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Effectiveness of post-fire salvage logging stream buffer management for hillslope erosion in the U.S. Inland Northwest Mountains

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Abstract

Active wildfire seasons in the western U.S. warrant the evaluation of post-fire forest management strategies. Ground-based salvage logging is often used to recover economic loss of burned timber. In unburned forests, ground-based logging often follows best management practices by leaving undisturbed areas near streams called stream buffers. However, the effectiveness of these buffers has not been tested in a post-wildfire setting. This experiment tested buffer width effectiveness with a novel field-simulated rill experiment using sediment-laden runoff (25 g/L) released over 40 min at evenly timed flow rates (50, 100 and 150 L/min) to measure surface runoff travel length and sediment concentration under unburned and high and low soil burn severity conditions at 2-, 10- and 22-month post-fire. High severity areas 2-month post-fire had rill lengths of up to 100 m. Rill length significantly decreased over time as vegetation regrowth provided ground cover. Sediment concentration and sediment dropout rate also varied significantly by soil burn severity. Sediment concentrations were 19 g/L for the highest flow 2-month post-fire and reduced to 6.9–14 g/L 10-month post-fire due to abundant vegetation recovery. The amount of sediment dropping out of the flow consistently increased over the study period with the low burn severity rate of 1.15 g L^{-1} m^{-1} approaching the unburned rate of 1.29 g L^{-1} m^{-1} by 2-year post-fire. These results suggest that an often-used standard, 15 m buffer, was sufficient to contain surface runoff and reduce sediment concentration on unburned sites, however buffers on high burn severity sites need to be eight times greater (120 m) immediately after wildfire and four times greater (60 m) 1-year post-fire. Low burn severity areas 1-year post-fire may need to be only twice the width of an unburned buffer (30 m), and 2-year post-fire these could return to unburned widths.

KEYWORDS
buffer, erosion, ground cover, riparian, salvage logging, soil burn severity

1 | INTRODUCTION

Large wildfires continue to present global challenges for forest land management as the size and total area burned has increased (Littell, McKenzie, Peterson, & Westerling, 2009; Westerling, Hidalgo, Cayan, & Swetnam, 2006). After wildfires, post-fire assessment teams evaluate the effects of the fire on the landscape with particular attention to potential watershed response after rain and snow-melt events (Robichaud & Ashmun, 2013). The soil burn severity after a wildfire is assigned based on remotely sensed changes in vegetation and surface conditions and is validated with ground measurements (Lentile et al., 2009; Parsons, Robichaud, Lewis, Napper, & Clark, 2010). The degree of alteration to the soil condition is indicated by one of three soil burn severity classes: high, moderate or low. Increased soil burn severity indicates more consumption of vegetation and organic material, greater exposure of mineral soil and soil charring, with the possible loss of soil structure and cohesion (Parsons et al., 2010). Depending on post-fire management objectives, salvage logging can be used as a forest management approach to recover and minimize the economic losses associated with the loss of timber (Peterson et al., 2009; Sessions, Bettinger, Buckman, Newton, & Hamann, 2004). However, this logging often occurs when the burnt landscape is most vulnerable to increased runoff and erosion, and presents the greatest risk to water quality (Emelko, Silins, Bladon, & Stone, 2011; Kunze & Stednick, 2006; Lane, Croke, & Dignan, 2004; Robichaud, Wagenbrenner, & Brown, 2010). There is growing evidence that ground-based post-fire salvage logging with heavy machinery may increase sediment production on impacted areas (Fernández et al., 2007; Prats, Malvar, Coelho, & Wagenbrenner, 2019; Robichaud, Lewis, Brown, Bone, & Brooks, 2020; Silins, Stone, Emelko, & Bladon, 2009; Smith, Sheridan, Lane, & Brena, 2011; Wagenbrenner, MacDonald, Coats, Robichaud, & Brown, 2015; Wagenbrenner, Robichaud, & Brown, 2016). Ground-based timber harvesting approaches use skid trails for hauling trees from their felled locations to landing areas where logs are loaded for transportation. High traffic on skid trails and landing areas increases physical disturbance to forest soils, mainly through soil compaction (Han, Han, Page-Dumroese, & Johnson, 2009) and vegetation removal (Wagenbrenner et al., 2016). The severity and extent of disturbance partially depends upon harvest methods (Klock, 1975; Morgan et al., 2015; Silins et al., 2009). The layout of harvest traffic routes becomes important during storm events when these disturbed areas may become sediment sources and hydrologic transport pathways which deliver sediment to downstream waterbodies (Haupt & Kidd, 1965; Megahan & Kidd, 1972). The potential for increased sediment production from traffic routes should influence their location and proximity to stream channels to avoid impairing water quality (Litschert & MacDonald, 2009).

Past studies have found concentrated flow and rilling to be influential hillslope erosion processes following wildfire and post-fire salvage logging (Moody & Kinner, 2006; Robichaud et al., 2010; Wagenbrenner et al., 2016). At the plot scale, bare soil on disturbed hillslopes tends to generate surface runoff which concentrates into rills (Benavides-Solorio & MacDonald, 2001; Larsen et al., 2009). Since rills have greater depth, concentrated flow in rills have greater hydraulic power and thereby more erosive energy than interrill flow (Pietraszek, 2006; Wagenbrenner, Robichaud, & Elliot, 2010). At the hillslope scale, rills may connect source areas (i.e., skid trails, landings and roads) with stream channels, providing dispersed flow paths for sediment transport (Olsen, Wagenbrenner, & Robichaud, 2020).

Forest practices have been developed to reduce the impact of harvest activities on sediment nonpoint source (NPS) delivery (Brown, Brown, & Binkley, 1993; Lee, Smyth, & Boutin, 2004). Prudent practices require a significantly reduced amount of traffic within close proximity to stream channels, effectively limiting source areas and flow paths that may deliver sediment (Rashin, Clishe, Loch, & Bell, 2006; Ward & Jackson, 2004). Best management practices (BMPs) use riparian zones as harvest buffers to mitigate upland source areas from affecting downstream waterbodies by relying upon undisturbed soils and vegetation in the buffer to infiltrate surface runoff and increase sediment deposition (Clinnick, 1985; Litschert & MacDonald, 2009). Buffers that have undisturbed vegetation, litter and woody debris ground cover may further improve infiltration rates by maintaining open pore spaces and encouraging sediment deposition by reducing runoff velocities because of increased surface roughness (Castelle, Johnson, & Conolly, 1994; Megahan & Ketcheson, 1996).

After wildfire, riparian zones likely lose some effectiveness in buffering streams from upland sediment sources due to reduced vegetation, litter, woody debris and soil organic matter (Johnstone & Kasischke, 2005). Wildfire can also cause a significant decline in soil infiltration rates due to water repellent soil conditions (Shakesby & Doerr, 2006). Wildfire often increases the severity and extent of soil water repellency with translocated hydrophobic organic compounds coalescing in shallow soil layers resulting in lower infiltration rates (Doerr, Shakesby, & Walsh, 2000; Jordán, Zavala, Mataix-Solera, & Doerr, 2013; Moody, Kinner, & Úbeda, 2009; Robichaud, 2000). Fire-induced alterations to the buffering characteristics of riparian zones may lead to higher volumes of surface runoff and higher sediment loads reaching stream channels. If the reduced buffering capacities of riparian zones are not considered during post-fire management, downstream waterbodies may be more vulnerable to sediment delivery.

Riparian zones with higher soil burn severity may not only experience reduced buffering capabilities but may be sources of sediment production during runoff events (Dwire & Kauffman, 2003). Therefore, remotely sensed imagery and soil burn severity maps may serve as useful tools to inform managers about locations where riparian zones may be less effective at reducing sediment delivery.

Forest managers lack information on how to compensate for the reduced mitigation effectiveness of riparian buffers under post-fire conditions as it is largely unknown the extent to which current harvest buffer BMPs are applicable after fire (McIver & Starr, 2001). Typically, harvest buffers are defined by lateral distances from a stream's bankfull width. In the U.S. Pacific Northwest, jurisdictions may specify various buffer width BMPs based upon stream size, or the water quality standards connected to the waterbody (Lee et al., 2004). Most BMPs in the Pacific Northwest region allow for selective harvest within riparian zones if specified stream shading requirements are achieved. Some BMPs require harvest buffer width extensions to include steep hillslopes or exposed, erodible soils that are adjacent to streams. Yet, in high soil burn severity situations, exposed, erodible soils may be widespread and stream shading requirements may no longer be achievable making pre-fire BMP guidance unclear.

Another challenge in the post-fire environment is determining the extent to which the hydrologic characteristics of a riparian buffer recover with time since the fire. Sufficient mitigation for NPS sediment pollution should depend upon the timing of salvage logging operations after the fire (García-Orenes et al., 2017; McIver & McNeil, 2006). As vegetative regrowth occurs within riparian zones, their buffering effectiveness may improve over time and required widths could decrease in response. Recovery time may vary by location due to a combination of factors including soil burn severity, soil type, climate, aspect, and vegetation regrowth (Halofsky & Hibbs, 2008; Pettit & Naiman, 2007). Additionally, pre- and post-fire management activities within riparian zones may influence their recovery (Dwire & Kauffman, 2003). Disturbance to influential factors, like vegetation regrowth and soil hydrology, may reduce buffering effectiveness and slow recovery times (Donato et al., 2006; Morgan et al., 2015; Wagenbrenner et al., 2015). Conversely, a strategy that allows sufficient time for riparian zone recovery should provide access to more timber as buffer width requirements decrease. However, fire-killed timber declines in quality and market value over time according to species, diameter, and deterioration (Lowell & Cahill, 1996). Therefore, a trade-off must be evaluated between harvesting higher quality timber immediately after fire and waiting for riparian zone vegetation recovery to reduce buffer widths requirements that allows for more timber access. An effective post-fire BMP should include a temporal component for the proper balance of harvest buffer widths according to resource extraction and mitigation needs.

