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Rill erosion in burned and salvage logged western montane forests: Effects of logging equipment type, traffic level, and slash treatment

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ABSTRACT

Following wildfires, forest managers often consider salvage logging burned trees to recover monetary value of timber, reduce fuel loads, or to meet other objectives. Relatively little is known about the cumulative hydrologic effects of wildfire and subsequent timber harvest using logging equipment. We used controlled rill experiments in logged and unlogged (control) forests burned at high severity in northern Montana, eastern Washington, and southern British Columbia to quantify rill overland flow and sediment production rates (fluxes) after ground-based salvage logging. We tested different types of logging equipment—feller-bunchers, tracked and wheeled skidders, and wheeled forwarders—as well as traffic levels and the addition of slash to skid trails as a best management practice. Rill experiments were done at each location in the first year after the fire and repeated in subsequent years. Logging was completed in the first or second post-fire year. We found that ground-based logging using heavy equipment compacted soil, reduced soil water repellency, and reduced vegetation cover. Vegetation recovery rates were slower in most logged areas than the controls. Runoff rates were higher in the skidder and forwarder plots than their respective controls in the Montana and Washington sites in the year that logging occurred, and the difference in runoff between the skidder and control plots at the British Columbia site was nearly significant ($p = 0.089$). Most of the significant increases in runoff in the logged plots persisted for subsequent years. The type of skidder, the addition of slash, and the amount of forwarder traffic did not significantly affect the runoff rates. Across the three sites, rill sediment fluxes were 5–1900% greater in logged plots than the controls in the year of logging, and the increases were significant for all logging treatments except the low use forwarder trails. There was no difference in the first-year sediment fluxes between the feller-buncher and tracked skidder plots, but the feller-buncher fluxes were lower than the values from the wheeled skidder plots. Manually adding slash after logging did not affect sediment flux rates. There were no significant changes in the control sediment fluxes over time, and none of the logging equipment impacted plots produced greater sediment fluxes than the controls in the second or third year after logging. Our results indicate that salvage logging increases the risk of sedimentation regardless of equipment type and amount of traffic, and that specific best management practices are needed to mitigate the hydrologic impacts of post-fire salvage logging.

1. Introduction

Following wildfires, land managers often consider salvage logging burned trees to recover the monetary value of the timber and to meet other objectives (Peterson et al., 2009). Although the effects of both wildfire and logging have been studied, there is relatively little known of the cumulative impacts of these forest disturbances on forest hydrologic and geomorphologic processes. Recently, research has been more focused on post-fire logging effects on the biotic components of the ecosystem, such as habitat loss, altered community composition or forest structure, delayed vegetation recovery, and increased colonization of non-native species (Beschta et al., 2004; D'Amato et al., 2011; Karr et al., 2004; Lindenmayer and Noss, 2006; McIver and Starr, 2000), than on the impacts to runoff, peak flows, erosion, and sedimentation.

The assessment of post-fire logging is particularly complex, as there are several logging techniques and the effects of a given technique may vary by equipment operator, time since fire, post-fire and/or post-logging weather, and other site conditions (Chase, 2006). Some studies have reported little or no difference in the sediment loss from comparable burned areas and burned and logged

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areas (Fernández et al., 2007; Marques and Mora, 1998; Marston and Haire, 1990; Spanos et al., 2005; Stabenow et al., 2006). Smith et al. (2011) compared the sediment exported from two adjacent burned eucalypt catchments and one pine catchment that was subsequently logged. The sediment exported from the logged pine catchment was 1–2 orders of magnitude greater than the sediment exported from the unlogged eucalypt catchments. However, given that the pine catchment was burned at higher severity than either of the eucalypt catchments, the differences in exported sediment cannot be solely attributed to the logging operation. As part of the Southern Rockies Watershed Project, total suspended sediment concentrations from five watersheds burned at high severity were compared (Silins et al., 2009). Two of the five watersheds were logged one year after the fire; during a very wet second post-fire year, the mean concentrations were greater from the burned and logged watersheds than from the burned only watersheds. During later, drier years, there were no significant differences in mean concentration, yet turbidity from the salvage logged watersheds remained higher than unlogged controls for four years after the fire (Emelko et al., 2011). A study in southwest Oregon found higher sediment production rates in salvage logged areas compared to unlogged burned areas, but other management activities or site differences may have affected the results (Slesak et al., 2015). In another study in the interior western US, sediment production from burned and logged hillslope plots were up to two orders of magnitude larger than the sediment production from burned controls (Wagenbrenner et al., 2015).

Rill formation and extension are dominant erosion processes on steep hillslopes with exposed mineral soil, especially in burned areas where decreases in soil organic matter, litter and vegetation cover led to highly erodible bare soil (Moody and Kinner, 2006; Robichaud et al., 2010; Shakesby et al., 2007). A study in the Colorado Front Range found that 60–80% of the post-fire erosion at the hillslope scale was associated with rill erosion (Pietraszek, 2006). In one study in the Sierra Nevada in California, rill formation was directly related to the amount of bare soil and rills were observed in burned plots with >60% bare soil (Berg and Azuma, 2010). In several other studies bare soil has been indicated as a significant controlling factor in post-fire sediment yields (e.g., Inbar et al., 1998; Benavides-Solorio and MacDonald, 2005; Larsen et al., 2009).

Studies of logging effects in unburned areas have reported that log skidding over bare ground can cause severe soil disturbance over more than a third of the logged area (Klock, 1975; Page-Dumroese et al., 2006; Steinbrenner and Gessel, 1955) and erosion can be from 5 to 100 times greater on skid trails as compared to undisturbed areas (Croke et al., 2001, 1999; MacDonald et al., 2004; Robichaud et al., 1993). Increased pressure from logging equipment was measured at depths of 20 cm in experiments in Germany (Horn et al., 2004). The impacts of disturbance by logging equipment such as compaction, rutting, and loss of macropores, can affect coarse soils for decades after logging operations (Cambi et al., 2015). A runoff modeling exercise showed that the log drag lines act as an extension to the drainage network, thereby increasing the potential of hillslopes to be hydrologically connected to the stream network (Smith et al., 2011). Reduced infiltration and the somewhat linear shape of skid trails suggest erosion from bare skid trails is likely to be dominated by rilling.

Simulated rill experiments have been used in laboratory and field experiments to compare different soil conditions and to develop the rill erodibility parameters needed to model erosion rates (Bryan, 2000; Elliot et al., 1989; Govers et al., 2007; Knapen et al., 2007; Merz and Bryan, 1993; Wirtz et al., 2012). Similarly, rill experiments on burned soils have helped improve our understanding and ability to model post-fire erosion rates (Al-Hamdan et al., 2012; Pierson et al., 2009; Robichaud et al., 2010; Wagenbrenner et al., 2010) and show that the amount and type of soil cover can affect erosion rates (Foltz and Wagenbrenner, 2010; Pannkuk and Robichaud, 2003; Robichaud et al., 2013). Simulated rill experiments have also shown that initial rill erosion rates are much greater than steady state erosion rates that occur just a few minutes into the simulations (Foltz et al., 2008; Pierson et al., 2008; Robichaud et al., 2010). Rill experiments in burned areas conducted over multiple years suggest that the pattern of high initial erosion rates followed quickly by lower steady state erosion rates is repeated, but the overall rill erosion rates decrease over the time scale of years (Pierson et al., 2008; Sheridan et al., 2007).

Previous studies have related the effects of bare soil, water repellency, time since burning, and soil compaction to increases in runoff and/or sediment yield in burned and logged areas (Chase, 2006; Slesak et al., 2015; Wagenbrenner et al., 2015), although post-fire logging does not always result in significant increases in sediment yields (Fernández et al., 2007; McIver and McNeil, 2006; Silins et al., 2009). Our objective was to determine the effects of different ground-based equipment and operational practices used in post-fire salvage logging on soil properties, runoff, and rill erosion rates. We used simulated rill experiments following earlier research methods (Robichaud et al., 2010) at three burned sites for two years after logging to determine if different site conditions, logging equipment, traffic levels, or the addition of wood slash resulted in: (1) differences in soil bulk density, soil water repellency, surface cover, or vegetation; and (2) changes in runoff rates, runoff velocities, or sediment flux rates during the first two–three years after logging.

