SOIL BURN SEVERITY AND ENVIRONMENTAL COVARIATE EFFECTS ON SOIL HEALTH TWO YEARS POSTFIRE IN THE SIERRA NEVADA

A Project Report

presented to

the Faculty of California Polytechnic State University,

San Luis Obispo

In Partial Fulfillment of the Requirements for the Degree

Master of Natural Resource Management and Environmental Science in Environmental Science

by

Julie Lewis

December 2023

COMMITTEE MEMBERSHIP

TITLE: Soil Burn Severity and Environmental Covariate Effects on Soil Health Two Years Postfire in the Sierra Nevada

AUTHOR: Julie Anna Lewis

DATE SUBMITTED: December 2023

COMMITTEE CHAIR: Stewart Wilson, Ph.D.

Assistant Professor of Soil Science

COMMITTEE MEMBER: Yamina Pressler, Ph.D.
Assistant Professor of Soil Science and Restoration Ecology

ABSTRACT

Soil Burn Severity and Environmental Covariate Effects on Overall Soil Health Two Years Postfire in the Sierra Nevada

Julie Anna Lewis

Wildfires have been steadily increasing in both size and severity in recent decades. This global trend is most evident in California, especially the North Coast and Sierra Nevada. Although these trends are rising, there is little known about the effects of these wildfires on forest soils. Soil is the 2nd largest C sink on the planet, and the largest terrestrial bank. Without understanding the implication of rising wildfire severities on these soils, we cannot understand how to help protect this resource in the future. Due to the rapid increase in wildfire size and intensity, there is little known about the effects these "megafires" have on overall soil health, and even less known about these effects in the California Sierra Nevada region. This study aimed to understand both how field SBS and soil forming factors, represented through environmental covariates, effects soil health. 117 samples at a depth of 0-5cm were collected, then processed for sampling. Twelve soil health indicators (Total C, POXC, MinC, C/N ratio, pH, Total N, NO_3N, K, P, CEC, Ca, and Mg) were selected and samples were tested in lab for each indicator. Field-validated SBS levels were assigned to each sample, and through ANOVA testing in R, where four variables were significant with SBS (Total C, C/N Ratio, NO_3N, and K). Linear regression modeling was then used to observe the effects of environmental covariates on the samples. Environmental covariates were chosen to represent various soil forming factors including terrain, climate, and organisms. Each of the twelve soil health indicators were significant with at least one, if not more, environmental covariates. This information is critical in that soils are the result of not one, but several soil forming factors and processes.

Keywords: Soil health, soil burn severity, Sierra Nevada, environmental covariates

ACKNOWLEDGMENTS

I would like to thank both Dr. Stewart Wilson and Dr. Yamina Pressler for their endless support, positivity, and guidance through this process. I would also like to thank Craig Stubler and the Soil Geoinformatics and Biochemistry Lab for their support with laboratory work. Additionally, thank you to Sam Prentice from the U.S. Forest Service for the soil burn severity mapping information. Funding for this project has been provided for by the USDA National Institute of Food and Agriculture (NIFA)

I would like to extend further thanks to the faculty of California Polytechnic State University who have supported me through the completion of this degree, as well as my fellow peers Lexi Heurer and Gracie Doolin for the limitless support. Finally, I would like to thank my friends, family, and coworkers for their support and kindness, as none of this would have been possible without them.

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CHAPTER 1 INTRODUCTION

1.1 Statement of Problem

Wildfires have always been a major component of California's ecosystem (Keeley et al., 2018). It has been observed that in the past two decades, changes in climate and land utilization caused by human activities have not only extended the wildfire season, but also significantly increased the severity and burned areas of wildland fires (Li & Banerjee, 2021). Fires exceeding 10,000km (megafires) are becoming more severe and more frequent (Goss et al., 2020; Khorshidi et al., 2020; J. D. Miller et al., 2009). Fire regimes have been altered by changes in climate and land use and are predicted to change further as temperatures rise and populations grow (Pellegrini et al., 2018). It is also known that fire leads to complex and varied effects on soil properties (Certini, 2014). Many studies have shown that fire affects both physical, biological, and chemical properties of soils (Certini, 2005). Soil chemical properties have substantial effects on soil organisms and microbial communities, plant growth, and carbon storage (Hernández et al., 1997). In a time where atmospheric C levels are rising, understanding the effects of fire on the health of our high C-sequestration areas is critical.