This study used a novel field simulation approach to evaluate the effectiveness of harvest buffer widths by using a variation of a concentrated flow experiment (Robichaud et al., 2010). Instead of measuring rill erosion rates with clean runoff (Robichaud et al., 2010; Wagenbrenner et al., 2015), sediment-laden runoff was applied and the reduction in sediment concentration and flux with distance was measured, in order to determine the effectiveness of post-fire harvest buffers. Our specific objectives were to compare the effects of post-fire soil burn severity and time since fire on the effectiveness of harvest buffer widths to mitigate sediment delivery.

2 | METHODS

2.1 | Site descriptions

Experiments were conducted at two locations, the 2015 North Star Fire and 2016 Cayuse Mountain Fire within the Okanogan Highlands of the Inland Northwest Region of the United States. Most of the plots were located within the perimeter of the 88,000 ha North Star Fire in Washington (Figure 1), the majority of which occurred on the Confederated Tribes of the Colville Reservation (post-fire report available at: https://forest.moscowfsl.wsu.edu/BAERTOOLS/baer-db/2500-8/2500-8_Northstar%20Fire_Okanogan.pdf; Accessed 16 January 2020). The remaining plots were located within the perimeter of the Cayuse Mountain Fire, on the Spokane Indian Reservation (Burned Area Report available from Spokane Tribe of Indians, Welpinit, WA).

The North Star Fire started on 13 August 2015 and burned in a temperate forest with dry open stands of predominantly Douglas-fir (*Pseudotsuga menziesii*) mixed with ponderosa pine (*Pinus ponderosa*) (Clausnitzer & Zamora, 1987; Williams, Kelley, Smith, & Lillybridge, 1995). Douglas-fir stands had an understory of bluebunch wheatgrass (*Agropyron spicatum*), Idaho fescue (*Festuca idahoensis*), or ninebark (*Physocarpus malvaceus*) at lower elevations, and pinegrass (*Calamagrostis rubenscens*) and mountain snowberry (*Symphoricarpos oreophilus*) at higher elevations, dependent upon microclimate (Clausnitzer & Zamora, 1987; Williams et al., 1995). The elevation of the fire ranged from approximately 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; however, most fires have been of low intensity (Williams et al., 1995).

Soils at the North Star Fire 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, Natural Resources Conservation Service, US Department of Agriculture, 2016). This soil is well-drained, and typically 50–100 cm in depth to a dense layer (Soil Survey Staff, Natural Resources Conservation Service, US Department of Agriculture, 2016). The A horizon is very thin. Clay content is 10%, sand is 21%, organic matter is 3% and hydraulic conductivity is 32 mm/hr (Soil Survey Staff, Natural Resources Conservation Service, US Department of Agriculture, 2017). This soil is naturally resistant to erosion, except when disturbed by compaction with heavy machinery (Williams et al., 1995).

The Cayuse Mountain Fire started on 23 August 2016 and burned more than 7,000 ha (Figure 1). Experimental sites at the Cayuse Mountain Fire were 200 m lower in elevation than those at the North Star Fire. Forest structure and composition at the Cayuse Mountain Fire resemble those of lower elevation sites at the North Star Fire, with dry open stands of Douglas-fir mixed with ponderosa pine. This suggests a similar fire history as found at the North Star Fire. Soils at the Cayuse Mountain Fire sites belong to the Dragoon Series, which is a mollisol derived from volcanic ash over granite, gneiss or schist, described as fine loamy, isotic, mesic *Vitrandic Argixerolls* (Soil Survey Staff, Natural Resources Conservation Service, US Department of Agriculture, 2016). Clay content is 13%, sand is 31%, organic matter is 3% with hydraulic conductivity reported as the same as the Nevine Series at 32 mm/hr (Soil Survey Staff, Natural Resources Conservation Service, US Department of Agriculture, 2017).

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, WA report mean annual precipitation of

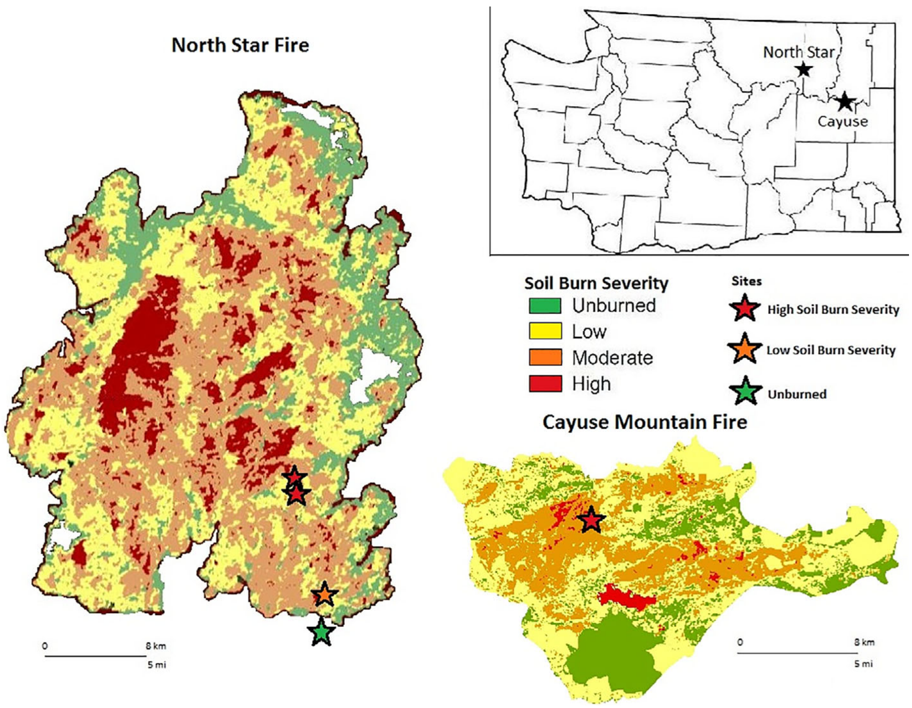

FIGURE 1 Soil burn severity map of the 2015 North Star Fire and the 2016 Cayuse Mountain Fire, WA and location of sampling points. Inset map of fire locations in Washington State

430 mm, mean maximum temperature of 13.4°C and mean minimum temperature of −0.1°C, averaged from 1981 to 2010 (Western Regional Climate Center, 2017). The Cayuse Mountain region is a slightly warmer climate area, with 530 mm of mean annual precipitation, mean maximum temperature of 14.7°C and mean minimum temperatures of 2.1°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. Site conditions are influenced by orographic effects.

2.2 | Site selection

After assessing the burned areas, the study sites were selected based upon consistent soil burn severity (Shakesby & Doerr, 2006), uniformity of hillslopes, proximity to stream channel, and road access. The soil burn severity map was produced by the Bureau of Indian Affairs Burned Area Emergency Response team (Figure 1). High soil burn severity is characterized by complete consumption of the forest floor and fine roots, altered soil structure, and water repellent soil conditions; whereas low soil burn severity has forest floor partially intact, no change in soil structure, with little to no water repellency (Parsons et al., 2010). At the North Star Fire, two high and one low soil burn severity locations were selected in addition to an unburned control site. The two high soil burn severity locations each had two rill experiments approximately 30 m apart on the same hillslope, for a total of four high severity rill experiments. At the low soil burn severity site all four rill experiments were conducted on the same hillslope; each rill experiment was about 30 m apart. Similarly, at the unburned site, located just outside the fire boundary, four rill experiments were conducted on the same hillslope about 20 m apart. A single high soil burn severity location was used for the four rill experiments which were about 30 m apart within the Cayuse Mountain Fire. All locations of the rill experiment were semi-permanently located for repeatability; in subsequent years the experiments occurred in the same locations.

All rill experiments occurred on hillslopes ranging from 20 to 40% as previous research has shown no slope-sediment concentration effect for slopes in this range (Robichaud, Lewis, Wagenbrenner, Ashmun, & Brown, 2013). Hillslope sites were chosen with similar physical characteristics (e.g., soil type, under- and over-story vegetation) across the different burn severity classes. In addition, all sites were undisturbed by any known land management prior to fire, post-fire, or scheduled for additional disturbance during the 2-year duration of the study.

To analyse the effect of time since fire, the Cayuse Mountain Fire sites were evaluated 2-month post-fire (October 2016) when the soils were bare due to the complete forest floor consumption, and 10-month post-fire (July 2017). The North Star Fire sites were evaluated 10-month post-fire (July 2016) and 22-month post-fire (July 2017). Because of the site similarities and the close physical distance between the two fires, study sites were largely analysed together by burn severity and time since fire rather than by fire site.