2. Methods

2.1. Site descriptions and experimental design

We used simulated rill experiments to compare areas recently burned at high severity (controls) to areas that were recently burned at high severity and salvage logged using ground-based logging equipment. Three high-severity burned areas classified according to burned area reflectance (White et al., 1996) or field assessments (Parsons et al., 2010) in forested mountainous areas were selected for rill experiments. All sites had coarse soils (Table 1), elevations between 1100 and 1800 m, annual precipitation between 545 and 1221 mm and pre-fire forests of firs and pines (Table 1). Hillslope gradients at the study sites were between 11 and 46%. Differences in logging equipment and practices among the three sites allowed us to compare some of the effects of these operations (Table 2).

The 2006 Red Eagle Fire burned 14,000 ha in northern Montana (Fig. 1). Sandy loam soil derived from argillite and mean annual precipitation of 1221 mm (1979–2009) supported a forest predominated by lodgepole pine (*Pinus contorta*) (Table 1) before the fire. Part of the burned area was logged in summer 2007 using feller-bunchers and whole-tree skidding. The grapple skidders had either steel tracks ("tracked") or rubber tires ("wheeled"). We compared runoff and erosion rates from unlogged controls to runoff and erosion from trails made by feller-bunchers, trails made by tracked skidders, and trails made by wheeled skidders (Table 2). We also measured the effect of adding logging slash to skid trails on runoff and erosion rates by manually adding wood ("slash") after skidding to achieve at least 50% wood cover on one track of each of the skid trail plots (Table 2). Five rill experiments were completed in the controls and treated plots in 2007, 2008, and 2009 (Table 2).

The 2005 School Fire burned 21,000 ha in southeastern Washington (Fig. 1). The study sites were located in volcanic ashy silt loam soils derived from basalt in an area with mean annual precipitation of 924 mm (2001–2012) (Table 1). Pre-fire vegetation was dominated by Douglas-fir (*Pseudotsuga menziesii*) and grand fir

Table 1
Long-term precipitation, elevation, soil series, taxonomic class, parent material, soil texture, percent clay, silt, and sand, and dominant overstory and understory species for each site.

Site	Precip. (mm) [elevation (m)]	Soil series [*Taxonomic class*]	Parent material [texture]	Clay (%)	Silt (%)	Sand (%)	Overstory	Understory
Red Eagle	1221[a] [1797]	Tenex [*Loamy-skeletal, mixed, superactive Spodic Dystrocryepts*]	Argillite [sandy loam]	3	33	64	Lodgepole pine (*Pinus contorta*) Douglas-fir (*Pseudotsuga menziesii*)	Grouse whortleberry (*Vaccinium scoparium*) Twinflower (*Linnaea borealis*)
School	924[b] [1500]	Klicker [*Loamy-skeletal, isotic, frigid Vitrandic Argixerolls*]	Basalt [ashy silt loam]	1	37	62	Douglas-fir (*Pseudotsuga menziesii*) Grand fir (*Abies grandis*)	Bluebunch wheatgrass (*Pseudorogneria spicata*) Pinegrass (*Calamagrostis rubescens*) Geyers sedge (*Carex geyeri*)
Terrace Mtn.	545[c] [1145]	Tunkwa [*Orthic Gray Luvisol*]	Morainal till [loamy sand]	3	14	83	Lodgepole pine (*Pinus contorta*) Subalpine fir (*Abies lasiocarpa*)	Pinegrass (*Calamagrostis rubescens*) Boxleaf (*Paxistima mysinites*)

[a] 1979–2009 data from Many Glacier station, elevation 1494 m, 23 km from study site. http://www.wcc.nrcs.usda.gov/nwcc/site?sitenum=613&state=mt accessed 8 July 2015.
[b] 2001–2012 data from Spruce Springs station, elevation 1740 m, 9 km from study site. http://www.wcc.nrcs.usda.gov/nwcc/site?sitenum=984&state=wa, accessed 13 May 2013.
[c] 1981–2009 estimated using ClimateBC software (Wang et al., 2006) for the study site.

Table 2
Logging equipment model and mass, number of passes by logging equipment, and slash treatment for the treatments at the three sites. There were five plots of each treatment at each site except seven control plots and six tracked skidder plots at Terrace Mountain.

Treatment name	Logging equipment[a]	Unloaded mass (Mg)	Number of passes[b]	Slash[c]
2006 Red Eagle Fire, logged in 2007, rill experiments done in 2007–2009				
Control	None	0	0	None
Feller-buncher	Timberjack 608L	26.8	1–2	None
Tracked, no slash added	Caterpillar 527	21.9	4–8	None
Tracked, slash added	Caterpillar 527	21.9	4–8	Yes
Wheeled, no slash added	Caterpillar 535B	19.0	6–10	None
Wheeled, slash added	Caterpillar 535B	19.0	6–10	Yes
2005 School Fire, logged in 2007, rill experiments done in 2006–2008				
Control	None	0	0	None
Low use forwarder	Valmet 890.3	19.1	1–2	None
High use forwarder	Valmet 890.3	19.1	3–6	None
2009 Terrace Mountain Fire, logged in 2010, rill experiments done in 2010–2011				
Control	None	0	0	None
Tracked, no slash added	Tracked skidder	–[d]	4–8	None

[a] Use of trade names does not imply endorsement by the US Department of Agriculture.
[b] Estimated from field observations: feller-buncher was unloaded, skidders were loaded in one direction, and forwarder loading was variable.
[c] Wood slash was added to provide at least 50% total cover.
[d] No data.

(*Abies lasiocarpa*) (Table 1). Logging of the sites occurred in 2007 using a feller-buncher outfitted with a processor head. Cut-to-length logs were carried to the landing using an eight-wheeled, rubber-tired forwarder with an open crib. The forwarder generally loaded several small piles of logs before returning to the landing to unload. We compared runoff and erosion rates from unlogged controls to runoff and erosion rates from forwarder trails with one or two round-trips ("low use"), and from forwarder trails with three to six round-trips ("high use") (Table 2). The number of round trips were estimated based on the number of trees removed from the hillslopes and the visual appearance of the forwarder trails. Five rill experiments were done in the controls in 2006 and in the controls and logged plots in 2007 and 2008 (Table 2).

The 2009 Terrace Mountain Fire burned 9300 ha in south-central British Columbia (Fig. 1). The soils were loamy sands derived from glacial till and the mean annual precipitation from 1981–2009 was 545 mm (Table 1). The continental climate supported lodgepole pine (*Pinus contorta*) and subalpine fir (*Abies lasiocarpa*) as the dominant pre-fire overstory vegetation (Table 1). Some of the burned area was logged in 2010 using a feller-buncher and a tracked skidder. We compared runoff and erosion rates from seven unlogged controls and six skidder trails in 2010 and 2011 (Table 2).

2.2. Rill experiments

Runoff and erosion rates were measured using the protocol described by Robichaud et al. (2010) in plots that were 9 m long by 1–3 m wide depending on flow characteristics. Water was released at the top of the plot through an energy dissipater at sequential controlled rates of 7, 22, 30, 15, and 48 L min^{-1} for 12 min each. The width and depth of flow in each rill were measured at 2 m and 7 m downslope of the release point, and the combined width and average depth of all rills at each downslope location were calculated and then averaged across the two downslope locations for each flow rate. Runoff velocity was measured using a saline solution and conductivity meters at 2 m and 7 m (King and Norton, 1992). Six timed runoff and sediment samples were collected during each flow rate at the outlet of the plot when the flow reached the 9-m point. Samples were later processed in the laboratory to determine runoff (overland flow) and sediment flux rates.

