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Effects of post-fire salvage logging and a skid trail treatment on ground cover, soils, and sediment production in the interior western United States

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ABSTRACT

Post-fire salvage logging adds another set of environmental effects to recently burned areas, and previous studies have reported varying impacts on vegetation, soil disturbance, and sediment production with limited data on the underlying processes. Our objectives were to determine how: (1) ground-based post-fire logging affects surface cover, soil water repellency, soil compaction, and vegetative regrowth; (2) different types of logging disturbance affect sediment production at the plot and small catchment ("swale") scales; and (3) applying logging slash to skid trails affects soil properties, vegetative regrowth, and sediment production. Four study areas were established in severely burned forests in the interior western USA. We installed plots at two study areas to compare burned but unlogged controls against skid trails, feller-buncher trails, and skid trails with added slash. Salvage logged and control swales were established at each study area, but only one study area had simultaneous measurements on replicated swales. Data were collected for 0-2 years prior to logging and from 2-8 years after logging.

The skidder and feller-buncher plots generally had greater compaction, less soil water repellency, and slower vegetative regrowth than the controls. Sediment production from the skidder plots was 10–100 times the value from the controls. The slightly less compacted feller-buncher plots produced only 10–30% as much sediment as the skidder plots, but regrowth was similarly inhibited. The relative differences in sediment production between the disturbed plots and the controls tended to increase over time as the controls exhibited more rapid regrowth. Adding slash to skid trails increased total ground cover by 20–30% and reduced the sediment yields by 5–50 times compared to the untreated skidder plots.

The replicated logged swales at one study area generally had higher sediment production rates than their controls but the absolute values per unit area were much lower than from the skidder and feller-buncher plots. Results from the swales at the other study areas indicated that logging did not increase runoff, peak flows, or sediment yields.

Vegetative regrowth and sediment production rates varied widely among the four study areas. This variation was largely due to differences in rainfall and soil properties, with the more productive sites having more rapid regrowth and thereby a more rapid reduction in sediment production. The susceptibility to surface runoff and erosion after high severity fires suggests that areas disturbed by ground-based salvage logging need additional mitigation practices.

1. Introduction

Moderate and high-severity wildfires in coniferous forests often kill nearly all of the trees, and there is a strong economic incentive to capture the market value of the timber before the wood decays. Salvage logging after such fires is a highly controversial activity, as

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there is little consensus on the extent to which post-fire salvage logging can either exacerbate or mitigate the effects of a wildfire on vegetative regrowth, soil water repellency, and/or surface runoff and erosion (McIver and Starr, 2001; Peterson et al., 2009). Previous studies on post-fire salvage logging have found widely varying impacts on physical soil properties, vegetative cover, and sediment production with only limited data on the underlying causal processes. More data and a better understanding of the effects of post-fire salvage logging is essential given the already-observed and projected increases in the number, extent, and severity of wildfires in the western U.S. and elsewhere (Flannigan et al., 2009; Littell et al., 2009).

The primary difficulties in predicting the effects of post-fire salvage logging are that the additional changes are being superimposed on a system that already has been highly altered by fire, and that logging can have counteracting effects on the underlying causal processes. A rapidly growing list of studies have documented the large increases in peak flows, erosion, and downstream flooding and sedimentation that can occur after moderate and high severity wildfires (e.g., Moody and Martin, 2009; Shakesby and Doerr, 2006). The increased runoff and erosion is driven by four main ecosystem state changes (Fig. 1), namely the: (1) reduction in plant canopy; (2) increase in soil water repellency; (3) loss of surface cover; and (4) consumption of soil organic matter. These state changes then cause a series of interacting process changes, such as reduced interception, higher soil erodibility, and increased runoff velocities (Fig. 1). The relative importance of each state and process change is still a matter of debate given the diversity in site conditions and multiple interactions among the observed changes. Our premise is that the net effect of post-fire salvage logging on runoff and erosion can only be understood and predicted by evaluating the incremental effects of the post-fire logging on the state variables and underlying processes.

With respect to the post-fire state changes, the loss of the plant canopy mainly affects the overall water balance and net precipitation, and this is generally regarded as having a smaller effect on post-fire peak flows and hillslope erosion than the changes in surface cover and soil properties (Shakesby and Doerr, 2006). The second state change is the commonly-observed increase in soil water repellency at or beneath the soil surface, and this has often been suggested as the primary cause for the observed increase in post-fire runoff and erosion (Doerr et al., 2006). Some recent studies have questioned the importance of soil water repellency given its extent prior to burning (Doerr et al., 2009), its high spatial variability (Woods et al., 2007), the relatively rapid decay of repellency compared to the duration of the increased runoff and erosion (Huffman et al., 2001; Larsen et al., 2009), and the loss of repellency at a threshold soil moisture content (Doerr and Thomas, 2000; MacDonald and Huffman, 2004).

Regarding surface cover, numerous studies have shown a strong, nonlinear relationship between the loss of surface cover and increase in surface erosion (e.g., Cerdà and Doerr, 2008; Larsen et al., 2009; Morris and Moses, 1987; Wagenbrenner and Robichaud, 2014). A loss of surface cover also will increase overland flow velocities, and this alone can increase peak flows, sheetwash, rilling and gullying (McGuire et al., 2013; Robichaud et al., 2010; Shakesby and Doerr, 2006). The relative importance of surface cover as a controlling process is further supported by the fact that mulching is generally the most effective treatment in reducing post-fire runoff and erosion (e.g., Robichaud et al., 2013a; Wagenbrenner et al., 2006). The loss of soil organic matter is the fourth state change, and this increases soil erodibility and thus

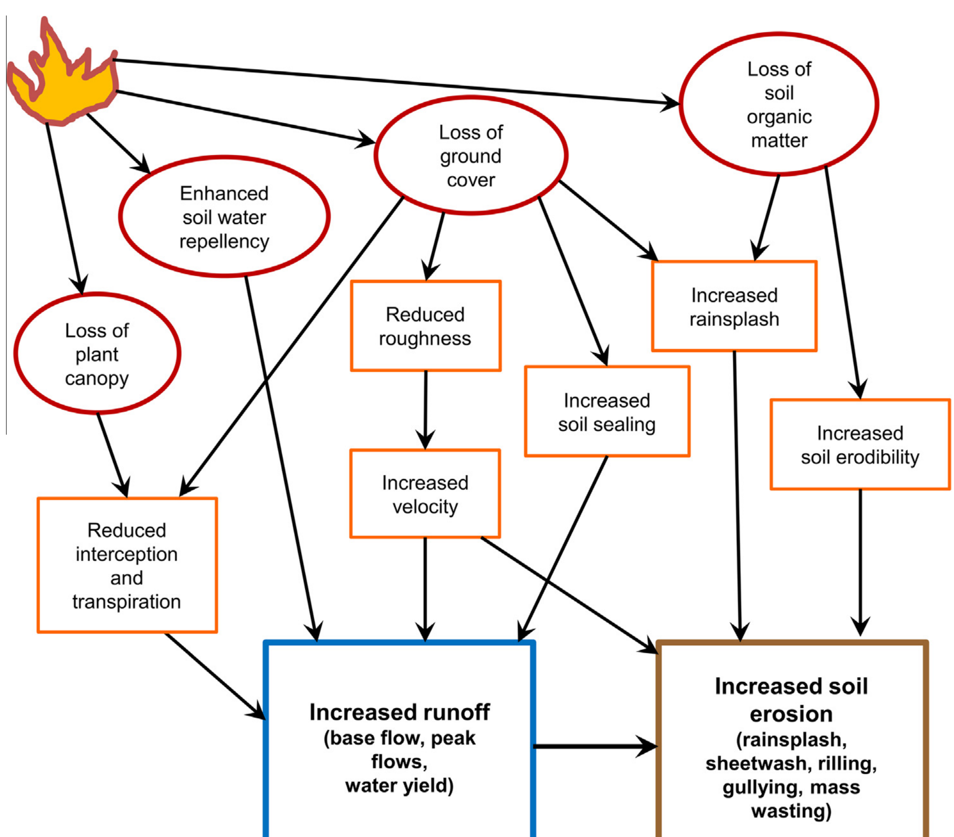

Fig. 1. Conceptual overview of the state changes and processes that can cause increased runoff and erosion after moderate or high severity wildfires. Ovals represent a change in state, and rectangles represent the resulting change in processes.

the susceptibility to rainsplash, soil sealing, and hydraulic erosion by surface runoff (Fig. 1).

The second difficulty—the multiple and counteracting effects of salvage logging—is illustrated by the conflicting claims and results from the literature. Some studies (McIver and Starr, 2001; Poff, 1989) have argued that salvage logging can be beneficial if it sufficiently disturbs the soil surface to break up the water repellent layer and increase infiltration; however, detailed data have not been provided to support these arguments. Similarly, if salvage logging increases surface cover by the addition of logging slash it should decrease erosion rates, but if logging reduces surface cover it should increase runoff and erosion (Silins et al., 2009). The effect of salvage logging on vegetative regrowth also is highly uncertain, as the effect of logging immediately after a fire may be quite different than the effect of logging two or more years after burning when vegetative regrowth is underway (Donato et al., 2006; Fernández et al., 2007). Finally, logging machinery may compact the soil and reduce infiltration, which could exacerbate a reduction in surface cover or counteract a net increase in surface cover from logging slash (Page-Dumroese et al., 2006).

Ground-based salvage logging typically is of greater concern than cable or helicopter logging due to the greater ground disturbance and potential for adverse effects from skid trails (McIver and Starr, 2001). Numerous studies under unburned conditions have shown that skid trails reduce infiltration and surface cover and can concentrate overland flow, resulting in increased surface runoff and erosion (Anderson et al., 1976; Ares et al., 2005; Megahan and Kidd, 1972). The relative effect of skid trails, or cableways in cable logging systems, will depend on their spatial extent, topographic orientation, and the relative change in runoff and erosion compared to undisturbed burned areas (Smith et al., 2011).

Studies of sediment production after post-fire salvage logging generally have not been able to discern an increase that can be directly attributed to logging, although the results have been mixed and complicated by various site-specific factors. In northeastern Oregon salvage logging increased soil disturbance, but most of the erosion and sediment transport was due to the existing road system (McIver and McNeil, 2006). The relatively low levels of hillslope sediment transport were attributed to low-to-moderate slopes, relatively erosion-resistant soils, logging over snow and dry ground, two years of recovery between burning and logging, and the absence of severe rainstorms during the post-logging period (McIver and McNeil, 2006). In Spain the absence of severe rainstorms and relatively low amounts of bare soil contributed to the lack of any significant effect of post-fire clear-cutting on sediment yields (Fernández et al., 2007; Marques and Mora, 1998). In Greece salvage logging had transient effects on soil pH, soil organic matter content and soil moisture, but only minor effects on sediment yields (Spanos et al., 2005). On the Apache National Forest in Arizona burning caused a 150-fold increase in sediment yield compared to unburned areas, but salvage logging did not have any detectable effect on sediment yields relative to burned but unlogged areas (Stabenow et al., 2006). Post-fire salvage logging did increase surface runoff and erosion in a rainfall simulation experiment on fragile granitic soils in Tasmania, and this increase was attributed in part to the disruption of a biotic crust by heavy equipment (Wilson, 1999).

Effects of salvage logging are more difficult to demonstrate at the catchment (>10 ha) scale. Data from seven catchments in the Canadian Rocky Mountains suggested that salvage logging increased total suspended sediment concentrations relative to burned and unlogged controls, but the differences were not statistically significant (Silins et al., 2009). In granitic terrain in New South Wales the combination of a severe fire and salvage logging caused a 100% increase in peak discharge and a 30–40% increase in stormflow volume, but this was based on only one storm in one catchment (Mackay and Cornish, 1982). Twelve percent of the logged catchment was severely compacted in roads, skid trails and landings, so these increases are consistent with some studies in unburned areas (Ares et al., 2005; Ziemer, 1981). In southeastern Australia salvage logging of a burned pine plantation increased total sediment yields by 180 and 33 times relative to two burned but unlogged catchments dominated by eucalypts (Smith et al., 2011). Although some of these differences could be attributed to the higher burn severity and greater soil water repellency under the pine, the increase was largely attributed to drag lines (skid trails) that concentrated runoff and converged downslope to form gullies (Smith et al., 2011). The effectiveness of mitigation treatments to reduce erosion, such as the placement of logging slash and mulch, has been proven in both burned and unburned areas, but the relative effectiveness of these treatments for post-fire salvage logging have not been rigorously studied.