2.3 | Experimental design

The effectiveness of a forest buffer was quantified on each slope by measuring runoff travel distance and downhill change in sediment concentration in rills generated by a novel method of applying a sediment-laden concentrated flow to each hillslope (Polyakov & Nearing, 2003; Figure 2). A calibrated flow meter and a sediment feed hopper, which added a known amount of sediment to the clear water

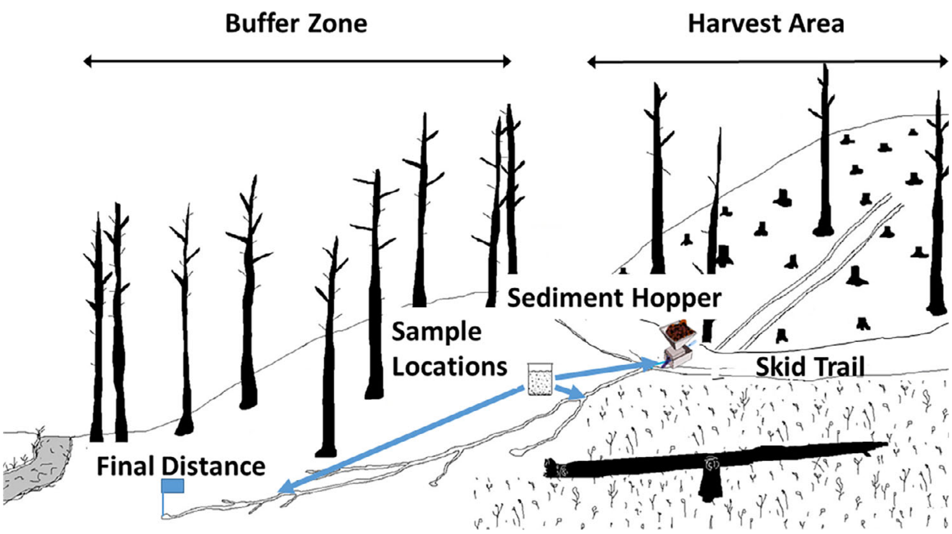

FIGURE 2 A sketch of a stream buffer experiment with rill flow entering from the above buffer area. Rills were measured for their travel length, and sediment concentration at the two sampling locations along the rill

FIGURE 3 Sediment being added to the sediment hopper which was calibrated to add 25 g/L of sediment to the three flow rates

flow, were used to simulate natural sediment-laden rills using three increasing flow rates (Figure 3). Flow is released through an energy dissipater onto the ground surface. Flow measurements and water samples were taken along each rill to quantify infiltration and sediment concentrations prior to entry into a stream channel at the base of the hillslope. The effects of burn severity and time since fire were also evaluated.

The basic experimental design was adapted from previous rill experiments after wildfires; however, changes were made to accommodate sediment-laden flow and greater runoff generated from skid trails and landings (Robichaud et al., 2010; Wagenbrenner et al., 2015). Each rill experiment was 40 min in duration with changing flow rates every 10 min. For the first 10 min, a clean water flow of 50 L/min (double filtered to 100 μm) was applied. The second 10 min included the addition of 25 g/L of sediment through the sediment feed hopper (Figure 3). The flow was then increased to 100 and 150 L/min for the last two intervals while maintaining the 25 g/L sediment concentration by adding twice as much sediment to the 100 L/min flow and three times as much sediment to the 150 L/min flow. The targeted sediment concentration (25 g/L) was based on typical sediment yields from disturbed forest soils (e.g., skid trails) from previous experiments (Robichaud et al., 2010). The three flow rates were chosen to represent typical rill flows from a wide range of upslope disturbed contributing areas (skid trails, up to 5 m wide by 20 m length and log landing areas, up to 0.25 ha) and storm durations for the Pacific Northwest. These disturbed upslope areas were similar to observations by Olsen et al. (2020) after salvage operations on the 2013 Rim Fire in California. Rainfall events of 1- to 5-year return intervals were used with the contributing areas to determine runoff volumes (flow rates). Sediment added to the rill flows consisted of burned soil (0–2 cm depth) collected from the North Star Fire area which were dried and sieved through a 6-mm sieve.

Runoff samples were taken in the rills during the onset of flow and during steady-state flow conditions for each flow interval similar to Robichaud et al. (2010). Three sample locations along each rill were identified for each of these flow intervals: the top of the rill at the outlet of the flow device to verify input sediment concentration, the midpoint (2–44 m), and near the endpoint (30–100 m) where sufficient flow could reasonably be sampled. The exact locations of the two downslope sample points were determined and adjusted in real time in response to the rill progression. Three runoff samples were taken at both downhill locations during each 10-min interval at the approximate beginning (1–2 min), middle (4–6 min), and end of the interval (8–10 min). Only one runoff sample was taken at the top of each rill during every flow interval for initial sediment concentration values. In total, 28 runoff samples were taken for each rill: one top sample for four flow rates, three middle samples for four flow rates, and three bottom samples for four flow rates. The entire flow was captured for each sample, so the time required to fill each bottle was used to estimate the runoff flow rates at each location. A flexible aluminium sheet with a slight conical shape was pressed firmly on the ground surface forcing all the runoff over the sheet which funnelled the runoff into timed-sample bottles (Figure 4).

The distance from the top of the rill to each runoff sample location was recorded. If rills divided into multiple flow paths or sub-rills, the sub-rill with the largest discharge was selected for sampling and flow width and depth measurements. Rills dividing into multiple paths occurred in about half of the rills, yet the flow volume in the sub-rills was often small or negligible. The smaller sub-rills were measured for flow width and depth. This resulted in a measured flow volume for the entire rill, and a discharge estimate for the largest of the sub-rills.

Runoff samples were later processed for water volume and sediment mass to calculate changes in sediment concentration. Overall mass of sediment in the sample was determined by oven-drying at 105°C for 24 hrs (Gardner, 1986).

2.4 | Other measurements

Additional measurements were taken to characterize the site and soil conditions for each hillslope (Bone, 2017). The Water Drop Penetration Time (WDPT) test was taken at three locations along each rill: upper, middle and lower. At each location, eight water drops were placed on the soil surface at 1 and 3 cm soil depths, and the time for each drop to infiltrate was recorded (DeBano, 1981; Robichaud, Lewis, & Ashmun, 2008). The median WDPT value from the eight drops was calculated for each soil depth. Surface soil moisture samples (0–2 cm) were taken prior to the rill experiment and were all below 8% moisture except the high severity Cayuse Mountain at 2-month post-fire which had a mean of 40% soil moisture due to a rain event the previous day (Bone, 2017).

Ground cover was measured adjacent to the top, middle and bottom sections of each rill (three repetitions). 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 and gravel, litter included woody debris and litter, vegetation included vegetation and moss, and other included rock and tree. Rill ground cover was calculated as the percentage of occurrence among sample points (Robichaud & Brown, 2002).

FIGURE 4 The rill flow being collected with a flexible aluminium sheet in the main rill about 15 m from the inlet at Cayuse Mountain site 2-month post-fire

Infiltration of the runoff was calculated by first calculating the wetted perimeter using the measured rill widths at sample locations, and the total rill length as the infiltration area. The total volume of water applied for the 40-min run time (3,500 L, 3.5 m^3) at a maximum flow rate of 150 L/min. This maximum flow rate was divided by the total wetted area of the main rill and sub-rills to calculate an infiltration rate (e.g., Cayuse Mountain 2-month post-fire, (0.150 m^3/min × 60 min/hr)/(43 m^2) × (1,000 mm/m) = 209 mm/hr; mean values are provided).

Sediment dropout rates (g L^{-1} m^{-1}), or change in sediment concentration per length, uses the slope of the linear model of sediment concentration and distance calculated for each flow rate. This was used to calculate a theoretical travel distance that all the sediment would be removed from the flow (e.g., 25 g/L / 1.29 g L^{-1} m^{-1} = 19.4 m for unburned conditions).

2.5 | Statistical methods

Statistical analysis was performed using R and SAS statistical software (R Core Team, 2016; SAS Institute, Cary, NC). Model diagnostics included testing residuals for normal distribution, plotting residuals versus predicted values, plotting residuals versus order, and quantile–quantile plots of residuals versus predicted values (Winters, 2013) to ensure model assumptions were met (details in Bone, 2017). Rill length and sediment concentration were log transformed to meet model requirements. Covariates were analysed for collinearity using a variance inflation factor method using the regression package *car* in R (Fox & Weisberg, 2011). Mixed effects models from the *lme4* package in R were used to compare means and variances of sediment concentration between soil burn severity classes (Bates, Maechler, & Bolker, 2012). Soil burn severity classes were: a) unburned, b) low soil burn severity, and c) high soil burn severity, time since fire was the number of months post-fire, and random variables were the rill plots and fire sites. Significance was determined with a likelihood ratio test at $p < .05$ (Winters, 2013). A general linear model with a contrast statement in SAS was used to evaluate the difference in sediment dropout rate (where dropout rate is equivalent to the slope of the linear model with sediment concentration the dependent variable and distance as the independent variable) between burn severity-time since fire groups. A linear mixed-effects model (Littell, Milliken, Stroup, Wolfinger, & Schabengerger, 2006) was also developed in SAS, using the post-fire month-burn severity group, flow rate, and their interaction as fixed effects, and rill plot nested in fire site as a random effect. The dependent variable was flow distance, which was considered significantly different between post-fire month-burn severity groups at a $p < .05$. Rill length was analysed as the rill's maximum travel distance during each flow rate. There were many significant differences between post-fire month-burn severity groups across the different flow rates; for ease of interpretation, only the model results of all flow rates together will be presented in the figure.

For water repellency analysis, another linear mixed-effects model was run in SAS using the site, post-fire year, soil burn severity, and depth, and the potential interactions between each as fixed effects and the fire site as a random effect. Least significant differences were used to compare differences in least squares means of median WDPT values among the interactions between site, post-fire years, soil burn severity and depth on all plots. All statistical results were considered significant at $p < .05$.

Another linear mixed-effects model used the post-fire month-burn severity group as fixed effects and fire site as a random effect. The dependent variables were total wetted perimeter, total wetted area and infiltration rate which were considered significantly different between post-fire month-burn severity groups at a $p < .05$.

