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WILDFIRE AND HYDROLOGICAL PROCESSES

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Hillslope sediment production after wildfire and post-fire forest management in northern California

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Abstract

High severity wildfires impact hillslope processes, including infiltration, runoff, erosion, and sediment delivery to streams. Wildfire effects on these processes can impair vegetation recovery, producing impacts on headwater and downstream water supplies. To promote forest regeneration and maintain forest and aquatic ecosystem functions, land managers often undertake active post-fire land management (e.g., salvage logging, sub-soiling, re-vegetation). The primary objective of our study was to quantify and compare sediment yields eroded from (a) burned, (b) burned and salvage logged, and (c) burned, salvage logged, and sub-soiled plots following the 2015 Valley Fire in the northern California Coast Range. We distributed 25 sediment fences (~75 m^2 contributing area) across four hillslopes burned at high severity and representative of the three management types. We collected eroded sediment from the fences after precipitation events for 5 years. We also quantified precipitation, canopy cover, ground cover, and soil properties to characterize the processes driving erosion across the three management types. Interestingly, during the second year after the fire, sediment yields were greater in the burned-only plots compared with both the salvage logged and sub-soiled plots. By the third year, there were no differences in sediment yields among the three management types. Sediment yields decreased over the 5 years of the study, which may have occurred due to site recovery or exhaustion of mobile sediment. As expected, sediment yields were positively related to precipitation depth, bulk density, and exposed bare soil, and negatively related to the presence of wood cover on the soil surface. Unexpectedly, we observed greater sediment yields on the burned-only plots with greater canopy closure, which we attributed to increased throughfall drop size and kinetic energy related to the residual canopy. While these results will aid post-fire management decisions in areas with Mediterranean climates prone to low intensity, long duration rainstorms, additional research is needed on the comparative effects of post-fire land management approaches to improve our understanding of the mechanisms driving post-fire erosion and sediment delivery.

KEYWORDS

erosion, forest fire, hillslope processes, runoff, salvage logging, sediment

1 | INTRODUCTION

The timing, extent, and severity of wildfire activity in many forested regions of the world, including western North America, has increased dramatically in recent years (Flannigan, Krawchuk, de Groot, Wotton, & Gowman, 2009; Moritz et al., 2012; Reilly et al., 2017; Westerling, 2016). While wildfire activity has intensified rapidly in the past three decades, historical evidence suggests the risks associated with high severity wildfire could continue to rise (Murphy, Yocom, & Belmont, 2018). As such, concerns have grown regarding the immediate and longer-term effects on forest resilience and the water supply originating in forests (Hallema, Robinne, & Bladon, 2018; Stevens-Rumann et al., 2018). Increasingly severe wildfires have produced substantial and long-lasting (>10 years) effects on annual streamflow and peak flows (Hallema, Sun, et al., 2018; Niemeyer, Bladon, & Woodsmith, 2020; Saxe, Hogue, & Hay, 2018), debris flows (Langhans et al., 2017; Nyman et al., 2015), physical and chemical water quality (Rhoades et al., 2019; Rust, Hogue, Saxe, & McCray, 2018), aquatic ecosystem health (Bixby et al., 2015; Emelko et al., 2016), and downstream drinking water supply (Emelko, Silins, Bladon, & Stone, 2011; Hohner, Terry, Townsend, Summers, & Rosario-Ortiz, 2017).

The broad range of impacts on water supply are attributable to the complex and interconnected effects of wildfires on soil water repellency, soil organic matter, canopy and litter interception, root reinforcement, sediment supply, and soil hydraulic properties (Ebel & Moody, 2017; Robichaud et al., 2016). In turn, these interrelated effects often result in increased surface runoff generation, faster runoff response, and increased erosion and sediment delivery to streams (Helvey, 1980; Malmon, Reneau, Katzman, Lavine, & Lyman, 2007; Moody & Martin, 2001b; Neary, Ryan, & DeBano, 2005). Amplified runoff and sediment delivery can lead to changes in stream geomorphology (Shakesby & Doerr, 2006), community structure of aquatic ecosystems (Arkle, Pilliod, & Strickler, 2010; McCormick, Riemen, & Kershner, 2010), and water supplies for downstream communities (Bladon, Emelko, Silins, & Stone, 2014; Emelko et al., 2011; Smith, Sheridan, Lane, Nyman, & Haydon, 2011).

Due to the broad range of post-fire threats, post-fire land management activities (e.g., emergency stabilization, rehabilitation, and restoration) are often applied in an attempt to promote regeneration and maintain forest and aquatic ecosystem functions (Leverkus et al., 2018; Robichaud, Beyers, & Neary, 2000). Salvage logging, or biomass harvesting, is one of the most common post-fire forest management practices (Karr et al., 2004; Lindenmayer et al., 2004). It is often justified as an approach to recover economic value from the burned timber resources, improve forest safety, reduce woody fuel loads and re-burn severity, lessen the potential for pest outbreaks, and facilitate reforestation efforts (Donato, Fontaine, Kauffman, Robinson, & Law, 2013; Malvar, Silva, Prats, Vieira, & Coelho, 2017; Müller et al., 2019). Land managers may also apply additional treatments to mitigate effects from wildfire and promote vegetation recovery. For example, ploughing furrows along the contours of hillslopes (sub-soiling) may be used with the objectives to decrease soil bulk density, break up hardpans, improve conditions for root development of newly established vegetation, and reduce runoff and erosion potential (Carlson et al., 2006; Morris & Lowery, 1988; Will, Wheeler, Markewitz, Jacobson, & Shirley, 2002). Similarly, contour-felled logs, straw wattles, and hand-dug contour trenches have been used as erosion barriers to mitigate post-wildfire runoff and erosion (Robichaud, Pierson, Brown, & Wagenbrenner, 2008; Robichaud, Wagenbrenner, Brown, Wohlgemuth, & Beyers, 2008). Land managers have also seeded grasses or planted trees to facilitate vegetation recovery on burned hillslopes (Ouzts, Kolb, Huffman, & Meador, 2015; Wagenbrenner, MacDonald, & Rough, 2006). Occasionally, herbicides are applied to re-planted, burned hillslopes to suppress competition from non-native vegetation or emergent understory vegetation (Munson et al., 2015; Powers & Ferrell, 1996; Powers & Reynolds, 1999).

Limited research into post-fire land management strategies has led to continued debate of their potential benefits and trade-offs (DellaSala et al., 2006; Donato et al., 2006; Leverkus et al., 2020; Leverkus, Puerta-Pinero, Guzman-Alvarez, Navarro, & Castro, 2012; McIver & Starr, 2000). For example, the removal of standing and downed large wood may eliminate important structural components that can help facilitate the recovery of terrestrial and aquatic systems (Lindenmayer & Noss, 2006; Maia et al., 2014; May & Gresswell, 2003). Moreover, salvage logging in recently burned areas has the potential to create additional site disturbance and soil compaction (McIver & Starr, 2001; Wagenbrenner, Robichaud, & Brown, 2016). These cumulative effects on the soil can further enhance post-fire runoff, erosion, and sediment delivery to streams with negative consequences for soil fertility, vegetation recovery, and aquatic ecosystem health (Karr et al., 2004; Wagenbrenner, MacDonald, Coats, Robichaud, & Brown, 2015). Similarly, the additional ground disturbance associated with sub-soiling or the suppression of ground cover due to herbicide application could extend the recovery period and further increase soil exposure to rainfall and runoff events and increase erosion rates (Benavides-Solorio & MacDonald, 2001; Robichaud, Wagenbrenner, & Brown, 2010).

Given the recent trends towards larger, more severe wildfires in many regions (Keyser & Westerling, 2019; Stevens, Collins, Miller, North, & Stephens, 2017), including California, it is crucial to improve our understanding of the efficacy of post-fire forest management approaches at mitigating runoff and erosion. While many post-fire management practices appear to have the potential to ameliorate the impact of fire on runoff and erosion (McIver & Starr, 2000), research on the short- and long-term effects is limited, especially in mixed-conifer forests of the northern California Coast Range. Studies on post-fire salvage logging have increased in recent years (Lewis, Rhodes, & Bradley, 2019; Lucas-Borja et al., 2019, 2020; Silins, Stone, Emelko, & Bladon, 2009; Slesak, Schoenholtz, & Evans, 2015), but we are only aware of one study assessing the effectiveness of either sub-soiling or herbicide application after salvage logging (James & Krumland, 2018). Further research on these practices will improve the understanding of the effects of the various post-fire land management strategies to facilitate informed land and water management decisions. Thus, our primary objective was to quantify differences in hillslope sediment yields among (a) burned-only, (b) burned and salvage logged, and (c) burned, salvage logged, and sub-soiled plots through the first 5 years after the 2015 Valley Fire in northern California. Our second objective was to quantify differences in precipitation characteristics, ground cover, canopy cover, soil bulk density, soil hydraulic properties, and soil water repellency to improve our understanding of the processes driving the differential erosion response across the three post-fire forest management types.

