

AN ABSTRACT OF THE THESIS OF

Ryan P. Cole for the degree of Master of Science in Water Resources Science presented on March 6, 2020

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Abstract approved: _____

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The extent and severity of wildfires in forested regions are increasing throughout many regions on the planet, including western North America. High-severity wildfires directly affect soils and vegetation by altering soil hydraulic properties, reducing soil organic matter, exporting carbon and nitrogen, and killing trees and understory vegetation. These impacts can increase runoff, erosion, and sediment delivery to streams, producing a range of impacts on terrestrial and aquatic ecosystems and downstream source water quantity and quality. Due to the broad range of post-fire threats, land managers often undertake active post-fire land management (e.g., salvage logging, subsoiling, revegetation) to promote forest regeneration and maintain forest and aquatic ecosystem functions. However, the magnitude and longevity of effects from wildfire and the various post-fire land management approaches remains uncertain.

Here, I present results from a study, which quantified and compared sediment yields eroded from (a) burned, (b) burned and salvage logged, (c) burned, salvage logged, and subsoiled, (d) burned, salvage logged, and pre-emergent herbicide, (e) burned, salvage logged, and foliar herbicide, and (f) burned, salvage logged, subsoiled, and pre-emergent herbicide plots in the northern California Coast Range. We constructed 25 sediment fences on four hillslopes that burned at high severity in the 2015 Valley Fire. We quantified sediment yields from the upslope contributing area to each silt fence (~75 m²). We also quantified ground cover,

precipitation, soil properties, and canopy cover from areas representative of each plot. In the second year after the fire, sediment yields were ~4- to 10-times greater in the burned plots compared to the salvage logged or subsoiled plots. By the third and fourth years after fire, there was no statistical evidence for a difference in sediment yields among the six site types. In general, we observed higher sediment yields across all site types following precipitation events with greater amounts or intensities. Our results suggested sediment yields were greater from sites with greater canopy closure and more exposed bare soil. Salvage logging operations initially increased soil bulk density, but by the fourth year, bulk density was similar among all site types. Sediment yields in all site types decreased over the course of the study and stabilized by the fourth post-fire year, which may be indicative of site recovery or exhaustion of easily mobile sediment.

We also present results from a related study, in which we quantified differences in soil carbon, nitrogen, carbon/nitrogen (C:N) ratios, and vegetation recovery from three of the post-fire management strategies, including: (a) burned, (b) burned and salvage logged, and (c) burned, salvage logged, and subsoiled sites. We collected 180 soil samples from two depths (0–5 cm and 5–10 cm) and 27 vegetation samples across sites burned at high severity representative of the three post-fire management strategies. We also collected samples to quantify soil carbon, nitrogen, and C:N ratios in eroded sediment from hillslope sediment fences. The eroded sediment and soils in the burned sites had C:N ratios ~9–22 % lower than in the salvage logged or subsoiled sites. Interestingly, both carbon and nitrogen concentrations were greater in the more heavily disturbed sites. In other words, C and N concentrations in soils were generally ranked as: burned, salvage logged, and subsoiled > burned and salvage logged > burned. Vegetation biomass in burned sites was ~2.7-times greater than salvage logged and ~6-times greater than

subsoiled sites. Coarse wood cover in our erosion plots was positively correlated with C:N ratios from our eroded sediment, but vegetation cover was not correlated with carbon, nitrogen, or C:N ratios. These results suggest that salvage logging may increase soil C:N ratios by increasing the coarse wood input to the forest floor, which may negatively impact vegetation recovery three years after fire by suppressing plant available nitrogen. Post-fire forest regeneration is critically important to increase carbon storage and clean water supply in disturbed forest ecosystems, so forest management practices should not exacerbate sediment transport or limit vegetation recovery.

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Post-fire Forest Management Activities Alter Soil Properties, Sediment Yields, and Vegetation
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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request.

Ryan P. Cole, Author

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1 Introduction

Wildfire activity has increased dramatically in western North America according to recent analysis of burned area (Flannigan *et al.*, 2009; Moritz *et al.*, 2012; Reilly *et al.*, 2017; Westerling, 2016). In addition, fire seasons have expanded earlier in the fall and later in the spring, with some regions of the Earth, including parts of western North America, now prone to catastrophic wildfires year-round (Westerling, 2016; Westerling and Bryant, 2008). Increased wildfire activity in the western US has been linked to changing climate, past fire suppression, and increased forest stocking (Abatzoglou and Williams, 2016). Expansion of human development into the wildland-urban interface may also make these fires more destructive to human property (Mann *et al.*, 2014). Despite these shifts in the fire regime, dendrochronological evidence suggests forested landscapes in the western US have the potential to burn at greater rates than we are currently experiencing (Murphy *et al.*, 2018). Wildfire is an important disturbance shaping landscapes and ecosystems (Keeley *et al.*, 2011), but high-severity fire can negatively impact forest ecosystems and human infrastructure linked to ecosystem services forests provide (Emelko *et al.*, 2011; Keeley and Brennan, 2012; Vukomanovic and Steelman, 2019).

Fire impacts on soils and hillslope processes often amplify runoff and sediment delivery from burned forests can also lead to changes in stream geomorphology (Shakesby and Doerr, 2006), community structure of aquatic ecosystems (Arkle *et al.*, 2010; McCormick *et al.*, 2010), and water supplies for downstream communities (Bladon *et al.*, 2014; Emelko *et al.*, 2011; Smith *et al.*, 2011). Ash and sediment transported in streams after wildfire can impair water quality for many in-stream species including fish and aquatic invertebrates by reducing dissolved oxygen

and pH (Bixby *et al.*, 2015; Minshall, 2003). Furthermore, post-fire flooding can scour stream substrates and remove organisms, while flooding or sediment deposition can modify physical stream habitat (Bixby *et al.*, 2015). Sediment, ash and organics can also impair drinking water quality downstream of burned areas and create costly challenges for the drinking water treatment process (Bladon *et al.*, 2014; Hohner *et al.*, 2017, 2019).

Additionally, fire directly reduces ecosystem stocks of carbon and nitrogen through combustion and volatilization, (Bormann *et al.*, 2008; Grier, 1975; Neary *et al.*, 2005; Urbanski, 2014), while after fire carbon and nitrogen are exported from burned forests via high rates of sediment transport, leaching and increased soil respiration (Robichaud *et al.*, 2016; Smith *et al.*, 2011; Wondzell and King, 2003). Soil carbon and nitrogen are critically important predictors of forest productivity across the globe (Edmonds and Chappell, 1994; LeBauer and Treseder, 2008; McLaughlin and Phillips, 2006), so loss of these nutrients may negatively impact forest recovery after wildfire. Post-fire salvage logging may limit nutrient availability by removing wood, a critical source of carbon and nitrogen in burned forests (Marañón-Jiménez *et al.*, 2013; Marañón-Jiménez and Castro, 2013). In addition, post-fire salvage logging may inhibit recovery of soil microbes and depress genes related to nitrogen cycling, further impacting soil nutrient cycling in the critical post-fire period (Busse *et al.*, 2006; Pereg *et al.*, 2018).

Land managers have the critical job of mitigating these post-fire threats to water and soils. Active post-fire management often includes emergency hillslope stabilization, erosion mitigation, and vegetation restoration to promote regeneration and maintain ecosystem function (Leverkus *et al.*, 2018; Robichaud *et al.*, 2000). Post-fire salvage logging is a common management approach that is purported to recover economic value from burned timber resources, improve forest worker and visitor safety, reduce woody fuel loads, mitigate pest

outbreaks, and facilitate post-fire revegetation efforts (Donato *et al.*, 2013; Malvar *et al.*, 2017; Müller *et al.*, 2018). Some efforts, such as mulching or grass seeding, combine with salvage logging practices to limit erosion in post-fire environments (Foltz and Wagenbrenner, 2010; Prats *et al.*, 2019; Schmeer *et al.*, 2018). Plowing furrows along hillslope contour may be used 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 and Lowery, 1988; Robichaud *et al.*, 2000; Will and Jacobson, 2002). Land managers may plant trees to promote forest recovery and apply herbicide to suppress competition from non-native or emergent understory vegetation (Munson *et al.*, 2015; Ouzts *et al.*, 2015; Powers and Ferrell, 1996; Powers and Reynolds, 1999).

However, scientific research into post-fire forest management strategies is limited, leading to continued debate about potential benefits and tradeoffs (Dellasala *et al.*, 2006; Donato *et al.*, 2006; Leverkus *et al.*, 2012; McIver and Starr, 2000). For example, post-fire salvage logging may reduce woody fuel loads in burned forests (Donato *et al.*, 2013; Peterson *et al.*, 2015), but post-fire forest management has also been linked to increased re-burn severity (Thompson *et al.*, 2007). Furthermore, while removing standing and downed large wood may eliminate important structural components that can help facilitate the recovery of terrestrial and aquatic systems (Lindenmayer and Noss, 2006; Maia *et al.*, 2014; May and Gresswell, 2003), post-fire forest management practices can support hydrologic recovery in some ecosystems (Niemeyer *et al.*, 2020). Salvage logging on burned soils has the potential to compound site disturbance and soil compaction (McIver and Starr, 2001; Wagenbrenner *et al.*, 2016), leading to enhanced 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 *et al.*, 2015). Many post-fire management practices are purported to ameliorate the impacts of fire (McIver and Starr, 2000), but research on the short- and long-term effects is limited, especially in mixed-conifer forests of the northern California Coast Range.

Given recent trends towards larger, more severe wildfires in western North America and across the globe, it is crucial to improve our understanding of the efficacy of post-fire forest management approaches at mitigating erosion and facilitating vegetation recovery. The objective of this research was to understand the effects of post-fire forest management on erosion, soil hydraulic properties, soil carbon and nitrogen, and vegetation recovery on hillslopes burned at high severity. To accomplish this we set up two studies on hillslopes burned at high severity by the Valley Fire in northern California Coast Range. We present the results of these studies in the following chapters:

In Chapter 2, we used data from 25 sediment fences on four burned and salvage logged hillslopes at Boggs Mountain Demonstration State Forest to compare impacts of five post-fire management treatments on sediment yields, soil physical and hydraulic properties, and ground cover compared to a burned and unlogged reference during the first four years (2016–2019) after the 2015 Valley Fire. For the first three post-fire years, we assessed the impact of two physical post-fire management site types: salvage logging, and salvage logging followed by subsoiling. In the fourth post-fire year, we assessed the additional impact of herbicide on salvage logged plots, and salvage logged and subsoiled plots. Our study quantified the differences in sediment yields among management site types and analyzed the principal drivers controlling erosion at our site.

In Chapter 3, we present our study investigating the effects of wildfire and post-fire land management on key soil nutrients and vegetation recovery. As part of that study, we collected 360 soil samples for analysis of total carbon and total nitrogen, and 27 vegetation samples from a

burned reference and two physical post-fire management site types (salvage logged and subsoiled) on one burned hillslope. We also collected sediment subsamples for total carbon and total nitrogen from our sediment fences during water years 2018 and 2019. We used these data to analyze impacts of post-fire forest management on carbon, nitrogen, and carbon/nitrogen (C:N) ratios in soils and eroded sediment. We also attempted to link carbon and nitrogen in soils and sediments with vegetation cover, vegetation biomass, and downed wood cover to determine post-fire forest management impacts on soil nutrients and vegetation recovery.

In Chapter 4, we synthesize the overall findings from our two data chapters and draw conclusions from our research about the range of potential impacts of wildfire and post-fire forest management on sediment yields, soil physical and hydraulic properties, soil carbon and nitrogen, and vegetation recovery. We also discuss potential limitations of our research and recommend modifications to our research approach if these studies were repeated in different burned forests. Finally, we provide some future research directions regarding the effects of wildfire and the efficacy of post-fire forest management.

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

2.1 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 *et al.*, 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 *et al.*, 2018). As such, concerns have grown regarding the immediate and longer-term effects on forest resilience and the water supply originating in forests (Hallema *et al.*, 2018a; Stevens-Rumann *et al.*, 2018; Wagenbrenner *et al.*, In review). Increasingly severe wildfires have produced substantial and long-lasting effects on annual streamflow and peak flows (Hallema *et al.*, 2018b; Niemeyer *et al.*, 2020; Saxe *et al.*, 2018), debris flows (Langhans *et al.*, 2017; Nyman *et al.*, 2015), physical and chemical water quality (Rhoades *et al.*, 2018; Rust *et al.*, 2018), aquatic ecosystem health (Bixby *et al.*, 2015; Emelko *et al.*, 2016), and downstream drinking water supply (Emelko *et al.*, 2011; Hohner *et al.*, 2017).

Such impacts on water supply are attributable to the complex and interactive effects of wildfires on soil water-repellency, soil organic matter, canopy and litter interception, root reinforcement, and soil hydraulic properties (Ebel and Moody, 2017; Robichaud *et al.*, 2016). In turn, these impacts often result in faster runoff response, increased surface runoff generation, erosion, and sediment delivery to streams (Helvey, 1980; Malmon *et al.*, 2007; Moody and Martin, 2001b; Neary *et al.*, 2005). Amplified runoff and sediment delivery can lead to changes

in stream geomorphology (Shakesby and Doerr, 2006), community structure of aquatic ecosystems (Arkle *et al.*, 2010; McCormick *et al.*, 2010), and water supplies for downstream communities (Bladon *et al.*, 2014; Emelko *et al.*, 2011; Smith *et al.*, 2011b).

Due to the broad range of post-fire threats, land managers often undertake active post-fire land management (e.g., emergency stabilization, rehabilitation, and restoration) in an attempt to promote regeneration and maintain forest and aquatic ecosystem functions (Leverkus *et al.*, 2018; Robichaud *et al.*, 2000). Salvage logging 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 *et al.*, 2013; Malvar *et al.*, 2017; Müller *et al.*, 2018). Land managers may also apply additional treatments to mitigate effects and promote vegetation recovery. For example, plowing furrows along the contours of hillslopes (subsoiling) 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 and Lowery, 1988; Robichaud *et al.*, 2000; Will and Jacobson, 2002). Similarly, contour-felled logs, straw wattle, and hand-dug contour trenches have been used as erosion barriers to mitigate post-wildfire runoff and erosion (Robichaud *et al.*, 2008a, 2008b). Land managers have also seeded grasses or planted trees to facilitate vegetation recovery on burned hillslopes (Ouzts *et al.*, 2015; Wagenbrenner *et al.*, 2006). Occasionally, herbicides are applied to reseeded or replanted hillslopes to suppress competition from non-native vegetation or emergent understory vegetation (Munson *et al.*, 2015; Powers and Ferrell, 1996; Powers and Reynolds, 1999).

Limited scientific 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.*, 2012; McIver and 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 and Noss, 2006; Maia *et al.*, 2014; May and Gresswell, 2003). Moreover, salvage logging in recently burned areas has the potential to create additional site disturbance and soil compaction (McIver and Starr, 2001; Wagenbrenner *et al.*, 2016). These further effects on the soil can 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 *et al.*, 2015). Similarly, the additional ground disturbance associated with subsoiling or the suppression of ground cover due to herbicide application could also increase soil exposure to runoff events and increase erosion rates (Benavides-Solorio and MacDonald, 2001; Robichaud *et al.*, 2010; Wagenbrenner *et al.*, 2015).