3 | RESULTS

3.1 | Rill length

Soil burn severity and post-fire time significantly influenced flow distance (i.e., rill length; Figure 5). With all flow rates analysed together (model result in Figure 5), the mean rill length at 2-month post-fire (67 m), was significantly greater than all other soil burn severity and post-fire time periods. The mean rill length at high soil burn severity sites 10-month post-fire (22 m) was still significantly greater than the high soil burn severity sites at 22-month post-fire (12 m) and the low soil burn severity at either 10-month (8 m) or 22-month post-fire (6 m). By 22-month post-fire, the low burn severity mean rill lengths were statistically similar to unburned rill lengths.

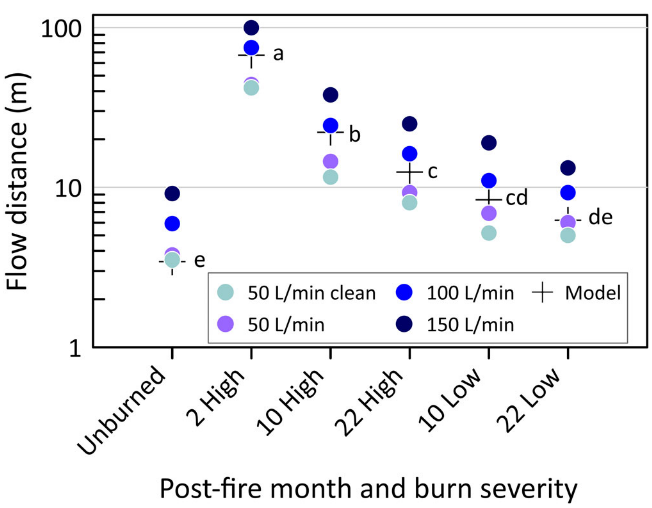

FIGURE 5 Means and ranges of flow distance (m) by post-fire time (2-, 10-, and 22-months), soil burn severity, and flow rate (50, 100, and 150 L/min). The model results (+) of all flow rates analysed together are also shown. Different letters indicate significant differences in flow distances at α = .05

As expected, rill length increased by 18 to 42% in response to increased flow rates for all soil burn severity conditions. Reduction in rill length over time was most pronounced on the high soil burn severity conditions.

3.2 | Sediment concentration

Sediment concentration of the rills significantly decreased with soil burn severity and time post-fire (Tables 1 and 2). Over an 8-month period mean sediment concentration from the high soil burn severity site at Cayuse Mountain Fire decreased by 63%, reducing from 15 to 5.5 g/L at the 100 L/min flow (Table 1). Similarly, at the North Star fire over a 12-month period spanning from 10- to 22-month post-fire, the mean sediment concentration from a high soil burn severity treatment declined by 80% dropping from 13 to 2.4 g/L.

The change in sediment concentration or sediment dropout rate with rill length also varied with soil burn severity and months post-fire (Table 2). The mean change in sediment concentration with distance down the rill was +0.05 g L^{-1} m^{-1} (e.g., scouring) for high soil burn severity plots 2-month post-fire and was less than zero (e.g., depositing sediment) for all other conditions and times. This indicates that immediately after the fire, the rills were still eroding, increasing sediment concentration over the rill length. While all other conditions dropout rates varied from 0.3 to 1.15 g L^{-1} m^{-1} for burned conditions and 1.29 g L^{-1} m^{-1} for unburned conditions. Dropout values indicate the rate at which sediment is being deposited along the rill and the sediment concentration in the flow is decreasing. Dropout rates from high soil burn severity hillslopes 2-month post-fire were significantly different than all other cases (Table 2). Interestingly, the dropout rate from the low soil burn severity at 10 months (0.73 g L^{-1} m^{-1}) was significantly less than low burn severity at 22-month post-fire (1.15 g L^{-1} m^{-1}) which was not statistically different than the unburned controls (1.29 g L^{-1} m^{-1}). Thus, the decrease in sediment concentration or dropout rate varied by soil burn severity and time since fire.

3.3 | Other measurements

Ground cover varied significantly with both soil burn severity and time since fire (Figures 6 and 7). At the high soil burn severity site bare soil cover declined from 75% 2-month post-fire to 50% 10-month post-fire, to only 15% 22-month post-fire in response to vegetative regrowth. Vegetation recovery varied by soil burn severity and can be seen in site photographs with grasses and forbs being the most prevalent vegetation (Figure 6b Cayuse Mountain 10-month post-fire). Both the unburned and low soil burn severity conditions had little bare soil (Figures 5 and 6d, f, g, h). Rill length was significantly correlated with ground cover (compare Figure 5 to Figures 6 and 7).

We measured strong water repellency on high soil burn severity rills 2-month post-fire at 1 cm soil depth (245 s) even though soil moisture was high (40%); for all other conditions, soil moistures were

TABLE 1 Means and ranges (in parentheses) of sediment concentration (g/L) using mid-point and end point samples by time since fire and flow rate. Different letters across a row indicate significant differences in soil burn severity treatment at α = .05

Time since fire (months), location and flow rate	Unburned	Low soil burn severity	High soil burn severity
2 months (Cayuse Mountain)			
50 L/min clean			2.6 (0.1–14)
50 L/min	—	—	8.5 (2.4–27)
100 L/min	—	—	15 (3.0–36)
150 L/min	—	—	19 (7.4–41)
10 months (Cayuse Mountain)			
50 L/min clean			1.3 (0–3.6)
50 L/min	—	—	3.2 (1.2–7.5)
100 L/min	—	—	5.5 (2.9–16)
150 L/min	—	—	6.9 (3.0–10)
10 months (North Star)			
50 L/min clean	0.1 (0.0–0.7) a	0.3 (0.1–1.2) b	2.8 (0–13) c
50 L/min	1.6 (0.1–4.2) a	2.5 (0.6–5.2) b	5.0 (1.1–30) c
100 L/min	5.6 (0.3–26) a	8.2 (2.3–30) a	13 (0.4–27) a
150 L/min	15 (0.7–27) b	7.0 (1.0–21) a	14 (4.4–38) b
22 months (North Star)			
50 L/min clean	0.6 (0.0–3.1) a	0.2 (0.0–0.7) a	0.7 (0–3.1) a
50 L/min	5.8 (1.8–19) a	3.3 (0.2–4.7) b	2.2 (0.5–6.9) b
100 L/min	10.4 (2.5–33) a	4.1 (1.2–12) b	2.4 (0.7–6.6) b
150 L/min	17.0 (0.7–33) a	5.2 (1.2–15) b	3.0 (1.2–15) b

TABLE 2 Statistical *p*-values of the slope of the line indicates the rate at which the sediment concentration changes along the length of the rill. Sediment dropout rate (g L^{-1} m^{-1}) is the rate at which sediment is being deposited by distance down the rill. The plus (+) sign indicates the rill is scouring and sediment concentration increased with distance down the rill. Bold *p*-values indicate significantly different dropout rates between treatment groups: 2-, 10- or 22- months post-fire; high or low soil burn severity with both fire locations analysed together

Time since fire (months) and soil burn severity	Slope comparison groups p-values					
	Unburned	2-month High	10-month High	22-month High	10-month Low	22-month Low
Sediment dropout rate (g L^{-1} m^{-1})	1.29	+ 0.05	0.3	0.49	0.73	1.15
2 High	**<0.0001**		**0.0002**	**0.0038**		
10 High	**0.0017**			0.34	**0.016**	
22 High	**0.022**					**0.0065**
10 Low	**0.033**					0.11
22 Low	0.61					

below 8% (Bone, 2017) (Table 3). Ten months post-fire, we measured moderate water repellency at the North Star Fire at 1 and 3 cm depths (95 and 73 s, respectively). Low water repellency was found on other plots within the North Star Fire at 10- and 22-month post-fire and at the Cayuse Fire at 1 cm depth 10-month post-fire (23 s) (Table 3) which contributed to a rapid recovery in soil wettability.

Water repellency was mostly non-existent on the North Star Fire after 22-months regardless of soil burn severity. While water repellency was measured at the soil surface after both fires in both years, WDPT averaged < 5 s on all low and high soil burn severity plots and was not included in the table for brevity. The moderate and strong water repellency we measured likely reduced infiltration and contributed to increased runoff and/or sediment yield.