2 | METHODS

2.1 | Site description

The Valley Fire burned approximately 30,700 ha of forested land and wildland–urban interface in southern Lake County, California, from September 12 to October 15, 2015. During the fire, approximately 98% (1,414 ha) of the Boggs Mountain Demonstration State Forest (BMDSF) was burned. BMDSF is a public forest, managed by the California Department of Forestry and Fire Protection (CAL FIRE) located about 10 km southwest of Clear Lake, CA, in the northern California Coast Range (38.83528° N, 122.70148° W). During the Valley Fire, about 48% of the BMDSF area burned at high severity, 34% at moderate severity, 15% at low severity, and 2% remained unburned/unchanged (Figure 1).

The climate of the region is Mediterranean with warm dry summers and cool, wet winters (Köppen Csb). Rainfall dominates the precipitation, although there are occasional snow events and transient snowpack during most winters. Mean annual precipitation is 1,408 mm with the majority falling between October and April (PRISM Climate Group, 2004).

The primary tree species across the region before the fire were ponderosa pine (*Pinus ponderosa*), sugar pine (*Pinus lambertiana*), Douglas-fir (*Pseudotsuga menziesii*), and California black oak (*Quercus kelloggii*). Pacific madrone (*Arbutus menziesii*) and canyon live oak (*Quercus chrysolepis*) were also present as minor components of the forest canopy.

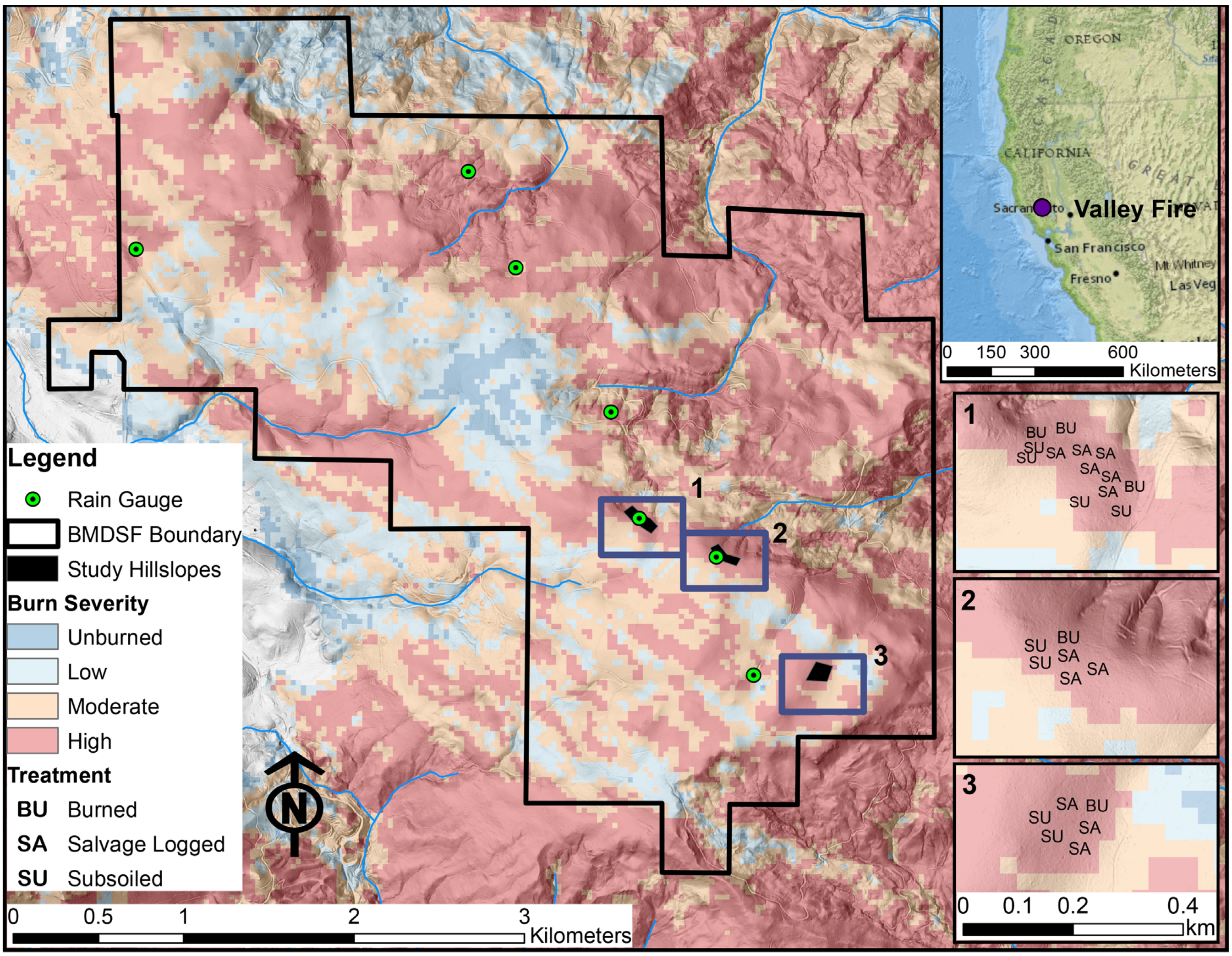

FIGURE 1 Burn severity map of Boggs Mountain Demonstration State Forest (BMDSF) with locations of rain gauges and silt fences. The upper inset map shows the location of the 2015 Valley Fire in California. Inset maps 1, 2, and 3 show sediment fence locations with a two letter code representing plot management types, where BU = burned, SA = burned and salvage logged, and SU = burned, salvage logged, and subsoiled

Study hillslopes were located on the upper slopes of BMDSF between elevations of 1,050–1,113 m, with mean slopes of 22–31%, and NNW and ESE aspects. The geology is representative of the Clear Lake Volcanic Field and consists of a cap of igneous andesite and dacite at high elevations over sedimentary sandstone and mudstone at lower elevations. In a soil survey that occurred after the fire, soils at our sites were classified as deep, well-drained, xeric andisols with a sandy loam texture (Edinger-Marshall & Obeidy, 2016).

2.2 | Post-fire forest management and hillslope sediment sampling

We studied four hillslopes across BMDSF to investigate the effects of post-fire land management treatments on hillslope sediment yields. We selected study hillslopes that had similar slopes and were representative of high burn severity as determined by remote sensing and field surveys (Parsons, Robichaud, Lewis, Napper, & Clark, 2010). We constructed 25 sediment fences modified from the methods of Robichaud and Brown (2002) to trap eroded sediment from hillslope plots. All plots were bounded to the approximate dimensions of 5 m wide along the contour and 15 m downslope (i.e., 75 m^2). Plots were "bounded" by rock and ditch barriers to limit variability in plot contributing area. However, the actual contributing area to each fence varied slightly due to microtopographic features. The sediment fences were distributed equally across the four hillslopes, except for one extra burned-only fence that was initially installed as the basis for a fifth hillslope location that was not pursued because of later operational constraints. Thus, we installed sediment fences at the base of: (a) five burned-only plots (one per hillslope plus one extra plot), (b) 12 burned and salvage logged plots (three per hillslope), and (c) eight burned, salvage logged, and sub-soiled plots (two per hillslope). Our original study design included herbicide treatments on a subset of the salvage logged and sub-soiled hillslopes that were to occur following the physical post-fire management treatments. However, herbicide application was delayed, leading to an extended monitoring period for those plots. Thus, we considered only the physical (i.e., non-herbicide) treatments in this study and reduced the number of sediment fences in the salvage logged and sub-soiled plots to one replicate per hillslope in 2019 and 2020. While this created an unbalanced experimental design, the additional statistical power during the early part of the study was warranted. A companion study on the effects of the herbicide application will be completed after additional data collection.

Most of BMDSF was salvage logged approximately 1 year after the fire (June–September 2016), excluding the riparian corridors. Burned trees were primarily hand-felled and skidded to landings with wheeled or tracked skidders, but some areas were harvested using feller bunchers. About 1–2 months after salvage logging (August–October 2016), some hillslopes were sub-soiled, which is a practice of breaking up the soil with the objective of reducing compaction to facilitate increased infiltration. Sub-soiling was accomplished by using winged blades mounted to the rear of a tracked Caterpillar D7H tractor, which ripped through the soil surface and ploughed furrows ~30 cm deep along hillslope contours. Furrow depth decreased over time as ridges eroded and sediment deposited in furrows. Approximately 2 years after salvage harvest operations (April 7, 2018), we planted four ponderosa pine seedlings (2-year-old plugs) in each of the 25 study plots following the protocol established by BMDSF managers. While we re-planted seedlings to be representative of a typical post-fire management practices, they did not substantially increase vegetation cover on site during this study.

Following major storm events or rainy periods, we quantified the mass of sediment that eroded into each sediment fence. During the first wet season after the fire (November 2015–April 2016), we measured sediment yield—the total mass of dry sediment eroded per unit area of the hillslope plots—from the five burned-only plots prior to the post-fire forest management activities to provide important context for the subsequent years in our study. Over the following 4 years of the study, we were able to capture the eroded sediment from 17 accumulation periods. Sediment stored in the fences was physically removed and weighed using portable scales by field crews. We then collected representative sub-samples (~0.5 kg), which were dependent on the amount of sediment trapped by the fence, which were dried in the laboratory at 105°C for 24 hours. Field-measured masses were multiplied by the dry fraction and divided by the total plot area upslope of each fence (~75 m^2) to produce *whole plot sediment yields*. We also calculated an *effective area sediment yield* for each plot because the post-fire sub-soiling treatment created furrows along the contour of the hillslope; thus, reducing the area contributing runoff and sediment to those fences (Figure 2). As such, we calculated the effective area sediment yields for the sub-soiled plots by dividing the mass of dry sediment by a field-estimated contributing area to each sediment fence.

