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WILDFIRE AND HYDROLOGICAL PROCESSES

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Evaluating post-wildfire logging-slash cover treatment to reduce hillslope erosion after salvage logging using ground measurements and remote sensing

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Abstract

Continuing long and extensive wildfire seasons in the Western US emphasize the need for better understanding of wildfire impacts including post-fire management scenarios. Advancements in our understanding of post-fire hillslope erosion and watershed response such as flooding, sediment yield, and debris flows have recently received considerable attention. The potential impacts of removing dead trees, called salvage logging, has been studied, however the use of remotely sensed imagery after salvage logging to evaluate spatial patterns and recovery is novel. The 2015 North Star Fire provided an opportunity to evaluate hillslope erosion reduction using two field experiments and coincidental remotely sensed imagery over 3 years. Simulated rill experiments with four flow rates were used to quantify hillslope erosion on skidder trails with and without added logging slash compared with a burned-only control. Seven replicated hillslope silt fence plots with the same treatments were also evaluated for natural rainfall events. WorldView-2 satellite imagery was used to relate ground cover and erodible bare soil between the two experiments using multi-temporal Normalized Differenced Vegetation Index (NDVI) values. Results indicate that the skid trails produced significantly more sediment (0.70 g s^{-1}) than either the slash treated skid trail (0.34 g s^{-1}) or controls (0.04 g s^{-1}) with the simulated rill experiment. Similarly, under natural rainfall conditions sediment yield from hillslope silt fence plots was significantly greater for the skid trail (3.42 Mg ha^{-1}) than either the slash treated skid trail (0.18 Mg ha^{-1}) or controls (0 Mg ha^{-1}). An NDVI value of 0.32 on all plots over all years corresponded to a ground cover of about 60% which is an established threshold for erosion reduction. Significant relationships between NDVI, ground cover, and sediment values suggest that NDVI may help managers evaluate ground cover and erosion potential remotely after disturbances such as a wildfire or salvage logging.

KEYWORDS

post-fire, recovery, rill erosion, salvage logging, silt fence, skid trail, WorldView

1 | INTRODUCTION

The broad impact of a wildfire disturbance depends on the intensity or severity of the fire, its spatial distribution and extent, and the history and resilience of the site (Neary et al., 2005). The summer of 2015 was the largest wildfire season in Washington state history. Cumulatively, more than 400,000 ha burned between June and September. The largest fires in eastern Washington that year were on the Colville Federation Tribal Reservation, primarily the Okanogan Complex Fires which consisted of several fires including the Tunk Block and North Star Fires. In total, these fires burned 209,000 ha; 91,000 ha of which was on the Reservation and covered nearly 20% of its land base.

The extent and degree of the soil and vegetation disturbance by wildfire may have serious ecological implications for hydrologic and biological processes for years. Economically, the cost of suppression, rehabilitation and loss of natural resources such as timber after a large wildfire can be in the tens or even hundreds of millions of dollars. This means significant lost revenue, particularly in land management organizations where timber harvest is expected to contribute a percentage of the natural resource budget. On the Reservation, there is a yearly timber harvest goal of 185,000 m^3 that provides 85% of the Tribal budget (Boyce et al., 1998; Klock, 2000).

One common post-fire management practice is to remove dead and damaged trees, called salvage logging (Sessions, Bettinger, Buckman, Newton, & Hamann, 2004). Salvage logging is intended to reclaim the economic value of the wood to offset the expenditures from fire suppression and provide rehabilitation dollars (Barker, 1989), and to optimistically reduce the danger of another fire (Collins, Rhoades, Battaglia, & Hubbard, 2012). However, burned forests are characteristically more sensitive to management activities (e.g., harvesting and the associated roads and skid trails) than unburned forests (Beschta et al., 2004; McIver & Starr, 2001). The dual disturbance of wildland fire and salvage logging can generate a variety of short- and long-term impacts (McIver & Starr, 2001). Post-wildfire salvage logging has the potential to exacerbate post-wildfire flooding and soil erosion over an area many times the extent of the original disturbance due to the connectivity of skid trails, logging roads and their proximity to streams (Beschta et al., 2004; Olsen, 2017). Elevated runoff and soil erosion can lead to adverse impacts on water quality (Lewis, Rhodes, & Bradley, 2018) and debris flows.

Previous studies reported that disturbances from logging traffic increases soil erosion by two orders of magnitude and that mitigating the erosion with a wood slash treatment significantly reduces soil loss at the plot scale (Olsen, 2017; Wagenbrenner, MacDonald, Coats, Robichaud, & Brown, 2015). Runoff and erosion vary with ground cover, soil properties, vegetation type and recovery, slope, and rainfall patterns (Peterson et al., 2009), and the response is often complicated by various time steps of when the salvage operations occurs in relation to burn severities (Robichaud, Bone, Brooks, & Brown, 2020). Due to soil compaction and rut formation from logging traffic, skid trails may also provide preferential flow paths where surface runoff may concentrate and accumulate erosive energy and sediment transport capacity (Ares, Terry, Miller, Anderson, & Flaming, 2005; Robichaud, Wagenbrenner, & Brown, 2010; Wagenbrenner, Robichaud, & Brown, 2016). Specific factors such as the effect of ground-based logging equipment on soil compaction have been studied (Han, Han, Page-Dumroese, & Johnson, 2009), but a holistic approach to quantify the disturbance effect of the entire network of salvage logging operations has never been completed. McIver and McNeil (2006) attempted this after the 1996 Summit Fire in Oregon, though salvage logging started 2 years post-fire and no large rainfall events occurred during their study.

Effects of salvage logging may impact the disturbed area over one to several years depending upon the areal extent, and the collective context of burn severity, vegetation, soils, climate, and pre- and post-fire management (Bone, 2017; McIver & Starr, 2001; Morgan et al., 2015; Peterson et al., 2009). The compounded effects of fire and logging may also delay vegetation recovery by a year or more (Morgan et al., 2015; Wagenbrenner et al., 2015, 2016). Because of the potential negative impacts of salvage logging, states or land management agencies often enforce a set of Best Management Practices (BMPs) to minimize detrimental effects. BMPs aim to provide a balance between ecological and economic interests; they allow for logging practices to occur within a set of rules designed to reduce the impact on the environment. Resource objectives that are part of the salvage logging protocol for the Colville Reservation include protection for soils, hydrology, and fish and wildlife (Colville Confederated Tribes, 2018) and to limit detrimental soil conditions to less than 25% of the logged area (Colville Confederated Tribes, 2006).

The most dominant factor influencing post-fire water-driven sediment loss is the amount of ground cover (or conversely, the amount of erodible soil that is unprotected). Vegetation regenerates after a wildfire at a rate that is highly dependent on the pre-fire environment, the fire severity, and the resilience of the ecosystem (Bright, Hudak, Kennedy, Braaten, & Khalyani, 2019; Falk, Watts, & Thode, 2019). Pannkuk and Robichaud (2003) suggest that 60% or more ground cover is needed to minimize soil erosion after wildfire. Such vegetation regeneration can take 2 or more years to regrow naturally, thus, ground cover can be supplemented in sensitive areas (e.g., high soil burn severity) to provide additional protection to the soil (Robichaud, Lewis, Wagenbrenner, Ashmun, & Brown, 2013). Salvage logging creates a disturbance that can essentially 'reset' productive vegetation growth that occurs after the fire, prior to salvage logging (Morgan et al., 2015). In order to minimize additional soil erosion from salvage logging, ground cover such as available logging slash (treetops, branches) can be distributed on skid trails and landings to protect from surface erosion (McIver & McNeil, 2006).