Given the recent trends towards larger, more severe wildfires in many regions, including western North America, 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 erosion and runoff (McIver and 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 (Donato *et al.*, 2006; Lewis *et al.*, 2018; Malvar *et al.*, 2017; Silins *et al.*, 2009; Slesak *et al.*, 2015), but we are only aware of one study assessing the effectiveness of either subsoiling or herbicide application after salvage logging (James and Krumland, 2018). Further research on these practices will improve understanding of the

hydrogeomorphic effects of different post-fire land management strategies to help facilitate informed land management decisions. Thus, our primary objectives were to quantify differences in hillslope sediment yields among (a) burned, (b) burned and salvage logged (salvage logged), (c) burned, salvage logged, and subsoiled (subsoiled) (d) burned, salvage logged, and pre-emergent herbicide application (early herbicide), and (e) burned, salvage logged, and foliar herbicide application (late herbicide) plots through the first four years after the 2015 Valley Fire in Northern California. Our secondary objective was to quantify the effect of the post-fire management site types on some of the primary runoff and erosion controlling variables, including precipitation characteristics, ground cover, canopy cover, soil bulk density, soil hydraulic properties, and soil water-repellency in order to understand how post-fire forest management influenced erosion.

2.2 Methods

2.2.1 *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 (Figure 2.1). 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). It is located about 10 kilometers southwest of Clear Lake, CA, in the northern Coast Range (38°50'00.07" N, 122°42'05.33" 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.

The climate of the region is Mediterranean with warm dry summers and cool, wet winters (Köppen Csb). Rainfall dominates the precipitation, though there are occasional snow events and transient snowpack during winters. Mean annual precipitation is 1,408 mm with the majority falling between October and April (PRISM Climate Group, Oregon State University, <http://prism.oregonstate.edu>).

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.

Mean slopes of Boggs Mountain are about 27 %, which are relatively shallow compared to surrounding mountains. The elevation ranges from 700 to 1,144 m. The geology consists of a cap of igneous andesite and dacite at high elevations over sedimentary sandstone and mudstone at lower elevations. After the fire, soils were classified as deep, well-drained, xeric andisols with a sandy loam texture (Marshall and Obeidy, 2016).

2.2.2 *Post-fire management*

We targeted four hillslopes (“blocks”) across BMDSF to investigate the effects of different post-fire land management treatments on hillslope sediment yields. These 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. We constructed 25 sediment fences modified from the methods of Robichaud and Brown (2002) to trap eroded sediment from hillslope plots. Each “plot” was approximately 5 m wide by 15 m long with a sediment fence constructed at the

downslope end to catch sediment transported during storm events, though the exact contributing area to each fence varied due to microtopographic features.

Most of BMDSF was salvage logged approximately one year after the fire, excluding unburned forest and riparian corridors. Burned trees were primarily hand-felled and skidded to landings with wheeled or tracked skidders and cable yarders. About one to two months after salvage logging, some hillslopes were subsoiled using winged blades mounted to the rear of a tracked Caterpillar D7H crawler tractor. The subsoiling blades ripped through the soil surface and plowed furrows into the soil along hillslope contour, and the wings extended lifted the soil adjacent to the blades, with the intent of increasing the porosity and laterally extending the effect of the subsoiling.

Approximately two years after salvage harvest operations (March 30, 2018) hexazinone herbicide (trade name: Velpar) was applied to four salvage logged and four subsoiled erosion plots to represent an “early” or “pre-emergent” herbicide treatment. Soon after the hexazinone application (June 22, 2018), four salvage logged plots received application of a glyphosate-based herbicide (trade name: Roundup) to represent a “late” or “foliar” herbicide treatment.

Approximately two years after salvage harvest operations (April 7, 2018), we planted four ponderosa pine seedlings (two-year-old plugs) in each of our study plots.

Following major storm events or rainy periods, we quantified the mass of sediment that eroded into each sediment fence. Over the four years of the study, we were able to capture the eroded sediment from 15 accumulation periods. Sediment stored in the fences was collected and weighed in the field. We then collected sub-samples of the sediment (~0.5 kg), which were returned to the laboratory for analysis. We dried the subsamples at 105 °C for 24 hours to determine the water content and then multiplied the field masses by the dry fraction to calculate

dry sediment masses. Dry sediment masses were converted to sediment yields by dividing the sediment mass by the total bounded area upslope of each fence. The post-fire land subsoiling treatment along the contour of the hillslope created furrows within these plots, which effectively reduced the contributing hillslope area to each of those sediment fences. As such, we also calculated effective area sediment yields by dividing the mass of dry sediment by a field-estimated contributing area to each sediment fence.

2.2.3 Plot Characterization

Rainfall near each block was measured from October 2016 to July 2019 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 (i_{30}), 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 identified if there was at least a six hour gap between rain gauge tips. When gauges occasionally malfunctioned, we used multiple linear regression relationships ($R^2 > 0.90$) between rain gauges to fill gaps in the data.

We quantified tree canopy cover before and after salvage logging using hemispherical photography in each plot (Chianucci and Cutini, 2012; Glatthorn *et al.*, 2014). A Nikon D7100 camera (Nikon Corporation, Tokyo, Japan) with a circular fisheye lens was installed facing up on a level tripod 1 m above the ground (Origo *et al.*, 2017). We took the hemispherical photographs from the center of each sediment fence contributing area during optimal lighting conditions. Digital photographs were processed using the Gap Light Analyzer (GLA) version 2.0 (Simon Fraser University, Burnaby, B.C., Canada; Institute of Ecosystem Studies, Millbrook, New York,

USA) to calculate percent tree canopy closure directly above the sediment fence contributing area.

We measured surface cover of the contributing area to the sediment fences starting at the beginning of the project and repeated measurements annually during the wet season. We used the point intercept method to quantify surface 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.

Surface soil samples were collected with a bulk density core sampler in two locations within the contributing area of each sediment fence and from two soil depths (0–5 and 5–10 cm). The soil core samples were dried in the lab at 105 °C for 24 hours and weighed to calculate bulk density. We measured soil water repellency 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). Field saturated hydraulic conductivity (K_{fs}) was measured using a SATURO infiltrometer (METER Group, Inc., Pullman, WA, USA) with a 5 cm insertion depth. We used bentonite clay to create a seal between the insertion ring and the soil surface. During the 2018 sampling period many infiltrometer trials resulted in non-detects of K_{fs} when the soil hydraulic conductivity exceeded the maximum quantifiable rate of the instrument. We filled these gaps in the dataset with the maximum rate of K_{fs} that the infiltrometer could effectively measure (0.0319 cm s⁻¹).

2.2.4 Statistical Analyses

We used linear mixed effects models to compare sediment yields among site types within the same water year at both the plot and effective contributing area scales. Sediment yields were log transformed to meet assumptions of our statistical model. We used random effects to adjust our comparisons by plot and block and allowed for unequal variances among site types. Pair-wise comparisons between site 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 principal drivers on erosion. We used random effects to adjust our comparisons by plot and block, allowed for unequal variances between groups, and used an exponential correlation structure to account for repeated measures through time.

We also used linear mixed effects models to compare ground cover among site types with block and plot as random effects and unequal variances among site types. Pair-wise comparisons between site types within each year were calculated using Tukey-Kramer adjustment (Driscoll, 1996). To analyze differences between site types in means for bulk density, precipitation intensity, hydraulic conductivity, and canopy closure we used linear regression with Tukey-Kramer adjustment for pairwise comparisons (Driscoll, 1996). All statistical analyses were conducted using R programming environment (R Core Team, 2018), and linear mixed effects models were created using the *nlme* package (Pinheiro *et al.*, 2018).

2.3 Results

2.3.1 Hillslope Sediment Yields

During the first wet season after the 2015 Valley Fire (WY 2016) in northern California, we collected preliminary sediment samples, which were eroded from the burned-only plots. These samples were collected prior to the post-fire forest management activities to provide important context for the subsequent years in our study. The mean annual sediment yield \pm *SD* from the burned-only plots during the first wet season was 13.3 ± 4.6 Mg ha⁻¹ yr⁻¹ (Figure 2.2).

Interestingly, mean annual plot sediment yields in the second year after the fire (WY 2017) were higher than the previous year (Figure 2.2). The geometric mean annual plot sediment yield and 95 % confidence interval [CI] from the burned-only plots was 28.8 [8.3, 99.2] Mg ha⁻¹ yr⁻¹. Comparatively, the geometric mean annual plot sediment yield from the salvage logged plots was 6.8 [3.0, 15.4] Mg ha⁻¹ yr⁻¹ and from subsoiled plots was 3.0 [1.1, 8.0] Mg ha⁻¹ yr⁻¹. Statistically, there was strong evidence that the geometric mean annual plot sediment yields from the burned plots were greater than the salvage logged plots ($t = 3.15$, $p = 0.008$) and the subsoiled plots ($t = 4.63$, $p < 0.001$). However, there was little or no statistical evidence for a difference in geometric mean annual plot sediment yields between the salvage logged and subsoiled plots ($t = 2.11$, $p = 0.10$).

During the third year after the fire (WY 2018), geometric mean annual plot sediment yields were substantially lower in all plots (Figure 2.2). Specifically, the geometric mean annual plot sediment yield from the burned plots was 0.9 [0.3, 3.2] Mg ha⁻¹ yr⁻¹, which was greater than from the salvage logged (0.4 [0.2, 1.0] Mg ha⁻¹ yr⁻¹) and subsoiled plots (0.4 [0.1, 1.0] Mg ha⁻¹ yr⁻¹). Statistically, there was no evidence for a difference in geometric mean annual plot sediment

yields between the burned and salvage logged plots ($t = 1.71, p = 0.21$), burned and subsoiled plots ($t = 1.96, p = 0.14$), or the salvage logged and subsoiled plots ($t = 0.45, p = 0.89$).

During the fourth year after fire (WY 2019), geometric mean annual plot sediment yields were similar to WY 2018 (Figure 2.3). Burned plots had a geometric mean annual plot sediment yield of $0.9 [0.2, 3.2] \text{ Mg ha}^{-1} \text{ yr}^{-1}$, which was similar to the salvage logged plots ($0.7 [0.2, 3.2] \text{ Mg ha}^{-1} \text{ yr}^{-1}$), but greater than the subsoiled plots ($0.4 [0.1, 0.5] \text{ Mg ha}^{-1} \text{ yr}^{-1}$). WY 2019 was also the first year we were able to quantify herbicide effects on the 12 plots treated in this manner. The geometric mean annual plot sediment yield was greatest in the subsoiled and early herbicide plots ($1.3 [0.3, 5.5] \text{ Mg ha}^{-1} \text{ yr}^{-1}$), followed by the salvage logged and early herbicide plots ($1.1 [0.2, 4.6] \text{ Mg ha}^{-1} \text{ yr}^{-1}$), and the salvage logged and late herbicide plots ($0.8 [0.2, 3.3] \text{ Mg ha}^{-1} \text{ yr}^{-1}$). There was no statistical evidence for differences in geometric mean annual plot sediment yields in WY2019 between any of the site types ($F = 0.94, p = 0.48$).

Subsoiling created a ridge-furrow microtopography, with the ridges trapping sediment from the uphill sections of the plot, preventing it from reaching the sediment fence. To account for the change in contributing area to the sediment fences, we adjusted the plot sediment yields by the estimated contributing area to each fence. This created a new metric of erosion we termed effective area yields. In WY 2017, the geometric mean of the annual effective area sediment yield from the burned plots was $28.9 [7.9, 105.8] \text{ Mg ha}^{-1} \text{ yr}^{-1}$. Comparatively, the geometric mean of the annual effective area sediment yield from the salvage logged plots was $6.8 [3.0, 15.8] \text{ Mg ha}^{-1} \text{ yr}^{-1}$ and from subsoiled plots was $21.1 [7.6, 59.0] \text{ Mg ha}^{-1} \text{ yr}^{-1}$. We found strong evidence that the geometric mean of the annual effective area sediment yields from the burned plots were greater than salvage logged plots ($t = 3.15, p = 0.009$). Similarly, there was strong evidence that the geometric mean of the annual effective area sediment yields in the subsoiled

plots were greater than the salvage logged plots ($t = -2.59, p = 0.035$). However, there was no statistical evidence for a difference in geometric mean of the annual effective area sediment yields between the burned and subsoiled plots ($t = 0.59, p = 0.83$).

During the third year after the fire (WY 2018), the geometric means of the annual effective area sediment yields were substantially lower in all plots. Specifically, the geometric mean of the annual effective area sediment yield from the burned plots was 0.9 [0.3, 3.5] Mg ha⁻¹ yr⁻¹, which was greater than salvage logged plots (0.4 [0.2, 1.0] Mg ha⁻¹ yr⁻¹), but less than the subsoiled plots (2.0 [0.4, 5.5] Mg ha⁻¹ yr⁻¹). Statistically, there was strong evidence that the geometric mean of the annual effective area sediment yields was greater from the subsoiled plots than the salvage logged plots ($t = -3.49, p = 0.003$). Comparatively, there was no evidence that the geometric mean of the effective area sediment yields was different between the burned and salvage logged plots ($t = 1.70, p = 0.22$) or between the burned and subsoiled plots ($t = -1.39, p = 0.36$).

2.3.2 *Precipitation*

Annual precipitation varied across the four water years of the study. While annual precipitation during the first year (WY 2016) and third year (WY 2018) after the fire was close to the long-term average for the region, the annual precipitation in the second year (WY 2017) and fourth year (WY 2019) was greater than normal for the region. Mean annual precipitation \pm *SD* was 1,511 \pm 59 mm in WY 2016, 3,105 \pm 231 mm in WY 2017, 1,010 \pm 65 mm in WY 2018, and 2,417 \pm 214 mm in WY 2019. Moreover, precipitation event intensities during WY 2016 and 2018 were lower than in WY 2017 and WY 2019. Maximum thirty-minute precipitation intensity

($i30$) was 28.5 mm hr⁻¹ in WY 2016, 32.0 mm hr⁻¹ in WY 2017, 27.4 mm hr⁻¹ in WY 2018, and 35.1 mm hr⁻¹ in WY 2019. Additionally, the geometric mean $i30$ and 95 % confidence interval [CI] was greatest in WY 2019 (5.2 [4.0, 6.8] mm hr⁻¹), followed by WY 2016 (4.5 [3.3, 6.0] mm hr⁻¹), WY 2017 (4.1 [3.1, 5.5] mm hr⁻¹), and WY 2018 (3.8 [2.7, 5.2] mm hr⁻¹). Statistically, there was no evidence for differences in the geometric mean $i30$ among all years sampled ($F = 0.85, p = 0.47$).

2.3.3 Ground Cover and Canopy Closure

In the first year after the fire (WY 2016)—before application of post-fire land management treatments—we measured the exposed bare soil in each of the burned plots. The wildfire produced a mean bare soil fraction $\pm SD$ of 60.4 ± 4.6 %.

In the second year after the fire (WY 2017) the mean bare soil fraction and 95 % confidence interval [CI] in the burned plots was 49.9 [32.9, 67.0] %, which was greater than in both the salvage logged (33.7 [15.4, 52.0] %) and subsoiled plots (26.9 [11.7, 42.1] %) (Figure 2.4). There was strong evidence mean bare soil fraction was greater in the burned plots than the salvage logged plots ($t = 2.95, p = 0.005$) or subsoiled plots ($t = 5.15, p < 0.0001$). There was no statistical evidence for a difference in mean bare soil fraction between salvage logged and subsoiled plots ($t = 1.37, p = 0.18$).