2.3 | Plot characterization

Rainfall near each hillslope was measured from November 2015 to May 2020 using tipping bucket rain gauges (Onset Computer Corporation, Bourne, MA, USA and Rainwise, Inc., Trenton, ME, USA) accurate to 0.25 mm. We used the rainfall data to calculate maximum 30-minute intensity (*i30*, mm hr^{-1}), storm duration (min), and total precipitation (mm) for each storm using RainMaxLaz software (R. Brown, US Forest Service, unpublished software). An individual storm event was defined if there was at least a 6 hour gap between rain gauge tips. We separated precipitation and other results by water years (WY), which span October 1 through September 30 of the index year. When gauges occasionally malfunctioned, we used multiple linear regression relationships (R^2 > 0.90) between rain gauges to fill the gaps in the data.

We quantified tree canopy cover immediately after the fire in 2015 and after salvage logging and sub-soiling in 2018 and 2020 using hemispherical photography in each plot (Chianucci & Cutini, 2012; Glatthorn & Beckschäfer, 2014). Photographs were taken with a digital single-lens reflex camera (Nikon P5000 or D7100; Nikon Corporation, Tokyo, Japan) with a circular fisheye lens (Nikon

FIGURE 2 Photos of (a) burned-only, (b) salvage logged, and (c) salvage logged and sub-soiled management types. Photos were taken during the second post-fire water year (March 2017). The dashed lines in panel (c) highlight ridges created by sub-soiling that prevented runoff and sediment from reaching fences. We estimated effective area sediment yields in sub-soiled plots based on the plot area from the sediment fence upslope to the first ridge

FC-E8, Nikon Corporation, Tokyo, Japan; or Sigma 4.5 mm, f/2.8 EX DC HSM) installed facing up on a level tripod 1 m above the ground (Origo, Calders, Nightingale, & Disney, 2017). In 2015, we took the photographs from the centre of the burned-only plots and evenly spaced across the hillslopes where logging and sub-soiling were planned, while in 2016 and 2018, we took the photographs from the centre of each plot. Photographs were taken during optimal lighting conditions, based on the protocols suggested by Zhang, Chen, and Miller (2005) and Glatthorn and Beckschäfer (2014), and processed using the HemiView Software (Delta-T Devices Ltd, Cambridge UK) to calculate percent tree canopy closure, which is "the proportion of the sky hemisphere obscured by vegetation when viewed from a single point" (Jennings, Brown, & Sheil, 1999). Photographs collected with the circular fisheye lens captured imagery more than 2.5 m beyond the camera locations. As such, "edge effects" in the photos due to plot adjacency precluded comparisons of tree canopy closure in burned-only plots from pre- and post-salvage logging periods.

We measured ground cover of the contributing area to the sediment fences starting at the beginning of the project and repeated measurements each year during the wet season. We used the point intercept method to quantify ground cover on three 1 m quadrats in each plot (Bonham, 2013). Cover categories included bare mineral soil, litter, wood (>10 mm diameter), gravel (>5 mm), rock (>25 mm), and live vegetation. We measured cover at two levels: cover at the ground surface (surface cover) and cover suspended above the ground surface (suspended cover), since cover may affect erosion either by interception of precipitation (suspended cover) or by detention of water and sediment moving via sheetflow or rilling (surface cover). For our analysis, we combined bare soil and gravel surface cover into one category to minimize observer influence on cover class distinction. In addition, we added surface and suspended components of vegetation and wood cover together to account for their full influence on hydrologic erosion.

Field saturated hydraulic conductivity (K_{fs}) was measured in the burned-only plots in 2016 and in all plots in 2018, 2019, and 2020 using a SATURO dual-head ring infiltrometer (METER Group, Inc., Pullman, WA, USA). We measured K_{fs} at one highly disturbed and one undisturbed point within each salvage logged plot and at the top of a ridge and the bottom of a furrow in each sub-soiled plot. Burned-only plots only had one K_{fs} measurement per sample period. The infiltrometer was inserted into the ground to a 5 cm depth, and we used bentonite clay to create a seal between the ring and the soil surface. Each infiltration measurement was run for 75 minutes with two cycles at a low pressure (0 or 5 cm) and a high pressure head (5 or 10 cm). We used the lower pressure heads when water infiltrated too quickly into soils at the higher pressure heads. Many infiltrometer trials resulted in non-detects of K_{fs} when the soil hydraulic conductivity exceeded the maximum quantifiable rate of the instrument. We replaced these non-detects with the maximum rate of K_{fs} that the infiltrometer could effectively measure (0.0319 cm s^{-1}). After each measurement of K_{fs}, we used a 5 cm core sampler to remove a volume of soil (86.75 cm^3) from depths of 0–5 and 5–10 cm. These soil samples were dried in the lab at 105°C for 24 hr, and then weighed to calculate bulk density (g cm^{-3}). We also measured soil water repellency at one point in the contributing area of each sediment fence using the water drop penetration time (WDPT) test under dry soil conditions at the mineral soil surface, 1 cm depth, and 3 cm depth in September 2016 and 2017 (DeBano, 1981). The duration of the tests was up to 300 s, and in cases where the water drop did not infiltrate during that time, we assigned 300 s as the observed value.

2.4 | Statistical analyses

We used linear mixed-effects models to compare annual sediment yields among management types within the same water year at both the whole plot and effective contributing area scales. Sediment yields were log-transformed to meet assumptions of normally distributed residual for parametric statistics. We used random effects to adjust our comparisons by plot and hillslope and allowed for unequal variances among management types. Pairwise comparisons between management types within each year were calculated using Tukey–Kramer adjustment (Driscoll, 1996). Linear mixed-effects models with standardized coefficients were used to compare the influence of bulk density, ground cover, canopy cover, and precipitation depth and intensity (maximum *i30*) on sediment yields aggregated over each sediment cleanout period. We used random effects to adjust our comparisons by plot and hillslope for each year and allowed for unequal variances between each storm.

We used generalized linear mixed-effects models of the binomial family to compare proportions of bare soil, wood, and vegetation ground cover among management types across all study years. We added an observation level random effect to the model to account for overdispersion. Pairwise comparisons between site types within each year were calculated using Tukey–Kramer adjustment.

To analyse the differences in mean bulk density, K_{fs}, and canopy closure, we used linear mixed-effects models with Tukey–Kramer adjustment for pairwise comparisons among management types within the same water year. Prior to analysis, we log-transformed K_{fs} and canopy closure data to meet the assumptions of our statistical models. We used random effects to adjust our comparisons by hillslope and allowed for unequal variances among management types. All statistical analyses were conducted using the R environment (R Core Team, 2020). We interpreted *p* values from all statistical analyses based on their strength of evidence against the null hypothesis, as suggested by Arsham (1988) and Sterne and Smith (2001). Linear mixed-effects models were created using the *nlme* package (Pinheiro, Bates, DebRoy, Sarkar,, & R Core Team, 2020), while generalized linear mixed-effects models were created using the *lme4* package (Bates, Mächler, Bolker, & Walker, 2015).

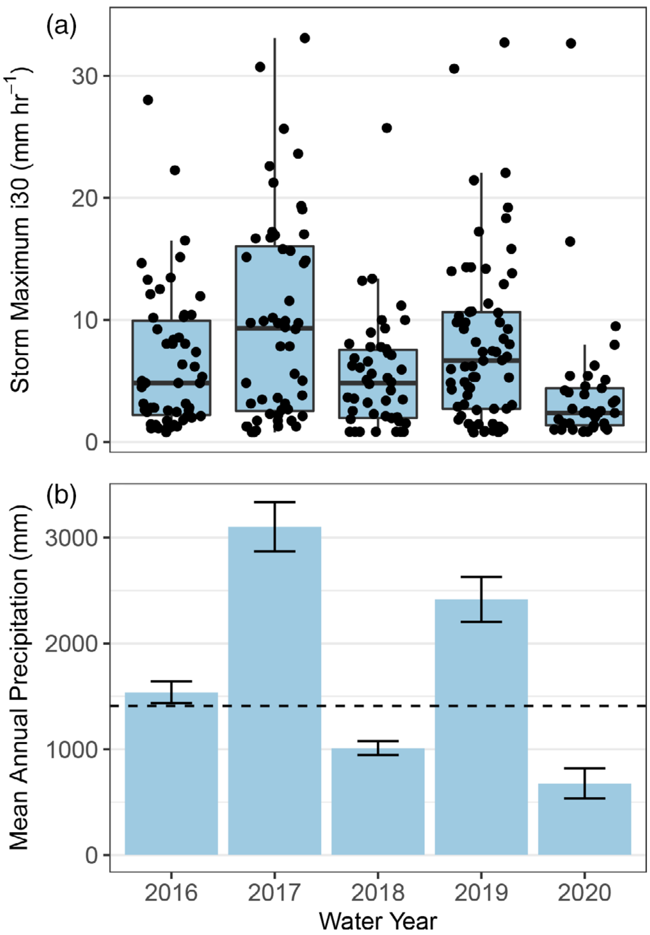

FIGURE 3 (a) Boxplots of the mean maximum 30-minute precipitation intensity for rain events occurring over the first five years after the 2015 Valley Fire. The boxplot central tendency line is the median, shaded boxes represent the interquartile range (IQR), whiskers represent the largest value up to 1.5-times the IQR, and the points show the individual data observations. (b) Mean annual precipitation and standard deviation for the three precipitation gauges. The horizontal dashed line in (b) indicates the long-term mean annual precipitation (1,408 mm) for the study site

