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Factors affecting connectivity and sediment yields following wildfire and post-fire salvage logging in California's Sierra Nevada

Will H. Olsen[1] | **Joseph W. Wagenbrenner**[2] | **Peter R. Robichaud**[3]

[1]College of Forest Resources and Environmental Science, Michigan Technological University, Houghton, Michigan

[2]US Department of Agriculture, Forest Service, Pacific Southwest Research Station, Arcata, California

[3]US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Moscow, Idaho

Correspondence
Will H. Olsen, California Department of Forestry and Fire Protection, 6105 Airport Road, Redding, CA 96002.
Email: will.olsen@fire.ca.gov

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Abstract

Sediment delivery following post-fire logging is a concern relative to water quality. While studies have assessed the effect of post-fire logging on sediment yields at different spatial scales, none have explicitly identified sediment sources. Our goal was to quantify post-fire and post-salvage logging sediment yields and use rill patterns to identify sediment sources. We measured the extent and type of logging disturbance, length of rills per unit area or "rill density", ground cover, and sediment yields in nine logged and five control small catchments or "swales", 0.09 to 0.81 ha, for 5 years after the 2013 Rim Fire in California's Sierra Nevada. The logged swales had a mean ground disturbance of 31%. After the first wet season following logging, there was no difference in either mean rill density (0.071 and 0.088 m m^{-2}, respectively) or mean transformed, normalized sediment yields between the control and logged swales. Untransformed mean sediment yields across three sites ranged from 0.11–11.8 and 1.1–3.2 Mg ha^{-1} for the controls and salvage-logged swales, respectively. Rill density was strongly related to sediment yield and increased significantly with the amount of high-traffic skid trail disturbance in logged swales. Rill density was not significantly related to the amount of bare soil despite a significant relationship between sediment yields and bare soil. Rills usually initiated in bare soil and frequently connected high traffic skid trails to the drainage network after being diverted by waterbars. Rill connectivity and sediment yields decreased in control and logged swales where vegetation or other surface cover was high, suggesting this cover disconnected rills from the drainage network. Increasing ground cover on skid trails and between areas disturbed by post-fire logging and stream channels may reduce sediment yields as well as the hydrologic connectivity between hillslopes and the drainage network.

KEYWORDS
hillslope erosion, hydrologic connectivity, post-fire erosion, rill erosion, salvage logging, sediment yield, wildfire

1 | INTRODUCTION

In the California Sierra Nevada and southern Cascade mountains, the frequency of stand replacing wildfires in forested regions has increased since the 1980's (Miller et al., 2009). Increases in burned area in the western United States has been linked to decreased summer precipitation and winter snowpack, and increased temperatures and vapour pressure deficits (Abatzoglou & Williams, 2016; Holden et al., 2018). Crockett and Westerling (2018) indicated that wildfire size and severity have increased during droughts relative to non-drought periods in recent decades. More frequent and extensive severe wildfires in mountainous forested watersheds in the western United States pose a potential threat to aquatic systems (Bladon et al., 2014), while expected wildfire severity and weather events under climate change may compound current watershed-scale sediment issues (Gould et al., 2016; Murphy et al., 2018).

Post-fire salvage logging, hereafter post-fire logging, the harvesting of dead or dying trees, may be implemented for economic, fuel management, or forest restoration goals. Post-fire logging presents a special concern due to potential impacts to watersheds beyond those induced by fire alone, particularly through interactive or additive effects (Karr et al., 2004; Leverkus et al., 2018; D. L. Peterson et al., 2009). Ground based harvesting techniques are frequently employed in logging operations, using tracked feller bunchers to cut and bunch trees, and wheeled or tracked tractors or "skidders" to drag whole trees along a "skid trail" to a central area for processing. In a review of the effects of post-fire logging on erosion, the use of ground-based harvesting techniques was identified as the method most likely to have detrimental impacts (McIver & Starr, 2001).

Wildfires can reduce ground cover and increase surface runoff, especially following severe wildfires, in turn increasing soil erosion rates at the hillslope scale (Benavides-Solorio & MacDonald, 2001; Larsen et al., 2009; Robichaud et al., 2016), in particular when subjected to high intensity precipitation (Kampf et al., 2016; Moody & Martin, 2009; Wilson et al., 2018). Runoff rates at the small watershed scale may not increase after post-fire salvage logging relative to rates from high severity burns (Wagenbrenner et al., 2015), but reduced ground cover can increase sediment yields on post-fire salvage logged hillslopes due to soil disturbance from logging operations (Chase, 2006; Slesak et al., 2015; Stabenow et al., 2016). Ground based logging disturbance during post-fire logging operations can be extensive, and the spatial variation of mechanical disturbance within and among catchments can be very high (Chou et al., 1994), which further complicates our ability to predict post-logging sediment yield responses. In one study of ground based post-fire logging of burned areas in Oregon, harvest intensities and soil disturbance increased together (McIver & McNeil, 2006). However, extensive soil exposure from post-fire logging does not always correlate to increased sediment yields, as the magnitude and variability of the responses to the wildfire may overwhelm any post-fire logging signal (Chou et al., 1994; Silins et al., 2009; Wagenbrenner et al., 2015).

Best management practices and post-fire precipitation events may strongly control erosional responses (Fernández et al., 2007; McIver & McNeil, 2006). Post-fire logging may significantly increase woody material that provides surface cover (Robichaud, Lewis, et al., 2020; Wagenbrenner et al., 2015), intercepting precipitation and overland runoff and potentially mitigating effects from fire and mechanical disturbance due to logging (Cole et al., 2020; Poff, 1989; Prats et al., 2020). Equipment design and weight can reduce salvage logging impacts to soils (Lucas-Borja et al., 2019, 2020; Wagenbrenner et al., 2016). Other alternative practices used to harvest unburned trees such as over-the-snow skidding may also reduce impacts caused by post-fire salvage logging, but to our knowledge these have not yet been tested in post-fire salvage logging operations. Post-logging site preparation activities may also increase or decrease erosion, adding yet another factor to the effects of wildfire and post-fire logging (Cole et al., 2020; James & Krumland, 2018; Slesak et al., 2015).

Given the complex combinations of disturbances, management, and precipitation, studies at the watershed scale have not always been able to discern direct links between post-fire logging disturbance and sediment yields. Burned and logged pine plantations in Australia had persistent, elevated runoff and sediment yields compared to adjacent burned and unlogged eucalypt forests, mainly in response to intense summer storm events, but also log drag-lines (Smith et al., 2011, 2012). Skid trails in Alberta, Canada were important sediment transport pathways that were not observed in burned and unlogged watersheds (Silins et al., 2009). Despite these pathways, geochemical fingerprinting indicated that the burned areas produced elevated suspended sediment in the stream network relative to unburned watersheds irrespective of logging status (Stone et al., 2014). Similarly, turbidity measurements in northern California indicated that suspended sediment was elevated following wildfire below both burned and burned and logged watersheds, with increased trends in turbidity below some of the logged watersheds. However, results were also strongly associated with road densities, confounding attribution of cause (Lewis et al., 2018). Within the same watersheds, reductions in sediment yield were observed on hillslopes following post-fire logging (James & Krumland, 2018), illustrating the complexity of assessing post-fire logging impacts at different spatial scales.

Because of these complexities, understanding the effects of post-fire logging on soil erosion and impacts to water quality requires the ability to determine sediment sources, how these sources influence sediment yields, and how these sources differ between burned and subsequently logged areas. There is some evidence that the spatial layout of skid trails and their connectivity to the drainage network are important factors in determining the effect of post-fire logging at the catchment scale (Wagenbrenner et al., 2015).

Rills can be both a dominant sediment source and a pathway link from the hillslope to catchment scales (Moody et al., 2013; Pietraszek., 2006), and may be relatively persistent features on the landscape following wildfires (Moody & Kinner, 2006). Increased sediment flux from rills in high soil burn severity patches (Robichaud et al., 2010) as well as from burned soils that were further disturbed by logging equipment during logging have been documented in controlled experiments (Wagenbrenner et al., 2016). Determining the sediment sources and connectivity of disturbance to drainage networks in burned and burned and logged areas will improve our ability to understand, predict and mitigate erosion from wildfire and post-fire logging. Our objectives were to: (1) identify rill initiation locations on burned and burned and logged hillslopes; (2) assess the effects of post-fire logging on the connectivity of rills to the drainage network; and (3) assess the relationships between sediment yields, rill networks, site characteristics, and types of logging disturbance following wildfire.