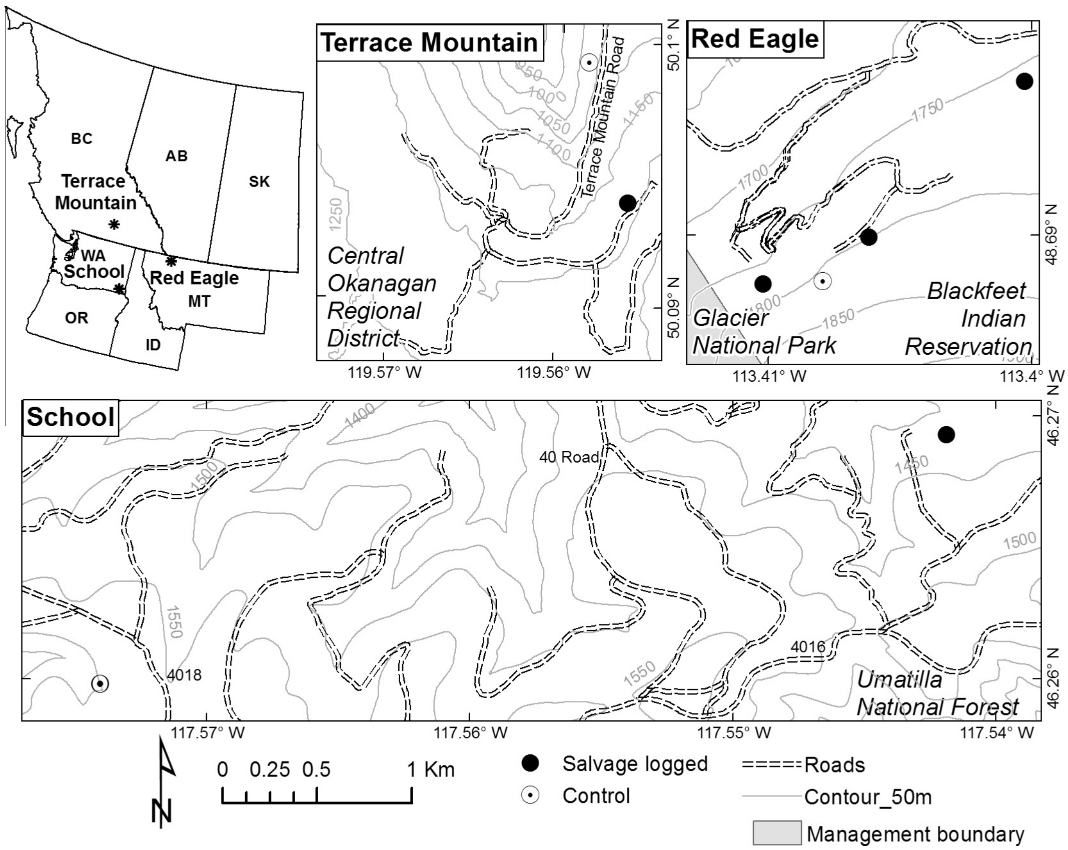

Fig. 1. Maps of the three study sites.

2.3. Plot characteristics

Dry soil bulk density was measured using a core sampler in areas adjacent to control plots and in each logging equipment track at Red Eagle and School in the year of logging and in areas adjacent to control plots at Terrace Mountain in the year of the fire. Bulk density was sampled at depths of 0–5 cm and 5–10 cm at each site and also at a depth of 10–15 cm at the Red Eagle and School sites. Soil particle size distributions were determined from samples of the top 1 cm of mineral soil (Gee and Bauder, 1986). Soil moisture content (Gardner, 1986) and soil water repellency using the water drop penetration time (WDPT) test (DeBano, 1981) were also measured before each rill experiment. WDPT was measured using 8 water droplets at the mineral soil surface and at depths of 1–2 cm and 3–4 cm below the mineral surface except at Terrace Mountain where no surface measurements were made. The median WDPT was determined for each depth in each plot (WDPT hereafter). Ground cover was classified in three 1-m^2, 100-point quadrats (after Chambers and Brown, 1983) in each plot before the experiments began. Ground cover classes were bare mineral soil, wood, gravel (>2 mm), cobble (>64 mm), live understory vegetation, or organic litter. The cover data from the three quadrats were averaged to determine the plot ground cover.

2.4. Analysis

The runoff and sediment flux rates approached a steady state condition by the fourth sample in each experimental flow rate, so only samples 4–6 were used to calculate means for each flow rate (Robichaud et al., 2010). The mean runoff rates, runoff velocities, flow widths, flow depths, and sediment flux rates were calculated only when runoff for that inflow rate reached the plot outlet. The mean plot values across all flow rates were used in statistical analyses.

Generalized linear mixed-effects statistical models (Schabenberger, 2005) were developed for response variables at each site using the type of logging equipment (none, feller-buncher, forwarder, wheeled skidder, or tracked skidder), level of impact (low or high use), slash treatment (no slash or slash added), and years after burning (one, two, or three) as fixed effects (Table 2). For each combination of fixed effects, plot was a random effect. Response (dependent) variables were soil bulk density, WDPT, surface cover, vegetation cover, runoff rate, flow width, flow depth, runoff velocity, and sediment flux rate. An autoregressive correlation structure was applied for each response except bulk density to account for the repeated measures in each plot through multiple years (Littell et al., 2006). We addressed heteroscedasticity in the model residuals for the vegetation cover, flow depths and widths, and sediment flux rates by square-root transforming the data. For the same reason we added 1 s to each WDPT and log-transformed these values before statistical analysis. Least-squares means with a Tukey-Kramer adjustment were used to test the significance of differences across multiple comparisons among fixed effects (Ott, 1993). The significance level was 0.05 except where otherwise indicated. Soil moisture was tested as a covariate for runoff rate, sediment flux rate, and runoff velocity, but it was not significant in any of the models.

3. Results

3.1. Soil properties and ground cover

3.1.1. Bulk density

The soil bulk density in the Red Eagle control plots increased with depth from 0.73 g cm^{-3} at 0–5 cm to 1.13 g cm^{-3} at 10–15 cm

(Fig. 2). In contrast, the bulk density in the controls at the School site was relatively constant with depth, and ranged from 0.84 to 0.89 g cm^{-3} across the three sampled depths. The soil in the Terrace Mountain controls was denser than the Red Eagle and School sites, and averaged 1.18 g cm^{-3} at 0–5 cm and 1.23 g cm^{-3} at 5–10 cm.

At Red Eagle, manually adding slash after skidding did not affect the soil bulk density ($p > 0.26$), so the slash added plots were combined with the no slash skidder plots. The feller-buncher, tracked skidder, and wheeled skidder plots had significantly higher bulk densities than the controls at the 0–5 cm depth. At the greater depths, only the wheeled skidder plots had significantly greater bulk density than the controls at the 10–15 cm depth (Fig. 2).

At the School site, there were no differences in bulk density between the low use and high use forwarder plots at any depth. The forwarder plots had higher bulk densities than the controls at both the 5–10 cm and 10–15 cm depths, but not at the 0–5 cm depth (Fig. 2).

3.1.2. Soil water repellency

At Red Eagle, the WDPT values in the control plots were highest at the surface but were more persistent at the 1–2 cm depth (Table 3). The mean WDPT at the surface in the control plots was 265 s in the first post-fire year, indicating strong soil water repellency. The surface WDPT in the controls did not change significantly in the second post-fire year, whereas the value decreased to 1 s in the third year. The surface WDPT in the logging equipment impacted plots ranged from 1 to 46 s right after logging in the first post-fire year, and from 1 to 2 s in the second post-fire year, and all of the logged values were significantly lower than the controls in the first two years (Table 3). There were no differences between any of the treatments and the controls in the third post-fire year (Table 3). WDPT results were similar at the 1–2 cm depth, except the values for the controls did not decrease in the third post-fire year and only the feller-buncher, slash-treated tracked skidder, and wheeled skidder plots had greater WDPT values than the controls in the first post-fire year (Table 3). The WDPT at 3–4 cm was relatively low in the first post-fire year in the control plots (31 s), and this did not vary through time or among the treatments (Table 3).

In contrast to the Red Eagle site, there was no water repellency at the surface in the School site. Some water repellency was observed in the first post-fire year at the 1–2 cm depth, and strong water repellency was measured at the 3–4 cm depth (203 s). The mean values at both depths decreased in the second post-fire year (2 s and 4 s, respectively) and did not change again in the third post-fire year (Table 3). There were no physically significant differences in WDPT among the control and forwarder plots at any depth in either the second or third post-fire year (Table 3).

The mean WDPT at 1–2 cm depth in the control plots at Terrace Mountain was only 6 s in the first post-fire year, and this was the lowest among the three sites (Table 3). The WDPT in the controls did not change in the second post-fire year, and the WDPT in the tracked skidder plots was not significantly different than the controls in either year (Table 3).