Given this background, the impetus for the present study came from the need for a more integrated and process-based approach to disentangle the interacting effects of fires and logging, the need to compare results from different study areas, and the need to combine process-based, plot-scale studies with more spatially-integrated effects at the swale or small catchment scale. The specific objectives were to determine: (1) how post-fire salvage logging affects surface cover, soil water repellency, and soil compaction in different study areas over time relative to burned but unlogged areas; (2) whether post-fire salvage logging increases sediment production at the plot and swale scales relative to burned but unlogged areas; and (3) how the application of logging slash to skid trails affects soil properties, vegetative regrowth, and sediment production over time.

2. Methods

2.1. Study areas

Site conditions and sediment production rates were measured in burned unlogged (control) and salvage logged sites in four study areas in the western U.S. (Fig. 2). At Red Eagle and Tripod we measured post-fire conditions and sediment production at both the plot (33–174 m^2) and swale (0.1–2.6 ha) scales for the first four years after burning (Table 1). At Hayman and Kraft Springs we measured post-fire conditions and sediment production from unreplicated control and salvage-logged swales 2.8–3.6 ha in size for nine and six years, respectively (Table 1).

Mean annual precipitation from the nearest long-term gage is more than 1200 mm at Red Eagle and from 340 to 400 mm for Tripod, Hayman, and Kraft Springs (Fig. 3). All of the study areas were dominated by ponderosa pine (*Pinus ponderosa* Dougl.), lodgepole pine (*Pinus contorta sbsp. latifolia* Engelm.), or Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco.). All of the sites also had relatively coarse-textured soils (Table 2). Logistical constraints resulted in variability among study areas with respect to the number and type of plots, and the timing of the salvage logging relative to the fire (Table 1).

Red Eagle, Hayman, and Kraft Springs were salvage logged in the year after burning, and each of these study areas included burned control and salvage logged sites. Tripod was logged after two years of post-fire monitoring, which permitted before and after logging comparisons in the same swales as well as comparisons among control and logged plots after logging occurred (Table 1). Similar ground-based logging systems were used during snow-free periods in each study area, with feller-bunchers cutting and piling the trees into bunches of 2–8 trees. Rubber-tired or steel-tracked skidders lifted one end of a bunch and dragged the trees to a staging area where the logs were processed for transport.

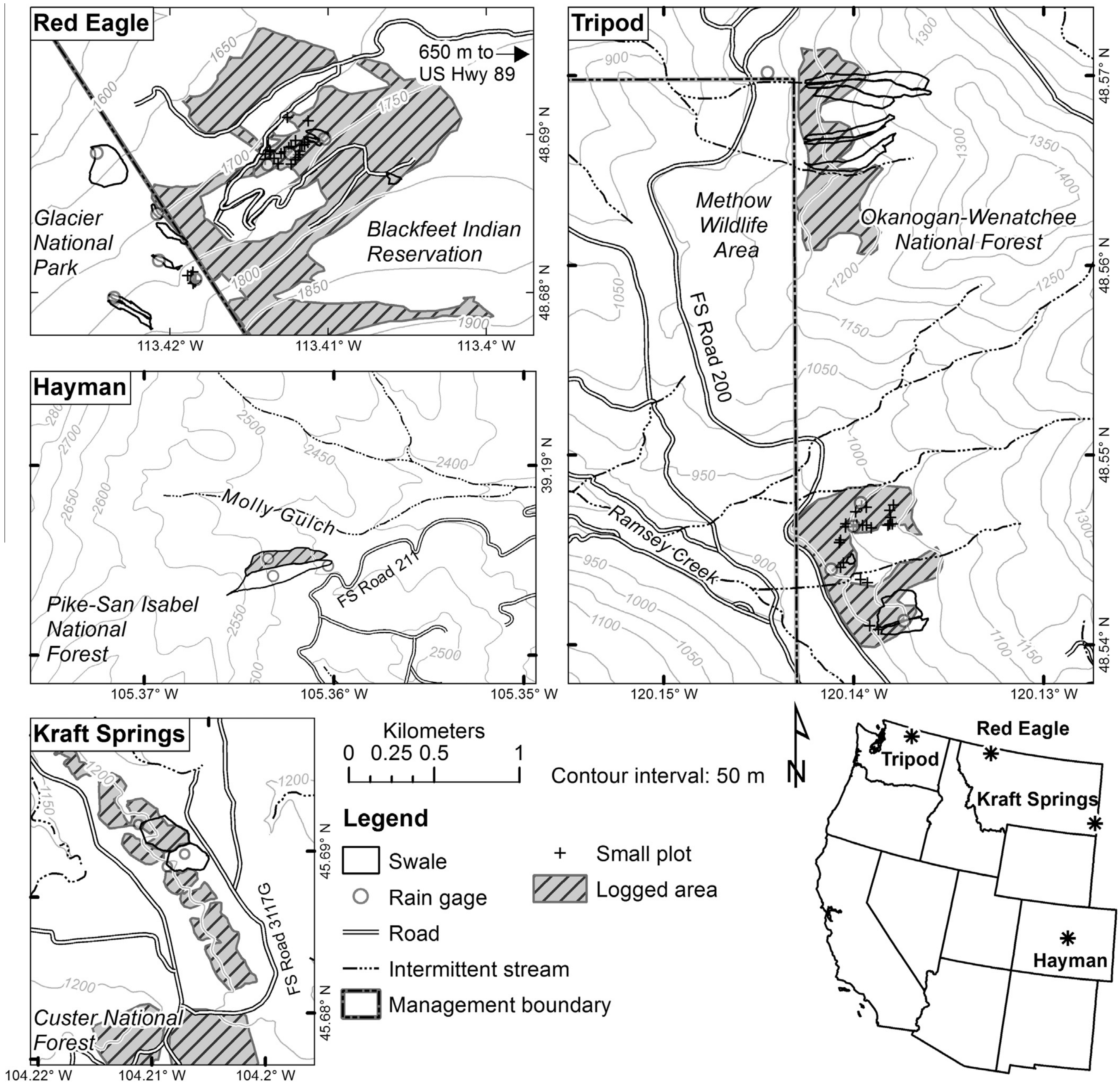

Fig. 2. Locations of the study areas in the western U.S. and detailed maps of each of the four study areas

The feller-buncher plots thus had only feller-buncher traffic, while the skidder plots had both feller-buncher traffic and at least two skidder passes. At Red Eagle and Tripod we hand-spread logging slash on half of the skidder plots to create slash-treated plots. These three types of disturbed plots were compared to burned, unlogged control plots. Upslope runoff was excluded from each plot by either a water bar or a diversion trench.

By virtue of their larger size, the logged swales included both feller-buncher and skid trails as well as areas with little or no ground disturbance. At Hayman and Kraft Springs skidders were used to install water bars and spread logging slash on the skid trails.

We measured several properties in each plot and swale and these methods are described following a brief description of each study area. Contributing area was measured in the field to express runoff and sediment yields per unit area. Plot slopes were measured with a clinometer and swale slopes were determined with a clinometer or from a 10-m digital elevation model.

2.1.1. Red Eagle

The 14,000 ha Red Eagle Fire burned in 2006, straddling the boundary between Glacier National Park and the Blackfeet Indian Reservation in northern Montana (Fig. 2). Rapid salvage logging on the Blackfeet Reservation in 2007 provided an opportunity to compare logged areas with adjacent burned but unlogged National Park land. The study area burned at high to moderate soil burn severity following Parsons et al. (2010). All sites were coniferous forest, but the logged sites were primarily lodgepole pine while the controls were more diverse with some subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.) and Engelmann spruce (*Picea engelmannii* Parry ex Engelm.). The 27 plots (33–174 m^2) and nine swales (0.1–2.6 ha) (Table 1) were established immediately after logging.

Table 1
Year of fire and logging, years of data collection, the number, type, and size of plots and swales, and the number of sediment clean-outs by study area. The values in parentheses are the minimum and maximum plot and swale sizes in m^2 and ha, respectively.

Study area	Fire year [logging year]	Years	Number of plots (min. and max. size in m^2)				Number of swales (min. and max. size in ha)		Sediment clean-outs
			Control	Feller-buncher	Skidder	Slash-treated	Control	Logged	
Red Eagle	2006 [2007]	2007–2010	6 (124–174)	7 (33–123)	8 (52–128)	6 (45–143)	6 (0.1–2.6)	3 (0.2–0.3)	8
Tripod	2006 [2009[a]]	2007–2008	0	0	0	0	6 (0.4–2.6)	0	3
Tripod	2006 [2009[a]]	2009–2010	6 (67–82)	6 (65–80)	6 (69–80)	6 (68–82)	0	9 (0.1–2.6)	6
Hayman	2002 [2003]	2002–2010	0	0	0	0	1 (3.0)	1 (2.9)	16
Kraft Springs	2002 [2004]	2003–2008	0	0	0	0	1 (2.8)	1 (3.6)	4

[a] Tripod swales were logged in spring 2009. Plots and three additional logged swales were installed at Tripod after logging.

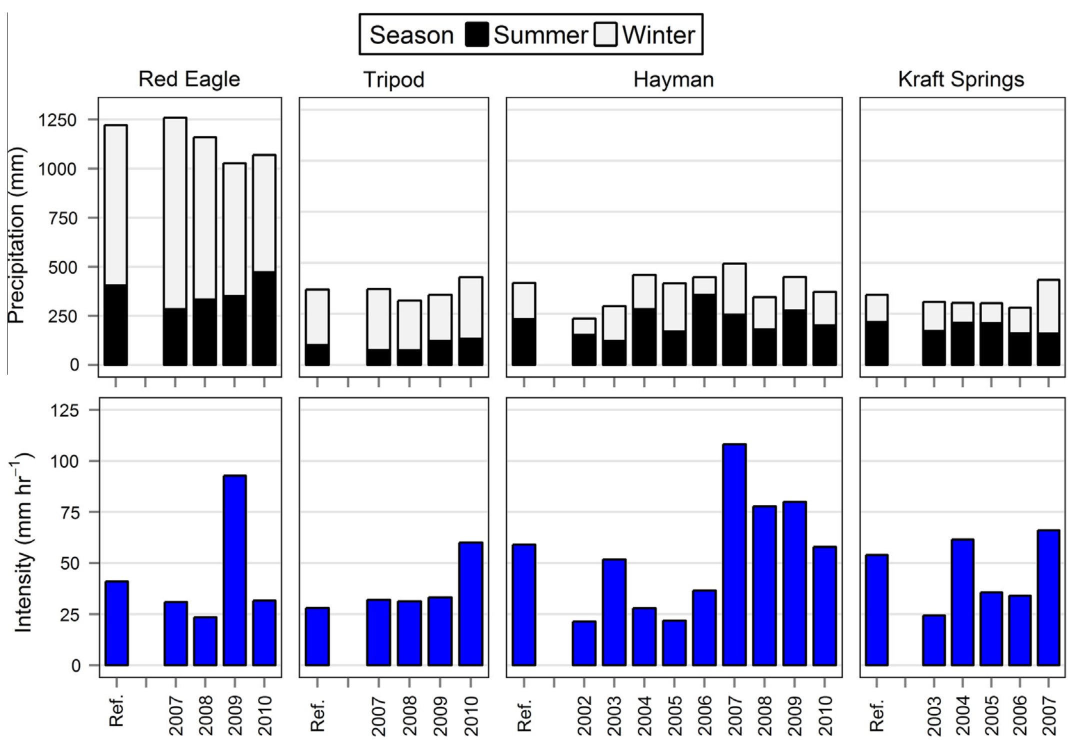

Fig. 3. Summer and winter precipitation from the nearest long-term gage and the maximum 10-min rainfall intensity averaged across the 3–8 rain gages in each study area for each year. The reference values ("Ref.") are the long-term averages from the nearest long-term precipitation gage and the 2-year I_{10} from rainfall-frequency atlases (Miller et al., 1973a,b; Perica et al., 2013), respectively. The 2007 Kraft Springs precipitation data are from the upslope gage.

The long-term precipitation data came from the Many Glacier precipitation gage that was 23 km away at an elevation of 1490 m.