1.2 Purpose of Study

Soil health is the continued capacity of soil to function as a vital living ecosystem that sustains plants, animals, and humans, and connects agricultural and soil science to policy. Historically, soil assessments focused solely on crop production, but, today, soil health also includes the role of soil in providing sustainable ecosystems including considerations water quality, climate change, and human health (Lehmann et al., 2020). Soil health has been measured in several different ways and includes a variety of chemical, physical, and biological indicators. While there is not a universal set of soil health measurements, several soil health indicators have been identified. The first of which include total C and N, which are required for sustained plant growth and healthy microbial communities, thus critical for carbon sequestration and water filtration and retention (Pandey, 2018). Additionally, cation exchange capacity (CEC) is used as an indicator of soil fertility, as it shows the soil's ability to supply three important plant nutrients: calcium, magnesium, and potassium (Cation Exchange Capacity, 2021). Phosphorous is often considered, as it is crucial for early root development in new plants, as well as hastening plant maturity (Pandey, 2018). The pH of the soil is also used as a soil health indicator as it controls the solubility, mobility, and bioavailability of trace elements (H. E. Allen et al., 1994).

Soil health has been recorded to be drastically affected by wildfire, yet there is variability within these studies. Often, total C, N, and P can be altered or even destroyed entirely by fire depending on the temperature and longevity of burning (fire severity) (Abney et al., 2019). The cation exchange capacity has been seen to decrease with fire at varying severities (Giovannini et al., 1990). The individual cations of extractable K_{ex}, Ca_{ex}, and Mg_{ex} are often left behind in soils post wildfire due to high volatilization temperatures, therefore often remaining unchanged or increasing in the soil (Abney & Berhe, 2018; Haby et al., 1990). Soil pH often increases post-wildfire (González-Pérez et al., 2004; Pereira et al., 2014). Although, some studies have shown no effect on pH from controlled burns (Murphy et al., 2006), indicating there may be differential effects on soil pH depending on burn severity, or that site-specific factors play a large role.

Soils are the result of a combination of soil forming factors and processes, (Jenny, 1946) which could lead to the variability in results of fire effected soils research. Where soil forming processes and factors are unique, soils will be unique, yet there is little accountability in this in fire effect soils research today. Soil forming factors of climate, organisms, relief and/or topography, parent material, and time can hide the effects that wildfire has on soil. Remote sensing data allows for the inclusion of raster layers to account for this variability across landscapes, yet little research has been conducted this way. With the increase of megafires, it is more crucial than ever to account for ecosystem variability, especially in areas as topographically diverse as the Sierra Nevada.

In an age where both fire severity, size, and frequency continue to increase, it is critical to have a strong understanding of the varying effects wildfire has on soil health indicators. Yet, there is a lack of research on soil health indicators at varying levels of burn severity after wildfire. While some existing studies relate soil burn severity to soil health (Huerta et al., 2020; Mehdi et al., 2012; Vega et al., 2013), we lack an understanding of fire impacts on soil health in the Sierra Nevada. This region consists of several national parks, protected lands, and tens of thousands of square miles of natural landscapes that aid in soil C storage and provide water to many California's cities as well as its billion-dollar agricultural industry. Therefore, it is important to evaluate the effects of megafires on the soil health in this region. To better understand the effects of megafire and soil burn severity on soil health in the Sierra Nevada region, we will measure soil total Carbon (C) and Nitrogen (N), permanganate oxidizable C (POXC), mineralizable C (MinC), extractable cations (K_{ex}, Mg_{ex}, Ca_{ex}), cation exchange capacity (CEC) Phosphorous (P), and extractable nitrate (NO_3N), in soil samples collected after the Creek Fire in the Sierra National Forest. We hypothesis that:

1) Higher burn severity areas will have a greater effect on soil C and N, extractable cations, P, MinC, and extractable NO_3N (Binkley et al., 1992; Huerta et al., 2020).
2) C, N, MinC, and CEC levels may gradually decrease with burn severity increase (Huerta et al., 2020; Xue et al., 2014).
3) Soil pH and NO_3N is expected to gradually increase with burn severity (Hernández et al., 1997; Turner et al., 2007).
4) We expect P in soils to decrease only at the highest burn severities considered in the study (Merino et al., 2019; Xue et al., 2014).
5) POXC is expected to decrease at higher burn severities (Bruun et al., 2013).
6) Inclusion of environmental covariates in the statistical model will enhance the effect of soil burn severity on soil health variables (Stavros et al., 2016).