In the third year after the fire (WY 2018), mean bare soil fraction was 54.3 [37.2, 71.3] % in burned plots, 42.9 [24.6, 61.3] % in salvage logged plots, and 39.9 [24.7, 55.1] % in the subsoiled plots (Figure 2.4). There was strong evidence mean bare soil fraction was greater in burned plots than salvage logged plots ($t = 2.03, p = 0.046$) and subsoiled plots ($t = 3.21, p =$

0.003). There was no statistical evidence for a difference in mean bare soil fraction between salvage logged and subsoiled plots ($t = 0.61$, $p = 0.544$).

In the fourth year after the fire (WY 2019), mean bare soil fraction was 33.3 [21.5, 45.1] % in burned plots, 43.5 [36.0, 51.0] % in salvage logged plots, and 40.4 [31.2, 49.6] % in subsoiled plots (Figure 2.5). There was no statistical evidence for pairwise differences in mean bare soil fraction between any of the post-fire management site types: burned vs. salvage logged ($t = -1.53$, $p = 0.30$); burned vs. subsoiled ($t = -1.00$, $p = 0.59$); salvage logged vs. subsoiled ($t = 0.54$, $p = 0.85$). WY 2019 also included the effects of herbicide treatments on ground cover in some of the salvage logged and subsoiled plots. Salvage logged plots that received an early application of herbicide (pre-emergent) had a mean bare soil fraction 22.6 % greater than the plots that received no herbicide application. Similarly, subsoiled plots that received the pre-emergent herbicide had a mean bare soil fraction 17.3 % greater than subsoiled plots that did not receive herbicide. .

Statistically, there was strong evidence that the bare soil fraction was greater on both the salvage logged sites ($t = -3.52$, $p = 0.008$) and the subsoiled sites ($t = -2.70$, $p = 0.04$) that received herbicide application relative to sites that did not receive herbicides. Comparatively, there was no statistical evidence for a difference in mean bare soil fraction between plots that received a late application herbicide (foliar) and those without herbicide application ($t = -1.72$, $p = 0.23$) or those that received the early herbicide application (pre-emergent) ($t = 1.80$, $p = 0.20$).

Measurements of canopy closure during WY 2018 also illustrated strong differences across the treatment types. Mean canopy closure and 95 % CI was greater in the burned plots (12.4 [8.1, 18.6] %) compared to both the salvage logged (1.3 [0.5, 2.9] %) and subsoiled (0.2 [0.01, 2.6] %) plots. There was strong statistical evidence the canopy closure was greater in the

burned plots compared to the salvage logged plots ($t = 4.95, p < 0.001$) and subsoiled plots ($t = 3.15, p = 0.014$). However, there was no evidence for differences in canopy closure between the salvage logged and subsoiled plots ($t = 1.37, p = 0.38$).

2.3.4 Soil Bulk Density, Hydraulic Conductivity, and Water Repellency

For WY 2018 mean average bulk density and 95 % CI at 0–5 cm soil depth was 0.65 [0.54, 0.76] g cm⁻³ in the burned plots, 0.83 [0.75, 0.90] g cm⁻³ in the salvage logged plots, and 0.73 [0.64, 0.82] g cm⁻³ in the subsoiled plots (Figure 2.6). Statistically, there was strong evidence for a pairwise difference in bulk density between the salvage logged plots and both the burned ($t = -4.62, p = 0.0005$) and subsoiled plots ($t = 2.78, p = 0.031$). There was no evidence for a difference in mean average bulk density between the burned and subsoiled plots ($t = -2.08, p = 0.12$).

Mean average bulk density in WY 2018 was higher at 5–10 cm soil depth in all three site types, but the relative differences among site types remained similar. The mean average bulk density at 5–10 cm soil depth was 0.83 [0.72, 0.93] g cm⁻³ in the burned-only plots, 0.86 [0.79, 0.94] g cm⁻³ in the salvage logged plots, and 0.79 [0.70, 0.88] g cm⁻³ in the subsoiled plots. There was no statistical evidence that the bulk density was different in the burned plots compared to either the salvage logged plots ($t = -0.97, p = 0.61$) or the subsoiled plots ($t = 0.95, p = 0.62$). However, there was suggestive evidence that the mean average bulk density was greater in the salvage logged plots compared to the subsoiled plots ($t = 2.31, p = 0.079$).

Mean average bulk density at 0–5 cm soil depth in WY 2019 was 0.75 [0.68, 0.82] g cm⁻³ in the burned plots, 0.80 [0.74, 0.86] g cm⁻³ in the salvage logged plots, and 0.87 [0.80, 0.94] g

cm^{-3} in the subsoiled plots. Statistically, there was strong evidence for a pairwise difference in mean average bulk density at 0–5 cm soil depth between the subsoiled plots and both the burned ($t = -4.57, p = 0.0006$) and salvage logged plots ($t = -3.07, p = 0.017$). There was no evidence for a difference in mean average bulk density between the burned and salvage logged plots ($t = -2.03, p = 0.13$).

Mean average bulk density at 5–10 cm soil depth in WY 2019 was 0.82 [0.75, 0.89] g cm^{-3} in the burned plots, 0.86 [0.80, 0.92] g cm^{-3} in the salvage logged plots, and 0.82 [0.75, 0.89] g cm^{-3} in the subsoiled plots. There was no statistical evidence for a difference in mean average bulk density at 5–10 cm soil depth among the three site types (burned vs. salvage logged: $t = -1.61, p = 0.27$; burned vs. subsoiled: $t = 0.04, p > 0.99$; salvage logged vs. subsoiled: $t = 1.73, p = 0.22$).

The mean field saturated hydraulic conductivity (K_{fs}) $\pm SD$ in the burned plots before the post-fire land management treatments was $0.0126 \pm 0.0074 \text{ cm s}^{-1}$. In WY 2018, the geometric mean of K_{fs} and 95 % confidence interval [CI] was 0.024 [0.0082, 0.068] cm s^{-1} in the burned treatment, 0.013 [0.0050, 0.032] cm s^{-1} in the salvage logged treatment, and 0.021 [0.0076, 0.060] cm s^{-1} in the subsoiled treatment. Statistically, there was no evidence for differences in K_{fs} between any site types (burned vs. salvage logged: $t = 1.41, p = 0.36$; burned vs. subsoiled: $t = 0.23, p = 0.97$; salvage logged vs. subsoiled: $t = -1.19, p = 0.47$). In WY 2019, geometric mean K_{fs} was substantially lower than 2018 in all site types. The geometric mean of K_{fs} was 0.0031 [0.0011, 0.0089] cm s^{-1} in the burned treatment, 0.0026 [0.0012, 0.0057] cm s^{-1} in the salvage logged treatment, and 0.0022 [0.00098, 0.0051] cm s^{-1} in the subsoiled treatment. Statistically, there was no evidence for differences in K_{fs} between any site types (burned vs. salvage logged: t

= 0.40, $p = 0.92$; burned vs. subsoiled: $t = 0.79$, $p = 0.72$; salvage logged vs. subsoiled: $t = 0.48$, $p = 0.88$).

Overall, soil water repellency ranged from slight to strongly persistent (Doerr *et al.*, 2006) in the first year after the fire (WY 2016), but soils were mostly wettable by the second post-fire year (WY 2017). In the first year after the fire, the soil surface was determined to be wettable with mean WDPT \pm SD of 1 ± 0 sec, but there was slight to strong water repellency at 1 cm (44.5 ± 93.6 sec), 3 cm (102.3 ± 136.4 sec), and at 5 cm (30.4 ± 70.6 sec) depth in the soil. In the second year, soils in the burned-only plots were determined to be wettable at the soil surface (<1 sec) and 1 cm depth (1.7 ± 2.7 sec)—water repellency remained only slightly persistent (12.4 ± 44.1 sec) at 3 cm depth. Soil water repellency was similar in the plots treated with the post-fire land management prescription compared to the burned plots. For example, soils in the salvage logged plots were wettable at the soil surface (<1 sec) and 1 cm depth (4.4 ± 30.5 sec) but were slightly water repellent (11.3 ± 52.4 sec) at 3 cm depth. Soils in the subsoiled plots were wettable at the soil surface (<1 sec), 1 cm (<1 sec), and at 3 cm depth (<1 sec).

2.3.5 *Relative importance of erosion controlling factors*

We found strong evidence that storm periods with higher precipitation depth recorded the highest rates of erosion—statistically precipitation depth was the most important driver of erosion across all three of the site types was depth with a standardized coefficient from our linear mixed effects model of 0.62 ($F = 61.7$, $p < 0.0001$). Not surprisingly, there was also very strong evidence that sites with greater canopy closure was had higher rates of erosion—statistically, it had a standardized coefficient of 0.45 ($F = 19.1$, $p = 0.0003$). There was also strong evidence

that sites with more bare soil (standardized coefficient = 0.39, $F = 33.9$, $p < 0.001$) and storm periods with higher maximum thirty-minute precipitation intensity (standardized coefficient = 0.32, $F = 21.5$, $p < 0.001$) also had higher erosion rates across the three site types. There was no evidence that soil bulk density was a factor influencing erosion rates across the three site types (standardized coefficient = 0.08, $F = 1.8$, $p = 0.18$).

2.4 Discussion

2.4.1 Sediment Yields

Contrary to our expectation, plot sediment yields following the 2015 Valley Fire in the northern Coast Range of California 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 plots were 4.2-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 year after fire (WY 2019), we did not measure any differences in mean annual sediment yields between burned 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 managed (Malvar *et al.*, 2017; Slesak *et al.*, 2015; Spanos *et al.*, 2005; Wagenbrenner *et al.*, 2015). Moreover, many other studies that have investigated hillslope rill development or 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.*, 2018; Smith *et al.*, 2011a; Wagenbrenner *et al.*, 2016). Few studies have reported

no change or decreased sediment yields from salvage logged hillslopes (Fernández and Vega, 2016; James and Krumland, 2018). Similar to our study, James and Krumland (2018) found ground cover in salvage logged swales had higher levels of wood and litter cover, a possible “mulch” that was created by logging activities. Fernandez and Vega (2016) reported mulching did more to limit soil erosion than salvage logging alone. Qualitatively, we also observed substantial wood and litter following salvage logging operations that may have functioned analogously to straw mulch, mitigating hillslope erosion (Foltz and Wagenbrenner, 2010; Prats *et al.*, 2012, 2016; Wagenbrenner *et al.*, 2006).

Comparatively, the mean annual sediment yields from the subsoiled plots were 9.6-times lower than burned plots in the second post-fire year and 2.3-times lower in the third post-fire year. However, there were no differences in sediment yields between the subsoiled and burned plots by the fourth post-fire year. At the plot scale, sediment yields were likely lower from the subsoiled plots compared to the burned or salvage logged plots due to the ridge-furrow microtopography created by subsoiling, which prevented sediment transport down the hillslope. While research on post-fire subsoiling is limited, our results were consistent with another Northern California study, which illustrated ~14.8-times greater sediment yields from burned plots and ~4.5-times greater sediment yields from salvage logged plots relative to subsoiled plots (James and Krumland 2018). James and Krumland (2018) attributed the lower sediment yields from the subsoiled plots to greater surface roughness in the headwater swales, which reduced sediment transport. 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 *et al.*, 1998; Römken *et al.*, 2002). While not directly analogous, post-fire studies from Portugal and Spain also found

rip-ploughed hillslopes produced ~2- to 5-times less sediment than reference hillslopes (Fernández *et al.*, 2019; Malvar *et al.*, 2011). However, results from other subsoiling experiments have not consistently led to reduced sediment yields. For example, in a study of unburned eucalypt plantations in Brazil soil loss was similar between hillslopes with and without contour subsoiling (Padilha *et al.*, 2018). In contrast to our results, soil tillage performed in agricultural settings, a practice analogous to subsoiling, has typically lead to greater soil loss than no-till agriculture (Montgomery, 2007).

Interestingly, when we adjusted plot sediment yields for the effective contributing area, the measurable erosion rates in the subsoiled plots increased substantially. These results suggest subsoiling had two contrasting effects on post-fire soil erosion. First, subsoiling increased small-scale erosion by creating ridges of disturbed and available (erodible) soil with microtopographic slopes steeper than the larger scale hillslope. However, subsoiling also created dramatic microtopographic roughness perpendicular to the downslope transport direction that limited sediment transport distances and provided areas for sediment detention. This result was not surprising to us as soil erosion is often negatively related to scale of measurement, reflecting increased opportunity for sediment storage at large scales (Shakesby and Doerr, 2006; de Vente *et al.*, 2007). However, we do not know whether we would have also observed lower effective area sediment yields from burned or salvage logged plots if they had similar contributing areas as the subsoiled plots. Thus, these results should be interpreted cautiously, as we are only aware of one other study that also quantified lower sediment yields from salvage logged and subsoiled hillslopes relative to burned hillslopes (James and Krumland 2018). Our results do suggest subsoiling could potentially be effective at disconnecting post-fire hillslope sediment from

streams, addressing a critical research gap about hillslope sediment connectivity to streams after forest harvesting (Croke *et al.*, 2005; Litschert and MacDonald, 2009).

2.4.2 *Principal Drivers of Erosion*

We found precipitation depth to be the most important driver of sediment yields through the course of this study. This was not surprising since fluvial processes are often the primary sediment transport mechanism during the wet season in post-fire environments (Cerdà and Robichaud, 2009; Doerr *et al.*, 2006; Johansen *et al.*, 2001; Wondzell and 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 *et al.*, 2012; Moody and Ebel, 2014; Moody and Martin, 2001a; Wondzell and King, 2003). These differences may be due to the dominance of lower intensity frontal storms from the Pacific Ocean at our sites, instead of convective storms common in the interior west, which have been linked to high rates of erosion and runoff in burned forests (Kunze and Stednick, 2006; Robichaud *et al.*, 2008b).

We also found strong evidence that canopy cover in the burned only plots contributed to the greater sediment yields measured on these sites. Generally, rain splash detachment and overland flow have been attributed as dominant processes driving runoff and erosion on burned hillslopes with low topographic roughness (Rengers *et al.*, 2016). Similarly, we observed visual evidence of rain splash detachment and sheetflow on hillslopes from all site types, contributing to soil erosion. However, anecdotally the raindrops underneath our burned canopy appeared to be heavier due to temporary detention on the burned snags and accumulation of water before

becoming throughfall. As such, the rain drops under the burned canopy likely fell with greater kinetic energy (Geißler *et al.*, 2012), which impacted the soil with greater force than in the salvage logged plots. Interestingly, the differences in drop kinetic energy between site types was especially evident during low intensity rainfall. This theory was also supported by the presence of substantial erosion pedestals underneath the burned tree canopies (Figure 2.7) that were absent within salvage logged areas.