2.1.2. Tripod

The Tripod Complex Fire burned 64,000 ha of a ponderosa pine and Douglas-fir forest on the Okanagan-Wenatchee National Forest in northern Washington (Fig. 2) in 2006. The following spring we established six swales (0.4–2.6 ha) in areas burned at high severity. The lower portions of the six swales were logged in spring 2009, and in summer 2009 we established three more logged swales (0.1–2.5 ha), but no control swales were available (Table 1). At the same time we installed six replicates of each of the four plot types (Table 1). The long-term precipitation data came from Winthrop, which is 12 km from the study area at an elevation of 1490 m.

2.1.3. Hayman

In 2002 the Hayman Fire burned 56,000 ha of the Pike-San Isabel National Forest in central Colorado (Fig. 2). The predominant vegetation was ponderosa pine with some Douglas-fir on the more sheltered or northerly aspects. In July 2002 we established two swales (3.0 and 2.9 ha) in an area of high burn severity and 14 months later the lower two-thirds of the smaller swale was logged as the upper section was too steep for the ground-based equipment. Following local best management practices the skid trails were seeded and lightly mulched with straw (~0.5 Mg ha^{-1}) in October 2003. Since no straw remained in May 2004 we assumed that the mulching had no effect on our results. Long-term precipitation data came from Cheesman, which was 7.5 km away at an elevation of 2100 m.

Table 2
Soil series and taxonomic class, texture, parent material, bulk density from 0 to 5 cm, percent sand, silt and clay for the fine fraction, and percent larger than 2 mm for each study area.

Study area	Soil series[a]	Texture	Parent material	Bulk density ($g\ cm^{-3}$)	Sand, silt, clay (%)	>2 mm (%)
Red Eagle	Tenex[b]	Sandy loam	Argillite	1.01	64, 33, 3	7–10
Tripod	Wapal[c] Brevco[d]	Very stony ashy coarse sandy loam	Mixed volcanic ash over glacial outwash	1.18	92, 7, 1	1–19
Hayman	Legault[e] Sphinx[f]	Gravelly, coarse sandy loam	Granite	1.36	72, 25, 3	35–40
Kraft Springs	Dast[g] Vebar[h]	Sandy loam	Alluvium or residuum over semi-consolidated sedimentary rock	1.28	67, 23, 10[a]	0–18[a]

[a] The soil series and taxonomic classes were based on USDA-NRCS soil series maps and soil taxonomy keys.
[b] *Loamy-skeletal, mixed, superactive Spodic Dystrocryepts.*
[c] *Sandy-skeletal, isotic, frigid Vitrandic Haploxerepts.*
[d] *Loamy-skeletal, isotic, frigid Vitrandic Haploxerepts.*
[e] *Sandy-skeletal, micaceous, shallow, typic cryorthents.*
[f] *Sandy-skeletal, mixed, frigid, shallow, typic ustorthents.*
[g] *Coarse-loamy, mixed, superactive, frigid, typic calciustepts.*
[h] *Coarse-loamy, mixed, superactive, frigid, typic haplustolls.*

2.1.4. Kraft Springs

The Kraft Springs Fire burned 30,000 ha in 2002 on the Custer National Forest in southeastern Montana (Fig. 2). Vegetation was predominantly ponderosa pine with a grass understory. Similar to Hayman, two swales (2.8 and 3.6 ha) were established in June 2003 and the larger swale was logged in July 2003. The long-term precipitation data were from Camp Crook, which was 21 km away at an elevation of 950 m.

2.2. Precipitation

Rainfall was measured by three to eight tipping-bucket rain gages in each study area from 1 June to 31 October ("summer"); winter was the preceding seven months. Rain events were separated by six hours with no rainfall and for each event we determined the total rainfall and the maximum 10-min, 30-min, and 60-min rainfall intensities (I_{10}, I_{30}, and I_{60}, respectively). If multiple events occurred between site visits the sediment measured was attributed to the event with the highest rainfall intensity. The 2-year I_{10} for each study area was calculated from rainfall frequency atlases (Miller et al., 1973a,b; Perica et al., 2013).

2.3. Ground cover

Ground cover was measured annually in each plot or swale in 3–25 fixed locations using 1-m^2 quadrats. The quadrat locations in the unlogged Tripod swales were in the lower third of the swales, and these areas were subsequently disturbed by logging. At Hayman and Kraft Springs the quadrats were equally spaced along randomly located transects (25 and 50 m long, respectively). At Hayman only one of the four transects in the logged swale was directly affected by logging as the others were either in the riparian buffer or the steep upper portion of the swale. In each quadrat ground cover was classified along a 10 cm by 10 cm grid (Chambers and Brown, 1983). The ground cover classes were bare soil, litter (needles, leaves, or wood fragments <1 cm in diameter), live vegetation, wood (including dead standing trees), and rock (>25 mm). Ground cover measurements generally were made in late summer or early autumn. Ground disturbance was determined along three transects in each logged swale at Red Eagle, and by measuring the area of the skidder and feller-buncher trails at Hayman and Kraft Springs.

Linear mixed-effects models were used to compare differences in cover classes among treatments and years for each study area (SAS Institute Inc., 2008). Treatment and year were fixed effects and plot was a random effect. A second random effect was included in each model to accommodate the repeated measures within plots across years, and the covariance for this factor was modeled using an autoregressive function (Littell et al., 2006).

2.4. Soil water repellency

Water drop penetration time (WDPT) (DeBano, 1981) was measured at the soil surface in 2003 at Hayman and at depths of 1 cm and 3 cm for all other periods. The time for each of eight drops of water to fully infiltrate was recorded up to a maximum time of 300 s at four locations in each plot and swale at Red Eagle and Tripod. At Hayman and Kraft Springs WDPT was measured at 8–10 locations in each swale, but the measurements in the logged swales were not stratified by soil disturbance so soil water repellency could only be compared at the swale scale. WDPT measurements were not made when the surface soil was visibly damp.

In most cases we recorded the actual WDPT and these times were used in the analyses. In some cases WDPT values less than 5 s or more than 180 s were recorded as a class, and for analyses we assigned the following times for the different classes: 2.5 s for <5 s; 200.5 s for >180 s; 220.5 s for >200 s; and 320.5 s for >300 s. The median WDPT of the eight drops at each depth was used in subsequent characterization because of the high variability in WDPT at a given location. At each location the highest median WDPT across all depths was determined and we averaged these maxima for each ground condition (e.g., tracked area or nearby undisturbed area) for each plot and year.

Comparisons of the mean-maximum-median WDPT (hereafter WDPT) were made between conditions, treatments, and years using linear mixed-effects models (SAS Institute Inc., 2008) with model structures similar to the ground cover models at Red Eagle and Tripod. At Hayman and Kraft Springs WDPT location was nested within each swale and included in the model as a separate random variable. Values of 0 s were changed to 1 s so a lognormal distribution could be used in the generalized linear mixed-effects models.

2.5. Bulk density and soil strength

Soil compaction was assessed by measuring dry bulk density in each study area within eight months after logging occurred. Bulk density was measured at 0–5 cm and 5–10 cm at one location in each plot at Red Eagle and Tripod, and at one location in the Red Eagle control swales. For each of the Tripod logged swales bulk density was measured at one location in a skidder track and at one location in a nearby undisturbed area. At Hayman bulk density was measured at only the 0–5 cm depth for four locations in the control swale and at both depths for 11 locations in the skidder tracks. At Kraft Springs bulk density was measured at both depths for one location in the control swale, one location in a feller-buncher track, and eight locations in skidder tracks. Bulk densities were compared among treatments or conditions at Red Eagle, Tripod, and Hayman using linear mixed-effects models with the treatment as a fixed effect and plot or sampling location as the random effect (Littell et al., 2006).

Soil strength was measured only at Red Eagle in 2007 and 2009 using a pocket penetrometer with either a 6.4 or 25.4 mm diameter loading piston. Five transects were laid out in each skidder, feller-buncher, and slash-treated skidder plot, with two measurements in the tracks of the skid trail, one measurement between the tracks or in the middle if the whole plot was tracked, and one measurement in a relatively undisturbed location adjacent to the plot. A linear mixed-effects model compared soil strength among the fixed effects of treatments, tracked or undisturbed areas, and year with plot as a random factor (SAS Institute Inc., 2008). A second random factor accounted for the repeated measures within plots across years, and the covariance for this factor was modeled using an autoregressive function (Littell et al., 2006).

2.6. Sediment production at Red Eagle and Tripod

Sediment production was measured in all of the plots and swales using sediment fences (Robichaud and Brown, 2002) in the spring and again in early fall. To the extent possible, sediment production also was measured after each major rain event (Table 1). The sediment was weighed in the field and adjusted for moisture content (Robichaud and Brown, 2002). The dry mass was divided by the contributing area to obtain sediment production in Mg ha^{-1}.

The more extensive dataset at Red Eagle allowed us to test for correlations between log-transformed sediment yields and possible explanatory variables. Linear mixed-effects models were developed for the annual sediment yields at each study area with treatment and post-fire year as fixed effects, and plot or swale as a random effect. The repeated measures on plots or swales were modeled as random effects with an autoregressive covariance structure (Littell et al., 2006). Half of the minimum value for a given data set was added to each datum to allow for log-transformation and meet the normality assumption.

2.7. Runoff and sediment production at Hayman and Kraft Springs

The swales at Hayman and Kraft Springs were adjacent and closely matched in size, slope, aspect, soils, and pre-fire vegetation (Fig. 2), and the similarities allowed the use of a paired catchment approach to analyze the differences in unit-area runoff, peak flows, and sediment yields (Robichaud, 2005). Each swale had a ninety-degree v-notch weir with a weir pond. Weir stage was measured with a linear magneto-strictive device (MTS Systems, Eden Prairie, Minnesota), and the depth of water and sediment in the pond was measured with an ultrasonic sensor (Judd Communications, Logan, Utah). Flow through the weir was calculated using a standard rating equation (Grant, 1989), and the depth-volume relationship for each weir pond was used to determine flow rates as the pond filled.

The removal of large sediment accumulations at Hayman sometimes required the use of a small loader, and in these cases we determined the mass of sediment from a volumetric survey and the measured sediment bulk density, or by a tally system when removing the sediment. For the tally method we weighed the sediment in one loader bucket, counted the number of loader buckets, and took samples from each bucket to determine the water content and calculate the dry sediment mass. When multiple runoff events occurred between sediment measurements, the sediment for each storm was prorated according to the respective storm runoff volumes.

At Hayman sediment production was measured for seven events in 2002–2003 prior to logging (“calibration period”), but the stage recording instruments were not installed for the first two events, resulting in five calibration events for runoff and peak flows. Another thirteen sediment-producing events were measured in 2004–2010, so we were able to compare responses in the two swales before and after logging.

The runoff volumes, peak flows, and sediment yields from the control and logged swales at Hayman were compared using linear mixed-effects models with the responses in the logged swale being modeled as a function of the corresponding responses in the control swale. Each paired event was a repeated measure, and the model’s covariance structure was a spatial power relationship based on the event date (Littell et al., 2006). The peak flow rates and sediment yields were square-root transformed to improve the normality of the residuals for those models. The responses in the post-logging period were deemed significantly different from the calibration period if the respective 95% confidence intervals for the modeled intercepts or slopes did not overlap (Ott, 1993).

At Kraft Springs no runoff events were measured prior to salvage logging while four events occurred after logging. For the first event nearly all of the runoff from the logged swale was diverted away from the weir by a waterbar. Flows for subsequent events were directed into the weir pond. The small number of valid events ($n = 3$) precluded any statistical analysis.

2.8. Statistical methods

Means, plot differences, and treatment effects were evaluated at each study area using the Generalized Linear Mixed Model (GLIMMIX) procedure (SAS Institute Inc., 2008). Least-squares means with a Tukey–Kramer adjustment were used to compare differences in mean values of the fixed effects in the mixed-effects models (SAS Institute Inc., 2008). Normality of linear model residuals was assessed using quantile–quantile plots, and homogeneity of variance of the residuals was assessed by plotting the residuals against the predicted values; all of the models described above reasonably met these assumptions. The transformed data were used to determine the significance of differences among treatments; we present untransformed data in the text and figures. The significance level was 0.05 for all comparisons and confidence intervals.