Remote sensing can be used to map and monitor large-scale forest disturbance over time. Remotely mapped disturbances relevant to this study include fire, logging, and deforestation (Jin, Sung, Lee, Biging, & Jeong, 2016; Lentile et al., 2006; Lewis et al., 2017; Lewis, Robichaud, Hudak, Austin, & Liebermann, 2012; Margono et al., 2012). The post-fire and post-logging environments are highly heterogeneous; therefore, a fine spatial mapping scale is beneficial. The vegetation response in the first post-fire or post-salvage year is variable and depends on the severity of the disturbance and the resiliency of the affected ecosystem, as well as the timing of precipitation (Abella & Fornwalt, 2015; Neary, Klopatek, DeBano, & Ffolliott, 1999; Robichaud et al., 2013). Because of the high spatial and temporal variability in the post-wildfire, post-salvage environment, there is a need for high resolution imagery used to detect change at a relevant spatial and temporal scale. The WorldView-2 satellite (MAXAR Technologies, Longmont, CO; www.maxar.com; accessed 4 Sep 2019) has a pixel size of 1.8 m and collects spectral data across eight visible and near infrared bands. WorldView's commercial satellites must be requested to acquire desired imagery of interest (e.g., a wildfire) and images are delivered ready for analysis.

This imagery can be used to calculate the normalized difference vegetation index (NDVI) (Tucker, 1979) which is commonly used to map vegetation greenness, health, and density; it also has a positive relationship with aboveground biomass and total vegetation cover in forests (Hogrefe et al., 2017; Prabhakara, Hively, & McCarty, 2015; Ramsey, Wright Jr., & McGinty, 2004; Trout, Johnson, & Gartung, 2008). The NDVI is calculated by the following equation, where Band 5 is red (R) and Band 7 is near-infrared (NIR) in WorldView-2 images:

$$\mathrm{NDVI} = \frac{\mathrm{NIR}-\mathrm{R}}{\mathrm{NIR}+\mathrm{R}}. \tag{1}$$

NDVI can be used to approximate vegetation phenology across a landscape, and when multi-temporal data are available, in multiple years (João, João, Bruno, & João, 2018). NDVI values less than 0.3 are associated with non-vegetated, or very sparsely vegetated areas (Hivley et al., 2018). Forest disturbance detection is possible because of the reduction in chlorophyll due to vegetation consumption during a fire (Díaz-Delgado, Lloret, & Pons, 2003; Epting, Verbyla, & Sorbel, 2005), or tree removal from logging (Read, 2003), resulting in an increase in reflectance in the visible region of the electromagnetic spectrum and a decrease in the near infrared region (Jin et al., 2016). NDVI can be particularly useful for capturing large-scale biomass disturbances (Mayes, Mustard, Melillo, Neill, & Nyadzi, 2017). NDVI is frequently used to map the extent, severity, and recovery of burned areas (Chuvieco et al., 2004; Epting et al., 2005; Viedma, Garcia-Haro, Melia, & Segarra, 1997).

Our goal was to use NDVI to monitor the change in vegetation and soil cover over time as a proxy for measuring disturbance and recovery after wildfire and salvage logging activities. We compared machine application of logging slash on many-pass skidder (skid) trails within three high burn severity conditions: burned plots without any additional disturbance (control); skid trails without any slash applied (skid); and skid trails with slash applied (treated). Our specific research questions were: (1) did the logging significantly affect sediment loss from the plots compared to the burned-only controls and was the slash treatment effective in reducing the sediment loss? (2) what amount of slash cover was effective in reducing the sediment to an acceptable level? (3) can NDVI be used to remotely estimate the amount of cover on the ground and then used as a proxy to estimate potential erosion risk after a salvage logging disturbance? and (4) is the vegetation response different between salvaged vs not salvaged sites, and can this be measured remotely?

2 | METHODS

2.1 | Study sites

Experiments were conducted within the 88,000 ha North Star Fire (Figure 1) on the Colville Federation Tribal Reservation which started from a lightning strike on 13 August 2015 and was contained on 15 October 2015 (post-fire report available at: https://forest.moscowfsl.wsu.edu/BAERTOOLS/baer-db/2500-8/2500-8_Northstar%20Fire_Okanogan.pdf; accessed 16 Jan 2020). Salvage logging within the burned area began on 10 November 2015 and continued for 2 years post-fire due to the extensive area burned. Salvage logging unit 208-156 was logged during late May 2016. Selecting ideal plot locations and applying treatments required significant

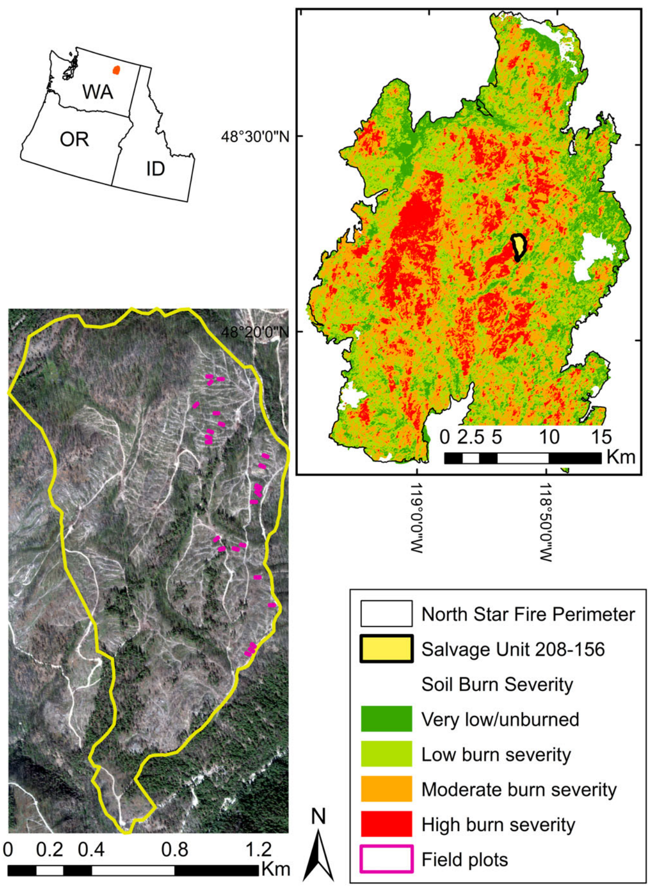


FIGURE 1 The North Star Fire, in north-central Washington State burned in summer 2015. All field plots shown in purple are in salvage unit 208-156, which are shown on a WorldView-2 image from November 2016 (1-year post-fire; 2 months post-salvage). The soil burn severity map was made by the Bureau of Indian Affairs Burned Area Emergency Response Team in September 2015

coordination with the timber sale officer and equipment operators. The burnt trees were cut with a feller-buncher, then a grapple rubber-tyre skidder dragged whole trees to a landing where a processor removed limbs and treetops. On each return trip of the skidder, slash was carried with skidder's grapple hook upslope and placed adjacent to the skid trail. After the last round trip, the stockpiled slash was placed onto the skid trails, about one load every 10 m of skid trail using the skidder's grapple hook. Thus, no additional trips were made to add the slash to the skid trail. The time to place the stockpiled slash on the skid trail was minimal. The skid trails in this study are considered 'many-pass' because there were at least four passes of logging equipment (Wagenbrenner et al., 2016).

The North Star Fire burned in a temperate dry forest with open stands of Douglas-fir (*Pseudotsuga menziesii*) mixed with ponderosa pine (*Pinus ponderosa*) (Clausnitzer & Zamora, 1987; Williams, Kelley, Smith, & Lillybridge, 1995). Douglas-fir stands had an understory consisting of bluebunch wheatgrass (*Agropyron spicatum*), Idaho fescue (*Festuca idahoensis*) or ninebark (*Physocarpus malvaceus*) at lower elevations, or pinegrass (*Calamagrostis rubenscens*), and mountain snowberry (*Symphoricarpos oreophilus*) at higher elevations (Williams et al., 1995). The elevation of the fire ranged from 650 to 2050 m. An analysis of historic fires suggest that occasional large, catastrophic fires have played a role in this region for many centuries, with an average return period of 25 years (LANDFIRE.gov; Williams et al., 1995).