3.1.3. Surface and vegetation cover

The surface cover among the control plots at the three sites varied considerably in the first post-fire year. The Red Eagle site had the most surface cover, 63%, and there was no significant change in this value through the third post-fire year (Table 3). The mean surface cover in the School control plots was only 20% in the first post-fire year, and this value also did not statistically change over time, despite more than doubling in the third post-fire year (Table 3). The surface cover in the Terrace Mountain controls was 52% in the first post-fire year, and this value increased to 84% in the second year. The increases in surface cover in the control plots can be accounted for by significant increases in vegetation cover (Table 3) among the measurement periods.

The wheeled skidder traffic at the Red Eagle site reduced the surface cover in the first post-fire year to 39%, but this difference was not observed in the second post-fire year. The other logging equipment (feller-buncher, tracked skidder, or forwarder), slash treatment, and traffic level did not significantly change surface cover in any year at any of the three sites.

In contrast to the general lack of difference in total surface cover among logged and control plots, the live vegetation cover in the year that logging occurred was significantly lower than the controls in all of the plots impacted by logging equipment at all three sites (Table 3). Adding slash did not affect the vegetation cover in the skidder plots at Red Eagle (Table 3). The lower rates of vegetation cover persisted through the third post-fire year in all of the plots impacted by logging equipment except the feller-buncher plots (Table 3).

3.2. Runoff

3.2.1. Runoff rate

Across all sites, treatments, years, and plots that produced runoff, the runoff rate averaged 16.1 L min^{-1}. This value was 66% of the water released on the plots, and the remainder infiltrated along the flow path. The mean runoff rate was only 7.1 L min^{-1} in the controls at Red Eagle in the first post-fire year (Fig. 3). The runoff rates in the controls at the School and Terrace Mountain sites were 17 L min^{-1} and 11 L min^{-1}, respectively (Fig. 3), indicating lower infiltration along the flow paths at those sites than Red Eagle. The runoff rates in the controls in the second post-fire year did not change at Red Eagle, but they significantly decreased by 29 and 65% at the School and Terrace Mountain sites, respectively (Fig. 3). There was no change in runoff rates in the controls in the third post-fire year at the Red Eagle or School site (Fig. 3).

At Red Eagle, the runoff rates in the feller-buncher plots did not vary among the three years, and the mean runoff was significantly greater than the control value in the third post-fire year (Fig. 3). The skidder plots produced runoff rates at least 2.5 times the runoff in the controls in each year, and the differences were significant except for the untreated tracked skidder plots in the second post-fire year (Fig. 3). There were no differences in runoff between the tracked and wheeled skidder plots, and the addition of slash did not reduce the runoff rates in the skidder plots (Fig. 3). There were no significant changes in runoff rate through time in the logging equipment impacted plots.

At the School site both levels of forwarder traffic produced significantly more runoff than the controls, and there was no difference in runoff rate between the two traffic levels (Fig. 3). There was no change in the runoff rates in the forwarder plots between the second and third post-fire years (Fig. 3).

At the Terrace Mountain site, the difference in runoff rate between the tracked skidder plots and the controls in the first post-fire year was nearly significant ($p = 0.089$) (Fig. 3). In the second post-fire year there was no change in the runoff in the tracked skidder plots, and this combined with the significant decrease in runoff in the controls led to significantly greater runoff rates in the tracked skidder plots (Fig. 3).

3.2.2. Runoff width and depth

In the first post-fire year, the mean flow widths in the control plots were 316 mm at Red Eagle, 453 mm at School, and 354 mm at Terrace Mountain and none of these flow widths varied in subsequent years (Table 3). The only logging equipment impacted plots with significantly narrower flows than their respective controls were the wheeled skidder plots at Red Eagle in the first and second post-fire years, the low use forwarder plots at the School site in the third post-fire year, and the high use forwarder plots

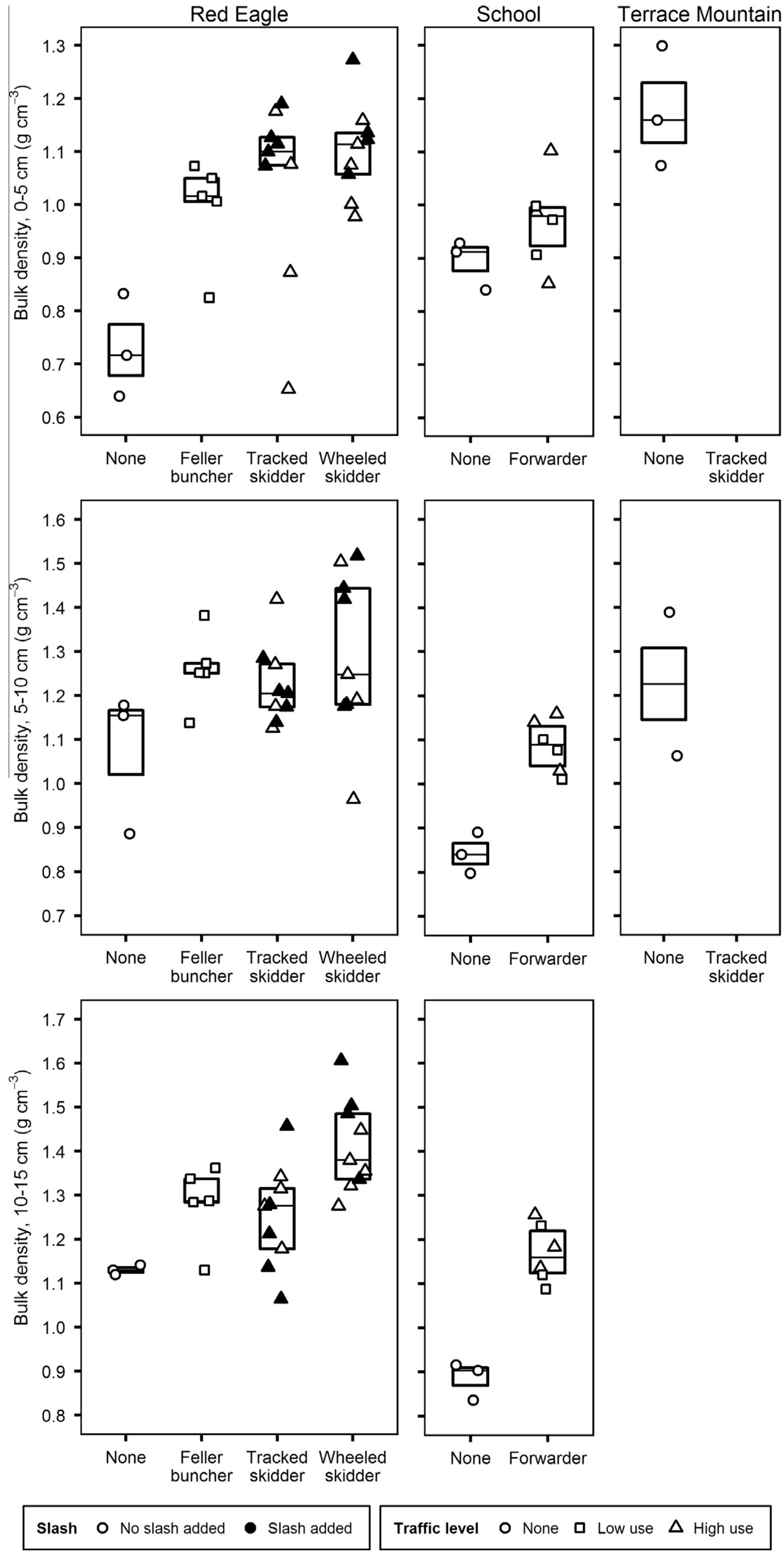

Fig. 2. Soil bulk density at depths of 0–5 cm, 5–10 cm, and 10–15 cm by type of logging equipment and site. Symbol fill indicates slash treatment and symbol shape indicates traffic level. At Terrace Mountain, bulk density was not sampled in skid trail plots or below 10 cm depth. Each box represents the median, first quartile, and third quartile; points are individual observations. Individual plot values are shown in Supplemental material and results of statistical analyses are shown in Table A1.

Table 3
Mean WDPT by depth, surface cover, vegetation cover, flow width, flow depth, and runoff velocity for each site, treatment, and post-fire year ("Yr"). Different superscript letters indicate differences in the (transformed) means among treatments and years within a site and measured variable.[a] Individual plot values are shown in supplemental material.