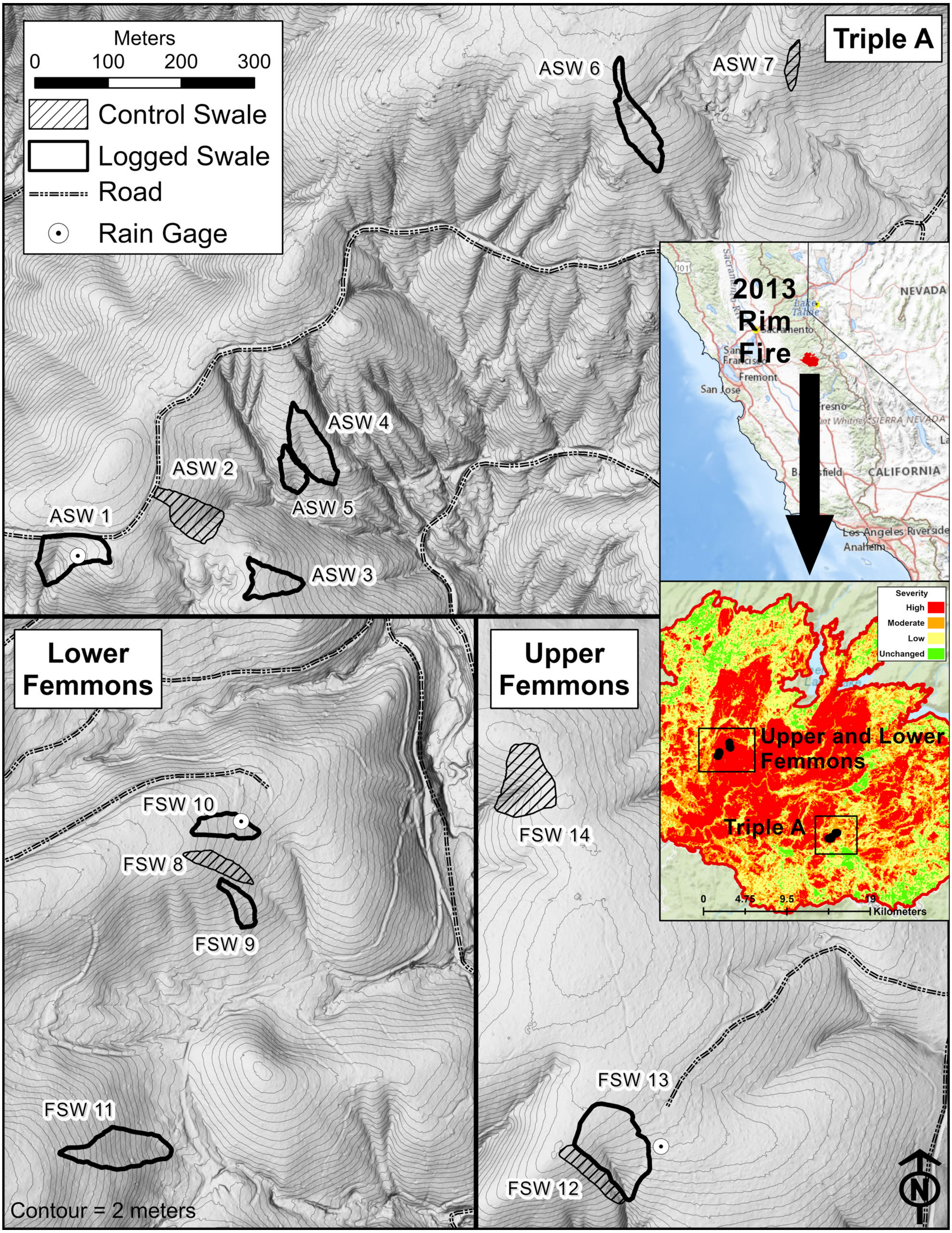

FIGURE 1 Location of Rim Fire (top inset), logging units (bottom inset), and study swales and rain gages in the three logging units. Contours and scale are the same for all three logging units

2 | METHODS

2.1 | Site description

The 2013 Rim Fire burned 104 131 ha, including 62 536 ha of the Stanislaus National Forest (USDA Forest Service, 2014) (Figure 1). Fourteen small catchments ("swales") of 0.09–0.81 ha were chosen across three planned post-fire logging harvest units, "Triple A" (ASW 1-ASW 7), "Lower Femmons" (FSW 8-FSW 11), and "Upper Femmons" (FSW 12-FSW 14), within the burned area within the Stanislaus National Forest in August 2014. Reference or control swales were randomly selected in each of the units and were excluded from logging operations. Swale outlets ranged from 1170–1447 m in elevation, and the mean slope of the swales ranged from 15 to 27%. The mean differenced normalized burn ratio (dNBR) ranged from 799 to 985, which we paired with field observations to categorize all the swales as high burn severity (Parsons et al., 2010) (Table 1).

The 1981–2010 mean annual precipitation for the nearby Cherry Valley Dam rain gage (37.975°N, 119.916°W, 1400 m elevation) averaged 1283 mm annually (Western Regional Climate Center, 2020). Soils are predominantly loams and gravelly loams weathered from granitic parent material, of the Holland, Sites, and Josephine series (National Resource Conservation Service, 2020). Pre-fire vegetation consisted of a mixture of ponderosa pine (*Pinus ponderosa*), incense cedar (*Calocedrus decurrens*), sugar pine (*Pinus lambertiana*), Douglas-fir (*Pseudotsuga menziesii*), and oak (*Quercus spp.*). Manzanita species (*Arctostaphylos spp.*) were also found interspersed within the study area, particularly in the Upper Femmons unit.

Five of the seven swales in Triple A were logged by the logging operators in November–December 2014; three of the four swales in Lower Femmons were logged in May 2015; and one of the three swales in Upper Femmons was logged in September 2015 (Figure 1, Table 1). The remaining swales in each unit were reserved as controls and were excluded from logging activities. Feller bunchers were used to cut and pile trees, except for larger trees, which were hand-felled. Cut trees were skidded to landings predominantly by rubber-tired skidders. Logging intensity generally decreased over time as the minimum diameter of trees selected by operators increased. Waterbars were constructed on high use skid trails (Table 2) by the operators to divert water off the skid trails as a best management practice, with spacing and frequency dependent upon skid trail gradient. Subsoiling (see Table 2) of skid trails was done in the swales ASW 3 and FSW 10 in September 2015 and May 2016, respectively. Subsoiling was the creation of furrows 0.4 m deep with the intent of decreasing soil compaction and increasing infiltration, and was completed by a tracked bulldozer with "winged" shanks along the path of the skid trails in the two swales. Sediment data were not collected in Lower Femmons for water year 2019, due to a prescribed fire in the unit that water year.

2.2 | Field methods

Precipitation was recorded for each unit using a tipping bucket rain gage (Rainwise, Inc., Trenton, Maine, USA) coupled to a HOBO data logger (Onset Corp., Bourne, Massachusetts, USA) (Figure 1). The annual precipitation was totaled by water year (October 1 to September 30). Precipitation was not adjusted to account for possible errors in magnitude and timing related to snow fall.

In the logged swales, patches of similar soil disturbance were surveyed using a mapping grade GPS unit (Trimble Geoexplorer,

TABLE 1 Swale characteristics

Unit	Number	Treatment	Logging date	Mean slope (%)	dNBR	Area (m^2)	Pre-salvage bare soil (%)	Logging disturbance (%)
Triple A	ASW 1	Logged	Nov–Dec 2014	26	814	4212	49	25
Triple A	ASW 2	Control	Nov–Dec 2014	20	799	3731	57	0
Triple A	ASW 3	Logged[a]	Nov–Dec 2014	15	875	2375	56	45
Triple A	ASW 4	Logged	Nov–Dec 2014	29	848	3911	43	41
Triple A	ASW 5	Logged	Nov–Dec 2014	24	871	1425	54	40
Triple A	ASW 6	Logged	Nov–Dec 2014	19	851	3705	65	29
Triple A	ASW 7	Control	Nov–Dec 2014	28	869	910	79	0
Lower Femmons	FSW 8	Control	May 2015	19	979	1782	58	0
Lower Femmons	FSW 9	Logged	May 2015	22	985	1418	57	20
Lower Femmons	FSW 10	Logged[b]	May 2015	15	943	2112	49	55
Lower Femmons	FSW 11	Logged	May 2015	27	877	5241	34	17
Upper Femmons	FSW 12	Control	September 2015	26	849	2305	82	0
Upper Femmons	FSW 13	Logged	September 2015	20	917	8103	70	9
Upper Femmons	FSW 14	Control	September 2015	20	865	5800	45	0

[a]Some of the high traffic skid trails in this swale were subsoiled in September 2015.
[b]Some of the high traffic skid trails in this swale were subsoiled in May 2016.