Across all our sites, bare soil was also positively correlated with higher sediment yields. 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 *et al.*, 2009) and higher sediment yields (Benavides-Solorio and MacDonald, 2001; Malvar *et al.*, 2017; Schmeer *et al.*, 2018; Stoof *et al.*, 2015; Wagenbrenner *et al.*, 2006). Interestingly, the significant influence of bare soil was only evident in WY 2019, after the addition of herbicide treatments to the plots. Herbicide application significantly decreased the amount of live vegetation cover in affected plots, indicating changes in ground cover may still influence erosion at our site in the future. Before herbicide application, it is possible that overall precipitation depth overwhelmed the signal of bare soil. Alternatively, it is possible that our sample size was not large enough to quantify the effect. Some studies have found threshold or other non-linear responses between bare soil and soil erosion in burned forests, with erosion significantly increasing at bare soil levels of approximately 60–70 % (Davenport *et al.*, 2016; Johansen *et al.*, 2001; Spigel and Robichaud, 2007). Highest mean bare soil cover in our plots was 53.9 %; if our site were to behave according to such an erosion threshold, it is possible the bare soil fraction was not large enough to influence the rates of hillslope erosion.

While we observed a statistical difference in soil bulk density at 0–5 cm depth between the burned plots and both the salvage logged and subsoiled plots, the actual differences were small. In the third year after the fire (WY2018), soil bulk density was only 1.3-times greater in the salvage logged plots and 1.1-times great in the subsoiled plots compared to the burned plots. While the differences were small, they were consistent with previous studies that have illustrated a ~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 (García-Orenes *et al.*, 2017; Malvar *et al.*, 2017; Parkhurst *et al.*, 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 and Schmidt, 2013). Furthermore, bulk density is inversely related to the amount of energy to detach sediment particles from the soil surface during concentrated overland flow (Ghebreiyessus *et al.*, 1994). Additionally, compacted soils tend to have lower infiltration rates due to decreased macroporosity (Kozłowski, 1999; Luce, 1997). If infiltration is substantially decreased, soil compaction can lead to increased runoff and erosion (Batey, 2009; Reynolds *et al.*, 2011). However, in comparison to these previous studies the absolute bulk density values were lower at our sites, which may have been due to hand felling of trees on our sites, with limited use of heavy logging equipment (Ares *et al.*, 2005; Bustos and Egan, 2011). As a result, we did not find any evidence that soil bulk density contributed to differences in plot sediment yields.

Burned soils demonstrated slight to strongly persistent water repellency at 1 and 3 cm depth before any management activities took place. Soil water repellency is the inability of water to wet or infiltrate into soil, usually due to hydrophobic organic molecules on the surface of soil particles (Doerr *et al.*, 2000). While unburned soils high in organics are often water repellent in

conifer forests in the western US, burning and the resulting volatilization of organic compounds in soil can exacerbate soil water repellency after wildfire (DeBano, 1981). Soil water repellency can exacerbate surface runoff and erosion in burned forests by preventing water from infiltrating into soil (Doerr *et al.*, 2006; Shakesby and Doerr, 2006), though hillslope scale effects of soil water repellency are difficult to assess due to high spatial variability of water repellency (Doerr and Moody, 2004).

After management and a winter with above average precipitation, we saw overall decreases in soil water repellency in our plots. Soil water repellency was only slightly persistent at 1 cm and 3 cm depth in burned, and burned and salvage logged plots. This was not surprising as rain splash, soil erosion, or freeze thaw action on the soil surface can decrease soil water repellency. We have no evidence salvage logging directly affected soil water repellency at our site, as we did not perform a WDPT test immediately after logging. However, we think subsoiling may have reduced soil water repellency as it was the only treatment in which no water repellency was detected after re-testing in summer 2017 (Figure 2.3).

2.4.3 Management Implications

Knowledge on the long-term efficacy of post-fire land management approaches at mitigating erosion and sediment transport from hillslopes to streams remains limited. Given the shifting wildfire regimes observed in many regions, it will become increasingly important to understand when and where these approaches are likely to be effective to facilitate post-fire forest and water management. Our study provided surprising evidence of lower sediment yields

during the first three years after the 2015 Valley Fire in northern California from both salvage logged and subsoiled hillslope plots compared to burned and unmanaged plots.

We posit that surface roughness created by wood and litter on salvage logged plot and furrows in subsoiled plots may reduce sediment transport distance in the short-term, which is typically the period of greatest risk for post-fire erosion (Robichaud *et al.*, 2000; Wagenbrenner *et al.*, 2006). This is consistent with research on the effectiveness of contour felled logs as erosion barriers in burned forests (Robichaud *et al.*, 2008b). However, the longer-term effectiveness of these post-fire management treatments remains uncertain. Recent observations from our sites suggested that in some cases sediment had filled the storage areas behind wood on the salvaged logged sites and between subsoiled ridges. Our results also suggest that the early application herbicide reduced cover and increased sediment yields in 2018. As such, longer-term research is needed to determine whether sediment yields remain lower in the post-fire managed areas.

Unsurprisingly, post-fire erosion at our site was highly variable both spatially and temporally. We recorded much higher mean plot sediment yields in the second post-fire water year (28.8 Mg ha⁻¹) than the third post-fire water year (0.9 Mg ha⁻¹). Furthermore, the burned-only plot that produced the highest cumulative sediment yield (55.7 Mg ha⁻¹ yr⁻¹) was located within 20 m of the burned-only plot that produced the lowest cumulative sediment yield (11.4 Mg ha⁻¹ yr⁻¹). Studies measuring post-fire erosion have recorded a wide range mean sediment yields ranging (from 0 to >10 Mg ha⁻¹) using the same methodology across sites (Wagenbrenner *et al.*, 2015). In general, measurement variability of sediment yields is related to sediment fence design, difference in spatial scale, and hydrologic connectivity to sediment fences (Boix-Fayos *et al.*, 2007). In post-fire environments, hillslope-scale and smaller variability in fire intensity or

severity can also lead to differences in surface runoff and erosion, with higher burn severity correlated with greater rates of erosion and runoff (Abrahams *et al.*, 2018; Benavides-Solorio and MacDonald, 2005; Vieira *et al.*, 2015).

Given the variable response of hillslope erosion to post-fire salvage logging and subsoiling, it also remains unclear where post-fire salvage logging or subsoiling for erosion mitigation is likely to be effective. Soil erosion following fire and post-fire land management activities depends on site-specific characteristics including slopes, soil burn severity, weather and climate, and logging equipment operation (Cambi *et al.*, 2015; Moody *et al.*, 2013; Nyman *et al.*, 2015; Perreault *et al.*, 2017; Shakesby and Doerr, 2006; Wagenbrenner *et al.*, 2016). Our results indicated that given certain site-specific characteristics and operational approaches (i.e., subsoiling parallel to the contour) that post-fire salvage logging and/or subsequent subsoiling may not always negatively influence soil erosion and downslope transport of sediment. As such, we strongly advocate for the need for additional research on post-fire land management approaches across a broader range of regions and post-fire conditions.

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2.6 Figures

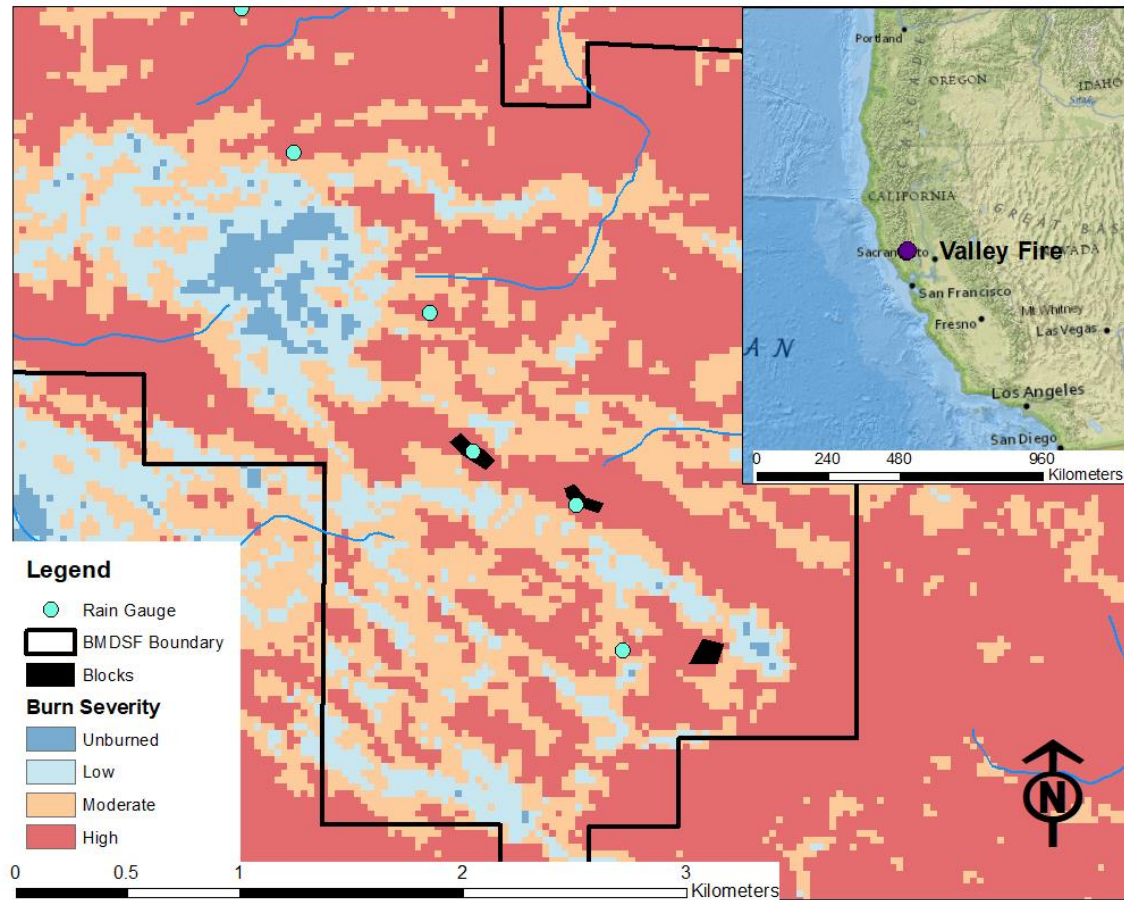


Figure 2.1 Burn severity map of Boggs Mountain Demonstration State Forest (BMDSF) with rain gauges and locations of blocks of silt fences. Inset map shows location of 2015 Valley Fire in California.

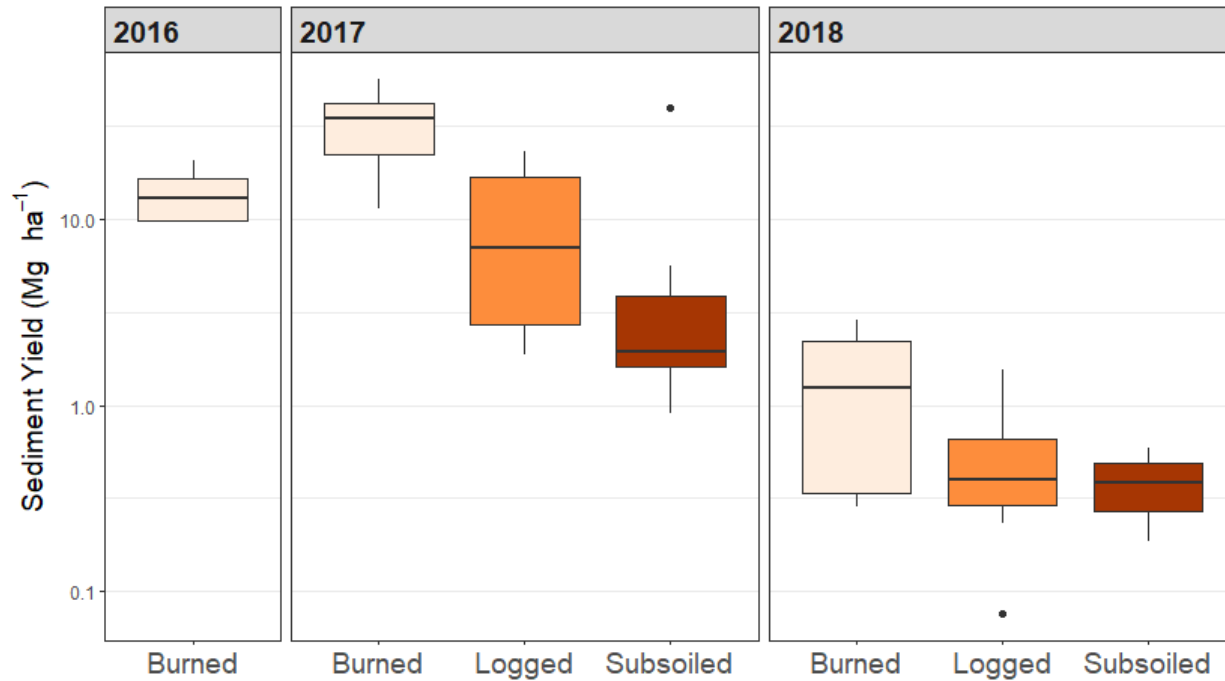


Figure 2.2 Annual plot sediment yields from silt fences from burned sites and two physical post-fire treatments for three water years (2016–2018). Water year 2016 was measured before post-fire management treatments were applied. All sediment yields were observed before influence of herbicide treatments in summer 2018. Y-axis is in the log scale.

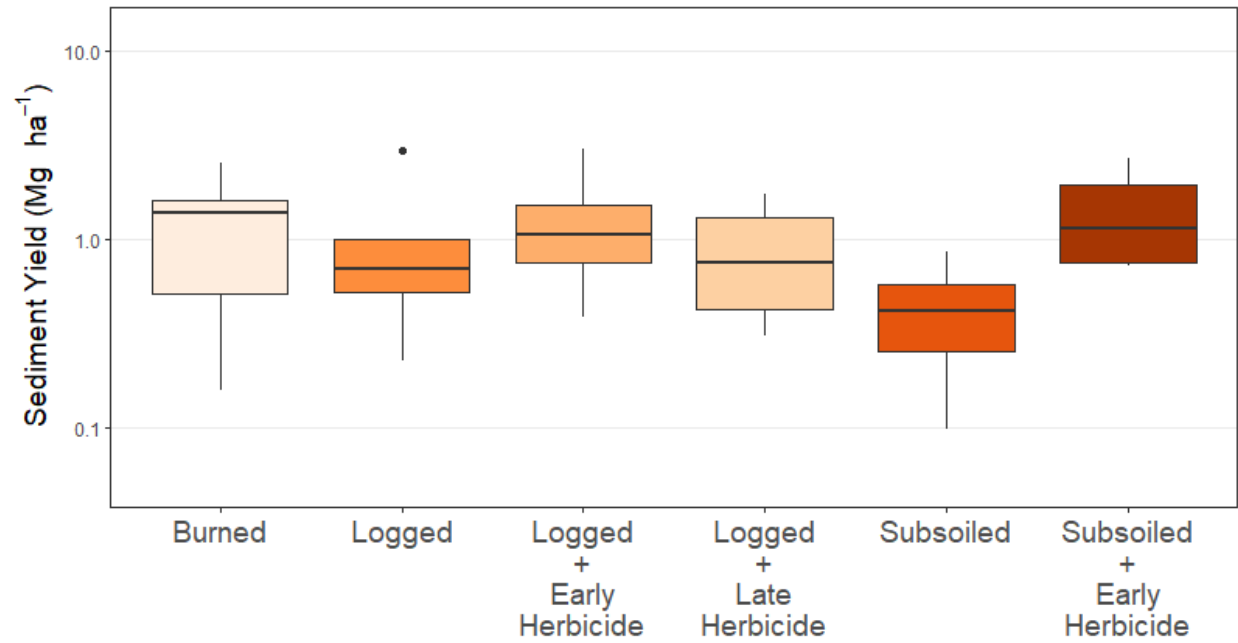


Figure 2.3 Annual plot sediment yields from silt fences from burned sites and five post-fire management treatments for water year 2019. Y-axis is in the log scale.