3. Results

3.1. Precipitation

The mean proportion of summer precipitation ranged from 26% at Tripod to 61% at Kraft Springs, and nearly all of the sediment production for each study area occurred as a result of warm season frontal or convective rainstorms rather than snowmelt. Mean summer precipitation was about 360 mm near the relatively wet Red Eagle study area, only 97 mm near Tripod, and about 180–210 mm near Hayman and Kraft Springs (Fig. 3). The variability of summer rainfall was greatest at Hayman (Fig. 3), where the values ranged from 117 to 343 mm.

The mean annual maximum I_{10} values among study areas ranged from 39–54 mm h^{-1}, and these values were generally consistent with the estimated 2-year I_{10} values (Fig. 3). As might be expected there was considerable variability in the maximum I_{10} among years (Fig. 3). The maximum I_{10} values were nearly double the mean values, and these ranged from 60 mm h^{-1} at the drier Tripod site to 108 mm h^{-1} at the monsoon-dominated Hayman site (Fig. 3).

3.2. Surface cover

3.2.1. Red Eagle and Tripod

In the first summer at Red Eagle (2007) the control plots averaged 60% bare soil, and the 40% surface cover consisted primarily of wood and litter with only 9% live vegetation (Fig. 4). The control swales initially had three times as much litter as the control plots and only 30% bare soil, but they still averaged only 14% live vegetation (Fig. 4). From 2007 to 2010 live vegetation cover progressively increased in the control plots and swales to about 60%. By 2010 both the control plots and swales averaged over 90% ground cover (Fig. 4).

In the summer after logging at Red Eagle (2007) the skidder plots averaged 52% bare soil, which was slightly less than the 60% for the control plots (Fig. 4). The skidder plots had more litter and rock cover and less vegetation than the controls, but only the difference in litter cover was significant. From 2007 to 2010 the skidder plots had very slow vegetative regrowth resulting in significantly less vegetation cover and more bare soil than the controls in 2009 and 2010.

At Tripod the unlogged swales averaged 39% bare soil in 2007, or slightly less cover than the swales at Red Eagle, and the cover consisted mostly of litter with only 14% live vegetation (Fig. 4). In contrast to Red Eagle, there was very little change in total surface cover in the unlogged swales from 2007 to 2008, and the amount of live vegetation only increased from 14% to 19% (Fig. 4). Logging had very little effect on total cover, and the main change was an increase in wood cover from 3% in 2008 to 14% in 2009. From 2009 to 2010 there was no increase in total cover and no substantial change in the makeup of the cover for the logged swales (Fig. 4).

All the plots at Tripod were installed in summer 2009, and the ground cover in the six control plots was similar to the swales prior to logging (Fig. 4). Like the logged swales, the control plots showed very little increase in total cover from 2009 to 2010, and there was still 29% bare soil in the control plots in 2010. This was a much lower rate of regrowth than at Red Eagle (Fig. 4).

In 2009 the skidder plots at Tripod averaged 45% bare soil, or 13% more than the control plots (Fig. 4), but this difference was not significant. The live vegetation cover was only 6% and this was significantly less than the 22% for the controls (Fig. 4). From 2009 to 2010 the amount of live vegetation increased slightly to 13%, but this was not significantly different than the mean of 17% for the control plots and did not significantly reduce the amount of bare soil (Fig. 4). The low vegetative cover, slow regrowth, and high bare soil on the Tripod skidder plots were consistent with the trends observed at Red Eagle.

At Red Eagle and Tripod the amount and type of ground cover on the feller-buncher plots followed similar trends over time as the skidder plots, except at Red Eagle the feller-buncher plots initially had only 41% bare soil as compared to the 52% in the skidder plots (Fig. 4). By 2010, the fourth summer after burning, the amount of bare soil in the feller-buncher plots had decreased in both study areas because of the increases in live vegetation (Fig. 4), but there were no significant differences between the feller-buncher and skidder plots in either study area. At Red Eagle there was significantly more bare soil and less vegetative cover in the feller-buncher plots than in the controls (Fig. 4). Overall, both

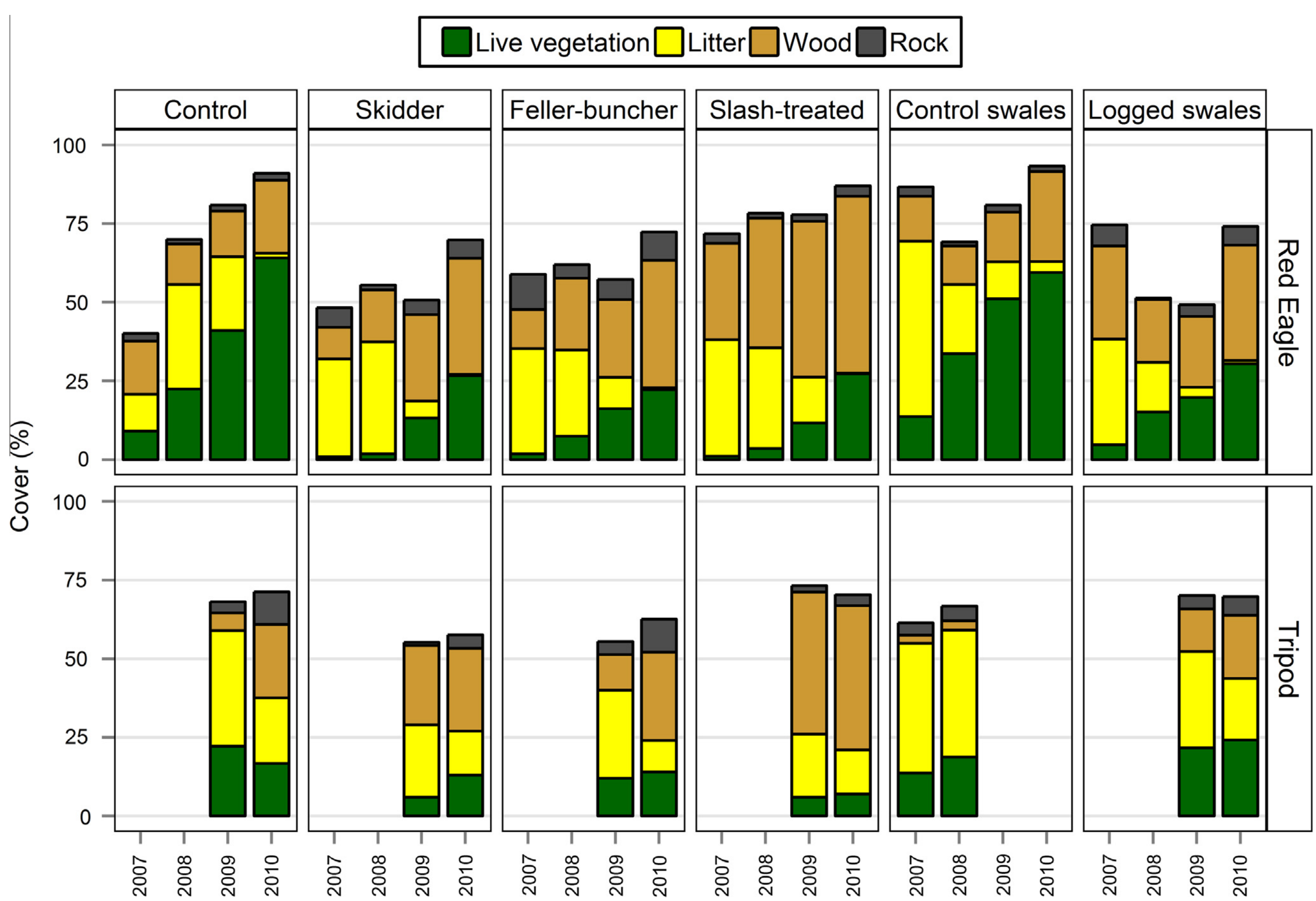

Fig. 4. Ground cover at Red Eagle and Tripod by plots or swales, treatment, and year. 100 minus the total ground cover yields percent bare soil. Logging occurred prior to the 2006 measurement at Red Eagle and after the 2008 measurement at Tripod.

the skidder and feller-buncher plots tended to have more wood than the controls (Fig. 4), but these differences were not significant. These results suggest that a relatively few passes of logging equipment can slow post-fire vegetative recovery.

At Red Eagle the manual application of logging slash to eight of the skidder plots increased the amount of wood cover from 10% to 31% (Fig. 4). This reduced mean bare soil to just 28% in 2007, and this was significantly less bare soil than both the control plots and the untreated skidder plots. From 2008 to 2010 vegetative regrowth on the slash-treated plots was similar to the untreated skidder plots, resulting in significantly less live vegetation in the slash-treated plots than the control plots in 2008–2010 (Fig. 4). The changes in wood, vegetation and litter cover through time in the slash-treated plots was similar to the untreated skidder plots, so from 2007 to 2009 the treated plots had significantly more total cover than the untreated plots.

At Tripod the slash treatment also caused a 20% increase in wood cover, and this again resulted in significantly more total cover than the untreated skidder plots (Fig. 4). The slash-treated plots at Tripod also had significantly less live vegetation than the controls and almost no change in the amount of live vegetation or total cover from 2009 to 2010.

The three logged swales at Red Eagle had 14%, 37%, and 50% of the area disturbed by logging. Total cover after logging averaged 75% in 2007, which was not significantly less than the 87% for the control swales (Fig. 4). However, the composition of the surface cover was quite different, as the logged swales had significantly more wood and less litter cover (Fig. 4). Vegetative regrowth in the logged swales was much slower than in the controls and only slightly faster than in the feller-buncher and skidder plots (Fig. 4). By 2009 the logged swales had less than half as much live vegetation as the control plots and swales (Fig. 4), and these differences were significant in 2009 and 2010. Despite the greater amount of wood cover in the logged swales, the logged swales consistently had less total cover than the controls, and the difference was significant in 2009 (Fig. 4). These results indicate that the spatially-averaged effects of salvage logging at Red Eagle reduced both vegetative regrowth and total cover, and increased wood cover. While the logging at Tripod also increased wood cover, at the swale scale logging had no clear effect on vegetative regrowth or total cover (Fig. 4).

3.2.2. Hayman and Kraft Springs

Three months after the Hayman Fire the control swale had 24% surface cover, and this consisted almost entirely of litter and rock (Fig. 5). Over the next eight years total cover gradually increased to 74%. Live vegetation increased each year until it reached 35% in 2006, and then stabilized at about 40% from 2008 to 2010 (Fig. 5). In contrast to Red Eagle and Tripod, there was almost no wood cover until 2007 when extensive treefall began, and the amount of wood cover increased to 20% by 2009.

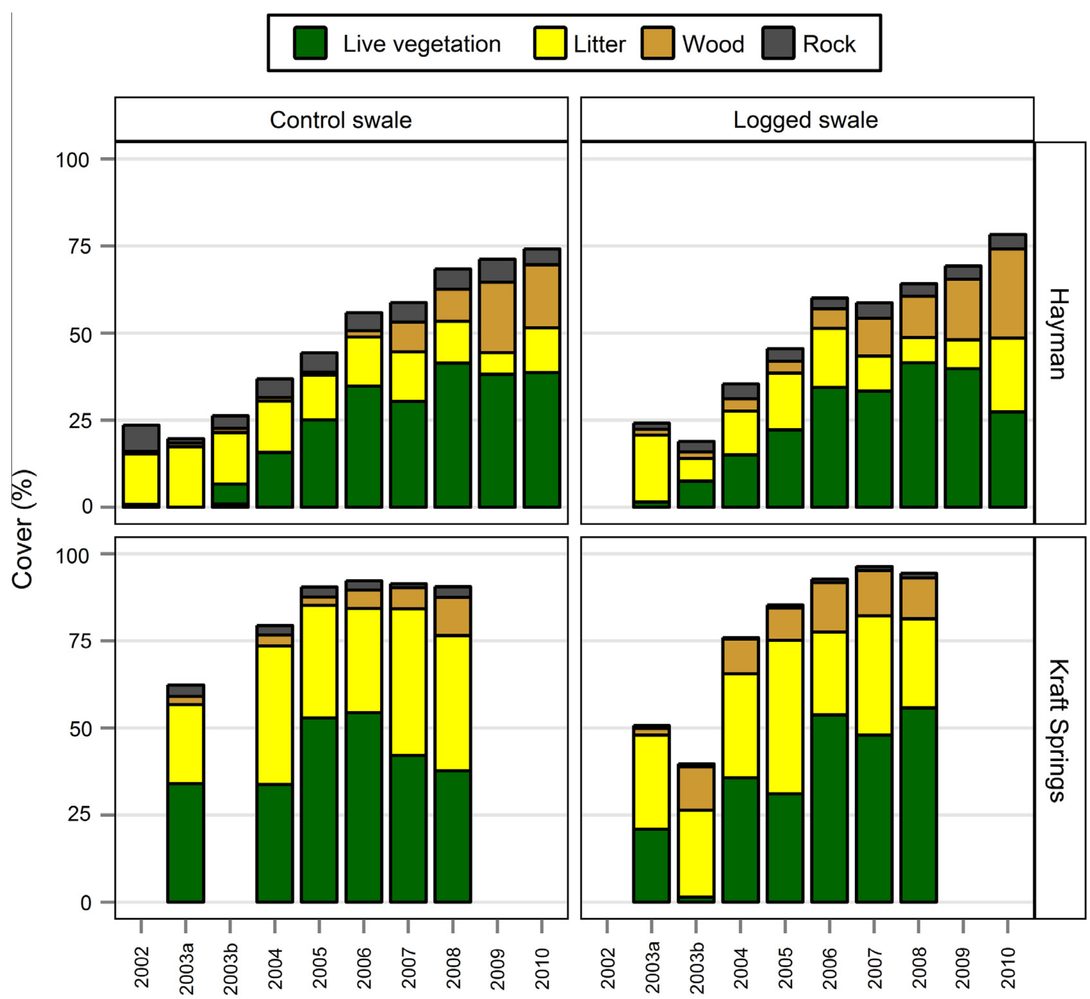

Fig. 5. Ground cover at the Hayman and Kraft Springs study areas for the control and logged swales by year. 2003a was in June 2003 prior to logging and 2003b was in October 2003 after logging. 100 minus the total ground cover yields percent bare soil.

