposed bare soil to an average of 29%. This is well below 60–70% bare soil threshold for high post-fire soil erosion hazards identified by Johansen et al. (2001) and below the 40% bare soil threshold at which erosion can be reduced to near negligible amounts in some ecosystems (Orr, 1970; Robichaud et al., 2000). First year mulch cover was probably higher, as noted above, thereby providing greater erosion control benefits. The prescribed mulching intensity also produced neutral to positive effects on live plant cover, plant species richness, and tree seedling densities, with only small increases in exotic plant species occurrence and density.

The relative need for and benefits of slope stabilization treatments are strongly linked with potential rates of natural vegetation recovery. Many plant species in fire-prone forests are well adapted to wildfire and can recover quickly following even severe fires (Lyon and Stickney, 1976; Turner et al., 2003; Kim et al., 2008). However, second year live plant cover on our study sites was often much lower than reported after other fires in the Eastern Cascades (Schoennagel and Waller, 1999; Robichaud et al., 2006; Dodson and Peterson, 2009; Peterson et al., 2009) and elsewhere (Robichaud et al., 2000; Kim et al., 2008). We note, however, the strong negative relationship we observed between site elevation and vegetation cover, species richness and tree regeneration, and the fact that the Tripod Wildfire burned a higher proportion of high elevation forests than some other fires in the region. High elevation lodgepole pine forests typically have less frequent wildfire and therefore fewer species with adaptations to fire (Turner et al., 2003); they may also support relatively sparse understory vegetation layers (Arno, 1980). Our study shows the need for slope stabilization treatments varies across elevation gradients, and that elevation may be useful in predicting post-fire vegetation recovery rates and the relative need for land surface treatments such as mulching.

5. Conclusion

Post-fire slope stabilization treatments should ideally provide protection against flooding and erosion without compromising long-term ecosystem recovery and function (USDA, 1995). In this study, we found that mulching had few negative effects on vegetation recovery, except in the relatively rare cases where mulch cover was very high or mulching resulted in deep piles. In contrast, moderate amounts of mulch appeared to facilitate growth and establishment of some species, including conifer regeneration. Although further study is needed to establish the generality of our findings, our study suggests that decisions to use mulching to control erosion and flooding after wildfire should focus on expected benefits, economic costs, and other potential ecosystem impacts, rather than on vegetation impacts.

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References

Arno, S.F., 1980. Forest fire history in the Northern Rockies. Journal of Forestry 78, 460–465.

Badía, D., Martí, C., 2000. Seeding and mulching treatments as conservation measures of two burned soils in the central Ebro Valley NE Spain. Arid Soil Research and Rehabilitation 13, 219–232.

Bautista, S., Bellot, J., Vallejo, V.R., 1996. Mulching treatment for postfire soil conservation in a semiarid ecosystem. Arid Soil Research and Rehabilitation 10, 235–242.

Bautista, S., Robichaud, P.R., Bladé, C., 2009. Post-fire mulching. In: Cerdà, A., Robichaud, P.R. (Eds.), Fire Effects on Soils Restoration Strategies. Science Publishers, Enfield, NH, pp. 353–372 (Chapter 13).

Benavides-Solorio, J., MacDonald, L.H., 2001. Post-fire runoff and erosion from simulated rainfall on small plots, Colorado Front Range. Hydrological Processes 15, 2931–2952.

Beyers, J., 2004. Postfire seeding for erosion control: effectiveness and impacts on native plant communities. Conservation Biology 18, 947–956.

Crawford, J.S., Wahren, C.-H.A., Kyle, S., Moir, W.H., 2001. Responses of exotic plant species to fires in *Pinus ponderosa* forests in northern Arizona. Journal of Vegetation Science 12, 261–268.

DeBano, L.F., Neary, D.G., Ffolliott, P.F., 1998. Fire's Effects on Ecosystems. John Wiley and Sons, New York.

Dodson, E.K., Peterson, D.W., 2009. Seeding and fertilization effects on plant cover and community recovery following wildfire in the eastern Cascade Mountains, USA. Forest Ecology and Management 258, 1586–1593.

Dodson, E.K., Peterson, D.W., Harrod, R.J., 2010. Impacts of erosion control treatments on native vegetation recovery after severe wildfire in the Eastern Cascades, USA. International Journal of Wildland Fire 19, 490–499.

Faust, R., 2008. Aerial mulching techniques—trough fire. In: Narog, M.G. (tech. coord.), Proceedings of the 2002 Fire Conference: Managing Fire and Fuels in the Remaining Wildlands and Open Spaces of the Southwestern United States, San Diego, California, December 2–5, 2002. General Technical Report PSW-GTR-189. USDA Forest Service, Pacific Southwest Research Station, Albany, California, pp. 91–97.

Freeman, J.P., Stohlgren, T.J., Hunter, M.E., Omi, P.N., Martinson, E.J., Chong, G.W., Brown, C.S., 2007. Rapid assessment of postfire plant invasion in coniferous forests of the western United States. Ecological Applications 17, 1656–1665.

Groen, A.H., Woods, S.W., 2008. Effectiveness of aerial seeding and straw mulch for reducing post-wildfire erosion, north-western Montana, USA. International Journal of Wildland Fire 17, 559–571.

Johansen, M.P., Hakonson, T.E., Breshears, D.D., 2001. Post-fire runoff and erosion from rainfall simulation: contrasting forests with shrublands and grasslands. Hydrological Processes 15, 2953–2965.

Kim, C.-G., Shin, K., Joo, K.Y., Lee, K.S., Shin, S.S., Choung, Y., 2008. Effects of soil conservation measures in a partially vegetated area after forest fires. Science of the Total Environment 399, 158–164.