Soils at the study sites belong to the Nevine Series which is an andisol derived from volcanic ash over glacial till parent material, described as ashy over loamy skeletal, glassy over isotic, frigid *Typic Vitrixerand* (Soil Survey Staff, 2016). This soil is well-drained, typically 50 to 100 cm in depth to a dense layer with a very thin A horizon. Clay content is reported at 10%, sand 21%, organic matter 3%, and hydraulic conductivity at 32 mm hr^{-1} (Soil Survey Staff, 2017). This soil is naturally resistant to erosion, except when disturbed, such as by severe wildfire or excess traffic and compaction with heavy machinery (Williams et al., 1995).

Climate in the Okanogan Highlands is generally xeric, due to rain-shadow effects from the North Cascades (Williams et al., 1995). Climate data from Republic, Washington (located 30 km east of the site) report mean annual precipitation of 430 mm, average maximum temperatures of 13.4°C and average minimum temperatures of −0.1°C, averaged from 1981 to 2010 (Western Regional Climate Center, 2017). Precipitation is relatively low from July to September, and most snowfall occurs from November to February. Precipitation records from the nearby Moses Mtn SNOTEL site (elevation 1,527 m, 17 km from sites) (https://wcc.sc.egov.usda.gov/nwcc/site?sitenum=644&state=WA; accessed 13 Dec 2019) are summarized in Table 1. The SNOTEL summary shows that precipitation throughout the study was near normal except the Spring of 2016, which was wetter than normal (Table 1).

TABLE 1 Precipitation data from the Moses Mtn SNOTEL site which includes snow water equivalent and the on-site rain gauges which measured rainfall events only

	Calendar year precipitation (mm)	Water year (October–October) precipitation (mm)	
Year	SNOTEL	SNOTEL	Measured (1 May-31 Oct)
2015	536	574	
2016	838	754	248[a]
2017	683	810	105
2018	648	671	132
2019	589	655	31[b]

Note: The average annual snow-adjusted precipitation during the measurement period (1991 to present) is 716 mm.
[a]15 June 2016 to 31 October 2016.
[b]1 May 2019 to 30 May 2019.

2.2 | Experiment descriptions

These experiments were designed to study the effects of post-fire salvage logging with two established field methods (Robichaud et al., 2010; Robichaud & Brown, 2002). We compare results from two independent experiments, simulated rills and natural rainfall hillslope silt fence plots, conducted on adjacent slopes (Table 2). In both experiments, sites were burned only (Control, or C); burned and salvaged skid trails (Skid, or S); and burned, salvaged skid trails treated with slash (Treated, or T). The two experiments were run concurrently to assess the effectiveness of logging slash cover to reduce soil erosion from disturbed field plots, and the effectiveness was determined through field measurements and remotely sensed imagery. The operational details are presented in Table 3.

2.2.1 | Rill simulation

Seven replicates of each treatment were located within the 208–156 harvest unit on the North Star Fire (Figure 1). We followed the experimental design of Robichaud et al. (2010) with plots that were 4 m long and 1–2 m wide. Some rill plots were located side by side on wheel tracks along the same skidder trail. Water was released at the top of the plot through an energy dissipater at sequentially controlled rates of 15, 30, 45, and 60 L min^{-1} for 15 min each as these inflow rates were used as a surrogate variable that integrated the effects of upslope accumulation area, rainfall intensity, and infiltration rate. The flow width and depth in each rill were measured at 1 and 3 m downslope of the release point, and the combined width and average depths of all rills at each downslope location were calculated and averaged for each flow rate. Rill depths were not significantly different by treatment and did not change over time, and thus were not analyzed further.

Runoff velocity was measured using a saline solution and conductivity metres at 1 and 3 m (King & 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 4-m point. Samples were processed to determine rill flow and sediment flux rates. The same rill sites, using the same water release point at the top of the plot, were used for re-measurement in post-fire years 2 and 3.

2.2.2 | Silt fences

Hillslope erosion was measured following Robichaud and Brown (2002) in close proximity to the rill experiment, but on separate field plots. Eight replicates of each treatment were located within the 208-156 harvest unit on the North Star Fire (Figure 1). All 24 fences were installed during June and July 2016 as soon as the area was accessible after salvage operations and remained in place for 3 years. The fences were designed to capture sediment from similar contributing area plots, typically 4-m wide by 25-m long (Table 2). Silt fences were cleaned out two times each year, initially in late spring to capture sediment associated with spring snowmelt and in late summer to capture sediment associated with higher-intensity rainfall events (Tables 3 and 4). Other studies have conclusively associated high 10- or 30-min rainfall intensities (I_{10} or I_{30}, respectively, mm h^{-1}) with elevated sediment yields after a disturbance (Moody & Martin, 2009; Robichaud et al., 2010). Four recording tipping-bucket rain gauges were used for analysis (Table 4). Rain events were separated by a 6-hr period with no rainfall. For each event the total rainfall (mm), duration (min), and I_{10}, I_{30}, and I_{60} were calculated. Return periods were calculated using a rainfall-frequency atlas (Arkell & Richards, 1986; Miller, Frederick, & Tracy, 1973).

TABLE 2 Plot characteristics by experiment and treatment

Treatment	Control	Skid[a]	Treated
Rills			
Plots (*n*)	7	7	7
Slope (%)	29 (19–39)	27 (21–36)	30 (26–38)
Silt fences			
Plots (*n*)	8	8	8
Slope (%)	22 (13–38)	19 (6–30)	17 (8–28)
Area (m^2)	92 (71–108)	110 (82–141)	95 (78–116)

Note: The mean slope (%) and area (m^2) are given and the range of each in parenthesis.
[a]Skid and treated plots contained rubber-tyre skidder tracks from the logging equipment. The rill plots were located in one of the tracks. The silt fence plots contained two tracks.

2.2.3 | Plot characterization

Representative ground cover plots were measured on the top, middle and bottom sections of each rill and silt fence. Ground cover was determined using a 1-m quadrat with a 100-point sampling grid (Chambers & Brown, 1983; Robichaud & Brown, 2002). Cover categories consisted of bare soil, litter, vegetation, and other. Bare soil included mineral soil, gravel, and ash; litter contained woody debris (including logging slash) and non-living organic debris; vegetation included live, attached vegetation and moss; and other included non-erodible larger rocks and trees. Because slash cover was mechanically added to the treated plots, 'cover' and 'treatment class' are not independent of each other.

Soil water repellency was measured with a water drop penetration time test on the soil surface and 1- and 3-cm depths below the soil surface, with eight drops each (DeBano, 2000; Robichaud, Lewis, & Ashmun, 2008). Time for each water drop to penetrate the soil was recorded in classes as either none/weak (0–60 s), moderate (61–180 s), or strong (>180 s) water repellency. Water repellency was classified by the percent of occurrence in each class. One bulk density measurement was taken along each rill and adjacent to each silt fence unless they were concurrently located then only one sample was taken using a soil core from the 0–5 and 5–10 cm soil depths. Soil cores were oven dried for 24 hr at 105°C and then weighed.

TABLE 3 WorldView imagery acquisition and field ground cover dates by experiment

Description	Date	Post-fire week	Image	Rills	Silt fences
Pre-fire	12 August 2015	0	X		
Post-fire Y0	30 September 2015	7	X		
Post-fire Y1	7 June 2016	43		X	
	17 August 2016	53			X
	21 September 2016	58	X		
Post-fire Y2	13 July 2017	100		X	
	31 August 2017	107			X
	4 September 2017	108	X		
Post-fire Y3	24 July 2018	154		X	
	8 August 2018	156			X
	11 October 2018	165	X		
Post-fire Y4	22 May 2019	197			X[a]
	5 August 2019	208			X[b]

[a]Sediment collected.
[b]Ground cover measured.