This work will advance our understanding of landscape-scale distributions of soil health following a megafire. The outcomes of the project can help inform post-fire management that aims to either maintain or increase soil health in fire-prone landscapes.

CHAPTER 2 LITERATURE REVIEW

2.1 Introduction

Soil health is the continued capacity of soil to function as a vital living ecosystem that sustains plants, animals, and humans. This includes not only utilization for crop production, but also the role of soil in water quality, climate change and human health (Lehmann et al., 2020). Healthy soils provide a habitat to ecosystems and allow for the growth of all living things. It is known that soils perform as C sinks, absorbing C from the atmosphere, utilizing it to build biomass, then transferring it into the soil through litter roots, and exudates. The soil C pool is 3.3 times the size of the atmospheric pool and 4.5 times the size of the biotic pool (Lal, 2004). In a world continuing to warm because of greenhouse effects, storing C into our soil is more vital than ever. While we understand that soil health overall is a key component to sustaining the life of organisms, there are factors today that are changing the nature of soil. While it is known that soil is formed from the connection of parent material, climate, biota (organisms), topography and time, factors outside of these parameters, such as fire and human influence, may have a much larger role in the trajectory of soil change than what was initially reported. The extent and longevity of these changes currently remains unknown. As we witness a drastic increase in fire patterns globally, we need to understand how these processes affect soil to understand how this massive C sink will operate in future scenarios. This review aims to break down what soil health and soil health indicators are, as well as the role of soil in C storage, the shifts in fire and fire severity in California, and the effects of fire and burn severities on soils.

2.2 Importance of Healthy Soil

While the production function of soil was recognized for centuries, the importance of conservation and enhancement of ecosystem services rendered by soil (i.e., C storage, water purification, recharge of ground water, control of populations of pathogens, biological nitrogen fixation and biodiversity conservation) has been accepted only in the recent past (Maikhuri & Rao, 2012). While criteria, indicators, and standards of water and air quality are universally defined (Karlen et al., 1997) the concept of soil quality, further elaborated as soil health is still changing, with soil quality legislations rarely drafted comparison to other natural resources. Consideration of soil as a finite and living resource led to the concept of soil health, defined as the continued capacity of soil to function as a vital living system within ecosystem and land-use boundaries; to sustain biological productivity, maintain or enhance the quality of air and water, and promote plant, animal, and human health (Karlen et al., 1997). Soil performs multiple functions: (i) providing physical support to terrestrial plants, (ii) supplying fundamental resources such as water, nutrients and oxygen required for terrestrial primary production, (iii) providing habitat to a variety of soil organisms, with taxonomic identity and functions of several organisms still unknown/lesser known to the scientific and wider community, (iv) regulating hydrological and mineral/nutrient cycling, with significant impacts on global climate, (v) detoxification of organic and inorganic substances, leading to purification of water resource, and (vi) resisting erosion (Maikhuri & Rao, 2012). Given all these functions, we can accept that soils act as a base of many global ecosystems and create a space for life to adapt, flourish, and grow. Although we do not fully understand the impacts of perfectly health or unhealthy soils, we can state that soils that are of a higher soil health standard will provide more resources to the surrounding environment, laying the groundwork for increased biodiversity and ecosystem resilience.