TABLE 2 Disturbance categories used in mapping disturbance in the logged swales

Type	Description
Feller buncher track	Traffic only by feller buncher; 1 or 2 feller buncher passes; minimal change in soil cover; no ruts
Low traffic skid trail	Branch skid route; ≤6 skidder passes; some residual soil cover; ruts less than 10 cm deep
High traffic skid trail	Main skid route; >6 skidder passes; no soil cover except for some residual slash; ruts sometimes exceeded 10 cm deep
Mixed traffic	Heterogeneous mixture of disturbed and undisturbed soil; no clear indication of equipment type, traffic level, or travel direction; wood or bark cover often present at high rates
Subsoiled	Furrows and ridges created by bulldozers with winged shanks (0.4 m deep) in high traffic skid trails; best management practice designed to decrease compaction and increase infiltration; no soil cover except for some residual slash
Waterbar	Best management practice created by bulldozer and designed to divert runoff from skid trails onto burned soil where no logging disturbance was present; no soil cover: deeper soil profiles were often exposed in adjacent upslope area

Sunnyvale, California, USA; or Juniper Archer 2, Logan, Utah, USA). Disturbed areas were classified into five categories, as described in Table 2: feller buncher tracks, low traffic skid trails, high traffic skid trails, mixed traffic, and subsoiled. Some feller buncher tracks were subsequently skidded over, and were mapped as low or high traffic skid trails. All remaining areas in the logged swales were classified as untrafficked.

Ground cover measurements were made along three transects that spanned the width of each swale at locations approximately 25%, 50% and 75% of the distance between the outlet and top of the swale. The transects varied in length, and we recorded the cover type at 0.25, 0.5 or 1.0 m intervals to achieve approximately 100 points per transect. At each point along each transect, we measured the cover that would first intercept precipitation from a height of about 1 m. The categories were vegetation or the understory canopy cover, litter, wood, rock, and bare soil. The bare soil category reflected bare ground exposed directly to precipitation or surface runoff and included gravel less than 75 mm in diameter because we observed gravel of this size class in the sediment yields. The cover data were averaged by cover class for each swale. The transects were installed and measured before logging in August 2014. In Lower and Upper Femmons, pre-logging cover was measured again in May 2015 except for FSW 11 where logging had commenced. Measurements were also made after logging in May 2015 in Triple A only, July 2015 in Lower Femmons only, May 2016, April–May 2018 and May 2019.

Rills and channels were mapped in each swale after the first post-logging winter wet period: May 2015 for Triple A and May 2016 for Lower and Upper Femmons. Rills in ASW 3 were also remapped in May 2016 following the first winter after subsoiling. Rills at least 3 cm wide, 1 cm deep, 1 m long and at least 1 m from the next closest rill were individually mapped. Channels were drainage features where hillslopes converged to a single flow path at least 20 cm wide. From the outlet of each swale, each channel and rill was mapped with a GPS unit to either the junction of the next upslope rill or its initiation point. All tributary rills were followed to their initiation point. Rills not connected to the outlet were mapped when their length exceeded 5 m. We classified and recorded the disturbance type at each rill initiation point as untrafficked or one of the five disturbance classes (Table 2), and we also identified where rills initiating in skid trails were diverted by waterbars to the outlet.

Silt fences, adopted from Robichaud et al. (2019), were installed at the outlet of each swale following logging in each unit. Sediment was periodically measured using a portable scale and removed during fence clean outs. Gravel and cobbles that were delivered into the silt fences were included in field masses unless there was no visual evidence of runoff for the period of accumulation. A sub-sample of sediment from each clean out was oven dried at 105°C for 24 h to determine the dry fraction, which was then used to calculate the dry sediment mass. Dry sediment mass was divided by the contributing area of the swale to calculate a sediment yield in Mg ha^{-1}. Sediment yields from periodic clean outs were totaled by water year to give annual yields.

2.3 | Data analysis

All GPS data were differentially corrected with sub-meter accuracy (Trimble GPS Pathfinder Office, Sunnyvale, California, USA; or Effigis EZSURV, Montreal, Quebec, Canada) and post-processed in ArcGIS 10.3 (Esri, Redlands, California, USA). Rill drainage density, or "rill density", was calculated in ArcGIS as the combined length of rill segments that were hydrologically connected to the outlet divided by the swale area (m m^{-2}). The channel density was calculated in the same way.

Statistical analysis was done using R software (R Core Team, 2020), using mixed effect models fit using the lme4 package (Bates et al., 2015) to compare treatments and determine effects. Significance of main effects were determined in the Anova function from the car package (Fox & Weisberg, 2019), using a Type-II F-test and Kenward–Roger approximation of degrees of freedom. The MuMIn package was used to calculate the coefficient of determination for "marginal" or fixed effects (Barton, 2019). Marginal means and trends were extracted and compared using the emmeans package in R (Lenth, 2019). In all models, the post-logging year was treated as a categorical variable. Model residuals were assessed for normality and homogeneity using quantile-quantile plots and by plotting the residuals against predicted values; all model assumptions were met. A significance level of 0.05 was used for all statistical tests.

To compare across different post-logging dates, cover in each category was first normalized by the 2014 pre-logging average for that unit. The estimated marginal means were compared across all units by treatment, with swale nested within logging unit as a random effect, for each period: 2014 and 2015 (pre-logging), and 2015, 2016, 2018 and 2019 (post-logging).

Rill densities were compared by treatment, using logging unit as a random effect. The effects of first-year post-logging ground cover, mean slope, and post-fire dNBR variables on rill density were assessed; an interaction with treatment was added to models to test for differences in responses between treatments for the variables. For the logged swales, rill density was assessed for relationships with logging disturbance variables, using logging unit as a random effect. The rill density in ASW 3 following subsoiling was excluded because of lack of replication.

For analysis, annual sediment yields were normalized by the mean first year control yield in each respective unit (Robichaud, Lewis, et al., 2013), in order to compare across units with different logging dates; as such, yields were divided by 4.28 Mg ha^{-1} in Triple A, 0.11 Mg ha^{-1} in Lower Femmons, and 11.8 Mg ha^{-1} in Upper Femmons. The normalized sediment yields were $\log_{10}$ transformed in order to normalize residuals. The transformed normalized yields were first compared by treatment and post-logging year, using swale nested within logging unit as a random effect. The relationship between the transformed normalized yields and rill density and channel density were assessed individually using an interaction between each density measurement and post-logging year. The effect of ground cover variables on the transformed normalized yields were similarly assessed. For the logged swales, the effect of logging disturbance variables was assessed using an interaction between the disturbance variable and post-logging year. ASW 3 and FSW 10 sediment yields were excluded after subsoiling due to lack of replication.

3 | RESULTS

3.1 | Precipitation

The 2014 water year, part of a longer-term drought that also included a few weeks of the Rim Fire (Griffin & Anchukaitis, 2014), produced 710 mm of precipitation at the Cherry Valley Dam rain gage. Cherry Valley Dam measured 650 mm of precipitation in 2015, putting the first 2 years of the study nearly 50% below the 1980–2010 annual average precipitation. In contrast, the 2016 water year was near average and precipitation ranged from 855–1031 mm in the three logging units. Precipitation in 2017, an El Niño year with above average precipitation, ranged from 1588–1973 mm in the units. Water year 2018 was near average and had precipitation totals of 775–859 mm, while 2019 produced a slight increase, with a range of 1088–1297 mm.

At the Cherry Valley Dam gage, the total precipitation from October 2013 to the start of logging measured 712 mm in Triple A, and because of the later logging periods, 1255 and 1357 mm of precipitation before logging in Lower and Upper Femmons, respectively. Precipitation following logging and preceding rill measurements measured 185 mm in Triple A, including three events with a 1-h rainfall intensity, or I_{60}, exceeding 10 mm h^{-1}. In contrast, because of the near-normal precipitation in water year 2016, the total precipitation between logging and rill measurements was 1095 mm in Lower Femmons, including two events with I_{60} over 10 mm h^{-1}, and 868 mm in Upper Femmons with one event exceeding an I_{60} of 10 mm h^{-1}. The lower total precipitation in Upper Femmons reflects a larger fraction of snow, which was less accurately measured than rain with the tipping bucket gages.

3.2 | Post-fire logging disturbance

Total post-fire logging related soil disturbance in the logged swales ranged from 9 to 56%, with a mean of 31% (SD = 15%) (Figure 2). In Triple A the total disturbance averaged 36% (SD = 8%), while in Lower Femmons the mean was 31% (SD = 21%), and the single logged swale in Upper Femmons had only 9% soil disturbance from logging (Figure 2). The most prevalent disturbance type was high traffic skid trail, which averaged 17% of swale area and occurred in all nine logged swales (Figure 2). Low traffic skid trails were present in seven of the nine swales and averaged 7% of the swale area (Figure 2). Feller buncher tracks were found in seven of the nine logged swales with a mean of 2% of the swale area (Figure 2). Mixed disturbance was found in only four of the nine logged swales and averaged 16% when present (Figure 2). Each logged swale had between two and four waterbars.

3.3 | Ground cover

Prior to logging operations in both 2014 and 2015, there were no significant differences in any cover category or bare soil between treatments (Figure 3). Before logging in the Lower and Upper Femmons units in 2015, vegetation cover was nearly significantly higher in the logged swales relative to control swales (p = 0.052) (Figure 3). There were no significant or nearly significant differences in any other cover category between to-be-logged and control swales.