TABLE 4 Mean rainfall data measured at the rain gauges and the associated cumulative sediment from the silt fences by treatment within each collection period

	Maximum rainfall intensity				Sediment total (Mg ha^{-1})		
Sediment collection date	$I_{10\ min}$ (mm hr^{-1})	$I_{30\ min}$ (mm hr^{-1})	$I_{60\ min}$ (mm hr^{-1})	Precipitation total (mm)	Control	Skid	Treated
July 2016	19.8	8.0	4.5	20.9	0	0	0
November 2016[a]	24.4	16.3	12.7	231.5	0	3.42	0.18
May 2017	28.6	13.1	8.3	327.2	0	0.30	0.20
August 2017	27.1	17.0	8.5	100.3	0	0	0
May 2018	43.4	15.2	8.5	418.7	0	0.24	0.31
July 2018[b]	71.1	51.6	30.4	61.1	0	1.83	0.32
May 2019	64.0	24.4	13.3	227.1	0	0.10	0.02

Note: The two greatest sediment-producing events are identified with footnotes.
[a]October 14, 2016 event.
[b]June 21, 2018 event.

2.3 | Image pre-processing

We used pre-collected, archived WorldView-2 (WV2) images, and we opportunistically selected the best available images with dates that aligned annually with dates of ground cover measurements on rill plots and silt fences (Table 3). Images were delivered georeferenced. A dark object subtraction was applied to each image to normalize the atmospheric effects between annual images. Residual geometric error was corrected by automatic orthorectification followed by manually geo-registration of all images to a 10-m DEM reference image using 12–20 ground control points per image and a polynomial warping method (ENVI 4.4, ITT Visual Information Solutions, Boulder, CO). The resulting root mean square error (RMSE) of position for each geometric correction was kept below one pixel. NDVI pixel values were extracted at all field plot locations and the mean NDVI for each rill and silt fence plot was calculated for each year.

2.4 | Statistics

Linear mixed-effects models were run (Littell, Milliken, Stroup, Wolfinger, & Schabengerger, 2006) in SAS 9.4 (SAS Institute, Cary, NC) using the post-fire year, treatment, and the potential interactions between post-fire year and treatment as fixed effects, the plot as a random effect, and post-fire week as the repeated measure unit. The plots were not re-measured weekly, rather the unit of time was scaled by week to account for seasonal differences (i.e., spring vs. fall measurements). The model was also run without treatment as a fixed effect to address continuous ground cover recovery as described below. Slope was tested as a covariate and was found not significant. The dependent variables were sediment flux rate, sediment concentration, runoff rate, runoff velocity, and runoff width. For modelling, half of the minimum value (0.001 Mg) was added to each sediment value to eliminate zero values.

For the rill study, the sediment flux and sediment concentration values were $\log_{10}$ transformed to improve the normality of the residuals; other variables met normality assumptions. Untransformed means are presented in tables and figures for ease of interpretation. Runoff and sediment flux rates approached a steady state condition by the fourth sample in each experimental flow rate, thus only Samples 4–6 were used to compare treatments (Robichaud et al., 2010). These 'steady state' dependent variables and flow velocities were averaged by plot for analysis.

Similarly, linear mixed-effects models were also run with the ground cover data. On all plots, the total cover fraction was the dependent variable and post-fire year, treatment, and the interaction between post-fire year and treatment were the fixed effects, the plot was a random effect, and post-fire week was the unit of repeated measure to address recovery. Least significant differences were used to compare differences in Tukey-adjusted least squares means of total ground cover amongst post-fire years on the control plots, and of vegetation cover amongst the interactions between post-fire year and treatment on all plots. All results were considered significant at $p < 0.05$.

A k-means (Hartigan & Wong, 1979) clustering algorithm (kmeans function, R) (R Core Team, 2017) was used to classify the data by similar NDVI and ground cover values in order to identify breakpoints in the NDVI and ground cover data. In k-means cluster analysis, objects (plot values) are initially assigned to a cluster and are iteratively re-clustered until a stopping criterion is satisfied. We used the Euclidean distance between each object, the cluster mean, as the measure and minimized the within-cluster sum of squares of the distances.

3 | RESULTS

3.1 | Plot characterization

Bulk density measurements at 0–5 cm depth were lower on the control (0.93 kg m^{-3}) plots than the skid and treated plots (1.10 kg m^{-3}). Bulk density at the 5–10 cm soil depth was the same on the control plots (0.93 kg m^{-3}), and greater on the skid plots (1.23 kg m^{-3}) and slash treated plots (1.14 kg m^{-3}). The median water drop time at 1 cm soil depth was low (<60 s) in the first year, and at 1 and 3 cm soil depths was low/none (0–5 s) across all other years and treatments.

3.2 | Sediment experiments

3.2.1 | Rill erosion

In the first post-fire year, the treated skid trail plots had significantly less sediment flux (0.34 g s^{-1}) than the skid plots (1.04 g s^{-1}) (Table 5). In post-fire year 2, the control plots (0.07 g s^{-1}) had the least amount of sediment which was an order of magnitude lower than in the first year. In post-fire year 3, the skid plots (0.29 g s^{-1}) had significantly more sediment than the control plots (0.08 g s^{-1}).

Sediment concentration was high on the control (1.7 g L^{-1}) and skid trail plots (2.3 g L^{-1}) in the first post-fire year and significantly decreased by the second year (Table 5). Interestingly, the concentration on the treated plots was relatively consistent over time yet was somewhat higher (not statistically) than either the control or skid treatments in the second year. By the third post-fire year, sediment concentration was low on all plots (0.35 g L^{-1} control, 0.66 g L^{-1} skid, and 0.52 g L^{-1} treated).

Runoff flow was significantly higher on the skid and treated plots than on the control plots in all years (Table 5). By the second year, runoff decreased significantly on the control and skid plots and remained consistent between post-fire years two and three. Runoff velocity was significantly higher on the control (0.24 m s^{-1}) and skid (0.22 m s^{-1}) plots than on the treated plots (0.13 m s^{-1}) in the first post-fire year. By the second and third years, velocity decreased on all plots and remained low (0.11 to 0.14 m s^{-1}).

The summed total width of the simulated rills was lowest in the first post-fire year regardless of treatment (0.24 m control, 0.19 m skid, 0.21 m treated). In the second post-fire year, rill width approximately doubled for each treatment, and remained statistically similar for the skid and treated plots in the third post-fire year (0.27–0.39 m) whereas the control width decreased (Figure 2; Table 5).

3.2.2 | Silt fences

There was no measurable sediment collected on the control hillslope silt fence plots in any year. In the first post-fire year, there was more sediment on the skid trail plots (3.42 Mg ha^{-1}) than either the control or the treated (0.18 Mg ha^{-1}) which were statistically equivalent (Table 6). In the second year, the skid trail plots again had significantly more sediment than the control plots. In the third post-fire year, there was an increase in sediment on both the skid trail plots (2.06 Mg ha^{-1}) and the treated plots (0.63 Mg ha^{-1}). This is likely due to one large rainfall event on 21 June 2018 which had the highest rainfall intensity of the entire study period (71.1 mm hr^{-1}) and produced 1.83 Mg ha^{-1} on the skid trails and

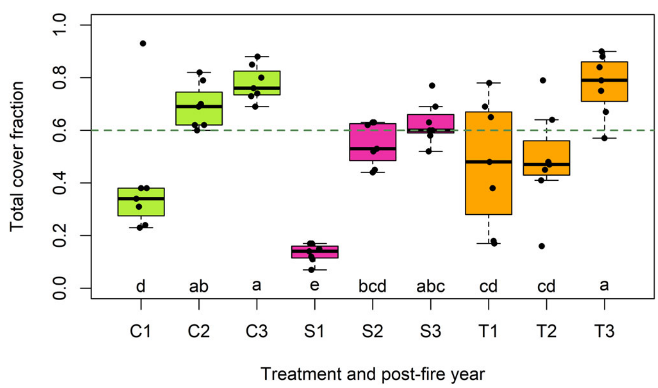


FIGURE 2 Rill total cover fraction by treatment and year. C: control, S: skid, and T: treated. Post-fire years are 1 (2016), 2 (2017), and 3 (2018). Different lowercase letters indicate significant differences between treatment and year

TABLE 5 Sediment and runoff variables from the rill experiment

Post-fire year (year)	Treatment	Sediment flux (g s^{-1})	Sediment concentration (g L^{-1})	Runoff flow (L min^{-1})	Runoff velocity (m s^{-1})	Rill width (m)
1 (2016)	Control	0.70 ab	1.70 a	24.3 c	0.24 a	0.24 cd
	Skid	1.04 a	2.32 a	28.5 a	0.22 a	0.19 d
	Treated	0.34 bc	0.91 ab	27.1 ab	0.13 bc	0.21 d
2 (2017)	Control	0.07 e	0.40 b	18.7 d	0.15 bc	0.48 a
	Skid	0.18 cd	0.48 b	26.1 bc	0.17 b	0.39 ab
	Treated	0.28 cde	1.05 b	25.5 bc	0.10 c	0.35 bc
3 (2018)	Control	0.08 de	0.35 b	18.5 d	0.11 bc	0.35 b
	Skid	0.29 c	0.66 b	25.2 bc	0.14 bc	0.39 ab
	Treated	0.13 cde	0.52 b	25.5 bc	0.14 bc	0.27 bcd

Note: Variable means are given and different letters within a column indicate statistically different values by year and treatment.