Yr	Red eagle						School			Terrace mountain	
	Control	Feller-buncher	Tracked skidder		Wheeled skidder		Control	Forwarder		Control	Tracked skidder
			No slash	Slash added	No slash	Slash added		Low use	High use		
WDPT (s) at mineral surface											
1	265^{A}	1^{C}	46^{BC}	3^{C}	2^{C}	5^{BC}	0^{B}			–[b]	–[b]
2	149^{AB}	1^{C}	1^{C}	1^{C}	2^{C}	2^{C}	0^{B}	0^{B}	0^{B}	–[b]	–[b]
3	1^{C}	32^{BC}	38^{BC}	1^{C}	3^{BC}	1^{C}	0^{B}	2^{A}	1^{AB}		
WDPT (s) at 1–2 cm depth											
1	226^{A}	8^{BC}	73^{ABC}	6^{BC}	8^{BC}	27^{ABC}	47^{A}			6^{A}	12^{A}
2	172^{AB}	56^{ABC}	81^{ABC}	7^{BC}	5^{BC}	3^{BC}	2^{B}	0^{B}	0^{B}	5^{A}	5^{A}
3	111^{ABC}	85^{ABC}	129^{ABC}	2^{C}	6^{ABC}	4^{BC}	1^{B}	2^{AB}	0^{B}		
WDPT (s) at 3–4 cm depth											
1	31^{A}	21^{A}	3^{A}	12^{A}	20^{A}	44^{A}	203^{A}			3^{A}	24^{A}
2	137^{A}	77^{A}	61^{A}	1^{A}	12^{A}	5^{A}	4^{B}	0^{B}	0^{B}	2^{A}	24^{A}
3	88^{A}	101^{A}	18^{A}	1^{A}	6^{A}	5^{A}	1^{B}	2^{B}	9^{B}		
Surface cover (%)											
1	63^{BCD}	53^{CDE}	49DE	68^{ABCD}	39^{E}	67^{ABCD}	20^{B}			52^{BC}	48^{C}
2	66^{BCD}	71^{ABC}	66^{ABCD}	73^{ABC}	61^{BCD}	76^{AB}	28^{B}	35^{AB}	27^{B}	84^{A}	70^{AB}
3	70^{ABC}	76^{AB}	75^{AB}	86^{A}	68^{ABCD}	61^{BCD}	45^{AB}	59^{A}	49^{AB}		
Vegetation cover (%)											
1	7.2^{BC}	1.4^{DEFG}	0.3^{FG}	0.2^{FG}	0.1^{G}	0.0^{G}	4.8^{CD}			26^{B}	0.5^{C}
2	4.9^{CDE}	1.8^{CDEFG}	0.9^{EFG}	0.5^{FG}	0.3^{FG}	0.0^{G}	14^{B}	0.5^{E}	1.1^{DE}	69^{A}	19^{B}
3	14^{A}	13^{AB}	4.0^{CDE}	5.9^{BCD}	2.1^{CDEF}	0.5^{FG}	28^{A}	0.7^{DE}	7.5^{BC}		
Flow width (mm)											
1	316^{AB}	229^{BC}	257^{ABC}	220^{BC}	146^{C}	267^{ABC}	453^{AB}			354^{A}	280^{A}
2	430^{A}	352^{AB}	284^{AB}	278^{AB}	238^{BC}	310^{AB}	463^{AB}	339^{BC}	141^{C}	325^{A}	352^{A}
3	363^{AB}	290^{AB}	262^{ABC}	329^{AB}	287^{AB}	273^{ABC}	754^{A}	407^{B}	468^{AB}		
Flow depth (mm)											
1	6^{C}	12^{AB}	12^{AB}	14^{A}	11^{AB}	9^{ABC}	5^{C}			5^{B}	10^{A}
2	5^{C}	8^{ABC}	7^{BC}	9^{ABC}	9^{ABC}	7^{BC}	6^{C}	16^{A}	19^{A}	7^{AB}	10^{A}
3	9^{ABC}	8^{BC}	8^{BC}	9^{ABC}	8^{BC}	8^{ABC}	4^{C}	10^{B}	10^{B}		
Velocity (m s^{-1})											
1	0.17^{A}	0.12^{BCD}	0.13^{ABCD}	0.12^{BCD}	0.17^{AB}	0.14^{ABC}	0.28^{A}			0.15^{A}	0.15^{A}
2	0.12^{BCD}	0.10^{CD}	0.11^{CD}	0.08^{D}	0.13^{ABCD}	0.10^{CD}	0.14^{B}	0.14^{B}	0.16^{B}	–[b]	–[b]
3	0.10^{CD}	0.11^{CD}	0.12^{BCD}	0.10^{CD}	0.13^{ABCD}	0.10^{CD}	0.18^{AB}	0.11^{B}	0.16^{B}		

[a] Sample interpretation: at Red Eagle, the WDPT at the mineral surface in the control plots in year 1 (265 s^{A}) was different at α = 0.05 from the WDPT in the feller-buncher plots in year 1 (1 s^{C}) but it was no different than the WDPT in the control plots in year 2 (149 s^{AB}).
[b] No data.

in the second post-fire year (Table 3). The flow widths in the logging equipment impacted plots generally increased through time, and the increases were significant for the wheeled skidder plots without slash and the high use forwarder plots. There were no differences in flow width among the feller-buncher or skidder plots, and neither the addition of slash nor the amount of forwarder traffic significantly affected flow width (Table 3).

The mean flow depths in the control plots ranged from 5 to 9 mm at all sites and years, and there were no significant changes among years (Table 3). All of the logged plots had greater flow depths than their respective controls in the year of logging, except for the slash-treated wheeled skidder plots. Neither the slash treatment nor the different amounts of forwarder traffic changed the mean flow depth, and there were no significant changes in the flow depths through time in the logged plots (Table 3).

3.2.3. Runoff velocity

At the Red Eagle and School sites, the mean runoff velocities in the control plots were highest in the first post-fire year and the values were 0.17 m s^{-1} and 0.28 m s^{-1}, respectively. The mean velocity decreased in the second post-fire year to 0.12 m s^{-1} at Red Eagle (Table 3). At the School site, the velocity decreased to 0.14 m s^{-1} in the second post-fire year. There was no significant change in the third post-fire year at either site (Table 3). The mean velocity in the controls at Terrace Mountain was 0.15 m s^{-1} in the first post-fire year, making it the lowest rate of the three controls in the first year after burning.

The velocities in the feller-buncher and slash-treated tracked skidder plots at Red Eagle were significantly lower than the controls in the first post-fire year, but these differences did not persist in the second or third post-fire years (Table 3). The velocities in the other equipment impacted plots were no different than the controls (Table 3). The velocities in the slash added plots were not significantly different than their comparable skidder plots.

3.3. Sediment

The mean sediment flux rate in the Red Eagle control plots was 0.90 g s^{-1} in the first post-fire year and the sediment fluxes in the second and third post-fire years were not significantly different (Fig. 4). In contrast, the School controls produced a mean sediment flux rate of 7.2 g s^{-1} in the first year, and while there were non-significant decreases in sediment flux in the second and third post-fire years, these values were still 7 or more times the rates from the Red Eagle controls (Fig. 4). The mean sediment flux from the controls at Terrace Mountain in the first post-fire year was only 0.38 g s^{-1}, or only about 5% of the sediment flux in the first post-fire year at the School site, and this value did not change significantly in the second year (Fig. 4).