The first cover measurements after logging showed that logging increased wood cover ($p \le 0.001$) except in the 2016 measurement (p = 0.07), possibly reflecting the decreased harvest intensity in Lower and Upper Femmons. In 2015, post-logging wood cover had a mean of 3% (SD = 2%) and 16% (SD = 4%) in control and logged swales, respectively. The significant differences in wood cover between treatments did not persist, as there were no significant differences in wood cover between treatments in 2018 or 2019 (Figure 3) as the rate of tree fall from the more abundant residual trees in the controls made up for the initial difference in wood cover. There were no significant differences between treatments in litter, rock, or vegetation cover (Figure 3).

Generally, overall cover increased in the swales regardless of treatments. There were no significant differences in exposed bare soil after logging (Figure 3). In 2014, before any logging and after the fire, there was 67% bare soil exposed to precipitation in the controls, and 58% bare soil in the to-be-logged swales (SD = 8% and 15%, respectively). Bare soil decreased in both treatments each year (Figure 3).

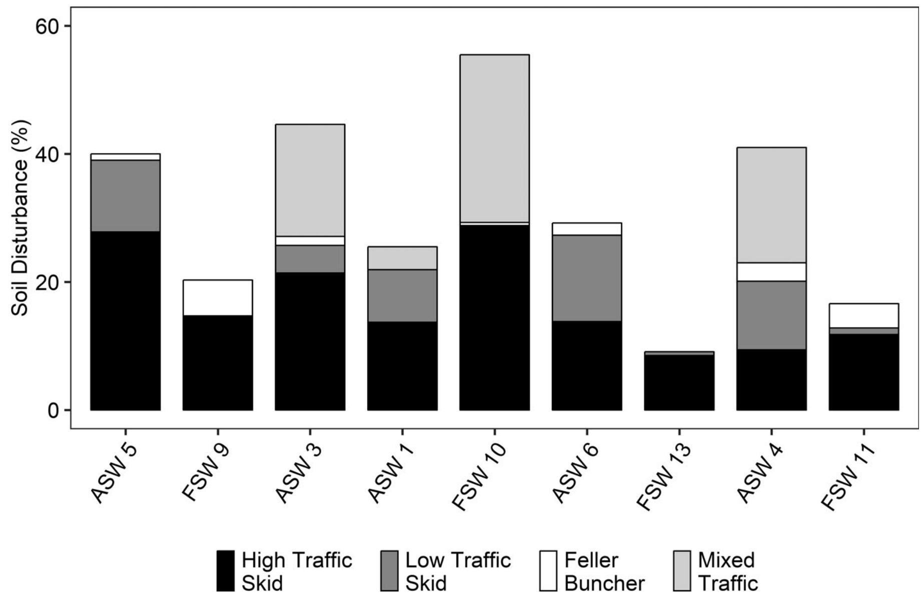

FIGURE 2 Logging disturbance in each logged swale following salvage operations, ordered from left to right by decreasing rill density (Figure 4a). ASW and FSW refer to swales in the Triple A and Femmons harvest units, respectively

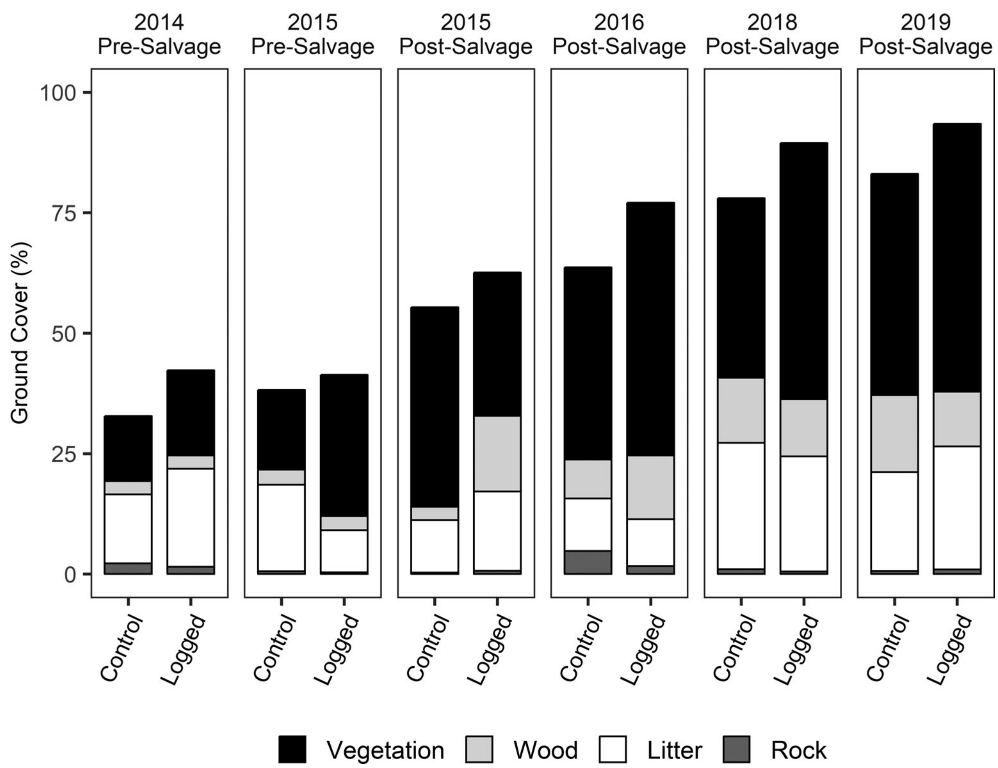

FIGURE 3 Ground cover averaged by year, pre- or post-salvage status, and treatment. Pre-salvage 2015 represents Lower and Upper Femmons, while the post-salvage 2015 represents Triple A and Lower Femmons only. 100 minus the total in each stacked bar equals the amount of exposed soil and gravel

In 2019, the controls had a mean of 17% bare soil (SD = 17%), related in part to a mean of 28% bare soil in Upper Femmons, while the logged swales had a mean of 7% bare soil (SD = 11%) (Figure 3).

3.4 | Rill networks and connectivity

Observations at the start of the study indicated that few rills had formed in the swales prior to logging because of the relatively low intensity precipitation between the fire and the logging. The disturbance from logging eliminated most of the pre-existing rills in the logged swales. After logging and a winter season, rills were identified in all 14 swales regardless of treatment, and in all 14 swales at least one rill was connected to the outlet (Figures 4a, 5 and 6). The density of connected rills ranged from 0.022 to 0.158 m m^{-2} in control swales, with a mean of 0.071 m m^{-2} (SD = 0.057 m m^{-2}), and from 0.004 to 0.218 m m^{-2} in logged swales with a mean of 0.088 m m^{-2} (SD = 0.069 m m^{-2}) (Figure 4a). At each logging unit, controls averaged 0.092, 0.051 and 0.060 m m^{-2} in Triple A, Lower Femmons, and Upper Femmons, respectively, and 0.10, 0.083 and 0.043 m m^{-2} in the