Kolb, P.F., Robberecht, R., 1996. High temperature and drought stress effects on survival of *Pinus ponderosa* seedlings. Tree Physiology 16, 665–672.

Kruse, R., Bend, E., Bierzychudek, P., 2004. Native plant regeneration and introduction of non-natives following post-fire rehabilitation with straw mulch and barley seeding. Forest Ecology and Management 196, 299–310.

Littell, R.C., Milliken, G.A., Stroup, W.W., Wolfinger, R.D., Schabenberger, O., 2006. SAS System for Mixed Models, 2nd ed. SAS Institute, Inc., Cary, NC.

Lyon, L.J., Stickney, P.F., 1976. Early vegetal succession following large northern Rocky Mountain wildfires. In: Proceedings of the Montana Tall Timbers Fire Ecology Conference and Fire and Land Management Symposium, Number 14, Tall Timbers Research Station, Tallahassee, FL, pp. 55–375.

McCune, B., Keon, D., 2002. Equations for potential annual direct incident radiation and heat load. Journal of Vegetation Science 13, 603–606.

Napper, C., 2006. Burned Area Emergency Response (BAER) treatments catalog. USDA Forest Service, National Technology & Development Program Technical Report 0625 1801-SDTDC.

Orr, H.K., 1970. Runoff and erosion control by seeded and native vegetation on a forest burn: Black Hills, South Dakota. Research Paper RM-60. USDA, Forest Service, Rocky Mountain Forest and Range Experiment Station, Fort Collins, Colorado, p. 12.

Peterson, D.W., Dodson, E.K., Harrod, R.J., 2009. Fertilization and seeding effects on vegetative cover after wildfire in North-Central Washington State. Forest Science 55, 494–502.

PRISM, 2009. Climatological Normals, 1971–2000. The PRISM Group, Oregon State University, Corvallis, OR.

Raudenbush, S.W., Bryk, A.S., 2002. Hierarchical Linear Models: Applications and Data Analysis Methods. Sage Publications, Thousand Oaks, CA.

Robichaud, P.R., 2005. Measurement of post-fire hillslope erosion to evaluate and model rehabilitation treatment effectiveness and recovery. International Journal of Wildland Fire 14, 475–485.

Robichaud, P.R., Beyers, J.L., Neary, D.G., 2000. Evaluating the effectiveness of post-fire rehabilitation treatments. General Technical Report. RMRS-GTR-63. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO, p. 85.

Robichaud, P.R., Lewis, S.A., Brown, R.E., Ashmun, L.E., 2009. Emergency post-fire rehabilitation treatment effects on burned area ecology and long-term restoration. Fire Ecology 5, 115–128.

Robichaud, P.R., Lillybridge, T.R., Wagenbrenner, J.W., 2006. Effects of postfire seeding and fertilizing on hillslope erosion in north-central Washington, USA. Catena 67, 56–67.

Robichaud, P.R., MacDonald, L., Freeouf, J., Neary, D., Martin, D., Ashmun, L., 2003. Postfire rehabilitation of the Hayman Fire. In: Graham, R.T. (technical editor), Hayman Fire Case Study. Gen. Tech. Rep. RMRS-GTR-114. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT, pp. 293–313, 396 pp.

Robichaud, P.R., Wagenbrenner, J.W., Brown, R.E., Wohlgemuth, P.M., Beyers, J.L., 2008. Evaluating the effectiveness of contour-felled log erosion barriers as a post-fire runoff and erosion mitigation treatment in the western United States. International Journal of Wildland Fire 17, 255–273.

SAS Institute, 2008. SAS/STAT 9.2 User's Guide. SAS Publishing, Cary, NC.

Schoennagel, T.L., Waller, D.M., 1999. Understory responses to fire and artificial seeding in an eastern Cascades *Abies grandis* forest, USA. Canadian Journal of Forest Research 29, 1393–1401.

Singer, J.D., 1998. Using SAS PROC MIXED to fit multilevel models, hierarchical models, and individual growth models. Journal of Educational and Behavioral Sciences 24, 323–355.

Turner, M.G, Romme, W.H., Tinker, D.B., 2003. Surprises and lessons from the 1988 Yellowstone fires. Frontiers in Ecology and the Environment 1, 351–358.

USDA, 1995. Burned Area Emergency Rehabilitation Handbook. USDA Forest Service, Forest Service Handbook FSH 2509.13-95-3. Washington, DC, USA.

USDA Forest Service, 2006. Tripod Fire Complex Burned Area Report. FS-2500-8. U.S. Department of Agriculture, Forest Service, Okanogan–Wenatchee National Forest, Wenatchee, Washington, USA.

USDA, NRCS, 2007. The PLANTS Database (http://plants.usda.gov, November 20, 2009). National Plant Data Center, Baton Rouge, LA, USA.

U.S. General Accounting Office (GAO), 2003. Wildland fires: better information needed on effectiveness of emergency stabilization and rehabilitation treatments. Report to Congressional Requesters GAO-03-430.

Wagenbrenner, J.W., MacDonald, L.H., Rough, D., 2006. Effectiveness of three post-fire rehabilitation treatments in the Colorado Front Range. Hydrological Processes 20, 2989–3006.

Wondzell, S.M., King, J.G., 2003. Postfire erosional processes in the Pacific Northwest and Rocky Mountain regions. Forest Ecology and Management 178, 75–87.

Wu, H., Pratley, J., Lemerle, D., Haig, T., 2001. Allelopathy in wheat (*Triticum aestivum*). Annals of Applied Biology 139, 1–9.