3 | RESULTS

3.1 | Precipitation

Annual precipitation varied across the five water years of the study (Figure 3). While mean annual precipitation across gauges during the first (WY 2016; 1,511 ± 59 (*SD*) mm) and third year (WY 2018; 1,010 ± 65 mm) after the fire was close to the long-term average for the region, the annual precipitation in the second (WY 2017; 3,105 ± 231 mm) and fourth year (WY 2019; 2,417 ± 214 mm) were both greater than the long-term normal for the region. The fifth post-fire year (WY 2020; 676 ± 143 mm) had much lower annual precipitation than the long-term average. Moreover, maximum 30-min precipitation event intensities (*i30*) during WY 2016 (28 mm hr^{-1}) and 2018 (27 mm hr^{-1}) were lower than in WY 2017 (32 mm hr^{-1}), WY 2019 (35 mm hr^{-1}), and WY 2020 (42 mm hr^{-1}). In addition, the mean maximum *i30* ± *SD* was greatest in WY 2017 (8.7 ± 8.5 mm hr^{-1}), followed by WY 2019 (8.0 ± 7.0 mm hr^{-1}), WY 2016 (6.5 ± 5.8 mm hr^{-1}), WY 2018 (5.4 ± 4.6 mm hr^{-1}), and WY 2020 (4.1 ± 5.9 mm hr^{-1}) (Figure 3).

3.2 | Hillslope whole plot sediment yields

Pre-treatment geometric mean annual sediment yield ± *SD* from the burned-only plots during WY 2016 was 13.3 ± 4.6 Mg ha^{-1} yr^{-1} (Figure 4). Interestingly, mean annual plot sediment yield in the burned-only plots in the second year after the fire (WY 2017) was greater than the previous year (Figure 4), when the geometric mean annual whole plot sediment yield and 95% confidence interval (CI) was 26.4 [3.1, 223.5] Mg ha^{-1} yr^{-1}. Comparatively, the geometric mean annual sediment yield from the salvage logged plots was 6.4 [1.6, 25.5] Mg ha^{-1} yr^{-1} and from sub-soiled plots was 3.2 [0.6, 17.6] Mg ha^{-1} yr^{-1}. Statistically, there was strong evidence that the geometric mean annual sediment yields from the burned-only plots were greater than the salvage logged plots (t = 2.69, p = .025) and the sub-soiled plots (t = 3.72, p = .001). However, there was no statistical evidence for a difference in geometric mean annual sediment yields between the salvage logged and sub-soiled plots (t = 1.15, p = .29).

During the third year after the fire (WY 2018), geometric mean annual sediment yields were substantially lower in all plots (Figure 4). Specifically, the geometric mean annual sediment yield from the

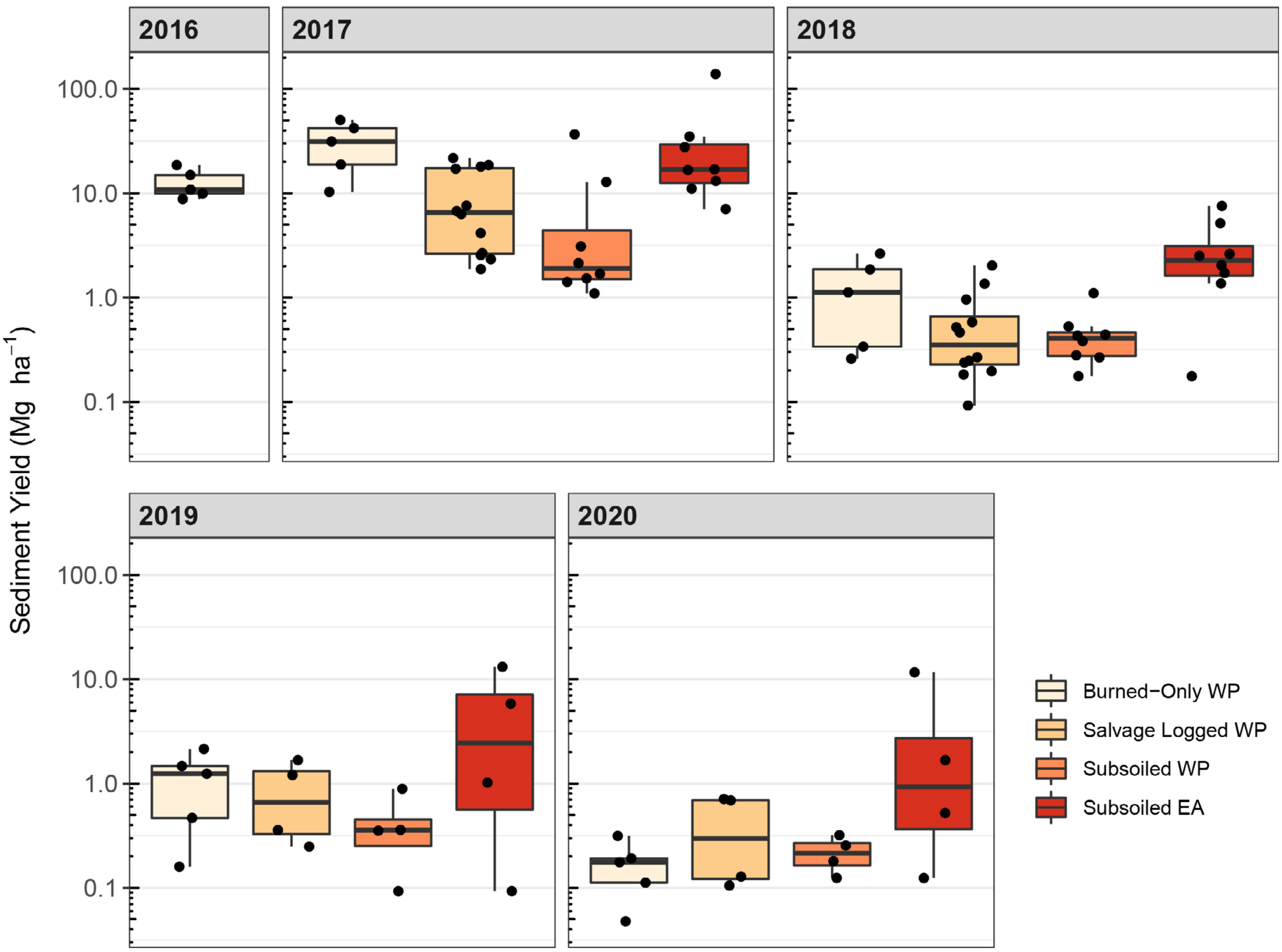

FIGURE 4 Annual whole plot (WP) and effective area (EA) sediment yields from silt fences in the burned-only, salvage logged, and subsoiled management types across five water years (WY 2016–2020). WP and EA sediment yields were the same in burned-only and salvage logged plots. Yields in WY 2016 were measured before post-fire management treatments (i.e., before salvage logging and subsoiling) were applied and the plots were installed. The number of plots was reduced to four in the salvage logged and subsoiled management types in WY 2019 and 2020 because some of the original plots were treated with herbicide as part of another study. The boxplot central tendency line is the median, shaded boxes represent the interquartile range (IQR), whiskers represent the largest value up to 1.5-times the IQR, and the points show the individual data observations

burned-only plots was 0.9 [0.1, 5.2] Mg ha^{-1} yr^{-1}, which was greater than from the salvage logged (0.4 [0.1, 1.3] Mg ha^{-1} yr^{-1}) and sub-soiled plots (0.4 [0.1, 1.6] Mg ha^{-1} yr^{-1}). Statistically, there was no evidence for a difference in geometric mean annual sediment yields between the burned-only and salvage logged plots (t = 1.71, p = .21), burned-only and sub-soiled plots (t = 1.66, p = .23), or the salvage logged and sub-soiled plots (t = 0.09, p = .996).

During the fourth year after fire (WY 2019), geometric mean annual whole plot sediment yields were similar to WY 2018 (Figure 4). Burned-only plots had a geometric mean annual sediment yield of 0.8 [0.1, 6.5] Mg ha^{-1} yr^{-1}, which was similar to the salvage logged plots (0.7 [0.1, 6.9] Mg ha^{-1} yr^{-1}), but greater than the sub-soiled plots (0.3 [0.0, 3.4] Mg ha^{-1} yr^{-1}). As with WY 2018, there was no statistical evidence for a difference in geometric mean annual sediment yields between the burned-only and salvage logged plots (t = 0.28, p = .96), burned-only and sub-soiled plots (t = 1.36, p = .37), or the salvage logged and sub-soiled plots (t = 1.03, p = .56).