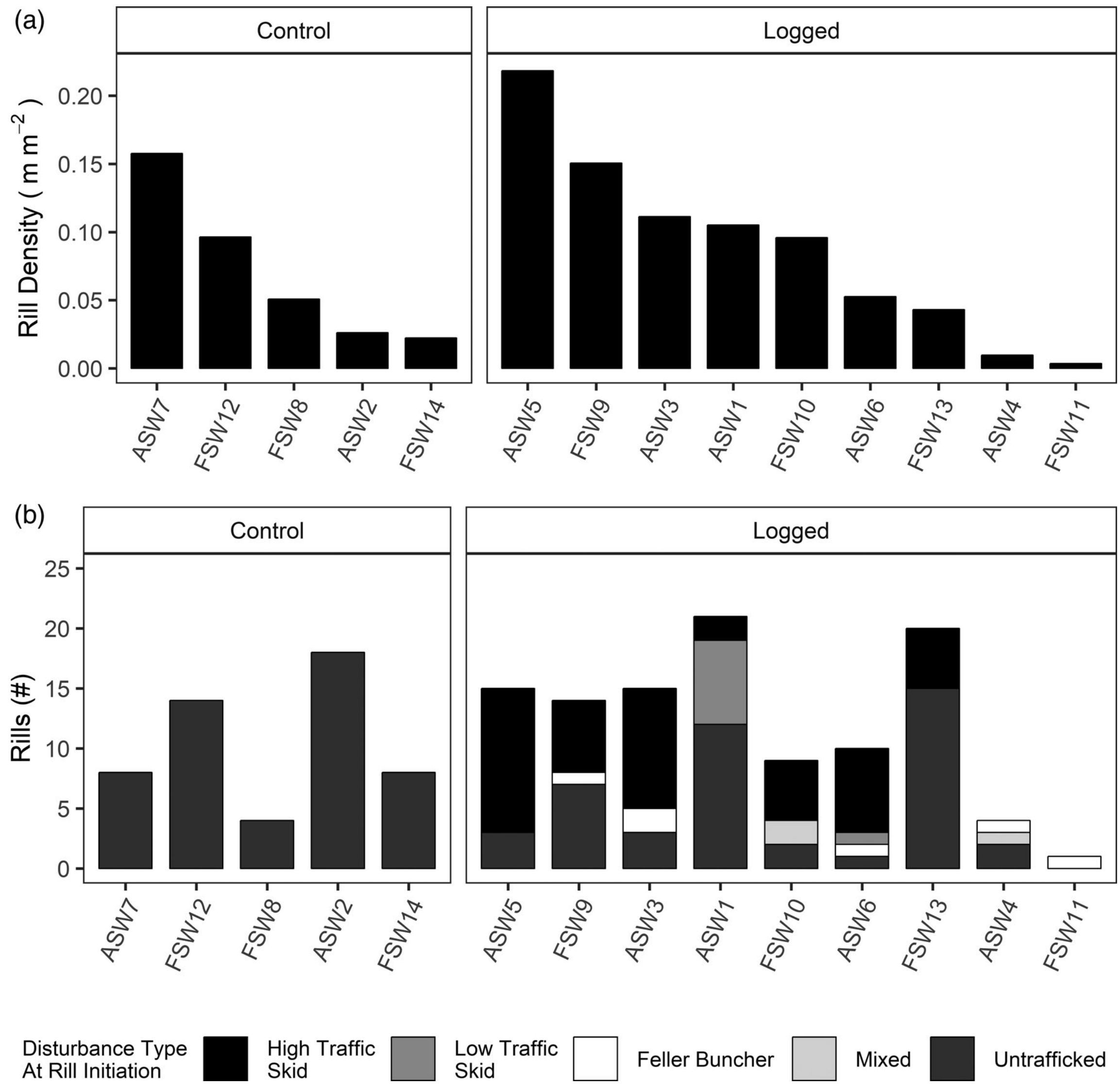

FIGURE 4 Rill density following salvage logging operations and the first winter wet season for each treatment and swale (a), and the number of hydrologically connected rills in each swale, coloured by the disturbance type at the point of initiation (b)

logged swales in the respective units. These results suggest rill density depended on site differences (Figures 5 and 6) including the timing of logging and logging intensity. Following subsoiling of skid trails in ASW 3 in fall of 2015 and the subsequent winter wet period, rill density increased from 0.11 to 0.15 m m^{-2} largely due to new rill formation in subsoiled areas, suggesting the additional soil disturbance in the swale led to increased soil erosion and connectivity to the channel.

In both logged and control swales, rills connected to the outlet initiated in bare soil. In the controls, all of the rills initiated in bare soil left by the fire, while in the logged swales there was bare soil from the fire as well as from logging disturbance. Within the controls, rill densities were comprised of 4 to 18 rills, with a mean of 10 rills. 85% of the 61 rills identified in the control swales were connected to the outlet (Figure 5). The total number of rills in control swales did not always reflect rill densities, as some of the combined rill lengths were relatively short. For example, ASW 7 had eight rills and a density of 0.158 m m^{-2}, while ASW 2 had 18 rills and a rill density of only 0.026 m m^{-2} (Figure 5). The bare soil averaged 67% in ASW 7 as opposed to 28% in ASW 2, and the dNBR in ASW 7 was 869 as compared to 799 in ASW 2, suggesting fire impacts in ASW 7 were greater than in ASW 2.

The number of rills that reached a channel leading to the outlet or connected directly to the outlet, hereafter "connected" rills, ranged from 1 to 21 per logged swale (Figure 4b). Between 0 and 15 connected rills initiated in untrafficked areas in logged swales, accounting for 41% of the total number of connected rills (Figure 4b). The connected rills in the logged swales that originated in soil affected by logging disturbance most frequently started in high traffic skid trails (43% of all connected rills) (Figures 4b and 6). All but two logged swales had connected rills initiated in high traffic skid trails, with up to 12 rills from skid trails in a single swale (Figure 4b). For the other disturbance classes in the logged swales, rill initiation and connection to the outlet

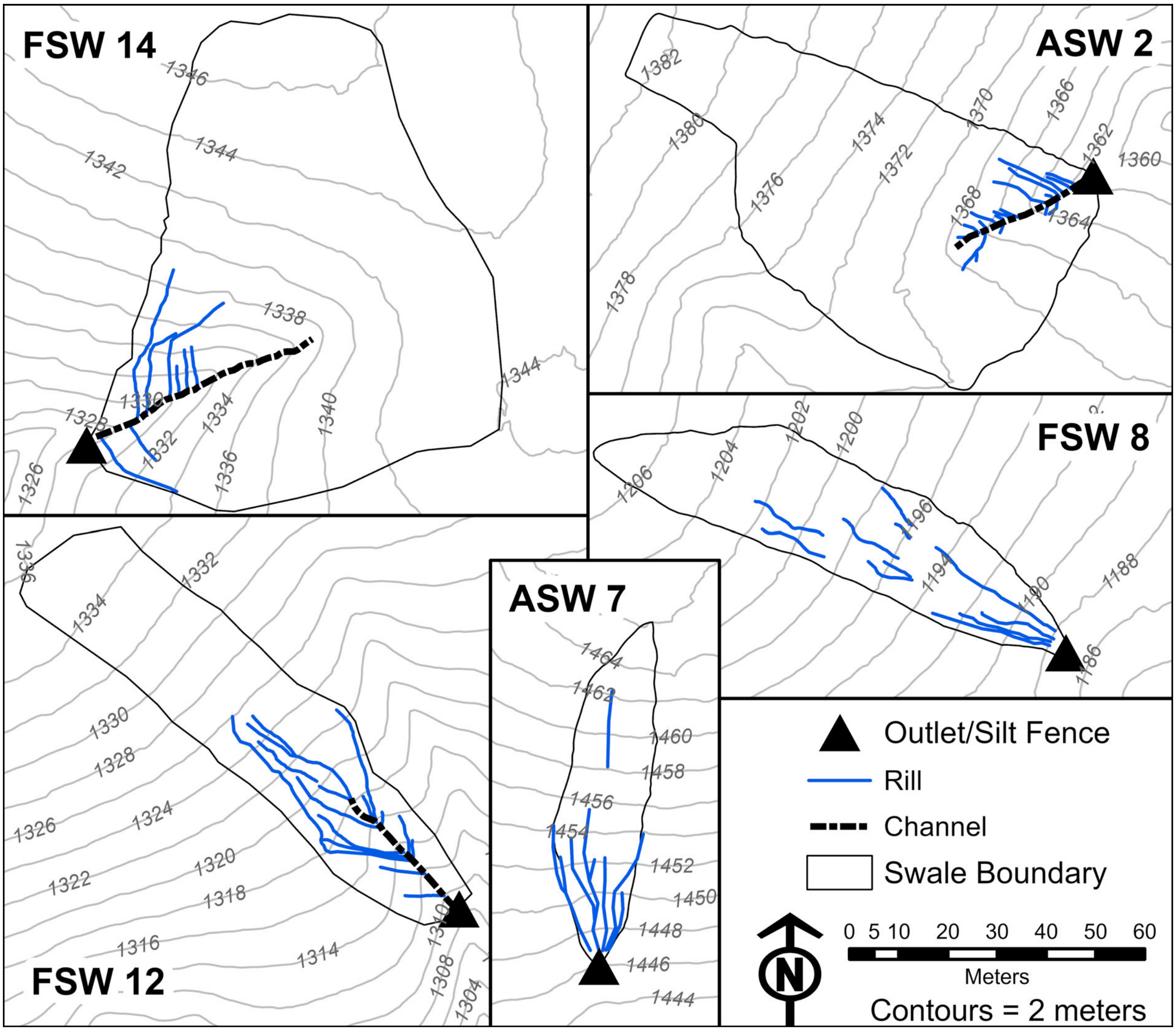

FIGURE 5 Rills and ephemeral channels, where present, in the control swales. ASW 2 and ASW 7 are in Triple A, FSW 8 is in Lower Femmons, and FSW 12 and FSW 14 are in Upper Femmons

was far more limited, with 8% of connected rills originating in low traffic skid trails, 5% originating in feller buncher tracks, and 3% originating in mixed traffic (Figure 4b). Following the subsoiling of skid trails in ASW 3, the total number of connected rills in ASW 3 increased from 15 to 25, with no change in the percentage originating in skid trails.

Three of the five controls and five of the nine logged swales had rills form that did not ultimately connect to the swale outlet, and the spatial location of rill initiation points affected the connectivity of rills. Rills in controls and logged swales were frequently observed entering areas of dense vegetative cover or stump holes, such as FSW 8 and FSW 11, leading to disconnection, and rills were more likely to be connected to the outlets when large patches of bare soil were present, particularly near channels (Figures 5 and 6). Rills in logged swales were also disconnected when their routes encountered areas of high surface roughness and slash accumulation from logging activity such as in ASW 4 (Figure 6), where we also observed sediment deposition. Conversely, rills in logged swales that were initiated by logging disturbance in close proximity to the swale outlet or a channel were frequently connected to the outlets, such as in ASW 5 (Figure 6). Waterbars often directed rills towards the channel and outlet of the swale (Figure 6).