Flow measurements using the entire rill length were used to calculate infiltration rate for each condition (Table 4). Total wetted perimeter of the high soil burn severity plots was significantly greater 2-month post-fire (0.9 m) than unburned plots (0.4–0.6 m). The mean total wetted area was significantly greater 2-month post-fire (43 m^2) and consistently decreased with time regardless of soil burn severity. The mean infiltration rate increased from 230 mm/hr 2-month post-fire to 540–600 mm/hr 10-month post-fire for high soil burn severity sites, although this increase was not statistically significant. The

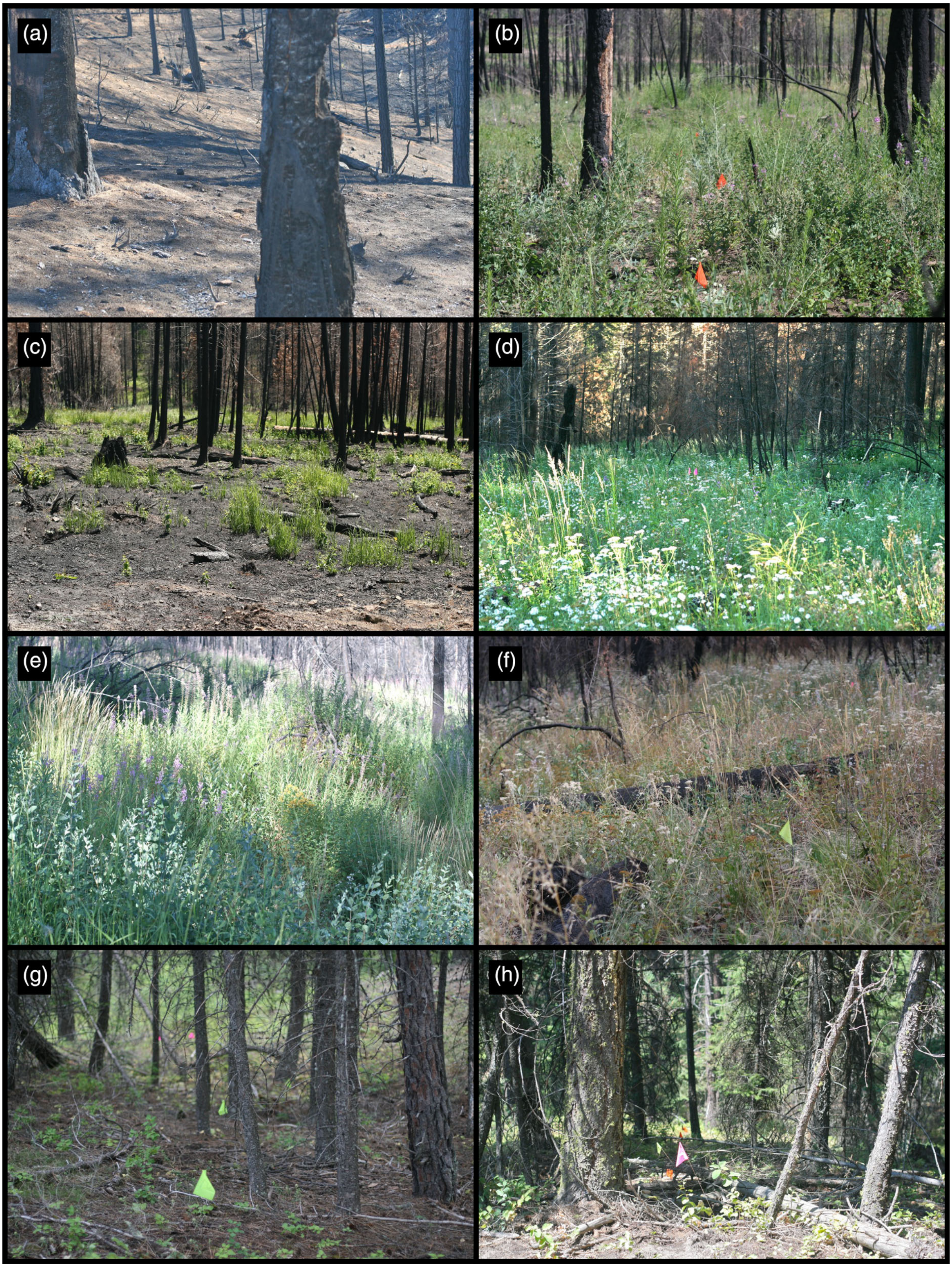

FIGURE 6 Site images. (a) Cayuse Mountain Fire high soil burn severity 2-month post-fire; (b) and high soil burn severity 10-month post-fire. (c) North Star Fire high soil burn severity 10-month post-fire; (d) North Star low soil burn severity 10-month post-fire; (e) North Star high soil burn severity 22-month post-fire; (f) North Star low soil burn severity 22-month post-fire; (g) North Star unburned site 1; (h) North Star unburned site 2

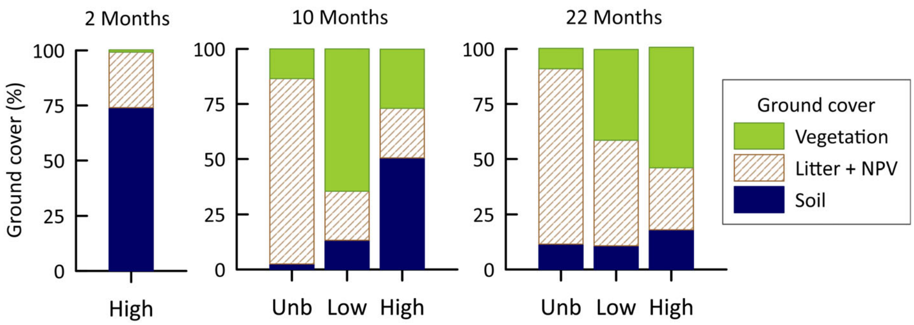

FIGURE 7 Vegetation, litter and other non-photosynthetic vegetation (NPV) which is brown or senesced, and soil ground cover by time post-fire and burn severity class (Unb is unburned) for both sites

TABLE 3 Median water drop penetration time (WDPT) at the Cayuse Mountain and North Star Fires by soil burn severity and time since fire. From the mixed model, significant differences in WDPT values are indicated by different lower case letters in the right column at α = .05

Location	Time since fire (months)	Soil burn severity	Soil depth (cm)	Average median WDPT (s)
Cayuse	2	High	1	245 a
Mountain	2	High	3	1 d
	10	High	1	23 cd
	10	High	3	4 d
North Star	10	Low	1	20 d
	10	Low	3	30 cd
	10	High	1	95 b
	10	High	3	73 bc
	22	Low	1	1 d
	22	Low	3	8 d
	22	High	1	2 d
	22	High	3	1 d
	—[a]	Unburned	1	26 cd
	—[a]	Unburned	3	3 d
	—[b]	Unburned	1	2 d
	—[b]	Unburned	3	1 d

[a]Sampled in 2016.
[b]Sampled in 2017.

TABLE 4 Mean infiltration rate during the maximum flow rate of 150 L/min over the total rill area. The total wetted perimeter, area, and infiltration rate means are grouped by location, time since fire, and soil burn severity. Different lowercase letters in a column indicate statistical (p < .05) differences between location-severity-time groups

Location	Time since fire (months)	Soil burn severity	Total wetted perimeter (m)	Total wetted area (m^2)	Infiltration rate (mm/hr)
Cayuse Mountain	2	High	0.9 ab	43 a	230 d
	10	High	1.1 a	16 b	600 cd
North Star	10	Low	0.6 c	8.7 c	1,260 b
	10	High	0.7 bc	17 b	540 d
	22	Low	0.6 bc	7.3 cd	1,370 b
	22	High	0.7 bc	11 c	1,040 bc
	—[a]	Unburned	0.4 c	3.4 c	2,720 a
	—[b]	Unburned	0.6 c	4.0 c	2,690 a

[a]Sampled in 2016.
[b]Sampled in 2017.

10-month low soil burn severity infiltration rate (1,260 mm/hr) was similar to the low soil burn severity infiltration rate (1,370 mm/hr) 22-month post-fire. However, these values were significantly less and about half of the unburned infiltration rates (~2,700 mm/hr).

4 | DISCUSSION

4.1 | Rill length

This study evaluates the ability of a riparian buffer to infiltrate runoff and deposit sediment from upslope areas and reduce rill flow distance or rill lengths after wildfire. Rill lengths were significantly influenced by burn severity and time since fire. High soil burn severity conditions had significantly longer mean rill flow lengths than low soil burn or unburned controls implying managed buffer widths should increase with increased soil burn severity (Figure 5). Soil burn severity may affect rill length for a wide range of runoff events. Even after 10- and 22-month post-fire, there was still a significant difference in rill length (Figure 5). Thus, a stream buffer designed for unburned conditions will likely not be effective after burning regardless of the soil burn severity. Castelle et al. (1994) suggested that water resource values and degree of upland disturbances should be considered when determining buffer sizes and greater widths may be necessary for buffers in poor condition (e.g., high or low soil burn severity).

The lowest runoff rate used in this experiment, 50 L/min, is similar in magnitude to the highest runoff (21 L/min) observed from skid trails in salvage logged areas having a small contributing area (9 m^2) (Robichaud et al., 2010). A typical disturbed area of a skid trail contributing flow can range from 30 to 100 m^2, depending on water bar spacing and terrain. Log landing contributing areas can range from 80 to 140 m^2 (Robichaud et al., 2020). Thus, the flow rates used in this study provide realistic evaluation of the effectiveness of a buffer down slope from salvage logging operations (Table 4).

As the site recovered after the fire, buffer effectiveness also improved. Rill lengths significantly decreased for both low and high soil burn severity sites with time (Figure 5); however, despite the improvements in infiltration over time, each soil burn severity condition still had significantly greater rill length than the unburned sites even 22-month post-fire, regardless of flow rate.

Infiltration rate may represent the most important functional asset of stream buffers managed for erosion mitigation. Similar to the findings of Robichaud (2000) and Larson-Nash et al. (2018), results from this study clearly indicate orders of magnitude differences in infiltration characteristics between unburned, low burn severity and high burn severity hillslopes (Table 4). At unburned sites, 3,500 L of simulated runoff added to hillslopes over 40 min travelled a maximum distance of only 11 m and is similar to the findings of Burroughs and King (1989) in unburned forests who measured mean rill travel distance of 8 m (Figure 5). The unburned forest floor provides ample macropores and preferential flow paths resulting in very high infiltration rates (Beven & Germann, 1982). Using total wetted area measurements and the maximum flow rate, the calculated infiltration rates 2 months after the fire (230 mm/hr) were an order of magnitude less from unburned conditions (~2,700 mm/hr; Table 4). Foltz and Elliot (2001) measured infiltration rates up to 1,400 mm/hr in unburned forests using similar concentration flow methods. The infiltration rates 2-month post-fire (230 mm/hr) resulted in a maximum flow distance of 100 m (Table 4). Thus, infiltration rates were directly related to the effects of the wildfire, likely from fire-induced water repellency (mean WDPT 245 s) and other changes in soil structure such a reduce macropores, as well as low ground cover (25%) (Table 3 and Figure 7). Although during the 2-month post-fire study at the Cayuse Mountain Fire, rill lengths may have been influenced by a previous day's rain event, the data also show a strong water repellent layer at 1 cm. Widespread strong water repellency may indicate that the soil profile was not fully saturated, even though water repellency tends to weaken in saturated soils (Doerr, Shakesby, & MacDonald, 2009). The 1-cm water repellent layer limited infiltration, saturating the upper 1 cm of soil, leading to the 100 m rill distances 2 months after the fire. Mean soil moisture at all other sites at the time of the experiment were under 8% (Bone, 2017).