During the fifth year after fire (WY 2020), geometric mean annual sediment yields were lower than WY 2019 (Figure 4). Burned-only plots had a geometric mean annual sediment yield of 0.1 [0.0, 0.7] Mg ha^{-1} yr^{-1}, which was less than salvage logged plots (0.3 [0.0, 1.8] Mg ha^{-1} yr^{-1}) and sub-soiled plots (0.2 [0.0, 1.3] Mg ha^{-1} yr^{-1}). There was no statistical evidence for a difference in geometric mean annual sediment yields between the burned-only and salvage logged plots (t = –1.37, p = .36), burned-only and sub-soiled plots (t = –0.74, p = .74), or the salvage logged and sub-soiled plots (t = 0.60, p = .82).

3.3 | Effective area sediment yields

The reduced contributing hillslope area to each sediment fence in the sub-soiled plots led to higher sediment yields in those plots, but there were no changes in sediment yields in the other management types. In WY 2017, the geometric mean of the annual effective area sediment yield from the sub-soiled plots was 21.1 [1.9, 236.5] Mg ha^{-1} yr^{-1} (Figure 4). We found strong evidence that the geometric mean of the annual effective area sediment yields from the burned-only plots was greater than salvage logged plots (t = 2.98, p = .01). However, there was no statistical evidence for a difference in geometric mean values between the burned-only and sub-soiled plots (t = 0.36, p = .93) or salvage logged and sub-soiled plots (t = −2.10, p = .10).

During the third year after the fire (WY 2018), the geometric mean of the annual effective area sediment yield from the sub-soiled plots was 2.0 [0.2, 22.0] Mg ha^{-1} yr^{-1} (Figure 4). Statistically, there was strong evidence that the geometric mean of the annual effective area sediment yields was greater from the sub-soiled plots than the salvage logged plots (t = −2.10, p = .01). Comparatively, there was no evidence that the geometric mean of the effective area sediment yields was different between the burned-only and salvage logged plots (t = 1.60, p = .50) or between the burned-only and sub-soiled plots (t = −1.29, p = .41).

During the fourth year after the fire (WY 2019), the geometric mean of the annual effective area sediment yield from the sub-soiled plots was 1.6 [0.1, 50.1] Mg ha^{-1} yr^{-1} (Figure 4). Statistically, there was no evidence that the geometric mean of the annual effective area sediment yields was different among any of the site types (burned-only vs. salvage logged: t = 0.30, p = .95; burned-only vs. sub-soiled: t = −0.91, p = .63; salvage logged vs. sub-soiled: t = −1.10, p = .52).

During the fifth year after the fire (WY 2020), geometric mean of the annual effective area sediment yield from the sub-soiled plots was 1.1 [0.0, 32.3] Mg ha^{-1} yr^{-1} (Figure 4). Statistically, there was moderate evidence that the geometric mean of the annual effective area sediment yields was lower in burned-only plots than sub-soiled plots (t = −2.49, p = .04), but there was no statistical evidence for a difference among the other treatments (burned-only vs. salvage logged: t = −1.15, p = .49; salvage logged vs. sub-soiled: t = −1.55, p = .27).

3.4 | Ground cover and canopy closure

In the first year after the fire (WY 2016)—before application of post-fire land management treatments—we measured a mean proportion of exposed bare soil ± *SD* of 78 ± 11% (Table 1) in the burned-only plots. In the second post-fire year (WY 2017), exposed bare soil had decreased in the burned-only plots; however, there was ∼1.4-times more bare soil in the burned-only plots than salvage logged plots and ∼ 1.6-times more than sub-soiled plots. Statistically, there was strong evidence that burned-only plots had more bare soil than salvage logged plots (t = 3.50, p = .002) and sub-soiled plots (t = 4.77, p < .001); however, there was no evidence for a difference in bare soil between salvage logged and sub-soiled plots (t = 1.81, p = .17) (Table 1). In WY 2018, we observed greater exposed bare soil than the previous year, particularly in salvage logged and sub-soiled plots. There was strong statistical evidence that burned-only plots had ∼1.4-times more bare soil than sub-soiled plots (t = 2.65, p = .026), but there was no statistical evidence that burned-only plots had different bare soil than logged plots (t = 1.62, p = .24) (Table 1). By WY 2019, there was no statistical evidence for a difference in bare soil across management types, and this continued in WY 2020 (t = .28–1.81, p = .17–.96) (Table 1).

Due to the high severity of the Valley Fire and lack of vegetative recovery, we measured no vegetation cover during the first year after the fire (WY 2016) in the burned-only plots (Table 1). In the second post-fire year (WY 2017), vegetation cover had started to return in all plots; however, in the burned-only plots, there was ∼2.4-times more vegetation cover than in the salvage logged plots and ∼ 5.7-times more than in the sub-soiled plots. Statistically, there was strong evidence that the vegetation cover in the burned-only plots was greater than the sub-soiled plots (t = 3.68, p = .001). There was suggestive evidence for a difference in vegetation cover between burned-only and salvage logged (t = 2.19, p = .08), but no evidence for a difference

TABLE 1 Proportion of ground cover for three cover classes in the burned-only, salvage logged, and subsoiled management types in the first five years after the 2015 Valley Fire

		Percent cover by water year				
Ground cover	Management type	2016	2017	2018	2019	2020
Bare soil	Burned-only	78 ± 11	64 [56, 71]	66 [55, 76]	65 [53, 75]	55 [43, 67]
	Salvage logged		47 [41, 52]	56 [47, 64]	60 [47, 72]	38 [26, 51]
	Sub-soiled		39 [33, 45]	47 [37, 57]	58 [44, 70]	47 [34, 60]
Wood	Burned-only	4 ± 4	5 [3, 7]	6 [3, 11]	10 [5, 16]	9 [5, 15]
	Salvage logged		20 [16, 25]	22 [15, 29]	42 [28, 59]	32 [20, 48]
	Sub-soiled		31 [25, 38]	32 [22, 43]	46 [30, 62]	27 [16, 41]
Vegetation	Burned-only	0 ± 0	10 [5, 19]	14 [5, 33]	37 [16, 64]	15 [6, 35]
	Salvage logged		4 [3, 7]	7 [3, 14]	37 [15, 67]	11 [4, 30]
	Sub-soiled		2 [1, 3]	4 [1, 10]	26 [10, 55]	15 [5, 37]

Note: Values for water year 2016, which were measured prior to post-fire management activity and the salvage logged and subsoiled plots were installed, are means and standard deviations. Values in all other water years are estimated proportions and 95 % confidence intervals from generalized linear mixed models. For water years 2016–2018, the number of plots was n = 5 in the burned-only, n = 12 in the salvage logged, and n = 8 in the subsoiled management types. Due to herbicide application, we reduced the number of plots in water years 2019–2020 to n = 4 in the salvage logged and n = 4 in the subsoiled management types.

between salvage logged and sub-soiled plots (t = 2.01, p = .12) (Table 1). In WY 2018 and WY 2019, we observed a slightly higher proportion of vegetation cover in all plots compared with previous years (Table 1). Interestingly, vegetation cover was markedly lower in all management types in WY 2020 compared to the previous years but remained similar across management types (Table 1). During WY 2018 to WY 2020, there was no evidence for differences in vegetation cover across all three plot types (t = –.36–1.95, p = .13–1.0).

In the first post-fire year (WY 2016), before any post-fire management activities, we measured mean wood cover ± *SD* of 4 ± 4% in the burned-only plots. Across all other years of the study (WY 2017 to WY 2020), we observed ~3.6- to 4.5-times more wood cover in the salvage logged plots compared to the burned-only plots and ~4.8- to 6.5-times more wood cover in the sub-soiled plots compared to the burned-only plots (Table 1). Statistically, mean wood cover in the burned-only plots was lower than both the salvage logged (t = 3.64–6.40, p < .0015) and sub-soiled plots (t = 3.03–8.21, p < .01) during WY 2017 to WY 2020. However, we only observed greater wood cover in the salvage logged plots compared to the sub-soiled plots (t = –2.73, p = .02) in the second year of the study (WY 2017).

In November 2015, before any post-fire salvage logging took place, mean canopy closure ± *SD* in burned-only plots was 61.3 ± 2.8%, which was similar to canopy closure in sites subsequently salvage logged and sub-soiled (60.6 ± 2.7%). Measurements of canopy closure during WY 2018 illustrated strong differences across the treatment types. Mean canopy closure and 95% CI was greater in the burned-only plots (15.9 [4.2, 59.9] %) compared to both the salvage logged (4.7 [1.1, 19.9] %) and sub-soiled (4.4 [0.9, 21.1] %) plots. There was strong statistical evidence that the canopy closure was greater in the burned-only plots compared to the salvage logged (t = 8.45, p < .001) and sub-soiled plots (t = 6.77, p < .001). However, there was no evidence for differences in canopy closure between the salvage logged and sub-soiled plots (t = 0.23, p = .97). Canopy closure in WY 2020 was similar to WY 2018. Mean canopy closure and 95% CI was greater in the burned-only plots (13.7 [3.6, 51.1] %) compared to both the salvage logged (3.6 [0.8, 15.2] %) and sub-soiled (3.5 [0.7, 16.8] %) plots. There was strong statistical evidence the canopy that closure was greater in the burned-only plots compared to the salvage logged plots (t = 9.27, p < .001) and sub-soiled plots (t = 7.20, p < .001), but there was no statistical evidence for a difference between the salvage logged and sub-soiled management types (t = 0.05, p > .99).