TABLE 6 Annual sediment totals from the silt fence plots by year and treatment

	Annual sediment totals (Mg ha^{-1})		
Post-fire year	Control	Skid	Treated
1 (2016)	0 ef	3.42 ab	0.18 def
2 (2017)	0 f	0.30 cd	0.20 cdef
3 (2018)	0 f	2.06 a	0.63 bc
4 (2019)	**0 ef**	0.10 cde	0.02 def
Totals	0	5.88**	1.03*

Note: Lower case letters indicate significantly different sediment yields from the mixed model results.
Compared to control plots: ** significant at p = 0.0001, * significant at p = 0.002.

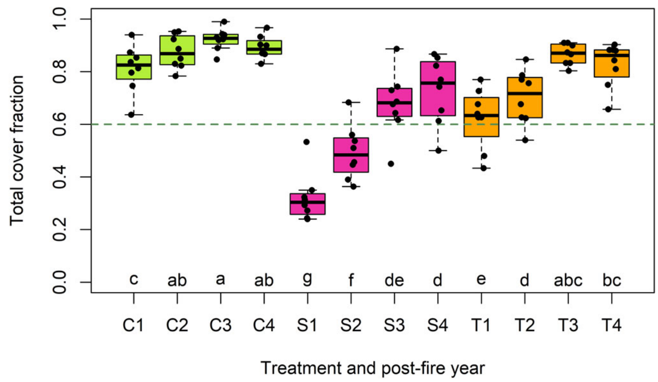


FIGURE 3 Silt fence total cover. C: control, S: skid, and T: treated. Post-fire years are 1 (2016), 2 (2017), 3 (2018), and 4 (2019). Different lowercase letters indicate significant differences between treatment and year

0.32 Mg ha^{-1} on the treated plots. By the fourth post-fire year, sediment values on all plots were low and were not statistically greater than zero; there were no statistical differences due to treatments.

Over all years, sediment yields from the skid trail plots (5.88 Mg ha^{-1}) were significantly greater than both the treated (1.03 Mg ha^{-1}) and the control plots (0 Mg ha^{-1}), and the treated plots had greater sediment yields than the control plots (Table 6). The sediment yield decrease with time was inconsistent (Table 6), likely due to interactions amongst the climate (Table 1) and vegetation regeneration (Figure 2).

3.3 | Ground cover

3.3.1 | Rill total cover

The mean total cover fraction (vegetation plus litter and moss) was low on the control plots (0.40) in post-fire year 1 (June 2016), and on the salvage plots treated with slash (0.48). The cover on the rill plots was significantly lower on the salvage skid trail plots (0.13) (Figure 2). In post-fire year 2 (July 2017), the mean total cover fraction had increased to 0.69 on the control plots and 0.55 on the skid trail plots, whereas the treated plots maintained nearly the same cover fraction (0.49) which was significantly less than the control plots. In post-fire year 3 (July 2018), the total cover fraction on the control and treated plots was 0.78 and 0.77, respectively, whereas the skid trail plots were still lower (0.63). The threshold for cover that significantly reduces erosion is often cited as 60% (Pannkuk & Robichaud, 2003), and all plots crossed that threshold by the third post-fire year, regardless of treatment or experiment.

3.3.2 | Silt fence total cover

Ground cover on the silt fence plots was measured later than the rill plots in each year: 10, 7, and 2 weeks later, respectively (Table 3; Figure 3). In the first growing season particularly, there is a distinct difference in ground cover between plots of the same treatment from post-fire week 43 (7 June 2016) and post-fire week 53 (17 August 2016) (Figures 2 and 3). On the silt fence plots, the total cover fraction on the control plots was 0.81, which was significantly greater than either the treated (0.62) or skid plots (0.32) (Figure 3). Cover increased on all plots in the second post-fire year (2017) and was highest on the control plots (0.88) followed by the treated (0.70) and skid plots (0.49). By the third post-fire year (2018), cover on the control and treated plots were similar (0.92 and 0.87, respectively), whereas the skid plots had 0.68 cover in August 2018. The control and treated plots had more than 60% cover for the duration of the study (Figure 3).

3.4 | NDVI results

Prior to the fire (12 August 2015 image), mean NDVI values ranged from 0.74–0.77 on the study plots; these are typical NDVI values for healthy green temperate forest vegetation (Price, 1993). Seven weeks after the fire (30 September 2015 image), the NDVI values averaged 0.1 on all plots (Figure 4). The similarity across plots of pre- and post-fire NDVI values suggests comparable vegetation density and health prior to the fire, as well as burn severity conditions after the fire. One year after the fire (21 September 2016 image), the NDVI values on the control plots averaged 0.44 indicating significant green vegetation response. Salvage logging occurred in the first post-fire year resulting in additional soil disturbance and vegetation removal. The plots that were located on skid trails (S) from the salvage logging had mean NDVI values of 0.25, whereas the plots that were on skid trails that had been treated (T) with slash cover had higher mean NDVI values of 0.31 (Figures 4 and 5). Interestingly, these disturbances and their spatial patterns can be easily seen by treatment (Figure 5c,d). In the second post-fire year (September 4, 2017 image), the mean NDVI value on the control plots (0.51) remained significantly higher than the skid-only or treated plots (0.37 and 0.38, respectively) (Figure 4). By the third post-fire year, (11 October 2018 image), mean NDVI values on all plots had somewhat equalized ranging from 0.34 to 0.38.

In the most immediate post-salvage imagery (September 2016) (Figure 5), the disturbance from salvage logging is visually apparent. Higher NDVI values (white and light grey) indicate healthy green vegetation, whereas dark grey and black areas have little or no green vegetation and may include senesced vegetation, logging slash, and bare soil from the skid trails (Figure 5a,c). Skid trail systems often have a central 'landing' from which the trails stem (Figure 5c,d) and are often the most disturbed because of the increased logging traffic. In the classified NDVI images (Figure 5b,d) the disturbance is apparent along the skid trails with the orange and red colours, whereas the areas of greatest disturbance are more concentrated red pixels.

Drawing from the positive relationship between total cover and NDVI for both silt fence and rill plots and the NDVI class breakpoint of 0.32 (Figure 6a), the cover was partitioned into low and high cover amounts. We further investigated the inverse relationship between cover and sediment yield from the silt fence plots and sediment flux from the rill plots and found a stronger relationship (R^2 = 0.78) between NDVI and rill plot sediment flux with a breakpoint of 0.30 g s^{-1} likely due to the timing of ground cover measurements, June 7 (post-fire week 43) for rill plots compared to August 17 (post-fire week 53) for silt fence when some vegetation may have senesced (Table 3; Figure 6b). The result was a significant, negative relationship (R^2 = 0.62) between NDVI values and sediment flux – as NDVI (and corresponding cover) decreased, sediment flux increased with the same breakpoint for high and low sediment flux (Figure 6c). This association makes physical sense, but it is novel to use a remotely sensed metric to predict the magnitude of a sediment response.