Total cover in the logged swale at Hayman prior to logging was slightly greater than in the control, and this was from slightly more litter and live vegetation (Fig. 5). Logging late in 2003 disturbed 55% of the area of the swale. The disturbance reduced the total cover from 24% to 19%, primarily due to a decrease in litter cover. Live vegetation increased from 2% in the spring before logging to 8% in the fall after logging, and the same increase was observed in the control swale. From 2004 through 2010 the type and amount of cover in the logged swale was very similar to the control swale (Fig. 5). The Hayman site is unusual in that the vegetation cover more or less stabilized at around 40% by 2008 in both the logged and control swales, and both swales still had more than 20% bare soil in 2010.

One year after the fire the control swale at Kraft Springs already had 62% surface cover, and this consisted of 34% live vegetation, 23% litter, and only 2% wood (Fig. 5). Total cover increased to 79% in fall 2004, and from 2005 to 2008 the control swale always had at least 90% surface cover. The increases were mostly due to increases in live vegetation and litter, but wood cover also increased from 1% in 2005 to 11% in 2008 due in part to treefall (Fig. 5).

The logged swale at Kraft Springs had 51% cover in June 2003 prior to logging, or slightly less than the control swale. Logging impacted 48% of the area of the swale, and this reduced the total ground cover to just 40% (Fig. 5). The decrease in cover was almost entirely due to the decline in the amount of live vegetation from 21% to just 1%, which was a very different response than at Hayman (Fig. 5). Total ground cover on the logged swale rapidly recovered to 76% in 2004 and 93% by 2006, and these increases were primarily due to rapid growth of grasses and forbs. From 2004 through 2008 the control and logged swales had relatively similar amounts and types of cover except for the greater amount of wood in the logged swale (Fig. 5).

3.3. Soil water repellency

Mid and late summer rains in the first summer at Red Eagle meant that we measured soil water repellency only in the skidder and feller-buncher plots. There was no significant difference in soil water repellency between the points in the tracks and the points in the middle of the tracks, so all the track points were grouped and compared to the adjacent undisturbed areas for each year and to the control plots for 2008–2010. In 2007 the undisturbed points near the skidder, feller-buncher, and slash-treated plots all had WDPT values greater than 110 s, indicating very strong soil water repellency (Fig. 6). WDPT values were less than 35 s in the tracks, but only the feller-buncher tracks had significantly less soil water repellency than the nearby undisturbed points.

Soil water repellency remained relatively strong through 2009 in the control and undisturbed points and generally quite weak in the skidder, feller-buncher, and slash-treated skidder tracks, but there were no significant differences between the tracked areas and any of the corresponding undisturbed points (Fig. 6). However, in the logged swales the tracks had less soil water repellency than the undisturbed points, and this difference was significant in 2008 (Fig. 6).

From 2009 to 2010 the WDPT in the control plots dropped significantly from 167 s to 22 s, and the WDPT values in the control swales showed a very similar decline. Soil water repellency was also weaker in the skidder and feller-buncher plots and their corresponding undisturbed points in 2010, with relatively little evidence of water repellency in the tracked areas (Fig. 6). The soil was relatively damp during the WDPT measurements in 2010, and this could have contributed to the lower values along with the usual decline in soil water repellency after burning (Doerr et al., 2009).

At Tripod no WDPT data were collected in 2007 or 2009 because of damp soils. In 2008 the WDPT in the unlogged swales was 25 s, indicating weaker water repellency than the comparable period at Red Eagle. In 2010 the WDPT for the undisturbed points near the skidder and feller-buncher tracks ranged from 6 to 59 s, while the WDPT in the tracked areas were less than 5 s. None of the differences in WDPT among plot types or track locations were significant.

At Hayman there was a relatively long-term record of soil water repellency, but it was not possible to relate the WDPT to logging impacts because of the few measurements in areas disturbed by logging. In summer 2003, one year after burning, the mean WDPT at the soil surface for both swales was 59 s. By summer 2004 the mean decreased to only 6 s, and low values were recorded through 2010.

At Kraft Springs soil water repellency was first measured in 2005 or three years after burning, and at that time the WDPT was only 8 s in the control swale and 11 s in the logged swale. There was no change in soil water repellency through 2008.

Taken together, these results indicate that post-fire soil water repellency was surprisingly persistent at Red Eagle, but had largely disappeared within 2–3 years at each of the other study areas. Water repellency was generally much less in the tracked areas than the undisturbed areas, but the high spatial variability meant that most differences were not significant.

3.4. Bulk density and soil strength

Mean bulk densities at 0–5 cm in the control sites ranged from 1.05 g cm^{-3} at Red Eagle to 1.37 g cm^{-3} at Hayman (Fig. 7). The highest bulk densities were in the skidder tracks with mean values at 0–5 cm of 1.29–1.52 g cm^{-3}, and the difference between the controls and the skidder tracks was significant at Red Eagle. The feller-buncher tracks generally had higher bulk densities than the controls and lower values than the skidder tracks (Fig. 7), but these smaller differences were not significant.

Mean bulk densities at 5–10 cm tended to be higher than at the 0–5 cm depth for all locations (Fig. 7). Once again the skidder tracks had the highest bulk densities, and at Red Eagle the mean bulk density of 1.40 g cm^{-3} was significantly greater than the mean of 1.13 g cm^{-3} for the controls. Again the mean bulk densities at 5–10 cm for the feller-buncher tracks generally were between the values for the controls and the skidder tracks, and the mean of 1.32 g cm^{-3} at Red Eagle was significantly greater than the mean for the controls (Fig. 7).

These results indicate that the compaction due to the logging machinery extended to a depth of at least 10 cm, and relatively few passes of the logging equipment resulted in substantial compaction. The significant differences at Red Eagle may be due to the finer texture of the soils in that area or the timing of the logging, which occurred shortly after snowmelt when the soils were relatively wet. Logging at Tripod also was done in the spring, but these sites had the highest sand content (Table 2). Logging at Hayman and Kraft Springs occurred in the summer or fall when the soils were relatively dry.

After logging the mean soil strength in the skidder tracks at Red Eagle was 2.6 kg cm^{-2}, and this was significantly greater than the mean value from between the skidder tracks (1.8 kg cm^{-2}) and the mean in the undisturbed locations near the skidder tracks (1.6 kg cm^{-2}). In contrast, there was no difference in the mean soil strength among the feller-buncher tracks (2.4 kg cm^{-2}), the points between the feller-buncher tracks (2.0 kg cm^{-2}), or the nearby undisturbed locations (2.5 kg cm^{-2}). It is not clear why there was a large discrepancy between the undisturbed areas for the feller-buncher tracks and the skidder tracks, but field observations indicated that the steeper areas adjacent to the feller-buncher tracks

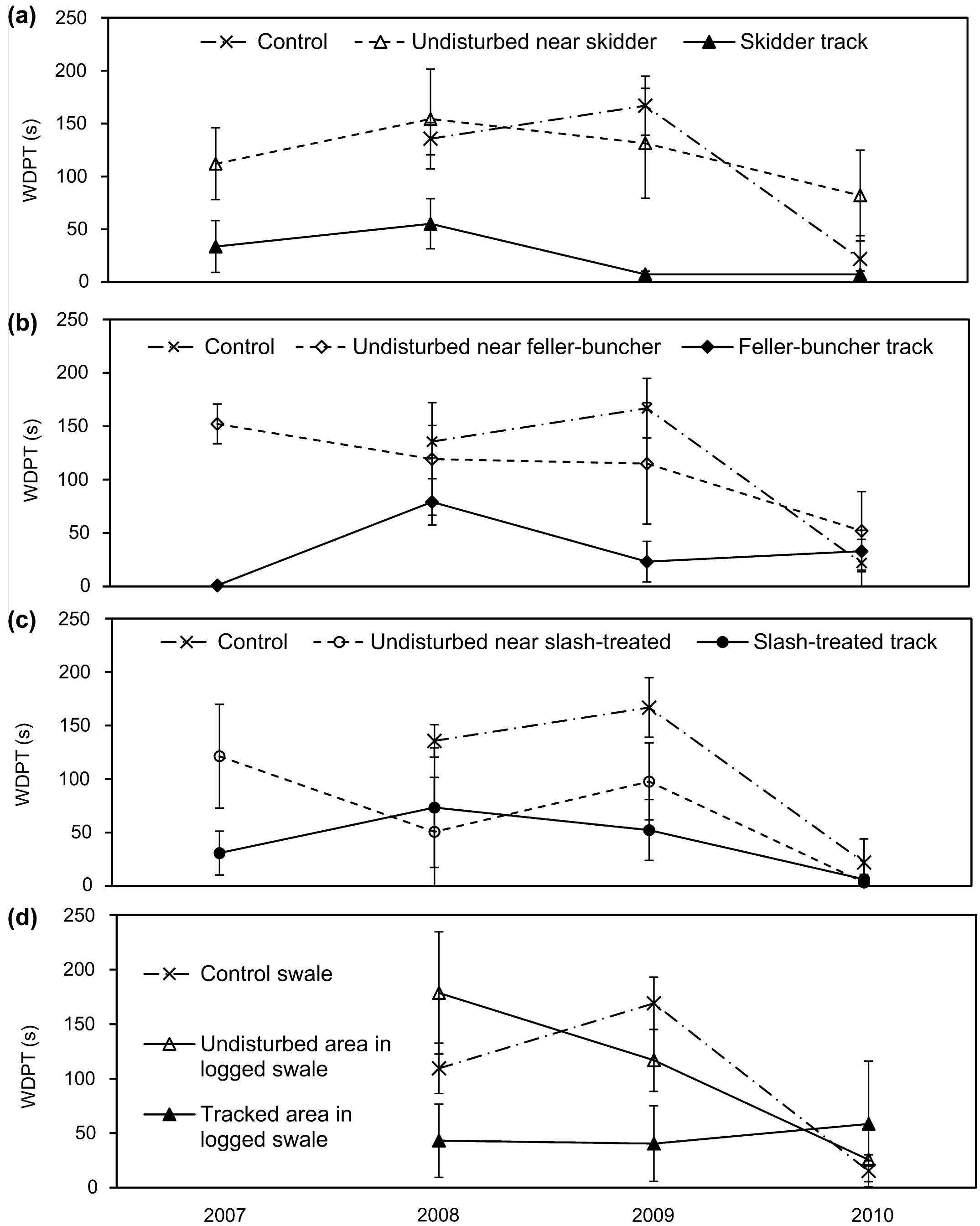

Fig. 6. Mean maximum median WDPT by year at Red Eagle for: (a) skidder tracks; (b) feller-buncher tracks; (c) slash-treated skidder tracks; and (d) tracked areas in the logged swales. Panels (a–c) compare tracked areas to the control plots and undisturbed areas near each treated plot. Panel (d) compares tracked areas in the logged swales to undisturbed areas in the logged swales and to the control swales. Error bars indicate one standard error.

had more bare soil and may have been more subject to soil crusting.