3.5 | Soil bulk density, hydraulic conductivity, and water repellency

Mean bulk density at 0–5 cm soil depth during WY 2018 was lowest in burned-only plots followed by the sub-soiled plots and the salvage logged plots (Table 2). Statistically, there was strong evidence for a pairwise difference in bulk density at 0–5 cm soil depth between the salvage logged plots and both the burned-only (t = –3.46, p = .002) and sub-soiled plots (t = 2.55, p = .032). There was no evidence for a difference in mean bulk density between the burned-only and sub-soiled plots (t = –1.16, p = .48). Mean bulk density was higher at 5–10 cm soil depth in all three site types (Table 2), and there was no evidence that the bulk density at 5–10 cm soil depth was different in the burned-only plots compared to either the salvage logged plots (t = –0.92, p = .63) or the sub-soiled plots (t = 0.91, p = .63). However, there was suggestive evidence that the mean bulk density at 5–10 cm soil depth was greater in the salvage logged plots compared to the sub-soiled plots (t = 2.19, p = .078).

Mean bulk density at 0–5 cm soil depth in WY 2019 was greatest in the sub-soiled plots, followed by salvage logged and burned-only management types (Table 2). Statistically, there was strong evidence for a pairwise difference in mean bulk density at 0–5 cm soil depth between the burned-only plots and sub-soiled plots (t = –2.67, p = .023), but there was no evidence for a difference between burned-only and salvage logged plots (t = –1.06, p = .54) or between salvage logged and sub-soiled plots (t = –2.08, p = .10). As with 2018, mean bulk density at 5–10 cm soil greater than at the surface in the burned-only and salvage logged sites (Table 2). However, there was no statistical evidence for a difference in mean bulk density at 5–10 cm soil depth among the three site types (burned-only vs. salvage logged: t = –0.28, p = .96; burned-only vs. sub-soiled: t = –0.56, p = .84; salvage logged vs. sub-soiled: t = –0.37, p = .93).

Interestingly, mean bulk density at 0–5 cm soil depth in WY 2020 was greater in all plots compared to WY 2019 or 2018 and there was no longer any statistical evidence for differences in mean bulk density at 0–5 cm or 5–10 cm soil depths among the three site types (p ≥ .30).

The mean field saturated hydraulic conductivity (K_{fs}) ± *SD* in the burned-only plots before the post-fire management in WY 2016 was 455 ± 267 mm hr^{-1}. In WY 2018, the geometric mean of K_{fs} in the burned-only plots was ~1.8-times greater than the salvage logged plots, and ~ 1.1-times greater than the sub-soiled plots (Table 3). However, there was no statistical evidence for differences in K_{fs} between any of the three management types (p ≥ .31). There was still no statistical evidence for a difference in K_{fs} between any of the management types in WY 2019 (p ≥ .69). In WY 2020, burned-only plots had geometric mean of K_{fs} ~ 1.7–1.8-times greater than salvage logged or sub-soiled plots (Table 3), but again there was no statistical evidence for differences in K_{fs} between any of the management types (p ≥ .27).

Overall, soil water repellency ranged from slight to strongly persistent (Doerr et al., 2006) in the first year after the fire (WY 2016). Specifically, in the burned-only plots, the soil surface was determined to be wettable with mean WDPT ± *SD* of 1 ± 0 s, but there was slight to strong water repellency at 1 cm (44.5 ± 93.6 s), 3 cm (102.3 ± 136.4 s), and at 5 cm (30.4 ± 70.6 s) depth in the soil. In the second year after the fire (WY 2017), soils in the burned-only plots were determined to be wettable at the soil surface (<1 s) and 1 cm depth (1.7 ± 2.7 s)—water repellency remained only slightly persistent (12.4 ± 44.1 s) at 3 cm depth. Soil water repellency was similar in the plots with the post-fire land management treatments. For example, soils in the salvage logged plots were wettable at the soil surface

TABLE 2 Mean soil bulk density (g cm^{-3}) and 95 % CIs over two depths (0–5 cm, 5–10 cm) from the burned-only, salvage logged, and subsoiled management types for water years 2018–2020

		Bulk density (g cm^{-3}) by water year		
Soil depth	Management type	2018	2019	2020
0–5 cm	Burned-only	0.68 [0.50, 0.85]	0.73 [0.56, 0.91]	0.83 [0.65, 1.00]
	Salvage logged	0.82 [0.71, 0.94]	0.78 [0.66, 0.89]	0.87 [0.76, 0.98]
	Sub-soiled	0.73 [0.59, 0.87]	0.86 [0.72, 1.00]	0.88 [0.74, 1.02]
5–10 cm	Burned-only	0.83 [0.65, 1.00]	0.83 [0.66, 1.00]	0.90 [0.73, 1.07]
	Salvage logged	0.87 [0.75, 0.98]	0.84 [0.73, 0.96]	0.88 [0.77, 1.00]
	Sub-soiled	0.79 [0.65, 0.92]	0.86 [0.72, 1.00]	0.83 [0.69, 0.97]

TABLE 3 Geometric mean saturated hydraulic conductivity (K_{fs}; mm hr^{-1}) and 95 % CIs from burned-only, salvage logged, and subsoiled plots for water years 2016 and 2018–2020

	K_{fs} (mm hr^{-1}) by water year			
Management type	2016	2018	2019	2020
Burned-only	455 ± 267	845 [180, 3,960]	111 [24, 519]	586 [125, 2,750]
Salvage logged		458 [128, 1,640]	95 [33, 273]	353 [123, 1,014]
Sub-soiled		767 [187, 3,150]	80 [26, 246]	319 [104, 974]

Note: Values for water year 2016, which were measured prior to post-fire management activity, are means and standard deviations. Comparisons between years are not appropriate due to poor instrument reliability.

(<1 s) and 1 cm depth (4.4 ± 30.5 s) but were slightly water repellent (11.3 ± 52.4 s) at 3 cm depth. Soils in the sub-soiled plots were wettable at the soil surface (<1 s), 1 cm (<1 s), and at 3 cm depth (<1 s).

3.6 | Relative importance of factors controlling sediment yield

As expected, precipitation was a dominant factor influencing the whole plot sediment yields throughout our study. The highest sediment yields across all three plot types generally occurred during storm periods with the greatest precipitation depth (β [standardized coefficient] = 1.13, t = 12.5, p < .001) and storms with higher maximum 30-min precipitation intensity (β = 0.24, t = 4.21, p < .001). Statistically, there was also strong evidence for higher sediment yields from plots with more exposed bare soil (β = 0.28, t = 2.53, p = .014) and greater canopy closure (β = 0.42, t = 4.38, p < .001), which were consistently the burned-only plots. There was no evidence that bulk density at 0–5 cm depth influenced sediment yields (β = 0.12, t = 1.27, p = .21).

4 | DISCUSSION

4.1 | Sediment yields

Contrary to our expectation, sediment yields following the 2015 Valley Fire in the northern California Coast Range were lower from hillslopes that had been salvage logged relative to hillslopes that were burned but not actively managed after the fire. Specifically, the mean annual plot sediment yields from the burned-only plots were 4.1-times greater than salvage logged plots during the second year after the fire (WY 2017) and 2.3-times greater in the third year after the fire (WY 2018). However, by the fourth and fifth years after the fire (WY 2019 and 2020), there were no differences in mean annual sediment yields between burned-only and salvage logged plots.

Our observations were surprising given that the majority of studies have observed 1.6- to 100-times greater sediment yields from salvage logged areas compared to sites that were burned but not actively managed (Malvar et al., 2017; Slesak et al., 2015; Spanos et al., 2005; Wagenbrenner et al., 2015). Moreover, other studies investigating hillslope rill development, in-stream turbidity, or sediment concentration have also found evidence for greater erosion and sediment transport from salvage logged hillslopes compared to burned hillslopes (Klock, 1975; Lewis et al., 2019; Smith, Sheridan, Lane, & Bren, 2011; Wagenbrenner et al., 2016). Elevated sediment yields after salvage logging have been attributed to increased ground disturbance, decreased infiltration capacity, and reduced surface cover, ultimately leading to more surface runoff and erosion (Malvar et al., 2017). Few studies have reported no change or decreased sediment yields from salvage logged hillslopes (Olsen, 2016; Wagenbrenner et al., 2015). However, in a similar study in northern California, sediment yields in the first year after the Ponderosa Fire were ~2.8-times greater from burned swales compared to salvage logged swales (James & Krumland, 2018). Lower sediment yields from the salvage logged sites in that study were attributed to reduced smoothness of the slope and higher levels of wood and litter cover, which facilitated increased infiltration capacity and hillslope sediment detention (James & Krumland, 2018). In our study, wood cover was more than five times greater on the salvage logged sites compared to the burned-only sites and may have functioned analogously to straw mulch, mitigating hillslope erosion by detaining sediment (Foltz &

Wagenbrenner, 2010; Prats et al., 2012; Prats, Malvar, Coelho, & Wagenbrenner, 2019; Prats, Wagenbrenner, Martins, Malvar, & Keizer, 2016; Wagenbrenner et al., 2006). It is also notable that the James and Krumland (2018) study, similar to ours, occurred in a region dominated by sandy loam andisols derived from quaternary volcanic parent material, which can exhibit rapid infiltration (Jefferson, Grant, Lewis, & Lancaster, 2010), but also provide an abundance of loose, erodible materials with relatively low cohesion when slope-stabilizing vegetation is absent (Esposito et al., 2017; Rodriguez, Guerra, Gorrin, Arbelo, & Mora, 2002).