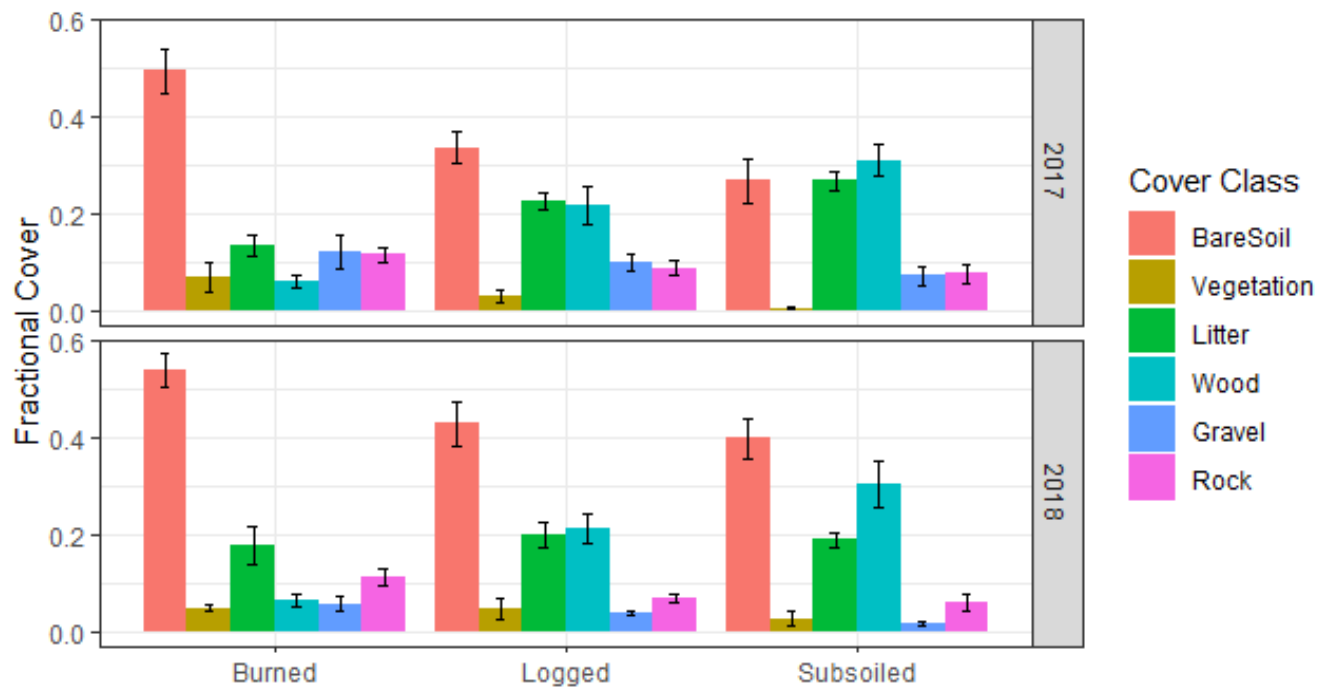


Figure 2.4 Mean fraction of ground cover and standard errors for burned sites and two physical post-fire management treatments from water years 2017 and 2018. Measurement of ground cover occurred before application of herbicide treatments in spring and summer 2018. Ground cover was measured during spring 2017 and 2018, early in the growing season.

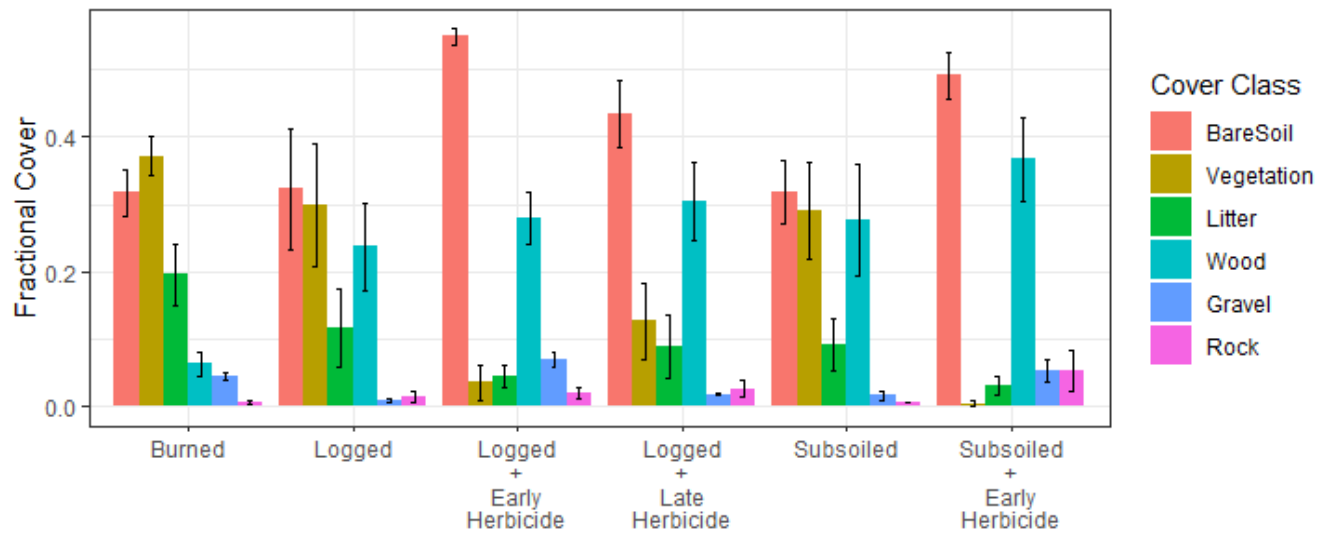


Figure 2.5 Mean fraction of ground cover and standard errors for burned sites and five post-fire management treatments from water year 2019. Unlike water years 2017 and 2018, ground cover was measured in June 2019, later in the growing season.

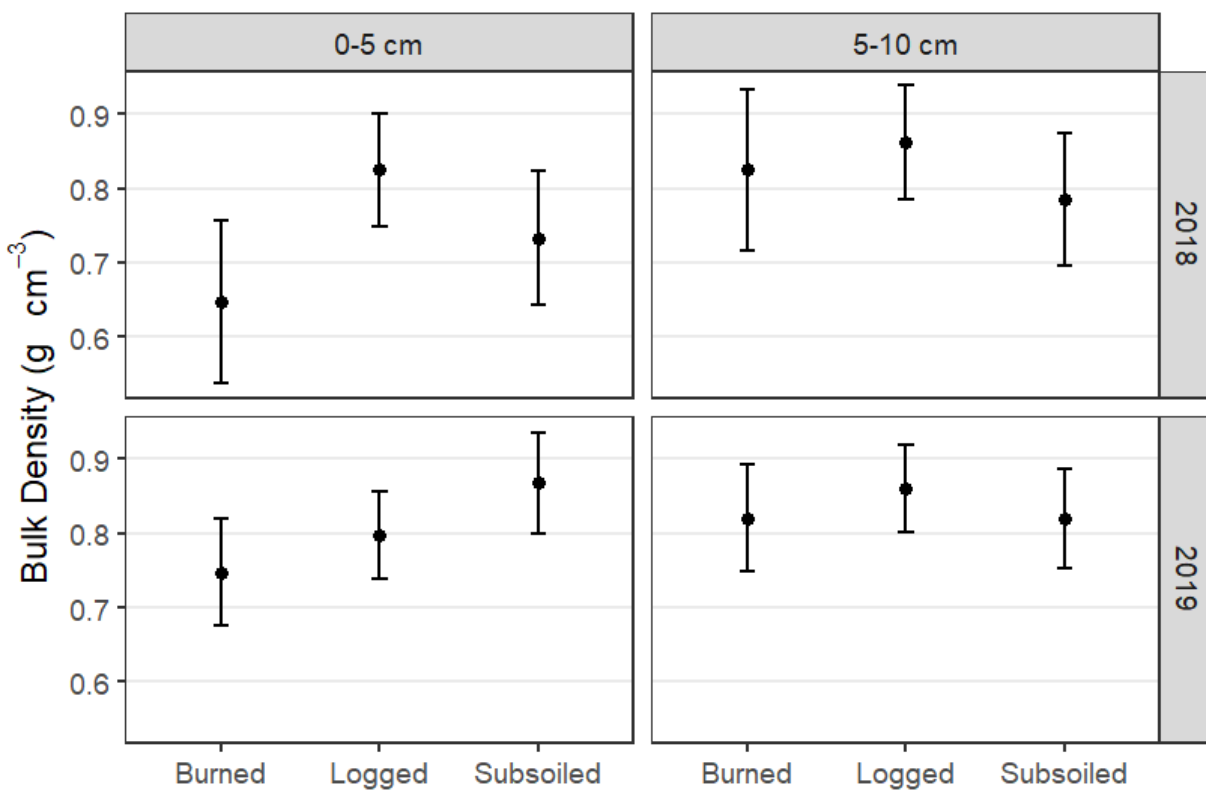


Figure 2.6 Mean soil bulk density and 95% CI from burned sites and two physical post-fire management treatments (Logged and Subsoiled) over two depths (0–5 cm, 5–10 cm) for water years 2018 and 2019.



Figure 2.7 Erosion pedestals near the trunk of a standing snag indicating raindrop splash erosion, which may have been an important driver of high sediment yields in burned plots.

3 Post-fire forest management influences mineral soil carbon, nitrogen, and vegetation recovery

3.1 Introduction

In forested regions of western North America the extent and severity of wildfires, along with the length of the wildfire season, have increased dramatically in recent decades (Flannigan *et al.*, 2009b; Moritz *et al.*, 2012; Reilly *et al.*, 2017; Westerling, 2016). These notable shifts in fire activity have been linked to climate change and 20th century land management policies (i.e., fire suppression and forest management activities) that have increased fuel loads in forests (Bowman *et al.*, 2013; Flannigan *et al.*, 2009a, 2013; Miller and Urban, 2000). Despite these increases in fire activity, historical reconstructions of wildfires prior to European settlement in the United States suggest western forests have a much greater potential to burn more area annually than presently observed (Murphy *et al.*, 2018). Wildfires are projected to burn increasing areas in the western US throughout the 21st century (Littell *et al.*, 2018). This raises uncertainties regarding the immediate and long-term effects of fire on a wide range of forest values, including water supply originating in forests, soil properties, and forest ecosystem resilience (Hallema *et al.*, 2018a; Stevens-Rumann *et al.*, 2018; Wagenbrenner *et al.*, In review).

Forest ecosystem resilience, the ability of an ecosystem to return to its pre-disturbance structure and function, is dependent on successful tree regeneration, live and dead legacy trees, species diversity of regenerating vegetation, and spatial soil and vegetation heterogeneity (Franklin *et al.*, 2002; Johnstone *et al.*, 2016; Lindenmayer *et al.*, 2008; Seidl *et al.*, 2014; Wine *et al.*, 2018). Historically, forests in many regions of the western US burned with frequent low- or mixed-severity fires, creating a heterogeneous mosaic of vegetation communities, which were

more resilient to disturbance (Franklin and Bergman, 2011; Hessburg *et al.*, 2005, 2016; Perry *et al.*, 2011). Forests that burned with mixed severity fire regimes are now burning with larger patches of high severity wildfire, which may lead to long-term changes to forest vegetation communities (Miller *et al.*, 2009; Stevens-Rumann and Morgan, 2019). Furthermore, vegetation communities recovering after a high severity fire may be less diverse than in low and moderate severity burns (Pingree and DeLuca, 2018). Recovery of diverse vegetation communities primes the forest for long-term recovery by reducing soil loss and fixing carbon and nitrogen into burned soils (Cerdà and Doerr, 2005; Johnson *et al.*, 2007; Morris and Moses, 1987; Qiao *et al.*, 2014; Shakesby *et al.*, 1993; Wagenbrenner *et al.*, 2015).

Vegetation recovery after a wildfire is dependent on nutrient availability for growth, including soil carbon and nitrogen (Cellier *et al.*, 2014; Hanley and Fenner, 2010). Soil carbon and nitrogen stocks exhibit predictive power on the productivity of forest ecosystems (Edmonds and Chappell, 1994; McLaughlin and Phillips, 2006). Carbon content and carbon/nitrogen (C:N) ratios of forest litter determine decay rates and thus, are linked to plant nutrient availability (Shi *et al.*, 2016; Taylor *et al.*, 1989). Wildfires reduce forest ecosystem stocks of soil carbon through combustion, volatilization, and increased net flux of carbon to the atmosphere (Bormann *et al.*, 2008; Urbanski, 2014; Van Der Werf *et al.*, 2010). Nitrogen, a critical limiting nutrient for forest growth, may also be lost to volatilization if soil burn temperatures exceed 200 °C (Grier, 1975; LeBauer and Treseder, 2008; Neary *et al.*, 2005; Vitousek and Howarth, 2016). Immediately after a fire, further carbon and nitrogen are exported by hillslope transport processes exacerbated by loss of erosion controls and increased surface runoff (Robichaud *et al.*, 2016; Smith *et al.*, 2011; Wondzell and King, 2003).

After wildfire, forest managers often attempt to promote regeneration and maintain forest ecosystem functions (Leverkus *et al.*, 2018; Robichaud *et al.*, 2000). One of the most common post-fire forest management approaches is salvage logging (Karr *et al.*, 2004; Lindenmayer *et al.*, 2004). This practice 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 *et al.*, 2013; Malvar *et al.*, 2017; Müller *et al.*, 2018). After salvage logging, forest managers may also apply additional treatments to mitigate erosion and promote vegetation recovery. Plowing furrows along the contours of hillslopes (subsoiling) 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 and Lowery, 1988; Robichaud *et al.*, 2000; Will and Jacobson, 2002).

Forest managers must consider the tradeoffs associated with the potential effects of the various post-fire land management options on soil hydraulic properties and nutrients due to potential impacts on vegetation recovery. Post-fire salvage logging removes carbon from burned forests in the form of dead wood, but it is uncertain how these activities influence soil carbon, nitrogen, and C:N ratio (Johnson *et al.*, 2005a; Powers *et al.*, 2013). Similarly, it remains uncertain how various other post-fire forest management activities impact recovery of early seral understory vegetation (Donato *et al.*, 2006; Peterson and Dodson, 2016). Standing snags left on site may serve some hydrologic function by providing shade for regenerating vegetation and therefore reducing water stress (Marañón-Jiménez *et al.*, 2013b). It is critical for researchers and forest managers to create management guidelines based on scientific understanding of soil nutrients and vegetation recovery to manage forests effectively after disturbance.