2.3 Soil Health Indicators

Currently, there is not a set standard for measuring if a soil is "healthy," yet we can point to several soil health indicators. This allows us to not only paint the picture of what is going on below the surface, but also help us predict how soils will produce in the future. As soils are the foundation to most ecosystems, it is essential that soils are "healthy" in the sense they contain all the minerals, nutrients, and properties that are necessary to support life. Without healthy soils, ecosystems cannot survive, as soils provide an ecosystem base, plants and habitat, and water among other resources that allow for ecosystems to function. What we turn to measure soil health are therefore a set of indicators. Soil health indicators are a composite set of measurable physical, chemical, and biological attributes which relate to functional soil processes and can be used to evaluate soil health status, as affected by management and climate change (D. E. Allen et al., 2011). When thinking about the soil's ability to sequester C, it is important to consider chemical attributes of soil, as without appropriate chemical levels, soil will not be able to provide nutrients for plants. Additionally, the soil's ability to retain chemical elements or compounds that are harmful to the environment can be weakened or lost entirely. Soil pH, total C and N, extractable cations, P, mineralizable C, and nitrates are considered strong indicators of soil health.

2.4 Soils and Climate Change

It is known that there has been a drastic increase in CO_2, and human activities have raised the atmosphere's C dioxide content by 50% in less than 200 years (Change, n.d.). Increases in CO_2 emissions are known to result in a plethora of climate shifts, including increases in fire weather seasons and fire size (*Chapter 2*, n.d.). C storage on land and preventing increased CO_2 in the atmosphere is more critical in this age than ever. It is widely accepted that soils function as the second largest C bank on earth, second only to the ocean, and is four times the biotic pool and about three times the atmospheric pool (Lal, 2004). The soil C pool comprises two components: soil organic C (SOC) and the soil inorganic C (SIC) pool. Biosequestration (the capture and storage of the atmospheric greenhouse gas C dioxide by continual or enhanced biological processes) temporarily removes C (C) from atmospheric cycling. Additionally, C sequestration is defined as the uptake of C-containing substances and CO_2, into living plant matter, then into another reservoir (such as SOM) with a longer residence time (Pachauri & Reisinger, 2008). Soil contains about 75% of the C pool on land- three times more than is found living in plants and animals (Cerda, 2009). Overall, mixed and coniferous forests had the highest average organic C stocks compared with broadleaved forests and grasslands, it is additionally known that forests cover 4×10^7 km^2 of the Earth's surface, equivalent to ~30% of the global land area (Luyssaert et al., 2008). We also know that soil biodiversity has a positive impact on the SOC pool. All other factors being equal, ecosystems with high biodiversity sequester more C in soil and biota than those with reduced biodiversity (Díaz et al., 2009).

Organic C is commonly stored in the form of SOM, which requires N to exist (Cotrufo et al., 2019). From this we can conclude that the ability of soils to store C is linked to N availability. This is crucial to understand, as SOM will sequester C for a much longer timescale than other means. When vegetation dies, C is released back into the atmosphere. This can happen quickly, in cases such as fire or other combustive episodes, or slowly through natural decomposition (Gorte, 2009). Unfortunately, the shifts in climate change are affecting these massive areas in ways that not only create the loss of SOM through decomposition (drought) but also in ways that lead to the rapid release of SOC (fire). Deforestation (Hatten et al., 2005), biomass burning (Shi et al., 2020), and cultivation of soil, specifically by tillage methods (Buragienė et al., 2019), enhances mineralization of SOC and releases CO_2 into the atmosphere. Soil temperature is the primary rate determinant of microbial processes (Cerda, 2009). Increases in soil temperature will exacerbate the rate of mineralization leading to a decrease in the SOC pool, yet some studies have found that there is an “upper-limit” to this process, where high enough temperatures will no longer increase the speed of C mineralization (Dalias et al., 2001). The decline in SOC pool will have an adverse effect on soil structure, with an increase in erodibility and the attendant increase in susceptibility to crusting, compaction, runoff, and erosion. (Lal, 2004). Several studies have indicated that substantial consumption of organic matter begins in the 200-250°C range, while combustion is around 460 degrees (Certini, 2005). SOC shifts by fire can be site specific, as the amount of SOC and SOM depend heavily on decomposition rates of underground mass (Pellegrini et al., 2022). To fully understand how fire affects SOC, a wide range of studies need to take place over a broad range of environments.