We also observed 8.3-times greater mean annual sediment yields from the burned-only plots compared to the sub-soiled plots in the second post-fire year (the first year after logging and sub-soiling). Comparatively, mean annual sediment yields from the burned-only plots were 2.3-times greater in the third post-fire year and 2.7-times greater in the fourth post-fire year compared to the sub-soiled plots. Despite these differences, there was no statistical evidence for differences in sediment yields between the sub-soiled and burned-only plots after the third post-fire year. At the whole plot scale, sediment yields from the sub-soiled plots were likely lower compared to the burned-only or salvage logged plots due to the ridge-furrow microtopography created by sub-soiling, which prevented sediment transport down the hillslope. While research on post-fire sub-soiling is limited, our results were consistent with the James and Krumland (2018) study, which illustrated ~10.2-times greater sediment yields from burned plots and ~3.6-times greater sediment yields from salvage logged plots relative to sub-soiled plots. The authors attributed the lower sediment yields from the sub-soiled plots to greater surface roughness in the headwater swales, which reduced sediment transport (James & Krumland, 2018). This is supported by laboratory experiments that have illustrated reduced soil erosion at the hillslope scale due to high soil surface roughness, which limited runoff velocity and sediment detachment, creating areas for sediment detention (Helming, Römkens, & Prasad, 1998; Römkens, Helming, & Prasad, 2002). While not directly analogous, post-fire studies from Portugal and Spain also found rip-ploughed hillslopes produced ~ two to five times less sediment than reference hillslopes (Fernández, Fontúrbel, & Vega, 2019; Malvar, Prats, Nunes, & Keizer, 2011). However, results from other sub-soiling experiments have not consistently demonstrated reduced sediment yields. For example, in a study of unburned eucalypt plantations in Brazil soil, loss was similar between hillslopes with and without contour sub-soiling (Padilha et al., 2018).

Interestingly, when we adjusted sediment yields for the effective contributing area, the sediment yields in the sub-soiled plots increased substantially. These results suggested that sub-soiling had two contrasting effects on post-fire soil erosion and sediment delivery. First, sub-soiling increased small-scale erosion by creating ridges of disturbed and available (erodible) soil, which also had local slopes that were steeper than the larger scale hillslope. However, sub-soiling also increased metre-scale roughness that limited sediment transport distances and provided areas for sediment detention. Due to the higher than expected sediment yields at the smaller spatial scales, sediment delivery probably would have been higher in our other management types if the slope lengths were comparable to the ridge spacing in the sub-soiled plots (de Vente, Poesen, Arabkhedri, & Verstraeten, 2007; Shakesby & Doerr, 2006; Wagenbrenner & Robichaud, 2014). However, the difference between our whole plot and effective area sediment yields from the sub-soiled plots exceeded the expected differences related to differences in slope length or area alone. Thus, sub-soiling can have a paradoxical role in temporarily increasing hillslope erosion by increasing soil erodibility, while simultaneously decreasing downslope delivery by increasing roughness and soil detention.

However, it remains uncertain whether post-fire sub-soiling can be effective at mitigating hillslope sediment yields over the longer-term. For example, we observed instances of sediment breakthroughs outside our measurement plots where sediment over-filled the storage capacity of lower furrows, leading to elevated sediment delivery down the hillslope. Moreover, in WY 2017, one of the sub-soiled plots over-filled and experienced sediment breakthrough, leading to effective area sediment yields that were ~2.8-times greater than the burned-only plots. Thus, while our results suggest that sub-soiling could be effective at disconnecting post-fire hillslope sediment from streams, the elevated effective area sediment yields and observations of breakthroughs suggest that additional research is necessary to improve our understanding of the efficacy of this post-fire land management approach.

Unsurprisingly, post-fire sediment yields were highly variable both spatially and temporally. We recorded much higher sediment yields in the burned-only plots in the second post-fire water year (26.4 Mg ha^{-1}) than in the first (13.3 Mg ha^{-1}) or third post-fire water years (0.9 Mg ha^{-1}). Furthermore, the two burned-only plots that produced the highest (55.7 Mg ha^{-1}) and lowest (11.4 Mg ha^{-1}) mean annual whole plot sediment yields were spatially located within 20 m of each other. Earlier studies have also recorded a wide range in mean post-fire sediment yields. For example, in the first year after the 2010 Twitchell Canyon fire in Utah, the annual hillslope sediment yields ranged from 6.8 to 103.5 Mg ha^{-1} (Robichaud, Storrar, & Wagenbrenner, 2019). Several studies in Colorado have observed hillslope sediment yields ranging from 6.0 to 112.7 Mg ha^{-1} in the first year after wildfire (Larsen et al., 2009; Rengers, Tucker, Moody, & Ebel, 2016; Schmeer, Kampf, MacDonald, Hewitt, & Wilson, 2018; Wagenbrenner et al., 2006). In general, variability of sediment yields may be related to a range of factors, including regional soil erodibility, hillslope topography, fire severity, post-fire weather and precipitation erosivity, hydrologic connectivity of hillslopes to sediment fences, sediment fence design, and spatial scale of measurement (Abrahams, Kaste, Ouimet, & Dethier, 2018; Benavides-Solorio & MacDonald, 2005; Boix-Fayos et al., 2007; Vieira, Fernández, Vega, & Keizer, 2015).

4.2 | Principal drivers of sediment yields

In our study, we found precipitation depth to be the most important driver of sediment yields. This was not surprising since fluvial processes are often the primary sediment transport mechanism during the wet season in post-fire environments (Cerdà & Robichaud, 2009; Doerr et al., 2006; Johansen, Hakonson, & Breshears, 2001; Wondzell & King, 2003). Precipitation depth had a greater influence on sediment yields than 30-minute precipitation intensity, despite previous research suggesting precipitation intensity may be more important in controlling surface runoff and erosion after fire (Ebel, Moody, & Martin, 2012; Kampf, Brogan, Schmeer, MacDonald, & Nelson, 2016; Moody & Ebel, 2014; Moody & Martin, 2001a). These differences may be due to the dominance of lower intensity frontal storms from the Pacific Ocean at our sites as compared to convective storms, which are common in the interior west and have been linked to high rates of runoff and erosion in burned forests (Kunze & Stednick, 2006; Robichaud, Wagenbrenner, et al., 2008; Wagenbrenner & Robichaud, 2014). In addition, periods between plot cleanouts ranged from weeks to months, and sediment data were aggregated into periods of events between cleanouts, which may have limited our ability to resolve the impact of any individual high-intensity precipitation event on sediment yields.

Counter-intuitively, we also found strong evidence that canopy closure in the burned-only plots contributed to the greater sediment yields measured on these plots. Generally, rain splash detachment and overland flow have been noted as the dominant processes driving erosion and sediment delivery on burned hillslopes, and we observed evidence of these processes on hillslopes from all management types (Rengers et al., 2016). However, we also observed larger raindrops underneath the burned snags, apparently due to interception, temporary detention, and coalescing of the drops in the residual canopy before becoming throughfall. As such, the raindrops under the burned canopy likely fell with greater kinetic energy (Geißler et al., 2012) and, therefore, impacted the soil with greater force than in salvage logged or sub-soiled sites where the tree canopy had been removed. Our observation, and this theory, was also supported by the presence of substantial erosion pedestals underneath the burned tree canopies (Figure 5), which were not present in either the salvage logged or sub-soiled plots. Dunkerley (2020) also noted the formation of soil splash pedestals after wildfires in Australia, which were facilitated by the co-occurrence of increasing mass of drops that accumulated on the defoliated tree branches and the presence of bare soil. Furthermore, a rainfall simulation experiment by Prats, Malvar, and Wagenbrenner (2020) at one of our study sites found that the erosion rates from the burned-only plots were only 56% of the values in the skid trails. While their skid trail plot condition did not replicate our salvage logged plots, the substantially lower sediment yields from the burned-only plots compared to the skid trail plot with the same precipitation rates and drop sizes applied to both conditions support our theory that drop size magnification can explain the higher sediment yields in our burned-only plots.