All the logged plots at each site had significantly greater sediment flux rates than their respective controls in the year logging occurred, except for the low use forwarder trails at the School site (Fig. 4). By the next year, none of the differences persisted, however, because there were significant decreases in sediment flux in

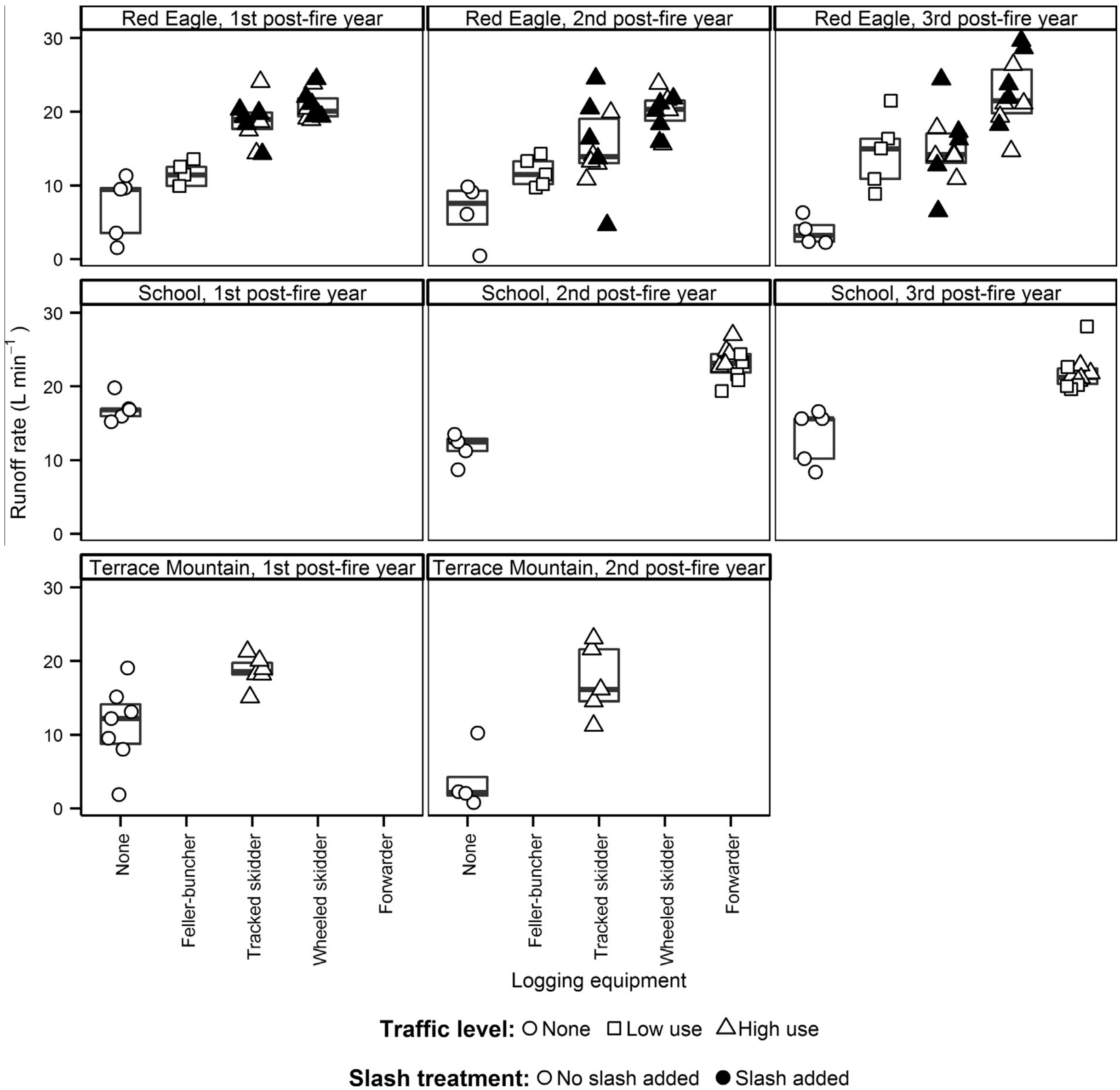

Fig. 3. Mean plot runoff rate by type of logging equipment and post-fire year for each site. Symbol fill indicates slash treatment and symbol shape indicates traffic level. Boxplot characteristics are the same as in Fig. 2. Individual plot values are shown in Supplemental material and results of statistical analyses are shown in Table A2.

all of the logging equipment impacted plots. The addition of slash did not significantly reduce the sediment flux rates in the tracked and wheeled skidder plots (Fig. 4). The low use forwarder plots had significantly lower sediment fluxes than the high use skidder plots in the year of logging, and the difference was nearly significant ($p = 0.086$) in the subsequent year.

4. Discussion

4.1. Effects of logging equipment on soil bulk density

The higher bulk densities in the logged plots than the controls were expected as ground-based logging equipment has been shown to increase soil compaction in unburned forests (e.g., Greene and Stuart, 1985; Horn et al., 2004; Page-Dumroese et al., 2006) and in burned forests (Wagenbrenner et al., 2015). The lack of differences in bulk density between the feller-buncher and the tracked or wheeled skidders at Red Eagle suggests that even a single pass by heavy logging equipment is sufficient to compact burned soil. Similarly, at the School site, there was no difference in bulk density between low use and high use forwarder trails, and both traffic levels compacted the soil (Fig. 2).

The School site had surprisingly uniform bulk density regardless of depth in the control plots, and the wheeled forwarders produced significant increases in density, but only at depths below 5 cm. Field observations suggest the surface soil at all sites showed signs of compaction where track or wheel traffic occurred but also showed signs of a surface layer of loose soil resulting from slipping or turning of the wheels or tracks. Our bulk density sampling design did not attempt to separate compacted or loose soil areas, and this may have resulted in more variability in the surface bulk density results. Wheeled equipment generally apply greater static pressures to the soil than tracked equipment of equal weight as the tires distribute the weight over a smaller surface area as compared to the area of tracks (Cambi et al., 2015). This may explain the slightly higher bulk density values in the wheeled skidder plots as compared to the tracked skidder plots at Red Eagle (Fig. 2).

Researchers in Germany tested a variety of tracked and wheeled timber harvesting equipment in unburned forests, including machines that were lighter and heavier than those used in our sites, and

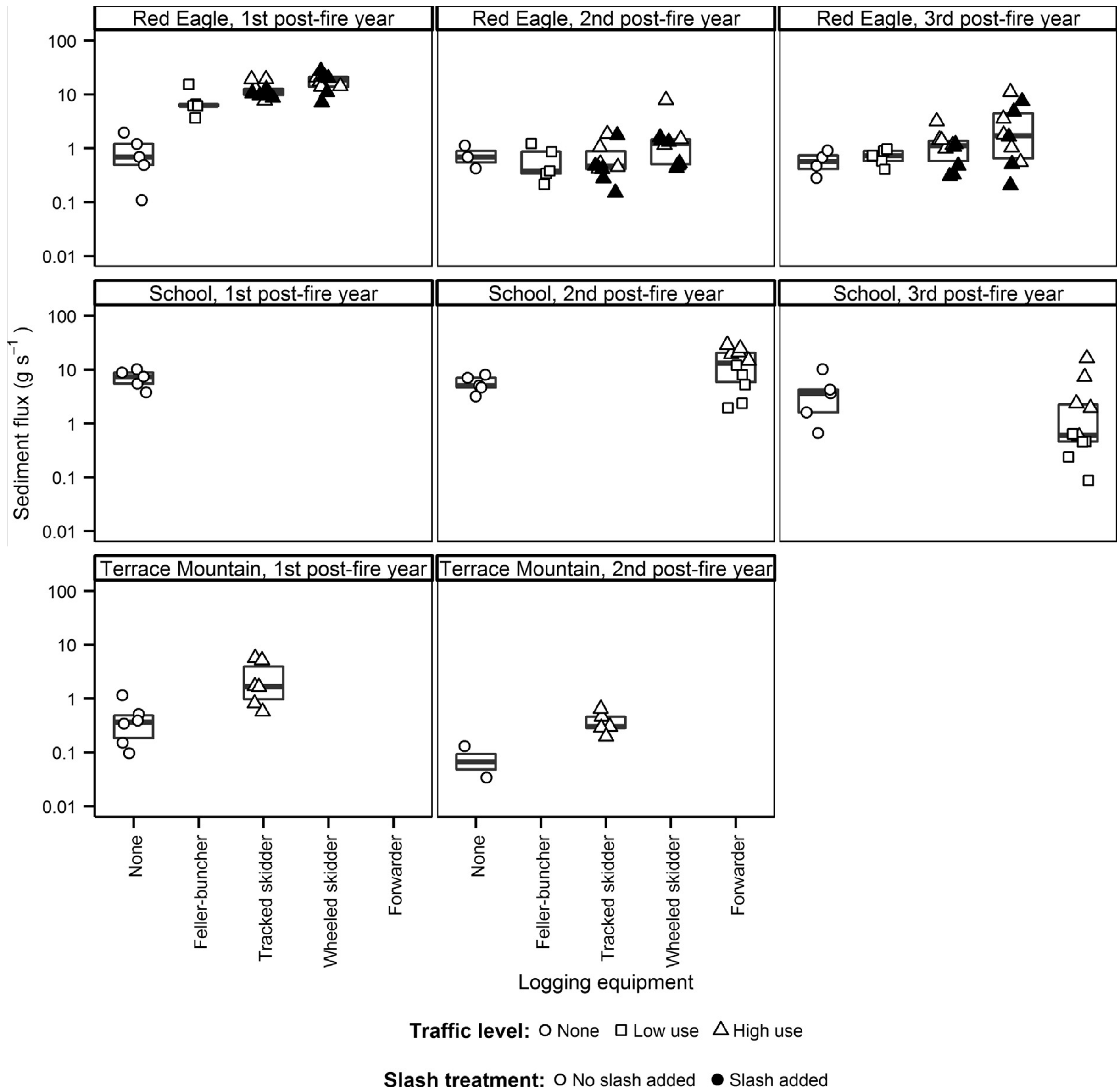

Fig. 4. Mean plot sediment flux by type of logging equipment and post-fire year for each site. Symbol fill indicates slash treatment and symbol shape indicates traffic level. Boxplot characteristics are the same as in Fig. 2. Individual plot values are shown in Supplemental material and results of statistical analyses are shown in Table A3.