Total soil C is both the organic and inorganic C found in soils. Soil C, including soil inorganic C (SIC) and organic C (SOC), is the largest C pool in the terrestrial ecosystem, thus playing a key role in the global C cycle and climate change (Eswaran et al., 1993). Increasing organic C storage in the soil not only sequesters atmospheric C but often enhances soil physical, chemical, and biological processes and properties (Blanco-Canqui et al., 2013). While SOC is the amount of C found in organic matter in soils, SIC is found primarily as calcium (and magnesium) (Guo et al., 2016). Studies have shown that increases in SOC can lead to an increase in SIC (Guo et al., 2016). Often, permanganate oxidizable C (POXC) has been used to measure the amount of active C in soils, along with analyzing mineralizable C (MinC) which is associated with recent additions of organic matter to the soil. The measurement of POXC is based on chemical oxidation of organic matter by a weak potassium permanganate solution (Weil et al., 2003), whereas mineralizable C measures flush of CO_2 from rewetted soils during a short-term aerobic incubation (Haney et al., 2008). This is critical as active C is an indicator of the small portion of SOM that can serve as a readily available food and energy source for the soil microbial community, thus helping to maintain a healthy soil food web. SOC is often lost in soils through the process of mineralization with following leaching, erosion, and loss (Cerda, 2009). Without understanding active C and total C, it is not possible to fully understand the health of soils, as well as their ability to affect climate change.

2.5 Fire Trends

While wildfire has played a role in shaping California’s landscape, the severity and frequency of fires have shifted drastically in the past one hundred years. The state's single deadliest wildfire, two largest contemporary wildfires, and two most destructive wildfires occurred during in the past decade (*CAL FIRE*, n.d.). Increased forest fire activity across the western continental United States (US) in recent decades has been enabled by many factors, including the legacy of fire suppression, human settlement, natural climate variability, and human-caused climate change (Abatzoglou & Williams, 2016). Overall, three primary factors have produced gradual yet significant change across western North American landscapes in recent decades: the warming and drying climate, the build-up of fuels, and the expansion of the wildland–urban interface (WUI; the zone where houses meet or intermingle with undeveloped wildland vegetation) (Schoennagel et al., 2017). Given these three factors, it is evident why there has been an increase in large wildfires in California.

Large fall wildfires became more frequent in California over the past several decades, mainly due to increases in the North Coast and Sierra Nevada regions (Williams et al., 2019). The observed increase in wildfire activity was due in many areas by reduced fuel moisture due to warming-induced increases in evaporative demand, reduced snowpack, and reduced warm-season precipitation frequency (Abatzoglou & Williams, 2016). One study found that there were state-wide increases in autumn temperature (∼1 °C) and decreases in autumn precipitation (∼30%) over the past four decades, and that this has contributed to increases in aggregate fire weather indices (+20%) (Goss et al., 2020). As a result, the observed frequency of autumn days with extreme (95th percentile) fire weather—which are shown to be preferentially associated with extreme autumn wildfires—has more than doubled in California since the early 1980s. Fire suppression over the past century allowed for artificial buildup of fuels in many regions that historically experienced frequent low-intensity fires, reducing fuel limitation as a constraint on fire activity and putting many areas into a so-called fire deficit (Higuera et al., 2015). Anthropogenic changes to the fire environment will increase the likelihood of such record wildfire years in the coming decades (Balch et al., 2018).

2.6 Soil and Fire

While several variables can impact the physical and chemical nature of soils, one factor of growing interest is fire. Due to the historical impacts of fire on soils, there has been discussion of considering fire as a 6th soil forming factor (in addition to parent material, climate, biota [organisms], topography, and time) (Certini, 2014). Although, the duration of effects of fire on soils are still greatly unknown. With increasing fire seasons due to climate change, the study of fire on soils has become more critical than ever. Wildfires have been ever prominent in California's history, yet there has been a massive increase in size and intensity of fires in the last one hundred years. While this can be partially accredited to climate change, in California specifically, poor fire management practices in the turn of the 20th century led to massive amounts of dense forest. This created forest that did not go through natural thinning processes and has far more burnable matter than was normal for these areas, creating potential for massive fires. The term "megafire" is defined as spatially and temporally continuous fire arising from single ignition or multiple related ignition events that exceed 10,000 ha in area and has been used to describe several fires in California in recent years. For context, 10,000 ha is approximately 40% bigger than Manhattan or ~14,000 football/soccer fields (Linley et al., 2022). Projections suggest an increase in global fire activity across vast portions of the Earth's surface in the coming decades (Masson-Delmotte et al., n.d.).