4 | DISCUSSION

4.1 | Soil properties

The skidder traffic increased soil bulk density by ~25% up to a soil depth of 10 cm. This has been observed in several studies after wildfire salvage operations (Silins, Stone, Emelko, & Bladon, 2009; Wagenbrenner et al., 2015) and green timber harvest without fires (Page-Dumroese et al., 2006). Increased soil bulk density can contribute to slower vegetation recovery (i.e., seedling establishment) in the short term (Morgan et al., 2015) and has been found to affect tree growth in the long term (Spanos et al., 2005). The high bulk density likely contributed to prolonged high runoff flow (Table 5) on both the skid trail and treated plots which is consistent with other research (Ares et al., 2005; Wagenbrenner et al., 2015). The compaction from the skidder traffic can compromise the large soil pores that convey water which can result in a longer recovery time to return soils to pre-fire hydraulic conductivity (Page-Dumroese et al., 2006).

Low to no water repellency was measured at any depth or in any year. High soil burn severity is often associated with decreased soil infiltration in the top 1–5 cm of the soil profile, but that did not seem to be a contributor to the increase in runoff we measured in the first post-fire years. Rather the increase in runoff is much more likely a result of skidder traffic, soil disturbance, bare soil, and compaction (Wagenbrenner et al., 2015).

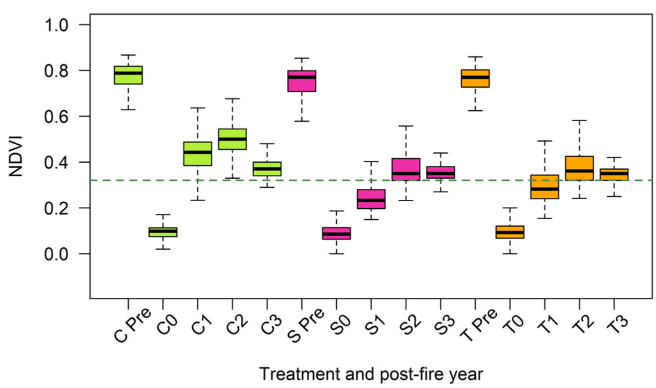
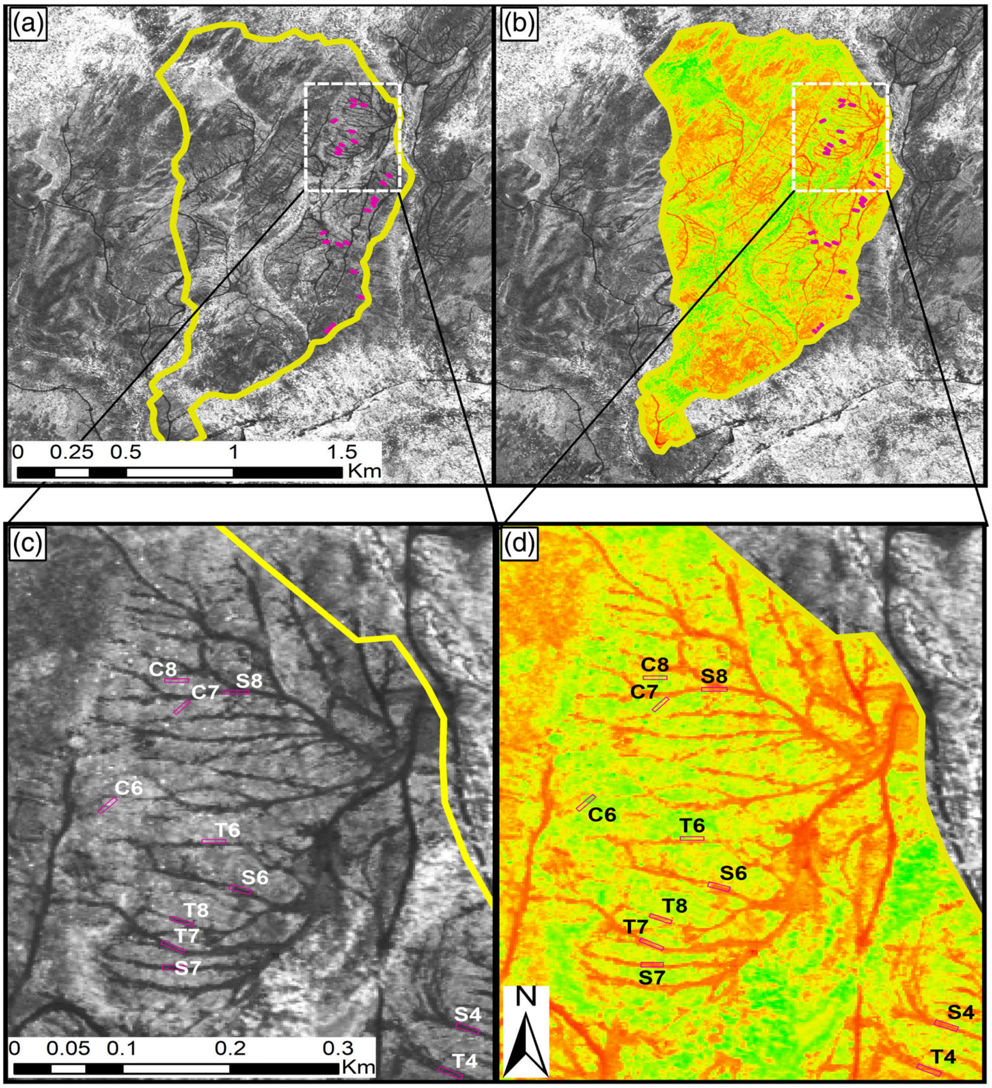


FIGURE 4 Mean NDVI by post-fire year and treatment. 'Pre' is pre-fire (12 August 2015); C: control, S: skid and T: treated. Post-fire years are pre-fire (August 2015), 0 (September 2015), 1 (2016), 2 (2017), and 3 (2018). The horizontal dashed line is at NDVI = 0.32, indicating a breakpoint between 'sufficient' cover and not. NDVI, Normalized differenced vegetation index

4.2 | Ground cover and NDVI

Total ground cover included logging slash, litter and vegetation; total ground cover is inversely correlated with bare soil cover, which is the primary factor affecting post-disturbance runoff and soil erosion (Benavides-Solorio & MacDonald, 2005; Cerdà, 1998; Moody & Martin, 2009; Pannkuk & Robichaud, 2003). As in other studies (Berg & Azuma, 2010; Pannkuk & Robichaud, 2003), we found that ground cover of 60% or more is needed for reducing runoff and sediment from disturbed soils. The highest sediment yields were observed when cover was below 60%, which occurred on all plots regardless of treatment in the first post-fire year. Spring of 2016 recorded above normal precipitation (Table 1) that promoted rapid vegetation recovery. Total cover on the control and skid plots at least doubled between the first and second post-fire years (Figure 2), whereas cover on the treated plots remained high (due in part to the added slash cover). Examining the variability in cover in the first year on the treated rill plots, it was apparent that 'treated' meant a wide variety of cover was applied (14–76% cover in year 1; Figure 2). This led us to investigate the significance of the amount of cover needed for reducing sediment, rather than just the broad classification of 'treated' (Figure 6).