To improve our understanding of the effects of wildfire and post-fire land management on soil nutrients and initial recovery of vegetation, we performed a study in a high severity burn area of northern California. Specifically, we collected soil and vegetation samples and data from three post-fire land management strategies, including: (a) burned, (b) burned and salvage logged (salvage logged), and (c) burned, salvage logged, and subsoiled (subsoiled). Our first objective was to quantify differences in soil carbon, nitrogen, and carbon/nitrogen (C:N) ratios at two depths (0–5 cm and 5–10 cm) among the three post-fire management types. We then quantified relationships between aboveground vegetation biomass and soil carbon, nitrogen, and C:N ratios and carbon, nitrogen, and C:N ratios in eroded sediment to ground cover in erosion plots.

3.2 Methods

3.2.1 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 (Figure 3.1). 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). It is located about 10 kilometers southwest of Clear Lake, CA, in the northern Coast Range (38°50'00.07" N 122°42'05.33" W). During the Valley Fire, about 48 % of BMDSF area burned at high severity, 34 % at moderate severity, 15 % at low severity, and 2 % remained unburned/unchanged.

The climate of the region is Mediterranean with warm dry summers and cool, wet winters (Köppen Csb). Rainfall dominates the precipitation, though there are occasional snow events

during the winter. Mean annual precipitation is 1,160 mm with the majority falling between October and April (PRISM Climate Group, Oregon State University, <http://prism.oregonstate.edu>).

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.

Mean slopes of Boggs Mountain are about 27 %, which are relatively shallow compared to surrounding mountains. The elevation ranges from 700 to 1,144 m. The geology consists of a cap of igneous andesite and dacite at high elevations over sedimentary sandstone and mudstone at lower elevations. After the fire, soils were classified as deep, well-drained, xeric andisols with a sandy loam texture (Marshall and Obeidy, 2016).

3.2.2 *Post-fire management*

We targeted four hillslopes (“blocks”) across BMDSF to investigate the effects of the different post-fire land management treatments on hillslope sediment transport. These 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. We constructed 25 sediment fences modified from Robichaud and Brown (2002) to trap eroded sediment from hillslope plots. Each “plot” was approximately 5 m wide by 15 m long with a sediment fence constructed at the downslope end to catch sediment transported during storm events, though the exact contributing area to each fence varied due to microtopographic features.

Most of BMDSF was salvage logged approximately one year after the fire, excluding unburned forest and riparian corridors. Burned trees were primarily hand-felled and skidded to landings with wheeled or tracked skidders and cable yarders. About one to two months after salvage logging, some hillslopes were subsoiled using winged blades mounted to the rear of a tracked Caterpillar D7H crawler tractor. The subsoiling blades ripped through the soil surface and plowed furrows into the soil along hillslope contour.

3.2.3 Soil and Vegetation Sample Collection

In September 2018, we collected soil samples for analysis of carbon concentration, nitrogen concentration, and carbon/nitrogen (C:N) ratios. We established sampling grids on three replicated hillslopes that were representative of each of the three types of the post-fire land management sites (burned, salvage logged, and subsoiled). All nine (three replicates of three site types) of the sample grids were located adjacent to previously established erosion plots, to facilitate comparisons between the soils and the deposited sediment. Each sample grid consisted of 20 sampling locations arranged in a 20 m x 25 m rectangle with 5 m spacing between adjacent points. At each sampling location, we collected mineral soil from two depths, 0–5 cm and 5–10 cm, for a total of 360 soil samples (120 for each site type). Litter or forest floor organic material were not sampled. After collection, all soil and samples were placed into Whirl-Pak bags and kept cold until we were able to prepare the samples for nutrient analysis.

We also collected samples of eroded mineral sediment from 25 sediment fences, which were located at the base of the hillslopes representing each of the three site types. Samples were collected following seven major storm events from January 2018 to July 2019. Following each

storm event, we collect a ~50 g sediment samples from each of the 25 sediment fences during routine sediment fence cleaning (see 2.2.2), which resulted in a total of 174 sediment samples.

During the September 2018 soil sampling, we also collected aboveground plant biomass and measured vegetation percent cover at a subset of the soil sampling locations. Three sampling locations within each of the nine soil sampling grids (27 total locations) were randomly selected for biomass collection. Before biomass collection, we estimated percent vegetation cover within a 1 m quadrat that was centered on the soil sampling location. We then clipped all aboveground biomass within the 1 m quadrat and placed it into plastic bags, which were transported to the lab and kept refrigerated until sample processing.

We also measured surface cover of the contributing area to each of the 25 sediment fences in March 2018 and June 2019. We used the point intercept method to quantify surface cover on three 1-m², 100-point 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.

3.2.4 Laboratory Processing

In the laboratory, soil and sediment samples were oven dried for 24 hours at 40 °C in paper bags. After drying, samples were sieved to collect the fine fraction (<2 mm) of soil. The fine fraction was then ground using a roller grinder for 24 hours or until samples were the texture of a fine powder. The samples were then dry combusted in an Elementar Vario Macro Cube from Elementar Analysensysteme GmbH (Langenselbold, Germany) to determine total carbon and nitrogen concentrations. Biomass samples were then oven dried at 60 °C until all moisture was

lost from the sample. After drying, biomass samples were weighed to determine aboveground vegetation biomass.

3.2.5 Statistical Analyses

We used linear mixed effects models to compare differences in soil and sediment C:N ratio, total carbon concentration, and total nitrogen concentration among treatments within the same water year. We used random effects to adjust our comparisons by plot number and block and allowed for unequal variances between groups. Pair-wise comparisons between site types within each year were calculated using Tukey-Kramer adjustment (Driscoll, 1996). We also used linear mixed effects models to analyze relationships between vegetation biomass, vegetation cover, wood cover, and soil carbon and nitrogen concentrations, and C:N ratios. We used linear mixed effects models to analyze changes in sediment carbon and nitrogen metrics over time within each treatment. All statistical analyses were conducted using R programming environment (R Core Team, 2018), and linear mixed effects models were created using the *nlme* package (Pinheiro *et al.*, 2018).

3.3 Results

3.3.1 Soil Carbon and Nitrogen

At 0–5 cm depth in the soil, the mean C:N ratio and 95 % confidence interval [CI] was 25.8 [20.2, 31.3] in the burned sites, 28.3 [22.7, 33.8] in the salvage logged sites, and 29.2 [23.7, 34.7] in the subsoiled sites (Figure 3.2). Statistically, there was strong evidence that the mean C:N ratio at 0–5 cm soil depth was lower in the burned sites compared to the salvage logged ($t =$

-2.50, $p = 0.0006$) and the subsoiled ($t = -3.43$, $p < 0.0001$) sites. Comparatively, there was no statistical evidence for a difference in mean soil C:N ratio between the salvage logged and subsoiled sites ($t = -0.92$, $p = 0.346$).

At 5–10 cm depth in the soil, the mean C:N ratio from the burned sites was 26.4 [20.9, 32.0] (Figure 3.2). Mean C:N ratio at 5–10 cm soil depth in the salvage logged sites was 29.8 [24.3, 35.4] and from the subsoiled sites was 31.4 [25.8, 36.9]. Statistically, there was strong evidence that the mean soil C:N ratio at 5–10 cm depth was lower in the burned sites compared to the salvage logged ($t = -3.41$, $p < 0.0001$) and the subsoiled sites ($t = -4.93$, $p < 0.0001$). Additionally, there was weak evidence for a statistical difference in mean soil C:N ratio between the salvage logged and subsoiled sites ($t = -1.53$, $p = 0.056$) at this soil depth.

There was statistical evidence mean soil C:N ratios were greater at 5–10 cm depth compared to 0–5 cm depth in both the salvage logged ($t = 2.39$, $p = 0.018$) and subsoiled ($t = 3.30$, $p = 0.001$) sites. However, there was no statistical evidence for a difference in C:N ratio between 0–5 cm and 5–10 cm depths in the burned soils ($t = 1.02$, $p = 0.308$).

At the 0–5 cm depth, mean soil carbon concentration was 4.88 [3.12, 6.64] % in the burned sites, 5.38 [3.63, 7.13] % in the salvage logged sites, and 6.30 [4.54, 8.05] % in the subsoiled sites (Figure 3.3). There was no statistical evidence for a difference between mean soil carbon concentration between the burned and salvage logged sites ($t = -0.50$, $p = 0.178$). However, there was strong statistical evidence for a difference in mean soil carbon concentration between the burned and subsoiled sites ($t = -1.42$, $p < 0.0001$) and between the salvage logged and subsoiled sites ($t = -0.92$, $p = 0.0036$).

At the 5–10 cm depth, mean soil carbon concentration were 3.63 [1.88, 5.38] % in the burned sites, 4.73 [2.98, 6.48] % in the salvage logged sites, and 5.15 [3.39, 6.90] % in the

subsoiled sites (Figure 3.3). Statistically, there was strong evidence for a difference between mean soil carbon concentration between the burned and salvage logged sites ($t = -1.10$, $p = 0.0003$) and between the burned and subsoiled sites ($t = -1.52$, $p < 0.0001$). Comparatively, there was no statistical evidence for a difference in mean soil carbon concentration between the salvage logged and subsoiled sites ($t = -0.42$, $p = 0.300$).

There was strong statistical evidence that the mean soil carbon concentration was greater at 0–5 cm depth as compared to the 5–10 cm depth in all sites (burned: ($t = -4.42$, $p < 0.0001$); salvage logged: ($t = -2.32$, $p = 0.021$); subsoiled: ($t = -4.09$, $p = 0.0001$).

At the 0–5 cm depth, mean soil nitrogen concentrations were 0.191 [0.156, 0.225] % in the burned sites, 0.190 [0.156, 0.225] % in the salvage logged sites, and 0.216 [0.181, 0.251] % in the subsoiled sites (Figure 3.4). There was no statistical evidence for a difference between mean soil nitrogen concentration between the burned and salvage logged sites ($t = 0.03$, $p > 0.99$). However, there was strong statistical evidence for a difference in mean soil nitrogen concentration between the burned and subsoiled sites ($t = -2.73$, $p = 0.018$) and between the salvage logged and subsoiled sites ($t = -2.77$, $p = 0.016$).

At the 5–10 cm depth, mean soil nitrogen concentration was 0.140 [0.106, 0.175] % in the burned site, 0.157 [0.122, 0.192] % in the salvage logged sites, and 0.160 [0.181, 0.195] % in the subsoiled sites (Figure 3.4). There was limited to no statistical evidence for a difference between mean soil nitrogen concentration between the burned and salvage logged sites ($t = -1.82$, $p = 0.17$) and between salvage logged and subsoiled sites ($t = -0.35$, $p = 0.93$). However, there was weak statistical evidence for a difference in mean soil nitrogen concentration between the burned and subsoiled sites ($t = -2.17$, $p = 0.078$).

There was strong statistical evidence that the mean soil nitrogen concentration was greater at 0–5 cm depth as compared to the 5–10 cm depth in all sites (burned: ($t = -5.44$, $p < 0.0001$); salvage logged: ($t = -3.62$, $p = 0.0003$); subsoiled: ($t = -6.03$, $p < 0.0001$)).

3.3.2 Sediment Fence Carbon and Nitrogen

Geometric mean sediment C:N ratio and 95 % confidence interval [CI] from the burned treatment was 28.6 [25.4, 32.2] (Figure 3.2). Geometric mean sediment C:N ratio from the salvage logged sites was 35.4 [32.6, 38.5] and from the subsoiled sites was 36.8 [33.5, 40.5]. Statistically, there was strong evidence that the geometric mean C:N ratio in the sediment from the salvage logged sites was greater than in the burned sites ($t = -4.76$, $p = 0.0004$). There was also strong evidence geometric mean C:N ratio in the sediment was greater in subsoiled sites relative to the burned sites ($t = -5.36$, $p = 0.0001$). Comparatively, there was no statistical evidence for a difference in geometric mean sediment C:N ratio between the salvage logged and subsoiled sites ($t = -0.99$, $p = 0.59$).

Geometric mean sediment carbon concentration was 6.3 [4.0, 9.9] % in the burned sites, 6.3 [4.4, 8.9] % in the salvage logged sites, and 5.6 [3.8, 8.3] % in the subsoiled sites (Figure 3.3). Statistically, there was no evidence for a difference in geometric mean sediment carbon concentration among any pairwise comparisons between the sites: burned vs salvage logged ($t = 0.04$, $p > 0.99$); burned vs subsoiled ($t = 0.76$, $p = 0.73$); salvage logged vs subsoiled ($t = 0.90$, $p = 0.65$).

Geometric mean sediment nitrogen concentration was 0.22 [0.14, 0.34] % in the burned sites, 0.18 [0.13, 0.25] % in the salvage logged sites, and 0.15 [0.11, 0.22] % in the subsoiled

sites (Figure 3.4). Statistically, there was no evidence for a difference in geometric mean sediment nitrogen concentration between burned and salvage logged sites ($t = 1.53, p = 0.30$) or salvage logged and subsoiled sites ($t = 1.20, p = 0.47$). However, there was suggestive evidence that the geometric mean nitrogen concentration in the sediment from the burned sites was greater than the subsoiled sites ($t = 2.39, p = 0.068$).

3.3.3 Aboveground Vegetation Biomass and Ground Cover

Geometric mean biomass and 95 % *CI*s from September 2018 were 167.6 [65.4, 429.9] g in the burned sites, 61.6 [24.0, 157.9] g in the salvage logged sites, and 27.9 [10.9, 71.5] g in the subsoiled sites (Figure 3.5). There was no statistical evidence for a difference in geometric mean biomass between burned and salvage logged sites ($t = 1.56, p = 0.284$) or salvage logged and subsoiled sites ($t = 1.24, p = 0.446$). However, there was strong statistical evidence that the geometric mean biomass was greater in the burned sites compared to the subsoiled sites ($t = 2.79, p = 0.027$).

We found no evidence that vegetation biomass was related to soil C:N ratio at 0–5 cm depth ($t = -0.68, p = 0.51$) or 5–10 cm depth ($t = -1.54, p = 0.14$). We also found no statistical evidence that vegetation biomass was related to total carbon concentration at 0–5 cm depth ($t = 0.01, p = 0.99$) or 5–10 cm depth ($t = -0.67, p = 0.51$). Similarly, there was no statistical evidence that vegetation biomass was related to total nitrogen concentration at 0–5 cm depth ($t = -0.05, p = 0.96$) or 5–10 cm depth ($t = 0.26, p = 0.80$).

In 2018, we found evidence sediment C:N ratio was positively correlated with wood cover ($t = 3.07, p = 0.006$), but not vegetation cover ($t = 0.41, p = 0.68$). Inversely, we found

evidence sediment carbon concentration was positively correlated with vegetation cover ($t = 2.48, p = 0.023$), but not wood cover ($t = 0.86, p = 0.40$). However, there was no evidence sediment nitrogen concentration was correlated with either wood cover ($t = -1.45, p = 0.16$) or vegetation cover ($t = 0.16, p = 0.88$).