To understand the effect fire has on soil, we must first understand fire trends and burn severity levels. Burn severity levels are used to describe the response that ecosystems, including soil, have with fire. Burn severity at its core is the correlation of fire intensity and residence time. Burn severity has been defined in the following ways: Low severity: less than 2% of the area is severely burned, less than 15% is moderately burned and the remainder of the area is burned at a low severity or unburned. Moderate burn severity is less than 10% of the area is severely burned, but more than 15% is moderately burned, and the remainder is burned at a low severity or unburned. High severity Is when more than 10% of the area has spots that are burned at high severity, more than 80% is severely or moderately burned and the remainder is burned at low severity (DeBano et al., 1998). Even so, burn severity measurements can vary from fire to fire, given the preexisting environment. There are several challenges with measuring the effects of various burn severities on soils, including lack of pre-fire data for comparison, as well as dangers of retrieving soil samples immediately post-fire.

What we do know from a multitude of studies surrounding fire and soils is that fire is capable of effecting physical, biological, and chemical properties of soils. Physical properties of soil, including, soil bulk density, soil porosity, soil temperature, soil structure, soil water repellency, water soil infiltration rate, water storage capacity of soils, and soil erodibility, are often minimally affected by most fires (Agbeshie et al., 2022) Physically changes of soil are strongly related to soil organic matter, and changes in both soil and plant matter are altered only after temperatures starting at 400^{o} C (Raison et al., 1990). Chemical properties of soils, specifically pH, soil electrical conductivity, nitrogen, phosphorous, soil organic matter, inorganic cations (calcium, sodium, potassium, magnesium), cation exchange capacity, and C are all critical components of healthy soil that are affected by fire in different ways and alter at different burn severities. This is due a variety of reasons, such as volitation at unique points, chemical breakdowns, physical structure shifts, and overall soil dynamic changes.

2.7.1 pH

Soil pH is often referred to as the "master soil variable" as it has major effects on many physical, chemical, and biological processes that affect plant growth and biomass yield (Minasny et al., 2016). It is known that soil pH is controlled by the leaching of basic cations such as Ca, Mg, K, and Na far beyond their release from weathered minerals, leaving H^+ and Al^{3+} ions to dominant exchangeable cations (Neina, 2019). Soil pH controls the solubility, mobility, and bioavailability of trace elements, which determine their translocation in plants (H. E. Allen et al., 1994). At low pH, trace elements are usually soluble due to high desorption and low adsorption. At intermediate pH, the trend of trace element adsorption increases from almost no adsorption to almost complete adsorption within a narrow pH range called the pH-adsorption edge. From this point onwards, the elements are completely adsorbed (Bradl, 2004). Therefore, we can see that the disruption of soil pH can lead to a ripple effect of what nutrients can be taken up by plants, what is left in soils, and what is leached away from soils and into water tables. Fire often leads to an increase in pH in soils, and the longevity of this change relates to wind and water erosion, as well as the longevity of ashes physically on site (Wells, 1979). While fire often leads to an increase in pH, there have been limited studies connecting the highest levels of burn severity and pH, especially on soils in the California region.

2.7.3 Extractable Cations

Inorganic cations, consisting of extractable Potassium (K_{ex}) , Calcium (Ca_{ex}) , and Magnesium (Mg_{ex}) are not typically volatized by fire due to high volatilization temperatures (Cerda, 2009) yet they may be transported from burn sites by other means such as leaching, wind, or surface erosion. CEC is known to show the soil's ability to supply Ca_{ex}, Mg_{ex}, and K_{ex} in soils. These nutrients are all essential to plant growth. Ca_{ex} is a key contributor to the structure of cells and the upholding of physical barriers against pathogens. Mg_{ex} is the central core of the chlorophyl molecule in plant tissues. Finally, K_{ex} is associated with the movement of water, nutrients, and carbohydrates in plant tissue. It is also involved with enzyme activation within the plant, which affects protein, starch, and adenosine triphosphate (ATP) production. The production of ATP can regulate the rate of photosynthesis (Pandey, 2018b). These three nutrients are critical for plant growth; thus, the CEC plays a strong role in understanding overall soil health. CEC has been studied on a range of different environments both post wildfire (Fernández et al., 1997; Huerta et al., 2020) and post lab experimentation (Fernández et al., 1997; Giovannini et al., 1990; Hatten et al., 2005; Huerta et al., 2020), and the overall theme is that CEC from fire overall decreases with temperature. Yet the lack of field studies indicate that CEC shifts can depend on organic matter and soil physical properties of the area. There are limited studies existing on the fire prone area of the Sierra Nevada region of California.