The difference between treatments in rill density was not significant. The dNBR and mean slope did not relate to rill density. However, there was a nearly significant interaction (p = 0.11) between treatment and mean slope. In the control swales, rill density increased 0.012 m m^{-2} for every percent increase in slope, while within the logged swales, rill density decreased 0.003 m m^{-2} units for every percent increase in slope. Generally, area of logging disturbance, including skid trail area, decreased as mean slope increased in the logged swales. None of the ground cover variables were significant predictors of rill density, with vegetation the closest to being significant (p = 0.06).

Within the logged swales, the percent of the swale with high traffic skid trails was positively correlated with rill density (p = 0.045, r^2 = 0.46), with an increase in rill density of 0.006 m m^{-2} for every percent increase in high traffic skid trails. The number of "connected

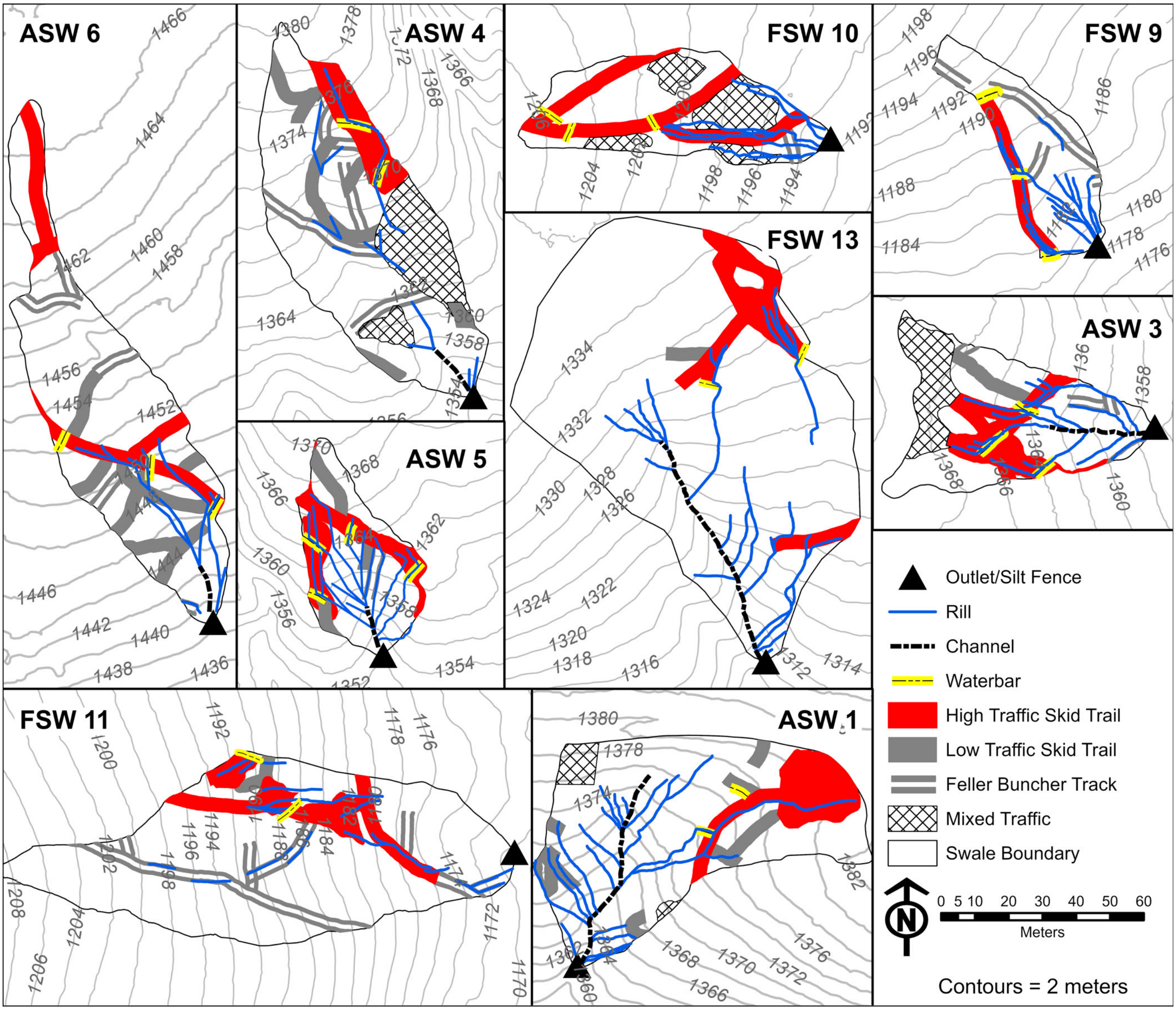

FIGURE 6 Logging disturbance, rills, and ephemeral channels in the logged swales. ASW 1, 3, 4, 5 and 6 are in Triple A; FSW 9, 10 and 11 are in Lower Femmons, and FSW 13 is in Upper Femmons. The map for ASW 3 and FSW 10 shows the logging disturbance before subsoiling occurred

waterbars", which are waterbars that diverted rills downslope to a channel or swale outlet, was nearly significantly correlated with rill density in logged swales (p = 0.060, r^2 = 0.59), with rill density increasing 0.038 m m^{-2} for each connected waterbar. The percent of swale impacted by all other disturbance categories including total disturbance and total skid trails across traffic levels were not significantly related to rill density.

3.5 | Sediment yields

In water year 2015 in Triple A, the first year after logging, sediment yields averaged 4.3 Mg ha^{-1} (SD = 3.6 Mg ha^{-1}) in the control swales and 3.0 Mg ha^{-1} (SD = 3.1 Mg ha^{-1}) in the logged swales (Figure 7a). Despite more precipitation in the second year after logging, the sediment yields decreased to a mean of 0.92 and 1.1 Mg ha^{-1} in control and logged swales, respective (SD = 1.2 Mg ha^{-1} for both treatments). ASW 3 produced more sediment in the second year after logging (3.2 Mg ha^{-1}) than in the first (2.7 Mg ha^{-1}) because of the large number of rills that formed in the furrows of the subsoiled skid trails (Figure 7a). Excluding this swale, the mean sediment yield for the logged swales decreased to 0.63 Mg ha^{-1} in the second year. The sediment yields continued to decrease over time, and the means were 0.081 Mg ha^{-1} in the controls (SD = 0.10 Mg ha^{-1}) and 0.24 Mg ha^{-1} (SD = 0.37 Mg ha^{-1}) in the logged swales (Figure 7a) in the third post-logging year, and 0.0076 and 0.081 Mg ha^{-1}, in the control and logged swales, respectively, in the fourth post-logging year (Figure 7a). By 2019, the fifth post-logging year, the control swales had a mean