The proportion of bare soil on the study hillslopes was also an important factor influencing sediment yields. During the first 4 years of our study, the proportion of bare soil in the burned sites was ~1.1- to 1.6-times greater than in both the salvaged logged and sub-soiled management types (Table 1). While these were comparatively small

FIGURE 5 Erosion pedestals near the trunk of a standing snag, indicative of raindrop splash erosion, which may have been an important driver of high sediment yields in burned-only plots

differences, they were statistically meaningful with strong, positive relationships between bare soil and sediment yields across the management types. This result was expected, given that high levels of bare soil have previously been related to rain splash detachment of soil particles (Rengers et al., 2016; Zavala, Jordan, Gil, Bellinfante, & Pain, 2009) and higher sediment yields (Benavides-Solorio & MacDonald, 2001; Schmeer et al., 2018; Stoof et al., 2015; Wagenbrenner et al., 2006). Some studies have found threshold or other non-linear responses between bare soil and soil erosion in burned forests, with a notable increase in erosion when the proportion of bare soil reached or exceeded ~60–70% (Davenport, Breshears, Wilcox, & Allen, 1998; Johansen et al., 2001; Spigel & Robichaud, 2007). Interestingly, the proportion of bare soil in the burned-only plots was within this range of possible threshold behaviour during the first 4 years after the fire, but only observed during 1 year in the salvage logged plots.

Sediment yields also appeared to be strongly governed by the presence of wood cover (e.g., twigs, branches, tree trunks) on the soil surface. Across the 5 years of our study, the salvage logged and sub-soiled sites had 3.6- to 6.5-times more wood cover than the burned-only sites. Increased wood cover was expected as others have previously noted elevated surface wood loads during the first 5 years after post-fire salvage logging (Donato et al., 2006; Peterson, Dodson, & Harrod, 2015). The presence of wood on the soil surface likely increased surface roughness and slowed erosion and downslope sediment movement in the salvage logged sites. This finding is comparable to previous research, which has illustrated declines in post-fire sediment erosion associated with the presence of wood and wood mulches (Prats, Gonzalez-Pelayo, et al., 2019; Robichaud, Lewis, Wagenbrenner, Brown, & Pierson, 2020). However, the longer-term efficacy of wood for reducing post-fire hillslope erosion remains uncertain, as Leverkus et al. (2020) found that the effect of fine wood (twigs and branches <7.6 cm diameter) on the soil surface had largely disappeared after approximately 5 years. Recent observations from our sites indicated that, in some cases, sediment had filled the storage areas upslope of wood pieces on the salvaged logged sites, suggesting that these pieces will be ineffective at storing additional sediment over the longer-term.

Contrary to our expectation, there was no evidence that the differences in sediment yields across our management types were driven by differences in soil bulk density. The highest soil bulk densities were found in the sites that were actively managed after fire; however, these sites generally had the lowest sediment yields, suggesting other factors were more important in driving the differences in erosion. Indeed, in the third year after the fire (WY 2018), soil bulk density was only 1.3-times greater in the salvage logged plots and 1.1-times greater in the sub-soiled plots compared to the burned-only plots. While the differences were small, they were consistent with previous studies that have illustrated a $\sim$ 1.2–1.4-times increase in bulk density associated with tracked logging equipment, falling trees, or skidding logs along the ground surface typical during post-fire salvage logging (Garcia-Orenes et al., 2017; Malvar et al., 2017; Parkhurst, Aust, Bolding, Barrett, & Carter, 2018; Wagenbrenner et al., 2015). Laboratory experiments have demonstrated soil bulk density influences rill formation, with higher bulk densities leading to fewer and shorter rills to transport sediment (Hieke & Schmidt, 2013). Furthermore, bulk density is often related to the amount of energy required to detach sediment particles from the soil surface during concentrated overland flow (Ghebreiyessus, Gantzer, & Alberts, 1994). Compacted soils also tend to have lower infiltration rates due to decreased macro-porosity (Kozlowski, 1999; Luce, 1997; Prats, Gonzalez-Pelayo, et al., 2019). If infiltration is substantially decreased, soil compaction can lead to increased runoff and erosion (Batey, 2009; Reynolds, Hessburg, Miller, & Meurisse, 2011). However, in comparison to these previous studies, the absolute bulk density values were lower at our sites, which are characteristic of andisols developed from volcanic parent material (Takahashi & Shoji, 2002). As a result, we did not find any evidence that soil bulk density contributed to differences in plot sediment yields.

Similarly, soil hydraulic properties, including soil water repellency and field saturated hydraulic conductivity, were not key factors driving differences in sediment yields across the site types. During the first year after the Valley Fire, the burned soils demonstrated slight to strongly persistent water repellency in the upper soil horizons (upper 5 cm). However, after a second winter, in which there was above-average precipitation, soil water repellency was only slightly present at 5 cm depth, while most soil layers became wettable, and the water repellency was similar among management types. Likewise, statistically, we found no evidence for differences in field saturated hydraulic conductivity (K_{fs}) between any of the management types. In addition, within-year differences in K_{fs} did not correspond with the differences in sediment yield, suggesting that either K_{fs} was not a good indicator of runoff generation or that other processes besides infiltration and overland flow more strongly controlled the sediment delivery responses. Indeed, enhanced soil water repellency and decreased hydraulic conductivity and infiltration are commonly noted after wildfires and have been linked to elevated surface runoff and erosion (Chen, McGuire, & Stewart, 2020; Doerr, Ritsema, Dekker, Scott, & Carter, 2007; Ebel & Moody, 2017; Woods, Birkas, & Ahl, 2007). However, our results agree with others who have observed strong declines in repellency in areas that have been salvage logged or where soils were disturbed post-fire (Bryant, Doerr, Hunt, & Conan, 2007; Wagenbrenner et al., 2015, 2016).

While our K_{fs} observations were highly relative to several studies that have quantified post-fire soil hydraulic properties (Ebel & Moody, 2017), they are consistent with observations from coarse-textured soils (Balfour, 2015) and several post-fire studies in locations with macro-porous soils capable of rapid infiltration (Blake et al., 2010; Nyman, Sheridan, Smith, & Lane, 2011; Sheridan, Lane, & Noske, 2007). Interestingly, despite high K_{fs}, both Sheridan et al. (2007) and Nyman et al. (2011) also observed high erosion rates, which were attributed to interrill processes, as well as localized locations of high rill erodiblity. We posit similar processes may have been present across our study hillslopes. In addition, our qualitative observations suggested that rainsplash erosion due to the direct impact of raindrops directly on soil particles may have been an important erosion mechanism at our sites, similar to other post-fire studies (Prats, Gonzalez-Pelayo, et al., 2019; Williams et al., 2020). Finally, the potential disconnect in the spatial scale of the experimental tools used to quantify K_{fs} compared to natural precipitation events could also have hindered our ability to capture the range of hillslope-scale soil hydraulic properties, which would have driven the erosional processes.

5 | CONCLUSIONS

As a result of shifting wildfire regimes observed in many regions across the planet, it is becoming increasingly important to understand the effects of wildfire on processes driving erosion and sediment delivery from hillslopes to streams, as well as when and where different post-fire land management approaches are likely to be effective at mitigating wildfire impacts on soil and water resources. In our study, sediment yields from burned hillslopes in the first year after the 2015 Valley Fire in northern California were $\sim$13.3 Mg ha^{-1} yr^{-1}. During the second year after the fire, sediment yields in the burned-only plots nearly doubled to 26.4 Mg ha^{-1} yr^{-1} because of greater precipitation inputs. However, sediment yields decreased through the remainder of the five-year study and were comparable to yields from other studies using similar-sized plots to measure sediment from high severity wildfires across the western US in areas with diverse geological and climatic settings.

Knowledge on the efficacy of post-fire land management approaches at mitigating erosion and sediment transport from hillslopes to streams remains limited. Our study provided surprising evidence of lower sediment yields from salvage logged and sub-soiled hillslope plots compared to burned and unmanaged plots during the first 3 years after the fire. While the evidence suggested that post-fire management resulted in lower rates of erosion and sediment delivery at the hillslope spatial scale, these results contradict many previous studies and must be interpreted with caution. Our results suggest that land managers and logging operators could potentially limit hillslope erosion when salvage logging, at least in the short-term, by distributing logging slash across harvested areas to detain sediment, rather than concentrating logging residues on landings. Similarly, our results also suggest that sub-soiling parallel to the hillslope contour may increase surface roughness by creating ridges and furrows and reduce sediment yields during the first several years after fire; however, these benefits may be short-lived in highly erodible locations. Salvage logging also appeared to have the unexpected consequence of reducing the kinetic energy of precipitation relative to the burned-only plots, where the rain appeared to coalesce on the branches of the standing snags, resulting in larger drop sizes. It is uncertain whether this is unique to regions with Mediterranean climates and persistent, low-intensity precipitation and this requires additional investigation. Due to the seemingly contradictory findings of this study, additional research is needed on the comparative effects of post-fire land management approaches, particularly to improve our understanding of the mechanisms driving post-fire erosion and sediment delivery.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are in preparation for submission to a US Forest Service data portal where they will be archived and publicly available.

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SUPPORTING INFORMATION

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