The inverse relationship between rill lengths and ground cover suggest that ground cover can be a useful indicator of buffer effectiveness. The amount of bare soil is usually correlated with soil burn severity and time since fire. Ground cover of any type reduced rill lengths in a similar study (Robichaud et al., 2010). The reduction in rill length across treatments indicates that vegetative regrowth greatly improves infiltration after wildfire, similar to findings of Cerdà and Doerr (2005). Forest floor or surface litter may have also decreased rill lengths by increasing surface storage and connectivity of macropores, reducing rill flow velocity, and allowing more time for infiltration (Lavee, Kutiel, Segev, & Benyamini, 1995; Megahan & Ketcheson, 1996; Pannkuk & Robichaud, 2003; Scott, Curran, Robichaud, & Wagenbrenner, 2009).

The temporal variability in buffer performance suggests the timing of salvage logging post-fire is important to minimize downstream impact. Allowing one growing season to pass before harvest disturbance may be an effective strategy to reduce sedimentation to downstream water bodies. However, Morgan et al. (2015) indicated that ground disturbance, like salvage logging, stalls vegetative recovery and sets it back by approximately 1 year, effectively increasing the overall recovery time.

4.2 | Sediment concentration

The novel design of adding sediment-laden runoff in a field study allows for direct measurement of effectiveness of the buffer or riparian widths by comparing sediment concentrations and sediment dropout rate. The change in sediment concentration with distance provides an innovative metric of buffer effectiveness (Table 2). The significant difference in the dropout rates with time since fire and burn severity was also a useful metric for distinguishing the changes in buffer effectiveness with time and burn severity. The effectiveness of a buffer can be gauged by its ability to reduce sediment

TABLE 5 Post wildfire stream buffer width recommendations for salvage logging. The adjustment factor is based on a 15-m stream buffer for logging activities often recommended for unburned forests (Washington State Department of Natural Resources, 2015)

Time since fire (post-fire year)[a]	Soil burn severity	Buffer width	Adjustment factor from 15-m buffer
2 months (0)	High	120 m (400 ft)	8×
10 months (1)	High	60 m (200 ft)	4×
22 months (2)	High	30 m (100 ft)	2×
2 months (0)	Low	60 m (200 ft)	4×
10 months (1)	Low	30 m (100 ft)	2×
22 months (2)	Low	15 m (50 ft)	1×

[a]Each year allows for one full growing season.

concentration in the flow with distance down the rill. Immediately after the fire in the high soil burn severity buffers, the sediment concentration increased with distance down the 100-m rill (0.05 g L^{-1} m^{-1}). The soil was easily eroded in 0.20-m wide rills (data not shown) having 74% bare soil and no vegetation (Figures 6 and 7). These results are similar to other researchers who have documented the importance of ground cover in the post-fire environment (Benavides-Solorio & MacDonald, 2005; Cerdà, 1998; Moody & Martin, 2009; Pannkuk & Robichaud, 2003). In contrast, 10-month post-fire, sediment concentration decreased with distance down the rill at a dropout rate of 0.3 g L^{-1} m^{-1} following vegetative recovery (Figure 6c, d).

After 10 months, the Cayuse Mountain Fire had lower sediment concentrations than the North Star Fire as result of more rapid vegetative recovery (Table 1). At that time ground cover in the high and low soil burn severity buffers was at 50% and 90%, respectively, which is relatively fast post-fire recovery (Dodson & Peterson, 2010). The low water repellency on the high soil burn severity sites at the Cayuse Mountain Fire just 10-month post-fire indicates a rapid recovery in soil wettability, and significantly higher infiltration rates (600 mm/hr) (Table 4).

As presented in the introduction, although the regulated unburned forest stream buffer widths for green timber harvest varies with jurisdiction, a 15-m stream buffer is the most widely accepted standard for non-fish bearing streams in the State of Washington. Our results suggest that this buffer distance is not adequate to allow infiltration of the runoff nor at reducing sediment concentration from an upstream disturbance typical of post-fire salvage logging operations.

This study suggests that a 15-m forested buffer is likely adequate for the unburned condition. Rill lengths with the flows used in this study were all less than 11 m in an unburned buffer. Based on a 1.29 g L^{-1} m^{-1} sediment dropout rate, adding 25 g/L sediment-laden runoff in unburned conditions would theoretically drop all sediment in 19 m (25 g/L / 1.29 g L^{-1} m^{-1} = 19.4 m; Table 2 and Figure 5). While in high soil burn severity conditions with increasing sediment concentration (+ 0.05 g L^{-1} m^{-1}) 2-month post-fire the buffer would continue to scour, and rill lengths may be as much as 100 m. Even 10-month post-fire with an initial sediment concentration of 25 g/L and a dropout rate of 0.3 g L^{-1} m^{-1}, the buffer would need to be 83 m to drop all the sediment, yet we observed rills only 24–35 m in length.

Overall, this study suggests that buffer widths should be temporarily increased based on soil burn severity (high and low) and time post-fire for salvage logging (Table 5). Based on both rill length and sediment dropout rate for salvage logging immediately after a fire, 120 m buffer (eight times the standard buffer width of 15 m) should be maintained. If the salvage logging occurs 1-year post-fire then buffer width can be reduced to 60 m as vegetation regrowth stabilizes the area. While in low soil burn severity areas immediately after a fire a 60-m buffer width (four times an unburned buffer width) should be maintained but this reduces to 30 m 1-year post-fire (Table 5).

These recommendations are most applicable for volcanic ash-derived soils in mixed conifer ecosystems with good regrowth potential for moderate salvage logging disturbances with skid trails and landings. Additional research may be warranted to evaluate post-fire buffer widths for different soils and site conditions. However, the derived adjustment factor may be appropriate to adjust BMPs buffer widths in similar settings.

5 | CONCLUSIONS

Post-fire forest management decisions require information that is relevant for wildfire-affected landscapes. Stream buffer effectiveness on high and low burn severity areas was quantified using a simulated sediment-laden rill experiment by measuring rill length and sediment concentration along the rill. Rill length and sediment concentration increased in both high and low soil burn severity conditions, indicating that the ability of a buffer to infiltrate runoff and decrease sediment concentration from upslope disturbances is greatly reduced following wildfire. Standard stream buffer widths may be insufficient immediately after high soil burn severity fires, but effectiveness is likely to improve with time as vegetative growth occurs. Rill length and sediment concentration were significantly greater in these burned landscapes but declined after 2 years due to vegetation regrowth. Rill lengths were 100 m immediately (2 months) after the fire in high soil burn severity areas and significantly reduced to 25 m after 1 year of vegetation recovery. Sediment dropout rate also varied with soil burn severity and time post-fire with no sediment reduction immediately after the fire. Sediment dropout rate increased from 0.3 g L^{-1} m^{-1} 1-year post-fire to 0.49 g L^{-1} m^{-1} 2- years post-fire for high soil burn severity conditions. These data suggest that stream buffer widths on high soil burn severity sites may need to be eight times greater (120 m) than unburned forest conditions (15 m) immediately after wildfire. One year after the wildfire, buffer widths at high burn severity sites may need to be four times the unburned width (60 m). Low soil burn severity sites 1-year post-fire may need twice the unburned buffer widths (30 m). These findings suggest that soil burn severity, time since fire, and the associated vegetation recovery in stream buffer zones may be useful considerations for future stream buffer management decisions.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from 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 P.R. Robichaud with the permission of Confederated Tribes of the Colville Reservation.

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REFERENCES

Bates, D. M., Maechler, M., & Bolker, B. (2012). *Lme4: Linear mixed-effects models using S4 classes*. R Package Version 0.999999-0.

Benavides-Solorio, J. D., & Macdonald, L. H. (2001). Post-fire runoff and erosion from simulated rainfall on small plots, Colorado front range. *Hydrological Processes*, *15*, 2931–2952. https://doi.org/10.1002/hyp.383

Benavides-Solorio, J. D., & MacDonald, L. H. (2005). Measurement and prediction of post-fire erosion at the hillslope scale, Colorado front range. *International Journal of Wildland Fire*, *14*(5), 1–18. https://doi.org/10.1071/WF05042

Beven, K., & Germann, P. (1982). Macropores and water flow in soils. *Water Resources Research*, *18*(5), 1311–1325.

Bone, E. D. (2017). *Effects of burn severity on stream buffer management for post-fire hillslope erosion in the inland Northwest Mountain ranges, USA*. (master's thesis). Moscow, ID: University of Idaho. 75 pp.

Brown, T. C., Brown, D., & Binkley, D. (1993). Laws and programs for controlling nonpoint source pollution in forest areas. *Water Resources Research*, *29*(1), 1–13.

Burroughs, E. R., Jr., & King, J. G. (1989). Reduction of soil erosion on forest roads. (Gen. Tech. Rep. INT-264). Ogden, UT: US Department of Agriculture, Forest Service, Intermountain Research Station. 21 pp.