The 2009 soil strength values in the undisturbed locations near the skidder and feller-buncher tracks (1.1 and 1.0 kg cm^{-2}, respectively) were lower than the 2007 values, suggesting a reduction in soil crusting. The skidder and feller-buncher tracks had substantially higher mean values (2.6 and 1.7 kg cm^{-2}, respectively) than the locations between the tracks and the undisturbed locations, and this suggests a continuing compaction effect. The soil strength was significantly greater in the skidder tracks than in both the undisturbed locations and the feller-buncher tracks. All groups except for the skidder tracks showed a decrease in soil strength from 2007 to 2009, and this suggests a slower recovery for the more heavily trafficked areas.

3.5. Sediment production

3.5.1. Red Eagle

At Red Eagle no sediment was generated from any of the plots or swales in 2007 due to low rainfall intensities (Fig. 3). In 2008 sediment production generally increased with increasing disturbance (Fig. 8) but sediment production rates were low in absolute terms as the maximum I_{10} was only 24 mm h^{-1} (Fig. 3). The control plots had a mean annual sediment production of only 0.11 Mg ha^{-1} yr^{-1} while the mean sediment production for the skidder plots was more than five times greater at 0.63 Mg ha^{-1} yr^{-1}. The sediment production from the feller-buncher plots was intermediate at 0.19 Mg ha^{-1} yr^{-1}, while the mean from the slash-treated plots was less than the control plots at only

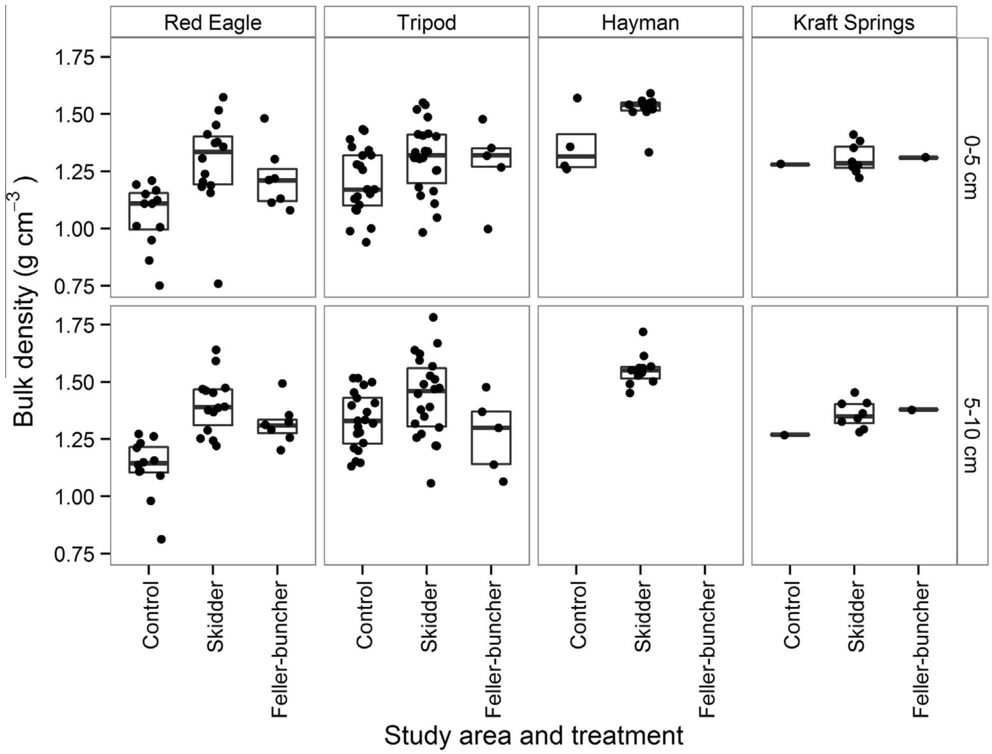

Fig. 7. Box plots of the soil bulk density at 0–5 cm and 5–10 cm by study area and treatment. For Red Eagle and Tripod, the controls include both plots and swales, and the skidder data include both untreated and slash-treated skid trails. Tripod controls are from undisturbed locations near skidder tracks in the logged swales. No data were collected from 5 to 10 cm for the control swale at Hayman and only one sample was collected from the control and feller-buncher tracks at Kraft Springs. The boxes are the 25th and 75th percentiles, the lines are medians, and the points are individual measurements.

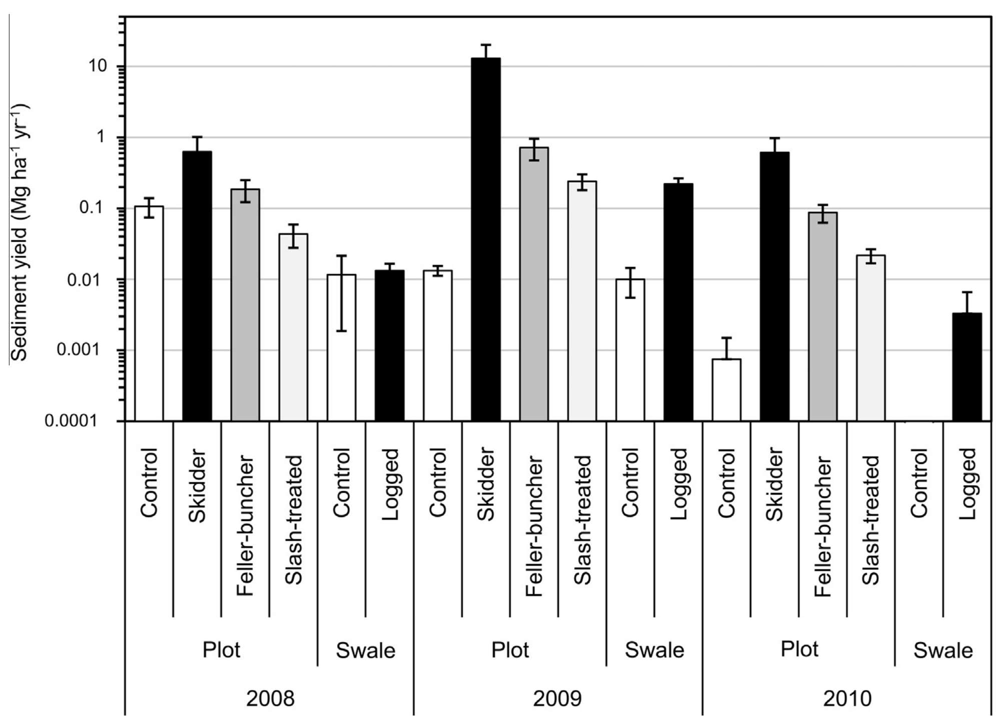

Fig. 8. Mean annual sediment yields from the Red Eagle study area by treatment, plot or swale, and year. There was no sediment yield from the control swales in 2010. Error bars represent one standard error.

0.04 Mg ha^{-1} yr^{-1}. None of the differences among plot types were significant. At the swale scale sediment production rates were much lower, as both the control and logged swales averaged just 0.01 Mg ha^{-1} yr^{-1} (Fig. 8).

The maximum I_{10} in 2009 was nearly four times higher than in 2008 (Fig. 3). In general, sediment production rates decreased in the controls but were much higher in the disturbed plots (Fig. 8). The mean sediment production rate from the untreated skidder plots was 13 Mg ha^{-1} yr^{-1}, two orders of magnitude greater than the 2008 value, while the mean for the control plots dropped to just 0.01 Mg ha^{-1} yr^{-1}, and both of these year-to-year changes and the skidder-control comparison were significant. Sediment production from the feller-buncher plots was again intermediate at 0.72 Mg ha^{-1} yr^{-1}, and this was significantly different than the means for the controls and the skidder plots. Mean sediment yield from the slash-treated plots was only 0.24 Mg ha^{-1} yr^{-1} or 2% of the value from the untreated skidder plots, but 20 times greater than the control plots; again, both of these differences were significant.

The 2009 sediment production from the logged swales averaged 0.22 Mg ha^{-1} yr^{-1}, or nearly 20 times more than in 2008 (Fig. 8), and we attribute this to the increased rainfall intensity (Fig. 3). Despite the greater intensity the control swales again averaged only 0.01 Mg ha^{-1} yr^{-1} so the 20-fold difference in sediment production between the logged and control swales was significant.

In 2010 the maximum I_{10} was only 32 mm h^{-1} (Fig. 3) and the amount of ground cover increased in all the plots and swales (Fig. 4), so the mean sediment production rates dropped by nearly an order of magnitude (Fig. 8). Again the sediment yields were highest from the skidder plots, intermediate for the feller-buncher plots, and lowest from the slash-treated plots, but each of these was significantly greater than the controls (Fig. 8). No sediment was generated from the control swales, while the mean sediment production from the logged swales was only 0.003 Mg ha^{-1} (Fig. 8).

The correlation analysis for the plots at Red Eagle showed that sediment production was positively correlated with rainfall intensity, event rainfall, and amount of bare soil (r = 0.40–0.47), and negatively correlated to the amount of litter plus vegetation (r = −0.37) (Table 3). The next strongest correlations were with cumulative rainfall between sediment clean-outs (r = 0.23) and slope (r = 0.18) (Table 3). Somewhat surprisingly there was a weaker but still significant negative correlation between sediment production and WDPT (r = −0.14) (Table 3), which can be explained by the lower WDPT and higher sediment production in the plots with more disturbance.

3.5.2. Tripod

Sediment production data from Tripod confirmed the general trends measured at Red Eagle (Fig. 9). In 2009 the maximum I_{10} was 33 mm h^{-1} and the mean annual sediment production rate from the skidder plots was 1.0 Mg ha^{-1} yr^{-1}, or about 16 times the mean value of 0.065 Mg ha^{-1} yr^{-1} from the control plots, and this difference was significant. The feller-buncher plots produced only 0.088 Mg ha^{-1} yr^{-1}, which was significantly less than the skidder plots. The slash-treated plots produced only 0.21 Mg ha^{-1} yr^{-1} or about 20% as much as the untreated skidder plots, and this difference also was significant.

In 2010 sediment production was significantly higher as a result of a rainfall event with an I_{10} of 60 mm h^{-1} (Fig. 3), but the relative values generally followed the same pattern as in 2009 (Fig. 9). The skidder plots produced 5.9 Mg ha^{-1} or about 14 times the mean value from the controls, and this difference was significant. The relatively large mean sediment production of 2.8 Mg ha^{-1} yr^{-1} from the feller-buncher plots was impacted by one plot's sediment production of 12 Mg ha^{-1} during this intense rainfall event. Nevertheless, the mean value from the feller-buncher plots was still not significantly different from the mean value from the control plots (Fig. 9). The slash-treated plots produced only 0.95 Mg ha^{-1} yr^{-1} or one-sixth as much sediment as the untreated skidder plots, and this difference was significant. Like Red Eagle, sediment production from the slash-treated plots was much greater than the unlogged control plots, but at Tripod this difference was not significant.

At the swale scale the mean sediment yields before salvage logging were only 0.17 Mg ha^{-1} yr^{-1} in 2007 when the maximum I_{10} was 32 mm h^{-1}, and no sediment was produced in 2008 (Fig. 9) despite the very similar maximum I_{10} (Fig. 3). In 2009, which was the first year after logging but the third year after burning, mean sediment production from the logged swales was only 0.001 Mg ha^{-1} yr^{-1} from nearly the same maximum I_{10} of 33 mm h^{-1}. As with the plots, in 2010 the mean sediment production from the logged swales increased to 0.15 Mg ha^{-1}, with nearly all of this sediment being produced from the 60 mm h^{-1} rain event. The 2010 sediment production rate for the logged swales was nearly identical to the value from the unlogged swales in the first year after burning from a series of lower intensity rainfall events. These plot- and swale-scale data showed increases in sediment production with increasing disturbance and rainfall intensity, and decreases with increasing surface cover.