determined that even the lightest equipment caused severe compaction (Horn et al., 2004). Logging equipment with lower weights or much lower ground pressures than traditional equipment might provide some reduction in soil compaction. Low-impact harvest methods, such as helicopter or sky line yarding, should also be considered to reduce soil compaction (Klock, 1975). Re-use of existing skid trails would reduce the added impact associated with salvage logging but may also greatly restrict the access to burned timber in a salvage harvest. Logging over ground covered in deep snow, where possible, may also reduce logging equipment impacts.

Slash was added to the slash-treated plots after skidding, so the slash treatment did not affect soil bulk density. Slash mats installed prior to equipment traffic in unburned forests can reduce soil compaction by logging equipment, especially with multiple equipment passes (Ampoorter et al., 2007; Eliasson and Wästerlund, 2007; McDonald and Seixas, 1997). However, the potential of slash mats to reduce soil compaction in burned systems, where fire has reduced soil cover, altered soil particle size and soil structure (Blake et al., 2005), and reduced soil organic matter, has not yet been demonstrated.

4.2. Effects of logging equipment and site characteristics on runoff and sediment flux

Several factors may have attributed to the increased runoff rates and sediment fluxes in the plots disturbed by the logging equipment. We attribute the increases in runoff in the plots with logging equipment traffic to lower infiltration rates resulting from reduced micro and macro porosity (Ares et al., 2005; Horn et al., 2004; Startsev and McNabb, 2000) associated with the measured soil compaction.

The change in infiltration was probably the dominant process change, but other factors also contributed to the higher runoff and sediment flux rates. The soil compaction by the equipment also produced shallow ruts, typically less than 5 cm deep, in the trafficked plots. Because our goal was to measure the effects of logging equipment traffic on rill runoff and erosion rates, we located the top of the logged plots in the tracks and applied the simulated runoff water in tracked areas. In most cases, the depressions served to constrain the flow and kept the flow from meandering outside the tracked areas, and this resulted in a smaller number of shorter flow paths with narrower and deeper flows (Table 3). In a natural runoff event, the track or wheel ruts would have a greater chance to concentrate sheet flow into rills, resulting in a shorter time of concentration and a greater chance of sediment delivery to the stream network. There may be an opportunity to reduce this concentrating effect during the layout of the skid trails, landings, and drainage control features.

The one main exception to the constraining effect of the ruts occurred in the skid trail plots with added slash at the Red Eagle site, where the slash sometimes caused the flow to disperse as compared to the untreated skid trail plots. The effect of the extra slash on rill erosion was less than expected because the slash was hand-placed on the plots after the skidding, and the smaller pieces of slash did not always have continuous ground contact. This allowed some rills to flow under the slash with less obstruction than if the slash had more complete ground contact.

Equipment paths in burned areas tend to avoid obstructions like stumps, stump holes, or large rock outcrops that can split or detain flow, thereby biasing the equipment tracks toward less opportunity for flow divergence or detention. The effect of this unintended bias would be to increase hydrologic connectivity between the hillslopes and the stream network. Water bars are intended to interrupt the flow in skid trails, but the flow diverted from the outlets of water bars in burned areas may not readily infiltrate because of the low infiltration rates found in severely burned areas in the first few years after the fire (Larsen et al., 2009; Moody and Ebel, 2013). Other best management practices are needed to allow the concentrated flow in skid trails to be dispersed, such as the addition of slash or mulch to skid trails or armoring water bar outlets. Methods to increase infiltration that can be implemented at operational scales are needed to disconnect the compacted skid trails and stream channel networks.

Despite the higher runoff rates in the logged plots as compared to the controls, the runoff velocities in the logged plots were generally similar to or lower than the velocities in the control plots. The lack of increase in velocity in the skidder and forward plots was somewhat surprising, and we suggest that the increases in surface roughness contributed by particle size and rill flow path irregularity in the equipment tracks compensated for the lack of vegetation and surface cover and its corresponding roughness contribution (Giménez and Govers, 2001; Nearing et al., 1997; Shakesby et al., 2007; Wagenbrenner et al., 2010). The increased erosion rates in the skidder and forwarder plots resulted in more rapid coarsening of the surface, leading to more grain roughness than in the control plots. Also, the tracked cleat and tire tread impressions created greater rill flow path variation and roughness in the trafficked plots than in the controls. Consequently, the combined net roughness in the trafficked plots was greater than in the controls, which led to the similar or lower runoff velocities in the trafficked plots despite their narrower and deeper flows. The slash treatment at Red Eagle increased the roughness contribution by surface cover, and this caused the velocities in the slash-treated skidder plots to be consistently but not significantly lower than velocities in the untreated skidder plots.

Several studies have concluded that fire-induced water repellency may increase hillslope runoff rates or erosion (Ahn et al., 2013; Benavides-Solorio and MacDonald, 2001; Doerr et al., 2009; Prats Alegre et al., 2016; Robichaud, 2000). At the Red Eagle site, we measured high soil water repellency in the control plots and low water repellency in the plots subject to logging equipment traffic. In apparent contrast to the water repellency results, the control plots had lower runoff and sediment flux rates than the logged plots at that location. We attribute the net increase in runoff and sediment fluxes to the soil compaction and loss of vegetative cover. Our results show that reducing post-fire water repellency by disturbance with logging equipment did not provide any benefit with respect to lower runoff or rill erosion rates.

The churning of the surface soil and transient layer of fine particles in the skidder and forwarder plots, confirmed by our visual observations of looser soil as compared to nearby untrafficked areas, led to greater availability of source material in the plots with equipment traffic. When combined with the greater runoff rates and depths in the trafficked plots, more soil detachment and transport occurred, greatly increasing the sediment flux rates in the plots with logging equipment traffic. Rainsplash and sheetwash may further increase rill sediment delivery rates under natural rainfall given the amount of exposed soil in the areas with logging equipment traffic (Bryan, 2000; Wagenbrenner et al., 2015).

4.3. Short-term post-fire and post-salvage recovery

In addition to the disturbance of the vegetation by logging equipment, vegetation recovery might have been hampered by the soil compaction (Page-Dumroese et al., 2006) or lower water availability (Marañón-Jiménez et al., 2013) caused by the equipment traffic. Site productivity may have also contributed to differences in vegetation recovery rates between the School and Red Eagle sites (Morgan et al., 2014). The lower vegetative regrowth rates in the logged plots meant that by the third post-fire year the vegetation cover values in the logged plots were closer to the second post-fire year values in the control plots. Reduced vegetation cover was also reported in the School Fire and other salvage logged areas in the western US (Morgan et al., 2014; Sexton, 1998).

The difference in runoff rates between the control plots and the logging equipment impacted plots persisted over time, and this suggests the short-term hydrologic recovery of the burned area was hampered in areas subject to logging equipment traffic. Although the vegetative recovery rate in the salvage logged burned areas was slower than the recovery rate in the unlogged areas, the vegetation recovery will probably be more rapid than the recovery of the soil compaction (Labelle and Jaeger, 2011). We anticipate the recovery of vegetation and the ensuing increases in surface cover will therefore have a great impact on reducing the medium and long term runoff responses in the salvage logged areas.