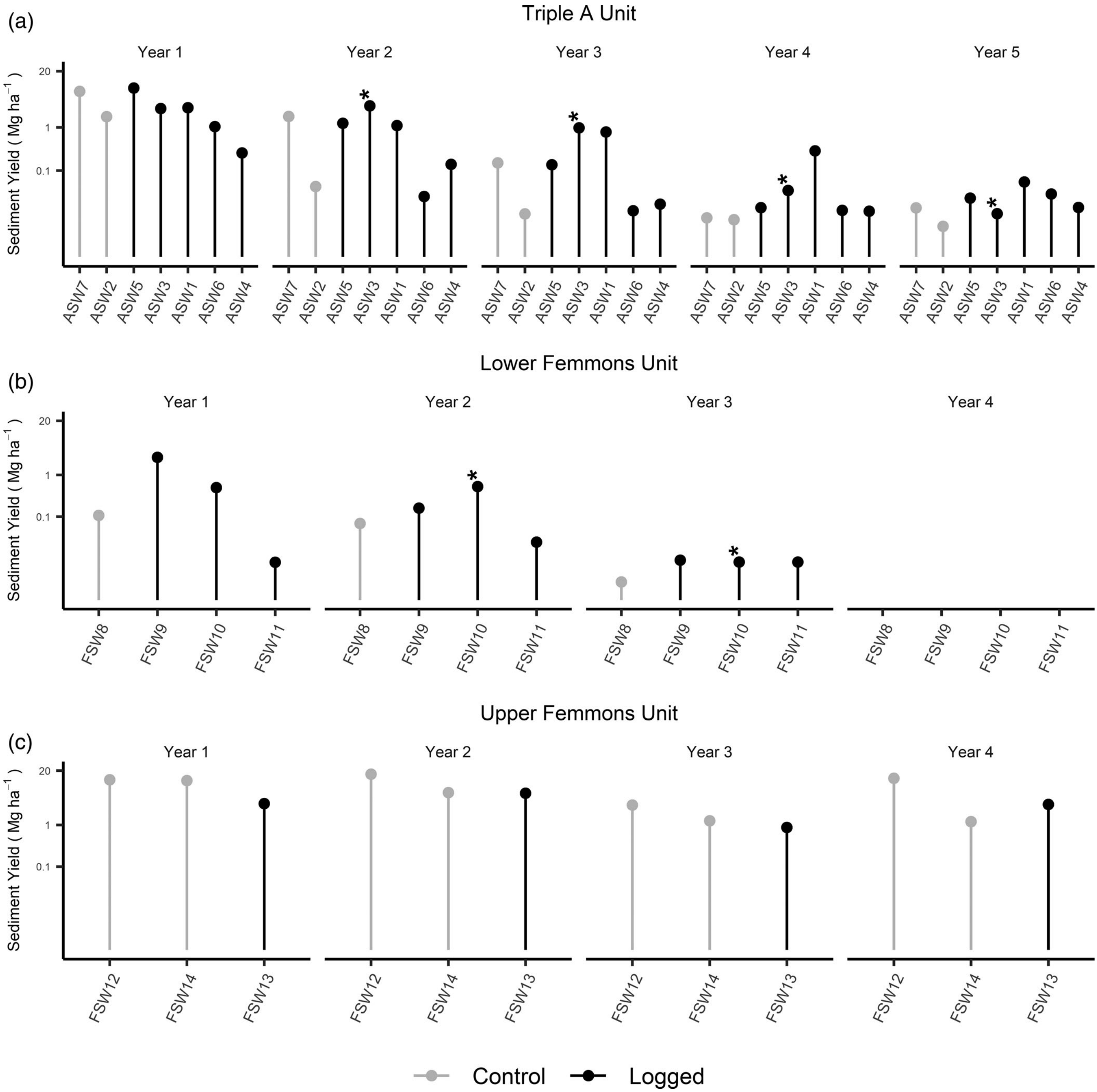

FIGURE 7 Annual sediment yields for each unit, post-salvage year, and swale: (a) Triple A, (b) Lower Femmons, and (c) Upper Femmons. The asterisks in (a) and (b) denote the years following subsoiling in ASW 3 and FSW 10. Logging was completed in Triple A in December 2014, in Lower Femmons in May 2015, and in Upper Femmons in September 2015. The fences in Lower Femmons were damaged in year 4 and data are not available

yield of 0.009 Mg ha^{-1} (SD = 0.0060 Mg ha^{-1}) while the logged swales averaged 0.03 Mg ha^{-1} (SD = 0.017 Mg ha^{-1}). These logged values exclude ASW 3, which produced 0.97, 0.034 and 0.01 Mg ha^{-1} in 2017, 2018 and 2019, respectively.

Sediment yields were lower in Lower Femmons than Triple A. In water year 2016, the first post-logging year, the control swale produced a sediment yield of 0.11 Mg ha^{-1}, while the logged swales averaged 1.1 Mg ha^{-1} (SD = 1.4 Mg ha^{-1}) (Figure 7b). In the second post-logging year the control yield was 0.070 Mg ha^{-1} and the logged swales mean was 0.24 Mg ha^{-1} (SD = 0.096 Mg ha^{-1}). In the third year, the means effectively decreased to zero, with 0.0027 Mg ha^{-1} in the control and 0.0085 Mg ha^{-1} in the logged swales (Figure 7b). Excluding FSW 10 following subsoiling, mean logged swale yields were 0.093 and 0.0090 Mg ha^{-1} in post-logging year two and three, respectively.

Upper Femmons produced more sediment than the other units, and the sediment production did not decrease over time as in Triple A and Lower Femmons (Figure 7c). The control swales produced a mean yield of 11.8 Mg ha^{-1} (SD = 0.34 Mg ha^{-1}) and the logged swale had a yield of 3.2 Mg ha^{-1} in the year after logging, water year 2016 (Figure 7c). Yields averaged 11.2 Mg ha^{-1} (SD = 7.5 Mg ha^{-1}) in the controls and 5.8 Mg ha^{-1} in the logged swale in the second post-logging year, and 2.1 Mg ha^{-1} (SD = 1.2 Mg ha^{-1}) in the controls and 0.87 Mg ha^{-1} in the logged swale in the third year after logging (Figure 7c). Water year 2019, the fourth year after logging, had increases in sediment yield, with means of 7.2 Mg ha^{-1} (SD = 8.4 Mg ha^{-1}) in the controls and 3.1 Mg ha^{-1} in the logged swale.

Once sediment yields were normalized to the unit-level control sediment yield, there were no significant differences between treatments in any post-logging year. The transformed normalized yields did show a non-significant relative difference, which were effectively equal between treatments until post-logging year three when the logged swales became and stayed higher. Regardless of treatment, the sediment yields did not significantly decrease from post-logging year one to two, but did decrease significantly from year one to three, one to four, and one to five. The inclusion of subsoiled swales in the analysis resulted in the logged swales having a non-significantly higher normalized yield starting in post-logging year two.

The transformed normalized yields were significantly related to bare soil (p = 0.03 and r^2 = 0.60). Sediment yields were also significantly and inversely related to vegetation (p = 0.02, r^2 = 0.53).

Within the logged swales, the effect of high traffic skid trails on yield was not significant, but was positive in post-logging year 1; however, in subsequent years, the overall effect of high traffic skid trail area was negligible. The total area of the swale in other disturbance categories was not significant, nor were the number of connected waterbars, although in the model the slope associated with total disturbance was positive.

Rill density was significantly and positively related to transformed normalized sediment yield (p = 0.004, r^2 = 0.48), most significantly in the first year after logging, although the relationship decreased over time to non-significant after year three. The inclusion of ASW 3 and FSW 10 after subsoiling did not alter results (p = 0.002, r^2 = 0.49). Channel density was not a significant predictor of sediment yield.

4 | DISCUSSION

4.1 | Effects of ground cover

Past research in California found that both the amount of exposed bare soil and percent disturbance have strong effects on sediment yields following wildfire and post-fire logging (Chase, 2006). Increases in cover from downed wood after logging may help mitigate soil erosion following the cumulative impacts of wildfire and post-fire logging disturbance on hillslopes (Poff, 1989; Prats et al., 2019, 2020; Wagenbrenner et al., 2015). We measured higher levels of wood cover in logged swales relative to controls following logging in the initial post-logging year, similar to other studies (Donato et al., 2006; D. W. Peterson et al., 2015). However, the increased wood cover did not have a significant effect on rill density or normalized sediment yields, and the lack of effect may have been due to the total cover being less than the cover needed to reduce surface erosion following logging (Berg & Azuma, 2010). One additional possible explanation for the lack of effect is that in our study the spatial distribution of wood cover may not have been consistent enough across the swales to affect swale-scale sediment yields. Relatively patchy wood cover across swales, concentrated near where trees were felled and removed, likely allowed rills to form in areas of bare soil outside of logging disturbance. Treefall increased the wood cover in the control swales, resulting in comparable wood cover rates between the treatments in the second post-logging year and reducing any potential relative benefit of the addition of slash from the logging operations.

Wood cover did not mitigate rainfall-induced sediment yields in our study, in contrast to some of the studies identified above. Wagenbrenner et al. (2016) found wood slash added to skid trails in a post-fire logging study using simulated rill experiments had no effect on rill erosion. The difference between these conflicting results may be explained by insufficient contact between the wood and the soil surface, which allowed the runoff to pass under the slash. A recent post-fire logging study in central Washington showed that wood slash added to skid trails during the skidding operation to increase ground contact produced a decrease in sediment yields (Robichaud, Lewis, et al., 2020).

While none of the ground cover variables significantly affected rill density, sediment yields were significantly related to bare soil and vegetation cover. This suggests that rill formation is related to discrete patches of fire- or logging-affected bare soil and that rill connectivity, as reflected in the sediment yields, is related to the size and downslope location of bare patches, rather than to a hillslope-averaged bare soil value. Bare soil in burned areas has been related to erosion mechanisms (Berg & Azuma, 2010) and post-fire sediment yields with and without logging (Larsen et al., 2009; Stabenow et al., 2016; Wagenbrenner et al., 2015). The subtle difference in the relationships of sediment yield and rill density with ground cover imply that the spatial arrangement of the effects of fire and post-fire logging disturbance are important to rill connectivity.

Earlier studies have concluded that increasing traffic levels of logging equipment, especially traffic by rubber-tired skidders, result in decreased ground cover, increased soil bulk density, and decreased infiltration rates (Croke et al., 2001; Wagenbrenner et al., 2015, 2016). These factors all increase the likelihood of surface runoff and the formation of rills under relatively common precipitation intensities. Cover in high traffic skid trails in our swales had a mean of 32% in 2015, while ground cover increased as traffic levels decreased, to a mean of 65% in untrafficked areas (Olsen, 2016). These areas were therefore more susceptible to rain splash detachment, raindrop induced soil sealing, and the initiation of rills. When the rills formed relatively low in our swales, they were more likely to be hydrologically connected to the outlets, increasing sediment yield (Figures 5 and 6).