Castelle, A. J., Johnson, A. W., & Conolly, C. (1994). Wetland and stream buffer size requirements—A review. *Journal of Environmental Quality*, *23*, 878–882. https://doi.org/10.2134/jeq1994.00472425002300050004x

Cerdà, A. (1998). The influence of geomorphological position and vegetation cover on the erosional and hydrological processes on a Mediterranean hillslope. *Hydrological Processes*, *12*, 661–671.

Cerdà, A., & Doerr, S. H. (2005). Influence of vegetation recovery on soil hydrology and erodibility following fire: An eleven-year investigation. *International Journal of Wildland Fire*, *14*(4), 423–437. https://doi.org/10.1071/WF05044

Chambers, J. C., & Brown, R. W. (1983). *Methods for vegetation sampling and analysis on revegetated mined lands* (Gen. Tech. Rep. INT-GTR-151). Ogden, UT: US Department of Agriculture, Forest Service, Intermountain Research Station. 57 pp.

Clausnitzer, R. R., & Zamora, B. A. (1987). *Forest habitat types of the Colville Indian Reservation* (Agriculture Research Center Publication No. MISC0110). Pullman, WA: Washington State University.

Clinnick, P. F. (1985). Buffer strip management in forest operations: A review. *Australian Forestry*, *48*(1), 34–45. https://doi.org/10.1080/00049158.1985.10674421

DeBano, L. F. (1981). *Water repellant soils: A state-of-the-art* (Gen. Tech. Rep. PSW-GTR-46). Berkeley, CA: US Department of Agriculture, Forest Service, Pacific Southwest Research Station. 21 pp.

Dodson, E., & Peterson, D. (2010). Mulching effects on vegetation recovery following high severity wildfire in North-Central Washington state, USA. *Forest Ecology and Management*, *260*(10), 1816–1823. https://doi.org/10.1016/j.foreco.2010.08.026

Doerr, S. H., Shakesby, R. A., & MacDonald, L. H. (2009). Soil water repellency: A key factor in post-fire erosion. In A. Cerdà & P. R. Robichaud (Eds.), *Fire effects on soils and restoration strategies* (pp. 197–223). Enfield, NH: Science Publishers.

Doerr, S. H., Shakesby, R. A., & Walsh, R. P. D. (2000). Soil water repellency: Its causes, characteristics and hydro-geomorphological significance. *Earth Science Reviews*, *51*, 33–65. https://doi.org/10.1016/S0012-8252(00)0011-8

Donato, D. C., Fontaine, J. B., Campbell, J. L., Robinson, W. D., Kauffman, J. B., & Law, B. E. (2006). Post-wildfire logging hinders regeneration and increases fire risk. *Science*, *311*(5759), 352. https://doi.org/10.1126/science.1122855

Dwire, K., & Kauffman, J. (2003). Fire and riparian ecosystems in landscapes of the western USA. *Forest Ecology and Management*, *178*, 61–74. https://doi.org/10.1016/50378-1127(03)0053-7

Emelko, M. B., Silins, U., Bladon, K. D., & Stone, M. (2011). Implications of land disturbance on drinking water treatability in a changing climate: Demonstrating the need for “source water supply and protection” strategies. *Water Resources Research*, *45*, 461–472. https://doi.org/10.1016/j.watres.2010.08.051

Fernández, C., Vega, J. A., Fonturbel, T., Pérez-Gorostiaga, P., Jiménez, E., & Madrigal, J. (2007). Effects of wildfire, salvage logging and slash treatments on soil degradation. *Land Degradation and Development*, *18*, 591–607. https://doi.org/10.1002/ldr.797

Foltz, R. B., & Elliot, W. J. (2001). Infiltration characteristics of forest road filter windrows. In J. C. Ascough, II & D. C. Flanagan (Eds.), *Soil erosion for the 21st century: Proceedings of the international symposium*. St. Joseph, MI: American Society of Agricultural Engineers Pub. 701P0007. 3 pp.

Fox, J., & Weisberg, S. (2011). *An (R) companion to applied regression* (2nd ed.). Thousand Oaks, CA: Sage. https://doi.org/10.1071/WF07091

García-Orenes, F., Arcenegui, V., Chrenková, K., Mataix-Solera, J., Moltó, J., Jara-Navarro, A. B., & Torres, M. P. (2017). Effects of salvage logging on soil properties and vegetation recovery in a fire-affected Mediterranean forest: A two-year monitoring research. *Science of the Total Environment*, *586*, 1057–1065. https://doi.org/10.1016/j.scitotenv.2017.02.090

Gardner, W. H. (1986). Water content. In A. Klute (Ed.), *Methods of soil analysis: Part 1* (pp. 493–507). Madison, WI: American Society of Agronomy.

Halofsky, J., & Hibbs, D. (2008). Determinants of riparian fire severity in two Oregon fires, USA. *Canadian Journal of Forest Research*, *38*, 1959–1973. https://doi.org/10.1139/X08-048

Han, S., Han, H., Page-Dumroese, D. S., & Johnson, L. R. (2009). Soil compaction associated with cut-to-length and whole-tree harvesting of a coniferous forest. *Canadian Journal of Forest Research*, *39*, 976–989. https://doi.org/10.1139/X09-027

Haupt, H. F., & Kidd, W. J. (1965). Good logging practices reduce sedimentation. *Journal of Forestry*, *63*(9), 664–670.

Johnstone, J. F., & Kasischke, E. S. (2005). Stand-level effects of soil burn severity of postfire regeneration in a recently burned black spruce forest. *Canadian Journal for Restoration*, *35*, 2151–2163. https://doi.org/10.1139/X05-087

Jordán, A., Zavala, L., Mataix-Solera, J., & Doerr, S. H. (2013). Soil water repellency: Origin, assessment and geomorphological consequences. *Catena*, *108*, 1–5. https://doi.org/10.1016/j.catena.2013.05.005

Klock, G. O. (1975). Impact of five postfire salvage logging systems on soils and vegetation. *Journal of Soil and Water Conservation*, *30*(2), 78–81.

Kunze, M. D., & Stednick, J. D. (2006). Streamflow and suspended sediment yield following the 2000 bobcat fire, Colorado. *Hydrological Processes*, *20*, 1661–1681.

Lane, P. N. J., Croke, J. C., & Dignan, P. (2004). Runoff generation from logged and burnt convergent hillslopes: Rainfall simulation and modelling. *Hydrological Processes*, *18*, 879–892.

Larsen, I. J., MacDonald, L. H., Brown, E., Rough, D., Welsh, M. J., Pietraszek, J. H., … Schaffrath, K. (2009). Causes of post-fire runoff and erosion: Water repellency, cover or soil sealing? *Soil Science Society of America Journal*, *73*(4), 1393–1407. https://doi.org/10.2136/sssaj2007.0432

Larson-Nash, S. S., Robichaud, P. R., Pierson, F. B., Moffet, C. A., Williams, C. J., Spaeth, K. E., … Lewis, S. A. (2018). Recovery of small-scale infiltration and erosion after wildfires. *Journal of Hydrology and Hydromechanics*, *66*(3), 261–270. https://doi.org/10.1515/johh-2017-0056

Lavee, H., Kutiel, P., Segev, M., & Benyamini, Y. (1995). Effect of surface roughness on runoff and erosion in a Mediterranean ecosystem: The role of fire. *Geomorphology*, *11*, 227–234. https://doi.org/10.1016/0169-55X(94)00059-Z

Lee, P., Smyth, S., & Boutin, S. (2004). Quantitative review of riparian buffer width guidelines from Canada and the United States. *Journal of Environmental Management*, *70*, 165–180. https://doi.org/10.3390/rs4072112

Lentile, L. B., Smith, A. M. S., Hudak, A. T., Morgan, P., Bobbitt, M. J., Lewis, S. A., & Robichaud, P. R. (2009). Remote sensing for prediction of 1-year post-fire ecosystem condition. *International Journal of Wildland Fire*, *18*, 594–608. https://doi.org/10.1071/WF07091

Litschert, S. E., & MacDonald, L. H. (2009). Frequency and characteristics of sediment delivery pathways from forest harvest units to streams. *Forest Ecology and Management*, *259*, 143–150.

Littell, J. S., McKenzie, D., Peterson, D. L., & Westerling, A. L. (2009). Climate and wildfire area burned in western US ecoprovinces, 1916-2003. *Ecological Applications*, *19*, 1003–1021. https://doi.org/10.1890/07-1183.1

Littell, R. C., Milliken, G. A., Stroup, W. W., Wolfinger, R. D., & Schabengerger, O. (2006). *SAS for Mixed Models* (2nd ed.). Cary, NC: SAS Institute.

Lowell, E., & Cahill, J. (1996). Deterioration of fire-killed timber in southern Oregon and northern California. *Western Journal of Applied Forestry*, *11* (4), 125–131.

McIver, J. D., & McNeil, R. (2006). Soil disturbance and hill-slope sediment transport after logging of a severely burned site in northeastern Oregon. *Western Journal of Applied Forestry*, *21*(3), 123–133.

McIver, J. D., & Starr, L. (2001). A literature review on the environmental effects of postfire logging. *Western Journal of Applied Forestry*, *16*, 159–168.

Megahan, W. F., & Ketcheson, G. L. (1996). Predicting downslope travel of granitic sediments from forest roads in Idaho. *Water Resources Bulletin*, *32*(2), 371–382.

Megahan, W. F., & Kidd, W. J. (1972). Effects of logging and logging roads on erosion and sediment deposition from steep terrain. *Journal of Forestry*, *70*, 136–141.