There were smaller relative changes in sediment flux rates in the control plots than in the logged plots over time. By the year after logging, there were no differences in sediment flux between the logged and control plots. The recovery in sediment fluxes probably reflects the change in sediment availability as the loose, easily transportable soil was removed (Shakesby et al., 2007), rather than a change in transport capacity which would remain relatively high with the high runoff rates in the logged plots.

4.4. Slash treatment effectiveness and other management concerns

In contrast to previous sediment delivery results from natural runoff events at the Red Eagle site (Wagenbrenner et al., 2015), the slash treatment had no significant effect on the runoff rates, runoff velocities, or sediment flux rates in the current study. We believe the lack of difference was because the hand-placed slash left some gaps between the slash and the soil surface, which allowed some of the runoff to pass unimpeded below the slash. The slash placed in the same manner would reduce soil sealing as well as rain splash and sheet erosion, and in our earlier study, adding slash resulted in a significant reduction in sediment delivery rates from natural rainfall events (Wagenbrenner et al., 2015). To be most effective at reducing rill erosion, slash should be placed on the slopes during the skidding operation so that logging equipment could break up and incorporate the slash into the soil and the slash would alleviate some of the ground pressure on the soil (Eliasson and Wästerlund, 2007), or the slash should be of sufficient size to allow for greater soil contact. Incorporating slash into skid trails may not reduce runoff generation. However, the likely effect of the increase in soil cover would be to disperse runoff and increase surface roughness, thereby reducing the erosivity of the flow. This approach might prove especially useful at water bar outlets, where relatively deep concentrated flow has the potential to scour burned soil. Mulching, slashing, or otherwise armoring skid trails and water bar outlets may reduce rill incision in these highly erodible areas.

Physically-based erosion models usually incorporate erodibility parameters developed from hydraulic descriptors such as shear stress, stream power, or unit stream power (Bryan, 2000). Our results will support parameterization of a physically-based runoff and erosion model for post-fire salvage logging that would complement earlier models on fire effects (Elliot, 2004) and mitigation practices (Robichaud et al., 2007).

5. Conclusion

We measured soil properties, surface cover, and runoff and sediment responses in three burned and logged forests for multiple years. Ground-based logging operations in recently burned areas compacted soils and reduced soil water repellency. Traffic by feller-bunchers, skidders, and forwarders in burned forests removed virtually all live vegetation within the equipment trails, even at the School site where one year of additional post-fire recovery had occurred prior to logging. The vegetation cover recovered at a slower rate after salvage logging than in the control plots. The changes induced by salvage logging to soils and vegetation led to reduced infiltration and increased runoff rates as well as increases in sediment fluxes in our simulated rill experiments. Rill sediment flux rates in almost all the plots disturbed by logging equipment were significantly greater than the rates in the control plots in the year logging occurred. Fewer statistical differences in sediment flux rates among logging treatments were observed in subsequent years despite sustained high runoff rates.

Among the different types of logging equipment used at the Red Eagle site, we found no significant differences in soil properties or cover until the third post-fire year, and only the feller-buncher plots had significantly lower runoff or sediment flux rates during some of the measurement periods. Manually adding slash to the skid trails after logging at Red Eagle did not affect runoff rates or sediment flux rates. There were no differences in soil properties, surface cover, or runoff rates between the two levels of forwarder traffic at the School site, but the sediment fluxes in the low use forwarder plots were lower than the sediment fluxes in the high use forwarder plots.

The increased runoff rates due to logging lasted for up to three post-fire years and runoff rates in the logged plots did not recover as in the control plots. Our results suggest the need to develop best management practices designed specifically for logging in burned areas to reduce the impacts of logging equipment on post-fire sediment production and reduce the risk of sediment delivery to the stream network.

Table A1
Tukey-Kramer adjusted p-values and t-values (in parentheses) for differences in least-squares means between treated and control plots for bulk densities shown in Fig. 2. There was no difference in the bulk densities between the slash added and no slash added skid trail plots at Red Eagle ($p > 0.26$) so these treatments were combined. No statistical comparisons were possible with Terrace Mountain data. Positive t-values indicate a smaller mean than the reference value.

Red Eagle			School	
Feller-buncher	Tracked skidder	Wheeled skidder	Forwarder	
			Low use	High use
0–5 cm				
0.043	0.007	0.001	0.62	0.45
(−2.8)	(−3.7)	(−4.4)	(−0.98)	(−1.3)
5–10 cm				
0.32	0.46	0.20	0.007	0.003
(−1.8)	(−1.5)	(−2.1)	(−4.8)	(−5.9)
10–15 cm				
0.22	0.32	0.002	0.005	0.002
(−2.0)	(−1.8)	(−4.1)	(−5.2)	(−6.2)

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Appendix A

See Tables A1–A3.

Table A2
Tukey-Kramer adjusted p-values and t-values (in parentheses) for differences in least-squares means between treated and control plots for runoff rates shown in Fig. 3. Year to year comparisons are shown for the control plots and comparisons between slash added and no slash added skidder are shown for the Red Eagle site. Positive t-values indicate a smaller mean than the reference condition.

Yr	Control	Red Eagle							School			Terrace Mtn.	
		vs. controls					vs. no slash		Control	Forwarder		Control	Tracked skidder
		Feller-buncher	Tracked skidder		Wheeled skidder		Tracked skidder	Wheeled skidder		Low use	High use		
			No slash	Slash added	No slash	Slash added	Slash added	Slash added					
1		0.94 (−1.8)	0.002 (−4.8)	0.004 (−4.6)	0.0003 (−5.4)	<0.0001 (−5.8)	1.0 (0.20)	1.0 (−0.35)		No data	No data		0.089 (−2.9)
2	1.0[a] (0.44)	0.75 (−2.2)	0.19 (−3.1)	0.038 (−3.8)	0.0002 (−5.5)	0.0006 (−5.1)	1.0 (−0.71)	1.0 (0.36)	0.03[a] (3.6)	0.0002 (−6.6)	<0.0001 (−8.2)	0.006[a] (5.1)	0.005 (−5.2)
3	1.0[b] (0.97)	0.014 (−4.2)	0.019 (−4.0)	0.005 (−4.5)	<0.0001 (−6.5)	<0.0001 (−8.0)	1.0 (−0.49)	0.98 (−1.6)	0.94[b] (−1.0)	0.0006 (−5.8)	0.001 (−5.4)	No data	No data

[a] Comparison between year 2 control and year 1 control values.
[b] Comparison between year 3 control and year 2 control values.

Table A3
Tukey-Kramer adjusted p-values and t-values (in parentheses) for differences in least-squares means between treated and control plots square-root transformed sediment flux rates shown in Fig. 4. Year to year comparisons are shown for the control plots and comparisons between slash added and no slash added skidder are shown for the Red Eagle site. Positive T-values indicate smaller means than the reference condition.

Yr	Control	Red Eagle							School			Terrace Mtn.	
		vs. controls					vs. no slash		Control	Forwarder		Control	Tracked skidder
		Feller-buncher	Tracked skidder		Wheeled skidder		Tracked skidder	Wheeled skidder		Low use	High use		
			No slash	Slash added	No slash	Slash added	Slash added	Slash added					
1		0.0016 (−4.9)	<0.0001 (−7.7)	<0.0001 (−6.4)	<0.0001 (−8.9)	<0.0001 (−8.7)	1.0 (1.3)	1.0 (0.2)		No data	No data		0.0010 (−4.0)
2	1.0[a] (0.6)	1.0 (−0.2)	1.0 (−0.6)	1.0 (−0.2)	0.97 (−1.7)	1.0 (−0.9)	1.0 (0.5)	1.0 (0.8)	0.99[a] (0.7)	1.0 (0.07)	0.0043 (−4.8)	0.62[a] (1.2)	0.58 (−1.3)
3	1.0[b] (−0.2)	1.0 (−0.2)	1.0 (−1.3)	1.0 (−0.1)	0.67 (−2.3)	0.92 (−1.9)	1.0 (1.2)	1.0 (0.5)	0.93[b] (1.1)	0.19 (2.6)	1.0 (−0.5)	No data	No data

[a] Comparison between year 2 control and year 1 control values.
[b] Comparison between year 3 control and year 2 control values.

Appendix B. Supplementary material

Supplementary data associated with this article can be found in the online version at http://dx.doi.org/10.1016/j.jhydrol.2016.07.049.

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