The evaluation of the relationship between total cover and the NDVI allowed us to approximate the amount of ground cover remaining after salvage logging and the slashing treatment, which we used as a proxy for estimating ground disturbance. Based on significant relationships between total cover, sediment flux, and NDVI (Figures 5 and 6) we concluded that NDVI can be used at an event-scale for approximating post-salvage disturbance as it relates to hydrologic events. Carlson and Ripley (1997) noted that the NDVI was suitable for small-scale change detection studies. Hogrefe et al. (2017) found that NDVI estimated landscape-scale biomass at



Fence	Treatment	2016 NDVI	Total cover
C6	Control	0.52	0.84
C7		0.44	0.81
C8		0.47	0.75
S4	Skid	0.21	0.32
S6		0.25	0.53
S7		0.20	0.29
S8		0.28	0.27
T4	Treated	0.22	0.73
T6		0.50	0.43
T7		0.28	0.64
T8		0.27	0.48

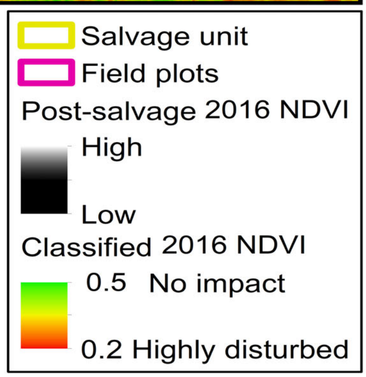


FIGURE 5 Post-Salvage Logging Unit 208-156 in 2016: (a) unclassified NDVI image, (b) classified NDVI image, (c) zoomed in unclassified NDVI image, and (d) zoomed in classified NDVI image. The silt fence plots are named by treatment (C: control; S: skid; T: slash treated) and are delineated along or next to the skid trails. NDVI, Normalized differenced vegetation index

R^2 = 0.67, and Prabhakara et al. (2015) found that the NDVI estimated total cover at R^2 = 0.85. These studies recognized the utility of NDVI measuring ground cover in sparsely vegetated environments, which corresponds with a post-fire, post-salvage environment. The novelty of our study was that we incorporated the remotely sensed measure NDVI to map total ground cover and relate it to post-disturbance hydrologic response. Martínez-Murillo and López-Vicente (2018) used high resolution imagery to model post-fire salvage hydraulic connectivity. We are unaware of any other studies that have suggested the NDVI-sediment correlation, however Kim, Kim, Li, Yang, and Cao (2017) successfully used NDVI to map urban greenness to predict runoff reduction using MODIS imagery. The premise is the same in that ground cover provides soil protection and NDVI can predict both greenness and total cover.

In the Colville Forest Practices Handbook (Colville Confederated Tribes, 2006) it suggests that 'detrimental soil conditions shall not be caused on more than 25% of the treatment area, which are the salvage harvest units in this situation. It is difficult to make an estimate

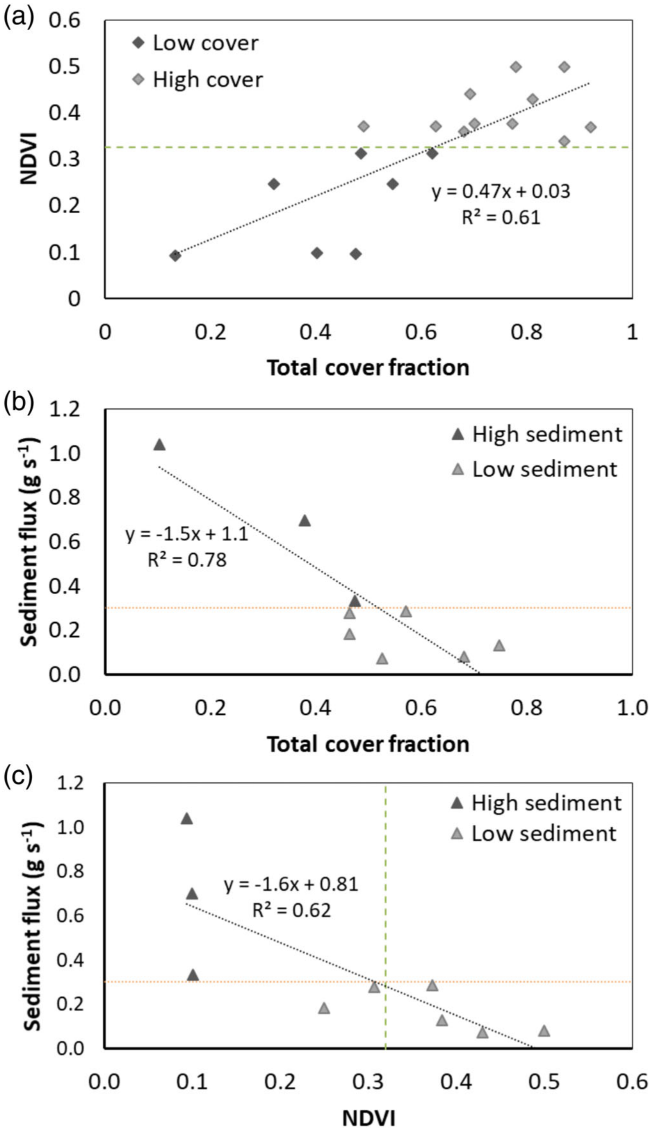


FIGURE 6 Results of the two-class *k*-means clustering algorithms: (a) the NDVI breakpoint of 0.32 (horizontal line) separates rill and silt fence plot cover into two classes; (b) the sediment breakpoint of 0.30 g s^{-1} (vertical line) separates the sediment flux values into two classes; and (c) the two breakpoints are shown together and illustrate how NDVI values can be used to approximate the amount of post-disturbance sediment. NDVI, Normalized differenced vegetation index

of areal disturbance from the ground point-of-view. We used the satellite imagery aerial view to estimate the amount of area that was disturbed due to the salvage logging (i.e., skid trails and landings). Using the NDVI value of 0.32 that corresponded to the breakpoint in the k-means analysis (Figure 6), we found that 29% of the harvest unit would be classified as 'disturbed'. This implies that there is ~30% disturbed bare soil in the first year after the harvest was completed, and that a precipitation event with sufficient intensity and rainfall would likely result in runoff and soil erosion. Based on the density of skid trails, this disturbance estimate seems reasonable and strengthens our claim that the NDVI can be used to estimate soil disturbances.

There is a need to monitor the disturbed environment after wildfire and salvage logging. We feel confident in our ability to scale our plot measures up to the spatial scale of the individual salvage unit, but we are unsure how this may relate to areas outside of the disturbed area. These results are from one salvage unit on one fire. The best results relating ground cover to NDVI and sediment yield were derived shortly after the salvage logging when the range of disturbance was the greatest (Figure 5).

It is often desirable to model continuous variables, particularly when attempting to identify a threshold or maximum value of the independent variable. However, it is also useful to know if a treatment (class) was effective to reduce sediment yields, particularly when considering management recommendations. The k-means classification that was used to produce the cover and sediment thresholds (Figure 6) separate the data into two classes, low and high cover, and low and high sediment. We feel that from a management perspective, these bimodal classifications are useful to help identify areas of concern for post-disturbance runoff and soil loss.

A challenge with remotely sensed imagery is coordinating the heterogeneity of the ground target with the spatial scale of the imagery (Read, 2003). 'Pure' target pixels are desirable but are practically nonexistent in a forested environment. A typical skid trail is 4-m wide, therefore it is likely to be captured in one or two WorldView pixels, making it easily detectable. Whereas, a 4-m skid trail in a 30-m Landsat pixel would not be captured. We were able to extract NDVI values from pixels with known ground cover properties that allowed us to draw strong relationships between NDVI and total cover. The result was a broad-scale estimate of ground cover, which is useful for approximating salvage unit disturbance. Minimizing the total area of disturbance is an operational strategy for minimizing the impact to forest soils and restoration expenses (Han, Page-Dumroese, Han, & Tirocke, 2006).