In 2019, we found evidence that C:N ratios in the eroded sediment were positively correlated with wood cover ($t = 2.52, p = 0.02$), but not vegetation cover ($t = 0.55, p = 0.59$). Comparatively, we also found no evidence that carbon concentrations in the sediment were correlated with vegetation cover ($t = -0.79, p = 0.44$) or wood cover ($t = -1.56, p = 0.14$). Similar to the C:N ratios, we found evidence that nitrogen concentrations in the sediment samples were negatively correlated with wood cover ($t = -2.49, p = 0.02$), but not vegetation cover ($t = -0.96, p = 0.35$).

3.4 Discussion

We found soil C:N ratios in burned sites were ~9–12 % lower than salvage logged sites and ~12–16 % lower than subsoiled sites at both soil depths (Figure 3.2). We saw even more dramatic differences in the C:N ratios of sediment eroded from hillslopes of the three post-fire site types—in the burned sites, the C:N ratio was ~19 % lower than the salvage logged sites and ~22 % lower than subsoiled sites. Unexpectedly, our results contrast with multiple studies that have found post-fire salvage logging did not affect soil C:N ratios in surface mineral soils across many environments and forest types (Francos *et al.*, 2018; Johnson *et al.*, 2005b; Parro *et al.*, 2019; Poirier *et al.*, 2014). For example, 20 years after wildfire and post-fire salvage logging in eastern Sierra Nevada there were no differences in soil C:N ratios between site types (Johnson *et*

al., 2005b). Johnson *et al.* (2005b) reported soil C:N ratios ~25–50 % lower than we measured, further suggesting N fixing vegetation may have had long-term impacts on soils beyond the first four years after fire and salvage logging. Studies in boreal forests reported thick O horizons with high C:N ratios overlaying mineral soil (Parro *et al.*, 2019; Poirier *et al.*, 2014); in contrast, our site lacked a developed O horizon at the time of sampling. Additionally, they reported C:N ratios (33.1–48.4) in surface mineral soils after fire greater than we observed (25.6–36.8) (Parro *et al.*, 2019; Poirier *et al.*, 2014), which may be related to immobilization of nitrogen in organic horizons or leaching of nitrogen past surface eluvial horizons found in boreal podzols (Pirainen *et al.*, 2002). C:N ratios in mineral soil two to ten months after fire in Spain ranged from 20–30 (Francos *et al.*, 2018), but no response of soil C:N ratios to post-fire wood management 2–10 months after the fire. Our results corroborate this idea, as we observed greater C:N ratios in sites with greater mechanical soil disturbance (burned < salvage logged < subsoiled).

Mean soil carbon concentrations also increased with disturbance (Figure 3.3), though those differences only met our threshold for statistical significance in the burned soils at 5–10 cm and subsoiled soils at 0–5 cm depth. Interestingly, we saw the opposite trend in eroded sediment, as sediment samples from the burned plots had the highest mean carbon concentration and subsoiled plots had the lowest. Our findings are similar to a study investigating post-fire salvage logging in the Sierra Nevada of California that reported soil carbon in the A horizon was higher in salvage logged sites than sites burned but not salvage logged 20 years after fire (Johnson *et al.*, 2005b). A study in Spain measured lower organic matter in surface mineral soils affected by salvage logging compared to unlogged soils two years after a moderate severity fire (García-Orenes *et al.*, 2017). Other studies have measured soil carbon unaffected by salvage logging in different forest ecosystems. Eucalypt forests in Australia had lower soil organic carbon

concentrations in forests with more fire and forest harvesting disturbance, but post-fire salvage logging specifically did not appear to influence organic carbon content in soils (Bowd *et al.*, 2019). Boreal forests in Quebec had similar soil organic carbon concentrations between logged and unlogged soils seven years after burning at low severity (Poirier *et al.*, 2014). Salvage logging also had no effect on soil carbon content in a boreal forest in Estonia (Parro *et al.*, 2019).

We did not find any correlation between vegetation biomass and soil C:N ratio, carbon concentration, or nitrogen concentration at either 0–5 cm or 5–10 cm depth. This was surprising considering we observed ~2.7-times greater vegetation biomass in the burned sites compared to the salvage logged and ~6-times greater biomass in the burned sites compared to the subsoiled sites (Figure 3.5). We expected vegetation biomass to be correlated to soil carbon and nitrogen since vegetation can influence soil carbon and nitrogen in multiple ways, including rhizodeposition of carbon (Hütsch *et al.*, 2002), litter decomposition, and symbiosis with nitrogen-fixing bacteria (Vessey, 2003). Inversely, soil carbon and nitrogen can influence the regrowth of vegetation in burned forests by providing essential nutrients for plants and soil microbial communities (Cellier *et al.*, 2014; Hanley and Fenner, 2010). Furthermore, multiple post-fire studies have linked elevated soil carbon and nitrogen concentrations with post-fire vegetation growth, particularly in sites with large communities of nitrogen fixers (García-Orenes *et al.*, 2017; Johnson *et al.*, 2007, 2005b; Johnson and Curtis, 2001a). The only vegetation response we saw was a positive correlation between vegetation cover and sediment total carbon concentration from 2018. Some of the mechanisms described above may explain this result, but we are unsure why vegetation was correlated with carbon concentrations in sediment fences for only one wet season.

Wood deposited on the soil surface by salvage logging appeared to drive differences in C:N ratios in sediment. High C:N ratios in our sediment samples were correlated with higher levels of wood cover in erosion plots in 2018 and 2019, and nitrogen concentrations were negatively correlated with wood cover in 2019. Wood has relatively high C:N ratios compared to herbaceous vegetation (Brady and Weil, 2010), and decay of wood has been linked to higher soil C:N ratios in post-fire environments (Black and Harden, 1995; Marañón-Jiménez and Castro, 2013). Forest harvesting practices that leave wood on site have been directly linked to higher levels of soil carbon and nitrogen in unburned conifer forests compared to harvest of whole trees (Johnson and Curtis, 2001b). Salvage logging practices at our site brought wood down to the forest floor and soil disturbance by tracked skidders and subsoilers may have promoted its incorporation into the soil matrix. It is possible that wood was linked to higher C:N ratios in our sediment fences by transport of wood or organic matter rather than decomposition. In fact, we observed wood in our sediment fences, particularly after storm events brought down large branches from standing snags. However, we think the C:N ratio of the sediment samples represent C:N ratios in the soil for three reasons: 1) Carbon concentration from sediment was not particularly elevated compared to the soil surface (Figure 3.3), 2) Carbon concentration in sediment was not related to wood in our plots, and 3) We observed greater amounts of wood in sediment fences at burned sites since branches could only fall into fences from sites with standing snags.

Our results suggest salvage logging and wood management have the potential to influence the short-term recovery trajectory of soil C:N ratios. Three years after wildfire and post-fire land management, the C:N ratios in the upper mineral soil at all our sites ranged from 25.8–31.4. These high ratios could lead to reduced nitrogen availability, as previous studies have

shown C:N ratios above 20–25 can increase nitrogen immobilization in the soil (Brady and Weil, 2010; Manzoni *et al.*, 2010). Similarly, soil disturbance by forest harvesting can decrease microbial biomass and depress genes related to nitrogen cycling in soil, further slowing decomposition and nutrient cycling two to six years after harvesting in forests with Mediterranean climates (Busse *et al.*, 2006; Pereg *et al.*, 2018). Additionally, salvage logging may slow recovery of soil microbes and basal soil respiration, decreasing nutrient cycling compared to burned but unharvested forests (García-Orenes *et al.*, 2017). Such a range of effects, leading to high C:N ratios, driven by low nitrogen availability, could potentially limit forest growth and post-fire vegetation recovery (Edmonds and Chappell, 1994; McLaughlin and Phillips, 2006). While we did not observe statistical evidence of a direct relationship between soil C:N ratios and vegetation biomass at our sites, mean vegetation biomass was ~63 % lower in the salvage logged sites and ~83 % lower in the subsoiled sites relative to the burned only sites, which may be indicative of vegetation recovery limited by nitrogen availability.

In the longer term, wood is a considerable pool of micronutrients that may eventually become available for use by recovering vegetation. Post-fire research in Spain found that decomposing coarse wood contained 2- to 9-times more nutrients than the soil, and decomposition of wood increased soil nitrogen by 26 % and phosphorous by 68 % (Marañón-Jiménez *et al.*, 2013a; Marañón-Jiménez and Castro, 2013). One meta-analysis of residue retention practices after harvest in unburned forests found tree growth was reduced by up to 7 % 33 years after harvest and removal of all residue compared to less intensive harvests, which they attributed to nutrient losses after residue removal (Achat *et al.*, 2015). There is some long-term evidence from Australia that soil nutrient concentrations are lower in eucalypt forests affected by fire and forest harvesting up to 80 years after disturbance (Bowd *et al.*, 2019). Removal of wood

by salvage logging after fire may lead to an overall loss of nutrients from a forest ecosystem, though results from our study show that salvage logging has the potential to influence nutrient cycling by increasing soil contact with wood.

Our study illustrated that wildfires and post-fire land management can affect soil nutrients and vegetation biomass. However, the long-term trajectory of effects of wildfires and post-fire management remains uncertain as there are comparatively few long-term studies in the literature (Donato *et al.*, 2016; Niemeyer *et al.*, 2020; Patel *et al.*, 2019). Moreover, the study by Johnson *et al.* (2005) illustrated that effects from post-fire salvage logging on soil carbon and nitrogen could potentially persist for multiple decades (Bowd *et al.*, 2019; Johnson *et al.*, 2007). Salvage logging exports carbon and nitrogen through removal of timber, erosion, and leaching (Johnson *et al.*, 2005b; Marañón-Jiménez *et al.*, 2013b; Shakesby, 2011). We found salvage logging might also lead to slower understory vegetation regeneration, supported by other research in Mediterranean environments (García-Orenes *et al.*, 2017). Additionally, salvage logging led to higher C:N ratios in soils and sediment at our site, likely due to the influence of wood. Nutrient concentrations in soil are not the only variables that influence plant growth after disturbance. Water stress can significantly limit the recovery growth of trees and understory vegetation after fire (Brzostek *et al.*, 2014; Restaino *et al.*, 2016). In xeric moisture regimes, such as our environment, controlling competing understory vegetation improved ponderosa pine growth more than soil nutrient additions in unburned forests (Powers and Reynolds, 1999). The interaction between soil carbon and nitrogen, understory vegetation growth, and plant water availability likely has a long-term impact on forest recovery after wildfire and salvage logging and is an important gap in our understanding of the full influence of post-fire salvage logging on vegetation recovery at our study site.

3.5 References

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3.6 Figures

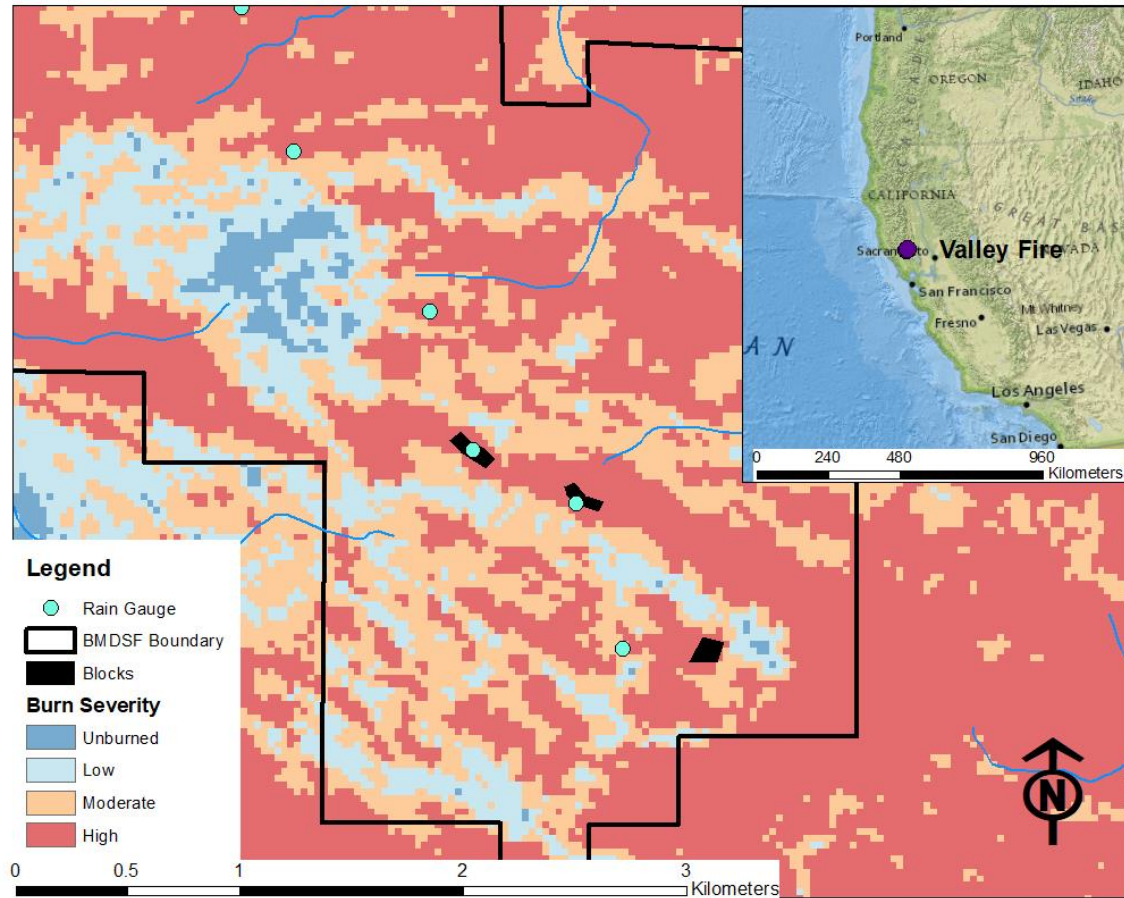


Figure 3.1 Burn severity map of Boggs Mountain Demonstration State Forest (BMDSF) with rain gauges and locations of blocks of silt fences. Inset map shows location of 2015 Valley Fire in California.

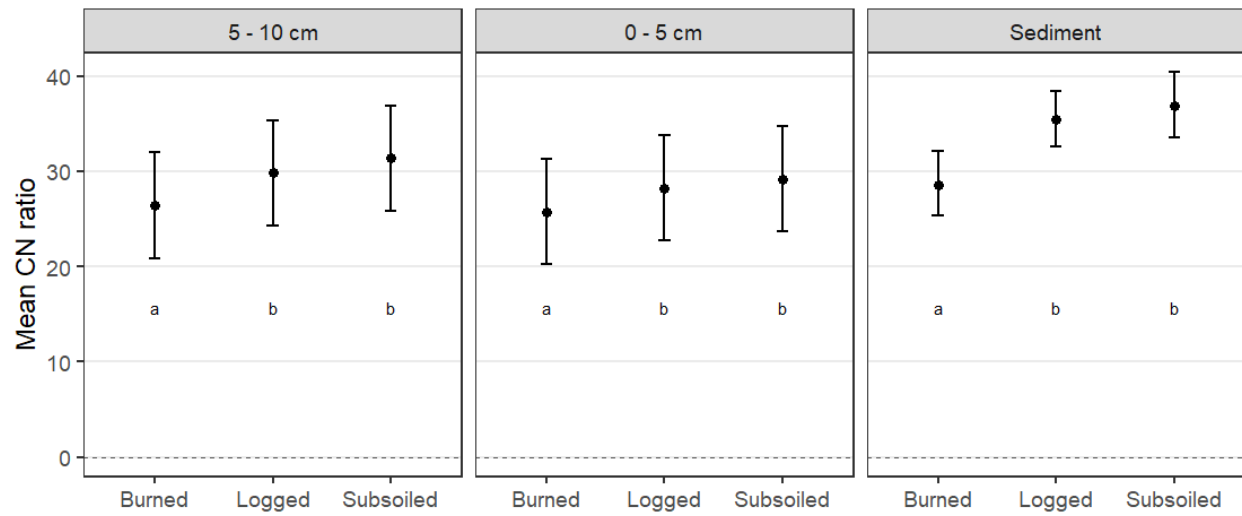


Figure 3.2 Mean CN ratio and 95 % confidence intervals of soil at two depths (September 2018) and eroded sediment from silt fences (2018–2019). Letters show statistically significant differences at $\alpha = 0.05$.