3.5.3. Swale-scale runoff and sediment yields at Hayman and Kraft Springs

In the 2002–2003 calibration period 15.3 mm of runoff was generated in the control swale at Hayman, while the soon-to-be logged swale produced 18.5 mm or 20% more runoff (Table 4). Between 2004 and 2010 the control swale produced 8.0 mm of runoff from 13 events totaling nearly 248 mm of rainfall, while the logged swale produced 7.5 mm or slightly less runoff (Table 4). There was no statistically significant effect of logging on total runoff.

Table 3
Pearson correlation coefficients between the log-transformed sediment yields for the plots at Red Eagle and various explanatory variables in order of increasing p-values.

Variable	n	Correlation	95% Confidence limits		p-Value
I_{60} (mm h^{-1})	215	0.47	0.36	0.57	<0.0001
I_{30}(mm h^{-1})	215	0.47	0.35	0.56	<0.0001
I_{10} (mm h^{-1})	215	0.43	0.31	0.53	<0.0001
Event rainfall (mm)	215	0.43	0.32	0.53	<0.0001
Bare soil (%)	203	0.40	0.27	0.51	<0.0001
Litter + live vegetation (%)	203	−0.37	−0.48	−0.24	<0.0001
Cumulative rainfall (mm)	215	0.23	0.10	0.35	<0.001
Slope (%)	215	0.18	0.04	0.30	0.01
WDPT (s)	211	−0.14	−0.27	0.00	0.04
Bulk density 5–10 cm	167	−0.14	−0.29	0.01	0.07
Soil strength in track (kg cm^{-2})	84	0.13	−0.08	0.34	0.22
Bulk density 0–5 cm	167	0.09	−0.06	0.24	0.26
Soil strength in undisturbed area (kg cm^{-2})	84	0.00	−0.21	0.22	0.97

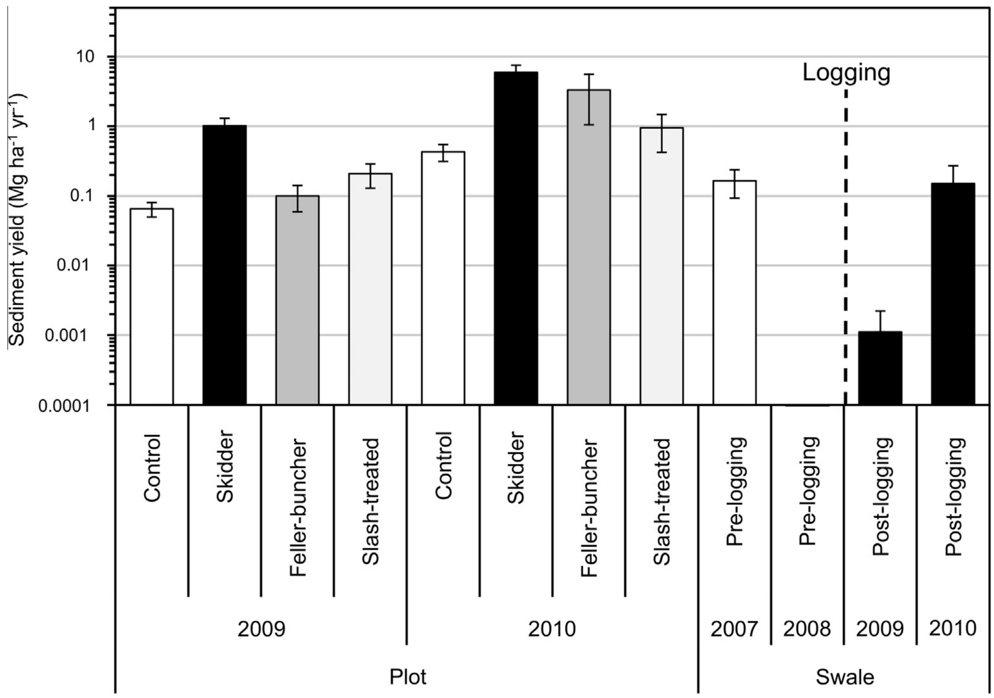

Fig. 9. Mean annual sediment yields from the Tripod study area by treatment, year, and plot or swale. No sediment was produced from the control swales in 2008, and all of the control swales were logged in winter 2008–09. Error bars represent one standard error.

Table 4
Date, rainfall, 10-min and 30-min maximum rainfall intensities, runoff, peak flows, and sediment yields for each sediment-producing event in the control (C) and salvage logged (L) swales at Hayman and Kraft Springs. ND indicates no data available.

Event date	Rain (mm)	I_{10} (mm h^{-1})	I_{30} (mm h^{-1})	Runoff (mm)		Peak flow (m^3 s^{-1} km^{-2})		Sediment yield (Mg ha^{-1})	
				C	L	C	L	C	L
Hayman: pre-salvage logging									
21 July 02	5.8	21	7.6	ND	ND	ND	ND	1.23	1.34
13 September 02	ND	ND	ND	ND	ND	ND	ND	0.94	0.59
1 October 02	18.0	9	6.6	2.1	1.1	2.4	2.09	0.74	0.60
18 July 03	7.4	15	7.1	0	0	0	0	0.01	0
9 August 03	18.0	52	27.9	8.6	12.1	5.0	5.4	19.80	16.70
30 August 03	18.0	14	9.7	1.5	1.8	1.1	0.9	1.49[a]	0.75[a]
30 August 03	10.9	23	15.7	3.1	3.5	2.7	2.6	3.08[a]	1.45[a]
Totals	78.1			15.3	18.5			27.29	21.45
Hayman: post-salvage logging									
18 June 04	4.3	17	6.3	0.02	0	0.002	0	0.07[a]	0
21 June 04	11.9	10	5.6	0.03	0	0.04	0	0.11[a]	0
27 June 04	19.1	28	16.6	0.8	0.5	2.6	1.4	2.88[a]	1.26
23 July 04	8.8	15	5.6	0.02	0	0.04	0	0.02	0.01
5 August 04	7.7	25	10.8	0.4	0.3	2.5	0.9	1.06[a]	0.39[a]
6 August 04	8.7	15	7.8	0.1	0	0.4	0	0.27[a]	0[a]
19 August 04	11.5	25	10.8	0.8	0.7	2.4	2.2	2.29	1.02
27 August 04	7.6	25	11.9	0.2	0.3	0.4	0.6	0.40	0.64
27 September 04	9.9	9	7.9	0.1	0	0.09	0	0.03	0.003
29 August 07	62.8	108	75.7	2.1	1.9	10.2	11.8	24.48	12.81
5 August 08	21.0	78	39.8	1.7	1.7	5.7	4.1	2.32	2.26
11 September 08	38.7	36	17.1	0.4	0.5	0.8	1.2	0.28	0.46
21 July 09	35.9	80	42.4	1.3	1.6	2.2	3.1	0.93	2.12
Totals	247.9			7.97	7.5			35.14	20.97
Kraft Springs: post-salvage logging									
19 August 03	10.9	24	17.3	ND	0.00[b]	ND	0.00[b]	1.76	0.00[b]
3 July 04	10.8	62	21.5	0.96	0.11	2.0	0.72	0.08	0.01
14 September 04	18.2	23	14.4	0.14	0	0.71	0	0.01	0.00
2 July 05	8.7	36	13.0	ND	ND	ND	ND	0.001	0.00
Totals	48.6			1.1	1.1			1.85	0.01

[a] Sediment produced by these consecutive events was measured during the same clean-out and prorated based on the events' runoff.
[b] Runoff and sediment were diverted out of the swale by a waterbar near the weir. Flow was directed toward the weir for subsequent events.

During the calibration period peak flows in both swales exceeded 5 $m^3 s^{-1} km^{-2}$ during the event with the highest I_{10} of 52 mm h^{-1}, and there was no consistent difference in peak flows between the two swales (Table 4). In the fifth year after burning a rain event with an I_{10} of 108 mm h^{-1} produced the maximum peak flows of 10.2 $m^3 s^{-1} km^{-2}$ in the control swale and 11.8 $m^3 s^{-1} km^{-2}$ in the logged swale. As with total runoff, logging did not significantly affect the size of the peak flows.

Sediment production varied greatly from year to year at Hayman but generally followed the variation in rainfall intensities (Fig. 3). In the control swale sediment yields were zero in 2005, 2006, and 2010 but exceeded 24 Mg $ha^{-1} yr^{-1}$ in 2003 and 2007 (Table 4). During the calibration period total sediment production in the swale to be logged was 21% less than the control despite its higher total runoff (Table 4). In 2004 and again in 2007 sediment production from the logged swale was only about half the value from the control (Table 4). In 2009, however, the logged swale produced 2.3 times more sediment than the control. As with the runoff and peak flow results, there was no significant effect of logging on sediment production.

At Kraft Springs there were only two rainfall events that produced more than 0.01 Mg ha^{-1} of sediment (Table 4). The control swale produced 1.8 Mg ha^{-1} from the first rainfall event in August 2003 (I_{10} = 24 mm h^{-1}), but in the logged swale a water bar directed the runoff away from the weir. Some sediment was trapped by the vegetation below the waterbar, but this could not be accurately quantified for comparison to the control swale. The second runoff event in July 2004 had I_{10} values of about 60 mm h^{-1} for both swales and the peak flow of 2.0 $m^3 s^{-1} km^{-2}$ in the control swale was three times the value from the logged swale, while the total runoff of 0.96 mm was nearly nine times greater. Sediment production in the control swale was 0.08 Mg ha^{-1} or eight times the value from the logged swale, indicating that the difference in sediment production can be attributed to the difference in runoff. Rain events in 2004 and 2005 produced minor amounts of sediment from the control swale and no sediment from the logged swale, but the I_{10} values in the control swale of 23 and 36 mm h^{-1} were respectively 75% and 34% greater than in the logged swale. No sediment was produced in either swale between 2006 and 2008. Although the number of events is limited, these results are consistent with the Hayman results in that logging did not increase runoff or sediment yields.

4. Discussion

4.1. Effects of salvage logging on post-fire sediment production

The effect of salvage logging on sediment production has been difficult to discern as the results from the few published studies often conflict (McIver and McNeil, 2006). At the plot scale our results generally showed large increases in sediment production with increasing ground disturbance due to the passage of logging equipment. At both the Red Eagle and Tripod study areas mean sediment production rates from the skidder plots were generally at least an order of magnitude higher and significantly different than the corresponding controls for all years. Even the less trafficked feller-buncher plots had significantly higher sediment production rates than the associated controls in two of the three years with measurable sediment production at Red Eagle, and for both years at Tripod. At Red Eagle the relative differences in sediment production increased over time as the control plots recovered more quickly than the skidder and feller-buncher plots.

From the soil and ground cover data we can infer the relative importance of the different controls on post-fire erosion (Fig. 1) and the primary causes for the observed differences in plot-scale sediment production. The bulk density and soil strength data indicated greater compaction with the increase in traffic from the feller-buncher tracks to the skidder tracks, and more compaction will cause less infiltration and lead to more surface runoff and erosion. Soil water repellency, however, showed the opposite tendency as WDPT tended to decrease with increasing traffic and less water repellency was correlated with higher erosion rates (Table 3). Since the amount of surface cover was relatively similar between the skidder and feller-buncher plots, the increase in sediment production with decreasing soil water repellency must be attributed to the overwhelming effect of more compaction and higher sediment production with increased traffic. A reduction in soil water repellency with compaction also was observed after post-fire salvage logging in the northern Sierra Nevada in California (Poff, 1989); laboratory experiments also have confirmed this effect (Bryant et al., 2007). The likely mechanism for this reduction in soil water repellency is the physical shearing and displacement of the soil particles that were coated with hydrophobic compounds during the fire.

The relative importance of compaction versus ground cover can be evaluated by comparing the sediment produced in the controls, the skidder plots, and the slash-treated skidder plots at Red Eagle in 2009–2010. The total ground cover on the controls was slightly more than on the slash-treated plots, yet sediment yields in the compacted slash-treated skidder plots were 20–30 times higher than the uncompacted controls. At the same time the untreated skidder plots had much less total cover and 14–50 times more sediment production than the slash-treated plots. These comparisons suggest that neither compaction nor lack of cover could individually explain the 6–1000 fold increase in sediment production from the skidder plots relative to the controls.