4.2 | Sediment yield

Hillslope erosion rates can increase with increasing disturbance from post-fire logging (Chase, 2006; Demirtaş, 2017; Wagenbrenner et al., 2015), although some studies also show that post-fire logging can have no effect or even reduce sediment yield (Cole et al., 2020; James & Krumland, 2018). The lack of significant difference in the transformed normalized sediment yields between logged and unlogged swales echoes past research that did not detect significant increases in sediment yield after logging in burned catchments (Chase, 2006; Chou et al., 1994; Stone et al., 2014; Wagenbrenner et al., 2015).

Our results include relatively high unit to unit variability in sediment yields, which are likely affected by variations among the units that are not apparent in some of our measurements. Upper Femmons, for example, had more manzanita than other sites, and the burning characteristics of manzanita and other chaparral species may cause greater burn severity than large conifers alone (Skinner & Chang, 1996). Even within the category of "high severity", there can be a broad range of fire effects. It is possible though highly uncertain that the manzanita was present before the Rim Fire due to previous high severity burning (Lauvaux et al., 2016). However, Upper Femmons also had higher rock content than the other sites, which also affects manzanita prevalence. The potential effects, either stand-alone or combined, of higher severity and higher rock content probably also contributed to the slower recovery in Upper Femmons as demonstrated by the still-high sediment yields in 2019 (Figure 7). In addition to differences in pre-fire vegetation, fire severity, and post-fire recovery at Upper Femmons, this unit also was the last of our three units to be logged, and because of decreasing timber value as time after the fire increased, there was less timber removed during the logging operation. This may have countered some of the effects of the vegetation type and fire severity on sediment yields, although given the small number of swales in Upper Femmons, we have insufficient data to test this relationship. Indications of recovery in the other units are demonstrated by the successively decreasing sediment yields in both treatments in Triple A and in the logged swales in Lower Femmons, despite increased precipitation amounts in later years of our study.

Upper Femmons also had more channelization and greater hillslope convergence than the other units, which likely contributed to higher sediment yields relative to the other units. As spatial scale increases from the hillslope to larger catchments (>10 ha), the importance of hillslope processes decreases and channel processes add to or dominate erosion (Moody & Martin, 2009; Robichaud, Wagenbrenner et al., 2013; Wagenbrenner & Robichaud, 2014). Although there is no consistent classification system, our swales are generally near the threshold of contributing area separating hillslope and catchment scales, suggesting that channel processes may be important to conveyance of rill-transported sediment from hillslopes and subsequent sediment yield. The more extensive and persistent bare soil, rill networks, and channelization in Upper Femmons help explain why Upper Femmons had such high sediment yields even 5 years after the wildfire and 3 years after post-fire logging.

4.3 | Effects of logging disturbance on connectivity

Rill erosion can dominate post-fire hillslope erosion (Moody et al., 2013) and can account for 60%–80% of sediment delivered from hillslopes following wildfires (Pietraszek., 2006), which explains the significant positive relationship we found between rill density and the normalized sediment yield. In this study the swale-scale sediment yields did not correlate to percent area disturbed, highlighting the importance of connectivity and sources. Some logged swales, such as ASW 1 and FSW 13, had a majority of rills initiated in untrafficked areas of bare soil similar to the control swales (Figures 4b and 6), such that sediment from untrafficked areas may have represented the majority of the sediment yield from the logged areas. Alternatively, the logged swales ASW 3 and ASW 5 had extensive rills initiated in skid trails and connected to the outlet which likely represented the majority of the first-year sediment yield (Figure 6). Control swales such as ASW 7 and FSW 12 exhibited extensive rill networks and sediment yields greater than some logged swales each water year. Other controls, such as ASW 2 and FSW 8, developed minimal rill networks and produced minimal sediment yield in the later water years (Figure 5). At the watershed scale, these results suggest that it would be difficult to differentiate signals from post-fire logging and wildfire in terms of sedimentation of stream networks (Lewis et al., 2018; Silins et al., 2009; Stone et al., 2014).

The proximity of logging disturbance or patches of untrafficked bare soil to ephemeral channels generally enhanced rill connectivity and sediment yield at the swale outlet. While the percent of the swale impacted by high traffic skid trails significantly increased rill density, the proportion of high traffic skid trails was not related to sediment yields. This result suggests that some deposition occurred before runoff and sediment reached the swale outlets. Logging disturbances located near sediment sinks such as areas of high vegetative cover were less likely to be connected to the swale outlet, even when rills formed in disturbed areas, such as in ASW 4 and FSW 11 (Figure 6). Areas of high surface cover downslope of the logging disturbances or patches of bare soil, in particular concentrated dense wood cover or grasses, forbs or shrubs, allowed sediment deposition to occur before the sediment could reach swale outlets or ephemeral channels.

While rill densities did not significantly vary between logged and unlogged swales, the disturbance patches connected by rills in the logged swales may represent sediment sources that persist longer than rills formed in soil exposed only by wildfire (Demirtaş, 2017). This persistence at least partly explains why the normalized sediment yields in the logged swales in Triple A and Lower Femmons were elevated relative to the control swales in later post-logging years.

Litschert and MacDonald (2009) found that skid trails and waterbars were commonly sources of sediment that reached streams in unburned harvest sites throughout the Sierra Nevada. They observed a lack of surface roughness downslope of waterbar outlets, which encouraged rill formation and erosion rather than the intended infiltration of runoff. In our logged swales, 56% of the waterbars directed rills towards the outlet or connected rills to the outlet via an ephemeral channel (Figure 6). While rills generally did not initiate at waterbars, waterbars often caused rills to merge as they travelled downslope. In the logged swales, 27% of all rills that were connected to the channel outlets were redirected by a waterbar (Figure 6), and these rills predominantly initiated within skid trails. The greater flow from the combined rills will have greater hydraulic power for sediment detachment and transport. Adding slash or other runoff-resistant surface cover to waterbar outlets could help disperse concentrated runoff, capture sediment, and reduce connectivity to the stream network, in particular when burned areas between skid trails and stream networks have little ground cover.

Reducing the extent of sediment sources and the hydrologic connectivity between sediment sources and the stream network are critical steps to minimizing the influence of post-fire logging on sediment yields. Logging operations often are prohibited in channels and restricted in riparian buffers. However, given the relative lack of surface cover adjacent to burned headwater channels and the potential distance over which rills may travel through bare soil, additional erosion mitigation may be needed in heavily disturbed areas or greater distance to the channel for salvage logging operations to prevent contribution of rills directly to stream channels (Figure 6) (Robichaud, Bone, et al., 2020). Adding surface cover that is in contact with the soil surface and will stay in place, such as mulch or wood slash, can be effective at reducing sediment yield from burned areas (Fernández & Vega, 2016; Prats et al., 2019; Robichaud, Lewis, et al., 2013; Robichaud, Lewis, et al., 2020). Supplementing logging-generated wood cover and naturally regenerating ground cover with mulch or additional slash may reduce sediment production from skid trails in burned and logged areas (Prats et al., 2020; Robichaud, Lewis, et al., 2020). Future research should consider mitigation treatments that can reduce rill initiation within skid trails or treatments that disconnect rills from the drainage network by increasing infiltration and sediment deposition, particularly downslope of waterbar outlets.

5 | CONCLUSION

We assessed the effects of post-fire logging on rill network development and sediment yields in swales burned by the 2013 Rim Fire and logged in late 2014 or 2015. Although logged and control swales had rills initiate in patches of untrafficked bare soil, rill initiation in the logged swales was related to the proportion of area in high traffic skid trails. Mean rill density did not significantly vary between logged swales (0.088 m m^{-2}) and controls (0.071 m m^{-2}), respectively, despite the influence of high traffic skid trails on rill initiation in the logged swales. Rills were frequently directed towards the swale outlets by waterbars on high traffic skid trails, although the interspersing of wood and vegetation cover among patches of bare soil also often disconnected rills from the outlets in logged and control swales.

Sediment yields varied widely regardless of logging, and there was no significant difference in log-transformed normalized sediment yields between the logged and control swales. Transformed normalized sediment yields significantly increased with increasing rill density in the first three post-logging years, and rill density explained 48% of the variability in the sediment yields. Increased bare soil and reduced vegetation cover resulted in significantly greater sediment yields. Transformed normalized sediment yields decreased at different rates over time in the control and logged swales, and this led to non-significantly higher transformed sediment yields in the logged swales than in the controls starting post-logging year three. Within the logged swales, the transformed normalized sediment yields were not related to the total amount of logging disturbance and were positively but non-significantly related to the percent of swale area with high traffic skid trails. These results suggest that the spatial layout of skid trails and surface cover are important factors in determining the effects of post-fire salvage logging on rilling and sediment yields, and are potential areas of focus to reduce hydrologic connectivity between post-fire logging-related disturbance and the drainage network.

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DATA AVAILABILITY STATEMENT

The data from this study are in preparation for submission to a US Forest Service data portal where they will be archived and made publicly available.

ORCID

Will H. Olsen https://orcid.org/0000-0003-4259-2924
Joseph W. Wagenbrenner https://orcid.org/0000-0003-3317-5141
Peter R. Robichaud https://orcid.org/0000-0002-2902-2401

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