4.3 | Rill experiment

Several simulated rill studies have shown increases in sediment yields after forested wildfires (Prats, Malvar, Vieira, MacDonald, & Keizer, 2013; Robichaud, Lewis, Wagenbrenner, Brown, & Pierson, 2019) and post-fire salvage logging (Wagenbrenner et al., 2015, 2016). Sediment flux values measured here were lower than in other studies (Robichaud et al., 2019). For example, immediately after 2002 Hayman Fire in Colorado, sediment flux from the control burned plots averaged 1.9 g s^{-1}, and 0.9 and 1.1 g s^{-1} on the treated mulched plots (straw and wood mulch, respectively). By post-fire year 10, the sediment flux rate from the control plots decreased to about 1 g s^{-1} (Robichaud et al., 2019). The control plots in the present study had sediment flux values of 0.07 and 0.08 g s^{-1} in post-fire years 2 and 3, the same as on the Hayman Fire unburned plots. The volcanic-parent material derived soils in this study were less erodible and similar to two nearby studies also in Washington state: the 1998 North 25 Fire, 125 km southwest (Robichaud, Lillybridge, & Wagenbrenner, 2006); and the 2006 Tripod Fire, 90 km to the southwest (Wagenbrenner et al., 2015), compared to the granitic soils of the Colorado Front Range (2002 Hayman Fire). Another similar post-salvage study with low post-fire sediment flux rates (0.05 g s^{-1}) occurred 1 year after 2019 Terrance Mountain Fire, approximately 190 km to the north in British Columbia, Canada, which also had rapid vegetation recovery on glacial till soil (Robichaud et al., 2013; Wagenbrenner et al., 2015).

While the sediment values from the rill plots decreased over time for all treatments, the runoff flow decreased the most between the first and second post-fire years on the control plots (Table 5), but then remained somewhat constant. The runoff flow was higher on the skid and treated plots compared to the control plots in every year. This is likely because of lower infiltration along the rill due to compaction, since the bulk density was consistently greater on the skid trails (1.1 kg m^{-3}) with and without treatment than the control plots (0.93 kg m^{-3}) at 0–5 cm. Interestingly, the runoff velocity was slowed by the slash treatment on the skid trail plots especially the first year (Table 5). Similar patterns in runoff flow and sediment flux were observed after the Hayman Fire between treatments over time (Robichaud et al., 2019).

The rill experiment was done early in the first growing season (7 June 2016), and vegetation cover was still low with a maximum vegetation cover of 48% on the treated plots, 40% on the control plots, and only 13% on the skid plots (Figure 2). By the second post-fire year, (July 2017), vegetation significantly increased on the control and skid plots to 69 and 55%, respectively, indicating a strong vegetation response, slower velocities, and wider rills (Table 5). These values suggest that the supplementary slash cover on the treated plots was serving to maintain a baseline amount of cover, but the vegetation response on the untreated plots was greater in the first two post-fire years (Figure 2). The first 1 or 2 years after fire or salvage disturbance is generally considered the greatest risk period for runoff and soil erosion (Robichaud et al., 2013; Wagenbrenner et al., 2015); thus, the slash cover was an effective erosion mitigation treatment during this time while the natural vegetation recovery caught up (Morgan et al., 2015).

4.4 | Silt fences

The lack of measurable sediment from the control hillslope silt fences was somewhat surprising but can be explained by the robust vegetation growth that occurred throughout the summer of 2016 (81% cover in post-fire week 53, on 17 August 2016). This is double the 40% cover that was measured on the nearby control rill plots 10 weeks earlier. Similarly, we measured nearly three times as much cover on the skid plots (32%) in post-fire week 53 (17 August) compared to 13% on the rill plots in post-fire week 43 (7 June). As we found in this study (Figure 6) and was established by Pannkuk and Robichaud (2003), and confirmed recently by Prats, Malvar, Coelho, and Wagenbrenner (2019) with slash, 60% ground cover is an effective breakpoint to significantly reduce hillslope erosion. Since total cover surpassed the 60% threshold during the first growing season on the control and slash treated plots it is not surprising the lack of measured sediment. Additionally, the first major rainfall event occurred on 17 October 2016 after the vegetation and slash treatment cover was established (Table 1). The skid trail silt fence plots produced more sediment than the slash treated plots in every year, although not all values were statistically higher (Table 6). This finding corroborates the results from the rill portion of the experiment – the additional slash cover was an effective erosion mitigation treatment. Beyond protective ground cover, rainfall amount and intensity are often the primary drivers of runoff and soil erosion (Moody & Martin, 2009; Robichaud et al., 2006). There was one storm on 21 June 2018, that had high rainfall intensity with an I_{10} of 71.1 mm hr^{-1} that was associated with the relatively large sediment values collected in the silt fences 2018, even though cover was greater than 60% (Tables 4 and 6).

For comparison, sediment production data after salvage logging on the 2006 Tripod Fire suggests that with a maximum I_{10} of 33 mm h^{-1} the mean annual sediment production rate from the skid plots was 1.0 Mg ha^{-1}, or about 16 times the mean value of 0.065 Mg ha^{-1} from the control plots in the first post-fire year with about 70% ground cover (Wagenbrenner et al., 2015).

The rill and silt fence experiments are independent approaches that we used at this site in order to answer multiple research questions. The results from both experiments largely agree and emphasize the importance of ground cover for reducing soil erosion as well as the importance of recovery time. The rill experiment is unique because we are able to replicate the experimental conditions (slope, applied water, time, treatment) at a small scale, and from these results we are able to directly make conclusions and inferences about treatments and soil properties. The silt fence experiment is spatially larger (~80 m^2 compared to the ~4 m^2 in the rill experiment) and is the experiment is temporally longer. Treatments are applied on the hillslope plots and the same plots are re-measured each year of the study. The increase in both scales of the silt fence study introduces variability and error, but also gives a more realistic representation of how a treatment behaves over time. The variability and heterogeneity increase with scale, yet the applied nature of the study yields results in which we have confident in and recommend to managers.

4.5 | Management implications

Logging slash applied at a rate to achieve a mean ground cover >60% was found to be an effective treatment to reduce post-salvage runoff and soil erosion. Depending on the size class of the slash, various amounts are needed to achieve 60% ground cover. For example, 100-hr fuels of ~8 t ha^{-1} and 1,000-hr fuels of ~100 t ha^{-1} both provide 60% ground cover. Han et al. (2006 and 2009) suggest about 70% slash cover or at least 150 t ha^{-1} of slash to reduce effects of heavy logging equipment on soil compaction. In this study at least 60% ground cover was attained with ~1 skidder's grapple hook load placed every 10 m of skid trail. As we worked with the salvage logging operators, we observed that they made no extra trips to bring the logging slash adjacent to the skid trail during logging operations. The time to place the stocked piled slash on to the skid trail was minimal yet the benefit in sediment yield reduction was significant. We recommend this treatment under similar conditions where post-disturbance hydrologic response is a concern.

5 | CONCLUSIONS

Using two established experimental methods, simulated rills and hillslope silt fences plots, we measured hillslope erosion rates after salvage logging. We evaluated the effectiveness of a logging slash treatment to reduce erosion on skid trails and found that it was effective at reducing runoff and soil erosion in the first 1–2 years after the disturbance. Adding logging slash to provide at least 60% ground cover immediately following logging operations significantly reduced sediment yield. Results from the rill experiment in the first post-fire year showed that we measured twice as much sediment on the control plots (0.70 g s^{-1}) than the treated (0.34 g s^{-1}), and three times as much on the skid (1.04 g s^{-1}). By the third post-fire year both the control and treated plots had similarly low sediment flux rates, whereas the skid plots were still significantly higher than the control plots.

Results from the silt fence experiment were similar, and our highest sediment yields were during post-fire year 1, which had one high rainfall intensity event; however total rainfall amount was also responsible for the increased vegetation cover which reduced sediment yield especially in the following years. Sediment yields from the skid trail silt fence plots was higher than the control and treated plots in all years.

High resolution WorldView satellite imagery demonstrated the utility of remote sensing to quantify the logging slash cover and the vegetation regrowth which both contributed to an increase in cover over time, and a significant reduction in erosion on the skid trails. Using NDVI values to quantify the disturbance and distribution of logging equipment skid trails shows how vegetation regrowth and logging slash both reduce hillslope erosion at a hillslope-disturbance scale.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the Confederated Tribes of the Colville Reservation. Restrictions apply to the availability of these data, which were used under agreement for this study. Data are available from Dr. P. R. Robichaud pending the permission of Confederated Tribes of the Colville Reservation.

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