Moody, J. A., Kinner, D., & Úbeda, X. (2009). Linking hydraulic properties of fire-affected soils to infiltration and water repellency. *Journal of Hydrology*, *379*, 291–303. https://doi.org/10.1016/j.jhydrol.2009.10.015

Moody, J. A., & Kinner, D. A. (2006). Spatial structures of stream and hillslope drainage networks following gully erosion after wildfire. *Earth Surface Processes and Landforms*, *31*(3), 319–337. https://doi.org/10.1002/esp.1246

Moody, J. A., & Martin, D. A. (2009). Synthesis of sediment yields after wildland fire in different rainfall regimes in the western United States. *International Journal of Wildland Fire*, *18*, 96–115. https://doi.org/10.1071/WF07162

Morgan, P., Moy, M., Droske, C. A., Lewis, S. A., Leigh, L. B., Robichaud, P. R., … Williams, C. J. (2015). Vegetation response to burn severity, native grass seeding, and salvage logging. *Fire Ecology*, *11*(2), 31–58. https://doi.org/10.4996/fireecology.1102031

Olsen, W. H., Wagenbrenner, J. W., & Robichaud, P. R. (2020). Factors affecting connectivity and sediment yields following wildfire and post-fire salvage logging in California's Sierra Nevada. *Hydrological Processes*, 1–16. https://doi.org/10.1002/hyp.13984.

Pannkuk, C. D., & Robichaud, P. R. (2003). Effectiveness of needle cast at reducing erosion after forest fires. *Water Resources Research*, *39*(12), 1333. https://doi.org/10.1029/2003WR002318

Parsons, A., Robichaud, P. R., Lewis, S., Napper, C., & Clark, J. (2010). *Field guide for mapping post-fire soil burn severity* (Gen. Tech. Rep. RMRS-GTR-243). Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. 49 pp.

Peterson, D. L., Agee, J. K., Aplet, G. H., Dykstra, D. P., Graham, R. T., Lehmkuhl, J. F., … Stuart, J. D. (2009). *Effects of timber harvest following wildfire in Western North America* (Gen. Tech. Rep. PNW-GTR-776). Portland, OR: US Department of Agriculture, Forest Service, Pacific Northwest Research Station. 51 pp.

Pettit, N., & Naiman, R. (2007). Fire in the riparian zone: Characteristics and ecological consequences. *Ecosystems*, *10*, 673–687. https://doi.org/10.1007/s10021-007-9048-5

Pietraszek, J. H. (2006). *Controls on post-fire erosion at the hillslope scale, Colorado Front Range*. (M.S. thesis). Fort Collins, CO: Colorado State University; 124 pp.

Polyakov, V. O., & Nearing, M. A. (2003). Sediment transport in rill flow under deposition and detachment conditions. *Catena*, *51*(1), 33–43. https://doi.org/10.1016/S0341-8162(02)00090-5

Prats, S. A., Malvar, M. C., Coelho, C. O. A., & Wagenbrenner, J. W. (2019). Hydrologic and erosion responses to compaction and added surface cover in post-fire logged areas: Isolating splash, interrill and rill erosion. *Journal of Hydrology*, *575*, 408–419. https://doi.org/10.1016/j.jhydrol.2019.05.038

R Core Team. (2016). *R: A language and environment for statistical computing*. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://www.R-project.org/

Rashin, E. B., Clishe, C. J., Loch, A. T., & Bell, J. M. (2006). Effectiveness of timber harvest practices for controlling sediment related water quality impacts. *Journal of the American Water Resources Association*, *42*(5), 1307–1327. https://doi.org/10.1111/j.1752-1688.2006.tb05303.x

Robichaud, P. R. (2000). Fire effects on infiltration rates after prescribed fire in Northern Rocky Mountain forests, USA. *Journal of Hydrology*, *231-232*, 220–229. https://doi.org/10.1016/S0022-1694(00)00196-7.

Robichaud, P. R., & Ashmun, L. E. (2013). Tools to aid post-wildfire assessment and erosion-mitigation treatment decisions. *International Journal of Wildland Fire*, *22*, 95–105. https://doi.org/10.1071/WF11162.

Robichaud, P. R., & Brown, R. E. (2002). *Silt fences: An economical technique for measuring hillslope soil erosion* (Gen. Tech. Rep. RMRS-GTR-94). Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. 24 pp.

Robichaud, P. R., Lewis, S. A., & Ashmun, L. E. (2008). *New procedure for sampling infiltration to assess post-fire soil water repellency* (Res. Note RMRS-RN-33). Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station. 14 pp.

Robichaud, P. R., Lewis, S. A., Brown, R. E., Bone, E. D., & Brooks, E. S. (2020). Evaluating post-wildfire logging-slash cover treatment to reduce hillslope erosion after salvage logging using ground measurements and remote sensing. *Hydrological Processes*, *34*, 4431–4445. https://doi.org/10.1002/hyp.13882

Robichaud, P. R., Lewis, S. A., Wagenbrenner, J. W., Ashmun, L. E., & Brown, R. E. (2013). Post-fire mulching for runoff and erosion mitigation. Part I. effectiveness at reducing hillslope erosion rates. *Catena*, *105*, 75–92. https://doi.org/10.1016/j.catena.2012.11.015

Robichaud, P. R., Wagenbrenner, J. W., & Brown, R. E. (2010). Rill erosion in natural and disturbed forests: 1. Measurements. *Water Resources Research*, *46*, W10506. https://doi.org/10.1029/2009wr008314

Scott, D. F., Curran, M. P., Robichaud, P. R., & Wagenbrenner, J. W. (2009). Soil erosion after forest fire. In A. Cerdà & P. R. Robichaud (Eds.), *Fire effects on soils and restoration strategies* (pp. 177–195). Enfield, NH: Science Publishers.

Sessions, J., Bettinger, P., Buckman, R., Newton, M., & Hamann, J. (2004). Hastening the return of complex forests following fire: The consequences of delay. *Journal of Forestry*, *102*, 38–45.

Shakesby, R. A., & Doerr, S. H. (2006). Wildfire as a hydrological and geomorphological agent. *Earth-Science Reviews*, *74*(3–4), 269–307. https://doi.org/10.1016/j.earscirev.2005.10.006

Silins, U., Stone, M., Emelko, M. B., & Bladon, K. D. (2009). Sediment production following severe wildfire and post-fire salvage logging in the Rocky Mountain headwaters of the Oldman River Basin, Alberta. *Catena*, *79*(3), 189–197. https://doi.org/10.1016/j.catena.2009.04.001

Smith, H. G., Sheridan, G. J., Lane, P. N. J., & Brena, L. J. (2011). Wildfire and salvage harvesting effects on runoff generation and sediment exports from radiata pine and eucalypt forest catchments, South-Eastern Australia. *Forest Ecology and Management*, *261*, 570–581. https://doi.org/10.1016/j.foreco.2010.11.009

Soil Survey Staff, Natural Resources Conservation Service, US Department of Agriculture. (2016). *Official soil series descriptions*. Retrieved from https://websoilsurvey.nrcs.usda.gov/app/WebSoilSurvey.aspx.

Soil Survey Staff, Natural Resources Conservation Service, US Department of Agriculture. (2017). Soil survey geographic (SSURGO) database for Okanogan County, Washington. Retrieved from https://data.nal.usda.gov/dataset/soil-survey-geographic-database-ssurgo.

Wagenbrenner, J. W., MacDonald, L. H., Coats, R. N., Robichaud, P. R., & Brown, R. E. (2015). Effects of post-fire salvage logging and a skid trail treatment on ground cover, soils, and sediment production in the interior western United States. *Forest Ecology and Management*, *335*, 176–193. https://doi.org/10.1016/j.foreco.2014.09.016

Wagenbrenner, J. W., Robichaud, P. R., & Brown, R. E. (2016). Rill erosion in burned and salvage logged western montane forests: Effects of logging equipment type, traffic level, slash treatment. *Journal of Hydrology*, *541*, 889–901. https://doi.org/10.1016/j.hydrol.2016.07.049

Wagenbrenner, J. W., Robichaud, P. R., & Elliot, W. J. (2010). Rill erosion in natural and disturbed forests: 2. Modeling approaches. *Water Resources Research*, *46*, W10507. https://doi.org/10.1029/2009wr008315

Ward, J. M., & Jackson, C. R. (2004). Sediment trapping within forestry streamside management zones: Georgia Piedmont, USA. *Journal of the American Water Resources Association*, *40*(6), 1421–1431.

Washington State Department of Natural Resources. (2015). *Forestry practices rules*. Retrieved from http://www.dnr.wa.gov/about/boards-and-councils/forest-practices-board/rules-and-guidelines/forest-practices-rules.

Westerling, A. L., Hidalgo, H. G., Cayan, D. R., & Swetnam, T. W. (2006). Warming and earlier spring increases western US forest wildfire activity. *Science*, *313*, 940–943. https://doi.org/10.1126/science.1128834

Western Regional Climate Center (WRCC). (2017). Cooperative climatological data summaries. Retrieved from http://wrcc.dri.edu/Climate/west_coop_summaries.php

Williams, C. K., Kelley, B. F., Smith, B. G., & Lillybridge, T. R. (1995). *Forested plant associations of the Colville National Forest* (Gen. Tech. Rep. PNW-GTR-360). Portland, OR: US Department of Agriculture, Forest Service, Pacific Northwest Research Station. 405 pp.

Winters, B. (2013). *Linear models and linear mixed effects models in R with linguistic applications* (arXiv:1308.5499). Retrieved from http://arxiv.org/pdf/1308.5499.pdf

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