These findings imply that sediment production after post-fire salvage logging is a function of several interacting factors, but for a given location and rainfall intensity the key concerns are the amount of soil compaction and loss of surface cover. Both of these lead to increases in infiltration-excess overland flow but by different mechanisms. The main effect of compaction is a decrease in the large pores that convey most of the water (Hillel, 1998). The reduction in cover can increase soil sealing (Larsen et al., 2009) and thereby decrease infiltration. More overland flow will produce greater soil detachment and transport via sheetwash and rilling (Fig. 1). The loss of ground cover also exposes more soil to detachment by rainsplash (Cerdà, 1998). Less surface cover, particularly litter, will reduce surface roughness, resulting in greater runoff velocity, more shear stress for soil detachment, and greater sediment transport capacity (Bryan, 2000; Foster et al., 1985; Robichaud et al., 2010; Torri et al., 2012).

4.2. Scale effects

A key concern is the extent to which these results can be scaled up to the watershed scale, and this can be assessed using our plot and swale-scale data from Red Eagle. In 2008 both the 150 m^2 control plots and the 0.1–2.6 ha control swales averaged 70% ground cover, but the plots produced about nine times more sediment per unit area than the swales. The higher sediment yield at the plot scale can be partly attributed to the more uniform slope, soil properties, and ground cover allowing more of the eroded sediment to be delivered to the sediment fences. The more heterogeneous conditions in the control swales suggest that more of the surface runoff and sediment was trapped en route, leading to localized deposition and a lower sediment yield per unit area (Cawson et al., 2013; Williams et al., 2014). Comparisons between scales were more difficult for 2009–2010 as the control plots and swales averaged at least 80% ground cover, and this alone can account for the very low sediment production rates.

The effect of spatial scale is much more complex in areas that have been subjected to ground-based salvage logging. In the first one or two years after a high-severity fire virtually the entire hillslope can generate large amounts of surface runoff and erosion given sufficient rainfall, and this tends to make the added impact of logging less pronounced in relative terms. Over time the surface runoff and erosion rates on burned hillslopes not affected by logging decrease, but the ground disturbance from salvage logging can change the spatial pattern of runoff pathways and also slow the expected decline in runoff and erosion rates. The heterogeneity within salvage logged areas therefore tends to increase over one to several years as the less disturbed areas recover while the more heavily trafficked areas continue to produce relatively large amounts of surface runoff and sediment. This discrepancy in recovery rates also means that the spatial organization and connectivity of the most highly disturbed areas becomes a progressively more important control on larger-scale runoff and sediment yields.

We can use the changes in sediment production rates over time from the control and logged plots and swales to test this conceptual model. In 2008 the Red Eagle logged and control swales produced very similar amounts of sediment, even though the logged swales had more bare soil than the controls. In 2009, however, the logged swales produced 20 times more sediment than in 2008 and 22 times more sediment than the controls. The larger response was related to the higher intensity rainfall in 2009, while the larger difference between the logged and control swales was a result of the reduced recovery of the logged swales as compared to the controls. In 2010, which had relatively low intensity rainfall, the control swales produced no sediment and only one logged swale had a measurable sediment yield.

We cannot uniquely identify the cause of the higher sediment production from the logged swales, but if we apply the sediment production rates from the skidder plots and assume that this sediment was delivered to the sediment fences, the 2009 and 2010 swale-scale sediment yields at Red Eagle could be produced by having just 1–2% of the swale area in skid trails. At Tripod less than 5% of the area would need to be in skid and feller-buncher trails to account for the measured sediment yields in the logged swales. Since the areas disturbed by logging were much more than 1–5%, these data indicate that only a relatively small proportion of the sediment produced from the tracked areas in the swales was being delivered to the sediment fences. Downslope delivery also will be affected by the spatial layout of the skid and feller-buncher trails. If the trails converge in the downslope direction a higher proportion of the sediment generated from these trails is likely to reach the stream (Smith et al., 2011), and trail layout will be increasingly important over time as the hillslopes produce less runoff and are more capable of slowing or trapping overland flow.

At yet larger scales one would expect a decreasing effect of post-fire salvage logging on sediment yields because of the increasing potential for sediment storage (e.g., de Vente et al., 2007; Parsons et al., 2006; Wagenbrenner and Robichaud, 2014). In addition, at larger scales the proportion of a watershed subjected to salvage logging will decline as more unlogged or less severely burned areas are included. In Alberta there was no difference in suspended sediment yields from burned catchments that were unlogged and from 700- and 1300-ha catchments that had 34% and 20% of their area disturbed by salvage logging (Silins et al., 2009).

4.3. Variability in responses among study areas

Our four study areas differed substantially in terms of vegetative regrowth, absolute sediment yields, and the rate of recovery to near-zero sediment yields, and this has direct implications with respect to the varying effect of salvage logging under different conditions. Vegetative regrowth and the decline in sediment yields was relatively rapid at Red Eagle and Kraft Springs, and we attribute this to a combination of more precipitation (in the case of Red Eagle) and finer-textured soils. High intensity storms after three years at Red Eagle and two years at Kraft Springs generated very little sediment from the controls.

Tripod had the driest summers, the sandiest soils, very slow vegetative regrowth, and low total cover. A relatively high intensity rain event in the fourth year after burning produced about 25 times more sediment than the more intense storm at Red Eagle in the third year after burning. The Hayman study area had the lowest regrowth rates, which we attribute to the very coarse-textured soils and relatively short growing season. The highest sediment yield from the control swale occurred in the fifth year after burning. These data clearly show the greater potential for continuing high sediment yields from sites with poorer growing conditions.

With respect to salvage logging, the feller-buncher and especially the skid trails have a much greater and longer-lasting potential to produce large sediment yields from high intensity rainfall. However, the areas with more rapid vegetative regrowth have less potential for this sediment to be delivered to the stream network, as shown by the similar logged-swale sediment yields from a very intense rain event at Red Eagle in 2009 and a lower intensity event at Tripod in 2010. We infer that post-fire salvage logging generally will have a shorter-term effect on larger-scale runoff and sediment yields in more productive sites, while areas with slower regrowth have a much greater potential for the additional sediment from salvage logging to be delivered to the stream network.

4.4. Minimizing the effects of post-fire salvage logging

Any effort to reduce the effects of post-fire salvage logging should start from an understanding of how salvage logging affects the various controlling processes, how the rate of recovery varies with site conditions, and how the rate of recovery can affect both the relative and absolute impacts of post-fire salvage logging. From an erosion perspective, the most important effect of ground-based salvage logging is the order(s)-of-magnitude increase in sediment production from the skid and feller-buncher trails relative to the burned controls. Assuming equal rainfall, the smallest relative difference in sediment yields between the controls and highly-disturbed areas will be in the first year after burning when the controls can produce relatively large amounts of surface runoff and erosion. The absolute sediment production rates will decrease over time, but the relative differences in sediment production will tend to increase over time because the more disturbed areas recover more slowly. Resource managers have to determine if the cumulative, absolute sediment production rates from highly-disturbed areas for different rainfall intensities at a given point in the recovery trajectory pose a significant threat to site productivity or downstream resources.

The skid trails are of greatest concern because these have the highest and most persistent increases in sediment production, but the less trafficked feller-buncher trails also had significantly higher sediment production rates than the controls. From a process-based perspective, our results show that sediment production rates after salvage logging are primarily a function of rainfall intensity, ground cover, and compaction. The reduction in soil water repellency due to logging traffic appears to provide no benefit in terms of reducing surface runoff or erosion, and this seemingly counterintuitive result is readily explained by the much more dominant effects of surface cover and soil compaction.

The effects of salvage logging are more difficult to predict at the swale and watershed scales, as these depend not only on the amount of area in skid and feller-buncher trails, but also on the extent to which the runoff and sediment are captured en route or delivered to the stream. The location and layout of the skid and feller-buncher trails is increasingly important at larger scales because this directly affects sediment delivery to the stream (Silins et al., 2009; Smith et al., 2011). Connectivity also will vary with the magnitude of rainfall, as larger or more intense rainfall events will generate more overland flow and cause greater connectivity between the hillslopes and stream network.

Our results indicate that the impacts of salvage logging can be mitigated by: (1) reducing runoff and erosion from skid and feller-buncher trails; and (2) reducing the delivery of runoff and sediment from the skid and feller-buncher trails to the stream. The slash treatment showed that sediment production from skid and feller-buncher trails can be significantly reduced by increasing ground cover, and this is consistent with studies showing that mulching can reduce post-fire erosion (e.g., Prats et al., 2013; Robichaud et al., 2013a; Wagenbrenner et al., 2006) and erosion from skid trails in unburned areas in the eastern U.S. (Sawyers et al., 2012). Further research is needed to determine to what extent the logging slash reduces erosion because of reduced rainsplash versus increased surface roughness.

Compaction was an equally important cause of the observed increases in sediment production from the tracked areas, so ground-based salvage logging generally should be done over the snow or when soils are dry in order to minimize compaction. Although not evaluated as part of this study, ripping may reduce compaction and increase infiltration, as it can reduce runoff and erosion from skid trails and roads (e.g., Luce, 1997), and contour ripping seems to have reduced erosion in salvage logged areas in northern California (James, 2014).

The potential for burned hillslopes to produce large quantities of runoff and erosion means that some standard best management practices may be less effective at reducing the impacts of post-fire salvage logging. Water bars are designed to divert runoff and sediment into areas with high roughness and high infiltration rates, and the water bars in the logged swales at Hayman and Kraft Springs may have helped reduce the delivery of sediment produced in the skid trails to the outlets of those swales. But after a high severity fire the concentration of runoff at the outlets of water bars may induce rilling in areas with little surface cover. Similarly, the use of riparian buffer zones may have little effect on sediment delivery immediately after severe burning because the hillslopes can be well-connected to the stream, particularly when the riparian areas also were severely burned (Pettit and Naiman, 2007). Placing slash at the outlets of the water bars or in the riparian buffer zones may help mitigate rilling and sediment transport, but these were not effective for a study in southern Spain (Marques and Mora, 1998). A more effective strategy would be to reduce surface runoff and erosion from the hillslopes by mulching. Efforts to minimize the effects of salvage logging on sediment production must emphasize intensive treatments of the largest and most persistent sources, which are the skid and feller-buncher trails.

5. Conclusions

The effects of ground-based post-fire salvage logging on surface cover, soil properties, runoff and erosion were measured relative to controls over 4–9 years at the plot and/or swale scales at four study areas in the interior western U.S. All four study areas were dominated by conifers, coarse-textured soils, and burned primarily at high severity. Both the more trafficked skid trails and less-trafficked feller-buncher trails had significantly less cover and vegetative regrowth than the controls. The passage of heavy logging equipment also increased bulk densities at depths to 10 cm, increased soil strength, and reduced soil water repellency.

Sediment production rates varied widely between plot types, study areas and years. Sediment production rates from the skidder plots were typically at least two orders of magnitude higher than the values from the control plots. Sediment production rates from the feller-buncher plots were usually intermediate between the skidder and control plots. The addition of logging slash to some of the skid trails reduced the amount of bare soil by about half, and this reduced sediment production to 2–20% of the values from the untreated skidder plots. The logging slash did not have any additional effects on soil properties or vegetative regrowth. Overall, sediment production was positively and strongly related to rainfall intensity and the amount of bare soil.

Sediment production rates in the control and logged swales were highly variable across the study areas, but direct comparisons were hindered by the lack of unlogged swales at one study area and few runoff events at another. At Red Eagle the logged swales had significantly higher sediment production than the controls in two of the three years, but the sediment production rates in both types of swales were relatively low. In contrast, the maximum sediment production rates at Hayman were much greater and more persistent, but there was no detectable effect of logging on swale-scale runoff rates or sediment yields. The swale-scale effects of salvage logging on sediment production decreased more rapidly in the more productive study areas with faster vegetative regrowth.

These results indicate that the compaction and reduced surface cover in the skid and feller-buncher trails caused large increase in erosion rates. The increase in sediment production from post-fire salvage logging can be reduced by the addition of surface cover such as logging slash, but the reduction in vegetative regrowth, increase in soil strength, and presumably the increase in bulk density are all relatively persistent over time. Salvage logging did not have as large an effect on ground cover, soil water repellency, or runoff and sediment production at the swale scale because much of the area was not directly affected by the logging equipment. The greater sensitivity of burned landscapes to surface runoff and erosion indicates that site-specific best management practices—particularly for areas disturbed by logging equipment—are needed to minimize the adverse impacts of post-fire salvage logging.

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