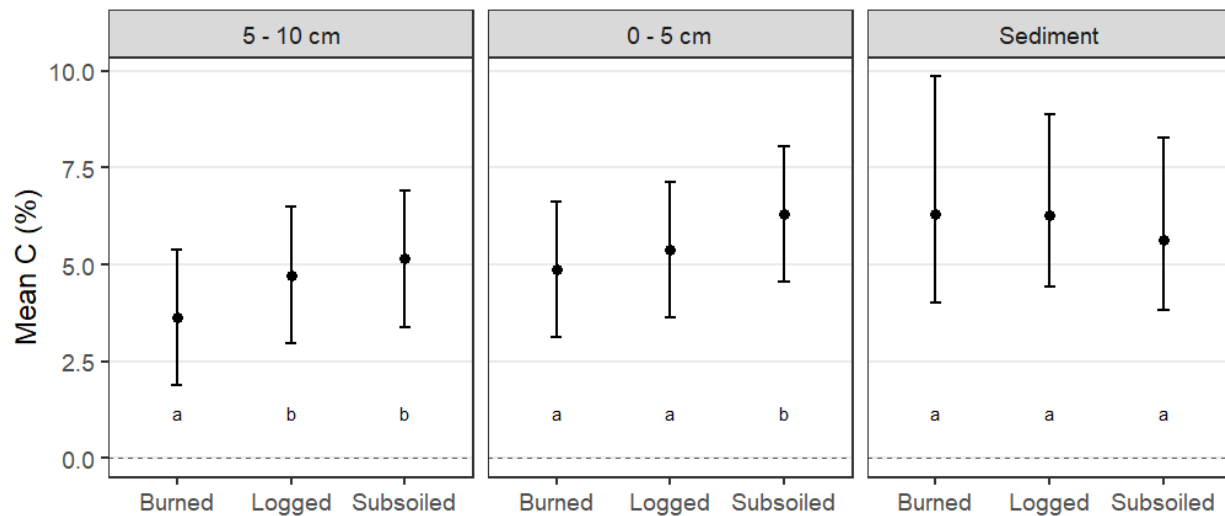


Figure 3.3 Mean carbon concentration (%) and 95% confidence intervals at two soil depths and in sediment. Letters show statistically significant differences at $\alpha = 0.05$

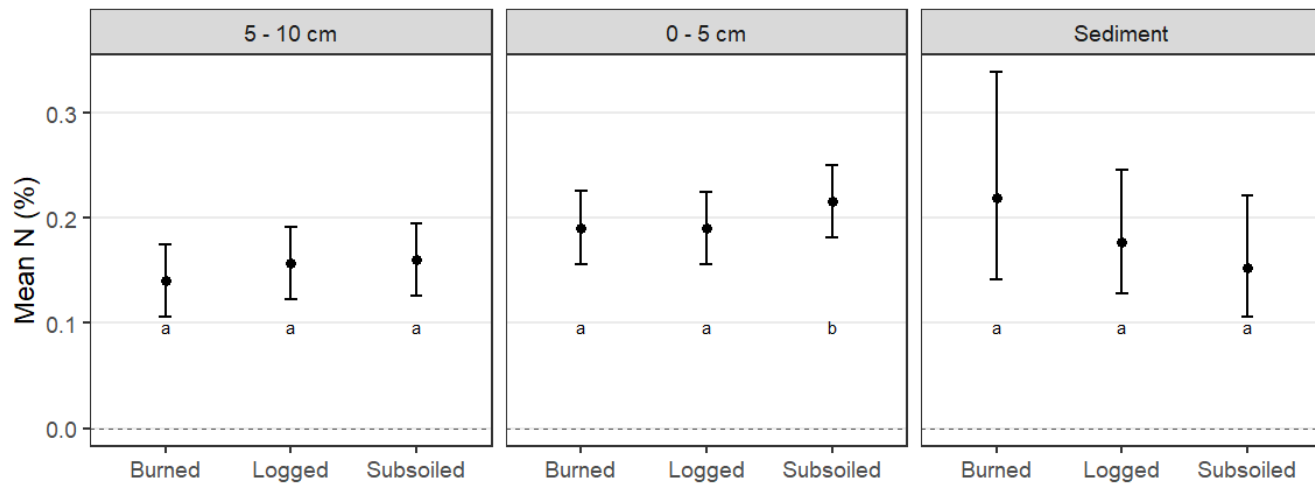


Figure 3.4 Mean Nitrogen concentration (%) and 95 % confidence intervals in soils at two depths and sediment. Letters indicate statistically significant differences at alpha = 0.05.

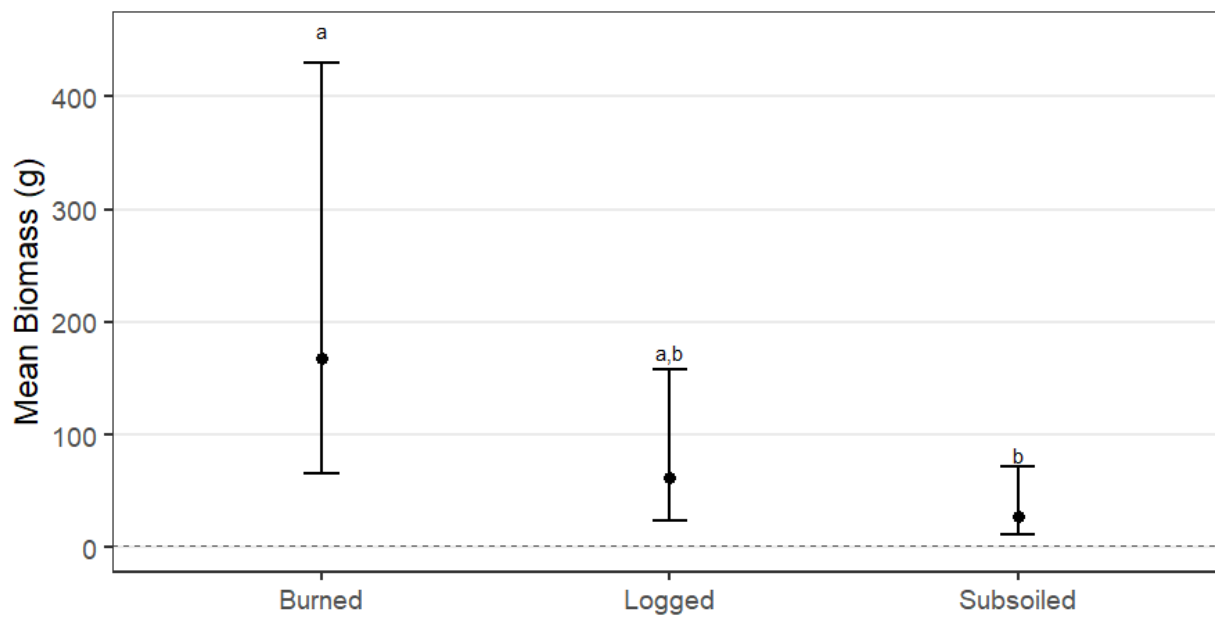


Figure 3.5 Mean vegetation biomass and 95 % confidence intervals for three post-fire site types.

Letters indicate statistically significant differences at $\alpha = 0.05$.

4 Synthesis

Dramatic shifts in wildfire occurrence and severity, along with area burned, in many regions across the planet have increased the challenges associated with forest and water management. As such, to facilitate informed land and water management decisions, managers and policy makers require information regarding the effects of wildfires and the efficacy of various post-fire land management treatments. Thus, the objectives of our research were to improve understanding of the impacts of wildfire and post-fire land management (salvage logging, subsoiling, and herbicide application) on soil erosion, soil physical and hydraulic properties, soil carbon and nitrogen, and vegetation recovery after high severity fire. To do this, we collected data and samples through the first four post-fire years from the Boggs Mountain Demonstration State Forest, which was burned during the 2015 Valley Fire in Northern California. We also analyzed linkages between sediment yields and various drivers of erosion, including exposed bare soil, precipitation characteristics, soil bulk density, and canopy closure. Additionally, we analyzed relationships of soil carbon, nitrogen, and C:N ratios with wood and vegetation recovery. Our principal findings included the following:

- Unexpectedly, sediment yields were greater from burned plots compared to salvage logged or subsoiled plots in water years 2017 and 2018. However, there were no differences in sediment yields between these site types four years after the wildfire 2019 (Figure 2.2). Plot sediment yields from salvage logged and subsoiled plots were similar in all years of measurement. We think higher rates of erosion in burned plots were related to lower ground cover and higher residual tree canopy cover. Burned plots had more exposed bare soil than salvage logged or subsoiled plots in 2017 and 2018, corresponding

with higher rates of erosion. Additionally, standing snags may have interacted with precipitation in a way that increased raindrop kinetic energy and promoted rainsplash erosion, which we observed during storm events. This idea is also supported by the presence of erosion pedestals, which we observed under burned snags and not in open areas (Figure 2.7).

- Subsoiling along the hillslope contour changed the microtopography of the hillslopes and created furrows, resulting in reduced sediment transport distances and sediment detention. Sediment yields from subsoiled plots, corrected for the contributing area to each fence, were higher than salvage logged plots, and in some instances were similar to burned (reference) sediment yields. We think soil disturbance caused by subsoiling has two influences on sediment yields: (1) subsoiling increased small scale erosion by increasing local slopes between ridges and furrows as well as by increasing soil availability for erosion, and (2) subsoiling created surface ridges and furrows that increased surface roughness and trapped sediment.
- We observed higher C:N ratios in soils and sediment in salvage logged and subsoiled plots compared to burned plots (Figure 3.2). We think elevated C:N ratios are related to increases in wood left on the soil surface by salvage logging operations, as evidenced by the strong positive correlation between wood cover and C:N ratios in sediment samples. Treatment-level differences in C:N ratios in the soil were primarily driven by differences in soil carbon (Figure 3.3), while differences in C:N ratios in the sediment were primarily driven by differences in sediment nitrogen (Figure 3.4).

We measured greater vegetation biomass in burned sites than salvage logged or subsoiled, but were not able to directly link vegetation biomass with carbon or nitrogen in the soils.

This was unexpected considering the influence soil carbon and nitrogen can have on vegetation regeneration after fire. However, the treatment-level differences we saw in vegetation biomass responses were similar to treatment-level differences in soil C:N ratio responses, suggesting there may be some relationship between vegetation biomass and C:N ratio were not able to capture in this study.

Some limitations of our study included:

- Sample size and number of replicates in each treatment limited our statistical inference in this study. Our 25 sediment fences were initially unbalanced between site types as we had 5 burned plots, 12 salvage logged plots, and 8 subsoiled plots. After herbicide application, we had 5 burned plots and 4 plots representing each of the 5 post-fire management treatments. Thus, we ended up with more different site types than we had replicates of each treatment. Increasing our sample size and number of replicates within each treatment would have allowed our statistical inference to be more robust.
- Another limitation of the study was our inability to quantify sediment transport distance or trace sediment movement from hillslopes into our sediment fences. If we would have traced sediment transported from plots to our fences, we may have learned more about the processes behind sediment transport and how transport differed between burned, salvage logged, and subsoiled hillslopes. For example, splash erosion typically moves sediment a short distance over a large area, whereas rill erosion may move sediment long distances with concentrated flow. We think splash erosion was a primary contributor to sediment yields in burned plots due to evidence of erosion pedestals (Figure 2.7), and tracing sediment movement could have supported this theory. Understanding the process

behind sediment transport may elucidate differences in erosion between site types we were not able to see by only measuring sediment yields.

- Limited spatial and temporal extent of soil carbon and nitrogen sampling during this study made it difficult to understand linkages between vegetation recovery and soil nutrients. We only measured soil carbon and nitrogen and vegetation biomass on one hillslope during one summer, so we are unsure if these relationships hold on other hillslopes that may have different vegetation communities and soil types. We are also unsure how soil carbon and nitrogen may have changed over the course of this study. In addition, if we had been able to expand vegetation sampling to a large spatial extent, we may have been able to see differences in vegetation communities. At the small spatial scale we measured, species diversity was difficult to quantify as many vegetation quadrats had only one species.
- Another limitation of the third chapter of this research was that we only measured total carbon and total nitrogen concentrations in the soil. Expanding our measurements to include other plant nutrients such as phosphorous, potassium, calcium, etc. would have allowed us to create a more complete picture of soil fertility. Additionally, these soil measurements could be supplemented by measurements of foliar nutrition and plant growth.
- We were unable to understand effects of herbicide on erosion over the course of our study due to the late application of herbicide treatments on erosion plots. We were only able to see the effect of herbicide treatments on our plots in water year 2019, but by this time sediment yields had equalized between site types. Herbicide effects may have been more dramatic if herbicide was applied before the first year of measurement.

We think future research projects could modify our approach in the following ways:

- Subsoiling warrants more investigation as an erosion mitigation practice after fire and salvage logging in the longer term. We are only aware of one other study that investigated subsoiling as a post-fire and salvage logging erosion mitigation technique. Our results suggest subsoiling could mitigate erosion on burned and salvage logged hillslopes, but many questions remain as to its efficacy. We saw contributing areas to sediment fences change in subsoiled plots as ridges eroded and furrows filled with sediment. It remains unclear whether subsoiled plots would continue to have lower plot sediment yields over the long term. Additionally we are unsure if subsoiling would be effective on hillslopes with different gradients or soil types.
- Future research could expand on our soil and vegetation sampling procedures by starting soil sampling immediately after fire and salvage logging and repeating sampling across multiple post-fire years. This would elucidate how soil nutrients and vegetation recovery are linked in the immediate post-fire years, and how they change through time during post-fire vegetation recovery.
- Researchers should focus on differences between various salvage logging practices to understand how salvage logging affects erosion. We found a variety of erosion responses to post-fire salvage logging during our search of the literature, and we think some of the discrepancies between studies could be explained by better description of salvage logging techniques. Felling of trees at our site was performed by hand and boles were skidded with rubber tired or tracked skidders. Burned forests logged with different equipment and

practices (e.g. helicopter or cable based yarding) may have differences in soil disturbance and erosion compared to our site.

- Finally, future research could focus on linking hillslope erosion and in-stream sediment transport in burned forests. Measuring sediment transport in streams affected by similar post-fire management site types could elucidate how hillslope sediment transport translates into streams. This would allow researchers to directly link salvage logging impacts on aquatic ecosystems and water quality.