2.7.4 Phosphorous

Phosphorous (P) is known to have a critical role in cell development in plants and is a crucial component of molecules in the plant that store energy, such as ATP (adenosine triphosphate), DNA and lipids (fats and oils). P is a macronutrient for plants and is critical to the continued growth of plant matter. P is also key for early root development in new plants, as well as hastening plant maturity (Pandey, 2018b). P is a macronutrient that is particularly limiting in soil systems and becomes insoluble in both low and high pH values. When pH values are higher, as after a fire, calcium compounds tend to immobilize it. Overall, there is controversy in the studies regarding the effects of fire on P levels. Some studies done observing low-intensity fires have shown an overall increase in extractable P in surface soils (Adams et al., 1994; McKee, 1982) while other studies have shown that only higher burn severity will have a sharp increase in inorganic P (Hernández et al., 1997; Raiesi & Pejman, 2021; Saa et al., 1993). Overall, the effects of soil on P are misunderstood, despite its critical nature to plant mass. This indicated that there are other factors at play, such as physical soil factors, burn severity, and original SOM and plant matter.

2.7.5 Nitrogen

Total soil nitrogen (N) is the mineral nutrient required in the greatest amount and its availability is a major factor limiting growth and development of plants (Kraiser et al., 2011). Although, it is also considered the most limiting nutrient in wildland ecosystems because it isn't produced by decomposition of rocks and minerals (Rosswall, 1976). This macronutrient for plant growth has been known to exist in soils in many forms, specifically studied as nitrate (ammonia from the decomposition process oxidizes forming nitrates, $NO3^-$), organic N (nitrogen that had organic parent material), and potentially mineralizable N (PMN; an indicator of the soil microbial community to break down nitrogen into a form that is able to be taken up by plants).

Additionally, PMN is an indicator of the capacity of the soil microbial community to convert (mineralize) nitrogen tied up in complex organic residues into the plant available form of ammonium. This often has to do with soil texture and composition, as clay particles often prevent mineralization of nutrients and prevent absorption.

All types of soil N in are affected by fire in unique ways. Volatilization (the loss of N through the conversion of ammonium to ammonia gas, which is released to the atmosphere) is the process which is most responsible for N losses during fire where temperatures frequently exceed 200^{o} to 300^{o} C and N loss by volitation gradually increases as temperature increases (Knight, 1966). N can also be lost from soils during particulate matter creation from fire and subsequent wind-borne transport (Bustamante et al., 2006). There has been conflicting reports on just how much N is lost during fire (Baird et al., n.d.; Cotrufo et al., 2019; Huerta et al., 2020, 2020; Knight, 1966; Miesel, 2009; Miesel et al., 2011; Prendergast-Miller et al., 2017), indicating that other variables, such as burn severity, site variables, and physical soil properties need to be taken into account when testing.

2.8 The Sierra Nevada Mountains and Creek Fire

There has been discussion regarding the history of fire in California, in that if it has always played a role shaping the landscape why should we not assume that this is a natural pattern taking its course. Unfortunately, fire has not yet impacted the California landscape to the severity or extent historically as it has been in recent years. It has been observed that a large area (approximately 120000 km^2) of California and western Nevada experienced a notable increase in the extent of high severity fire between 1984 and 2006. High severity forest fire is intricately linked to forest fragmentation, wildlife habitat availability, erosion rates and sedimentation, post-fire seedling recruitment, C sequestration, and various other ecosystem properties and processes. In this region, it is also observed that both mean and maximum fire size, and the area burned annually have also all risen substantially since the beginning of the 1980s and are now at or above values from the decades preceding the 1940s, when fire suppression became national policy (Miller et al., 2009). In 2020, the Creek Fire scorched the Sierra Nevada mountains, resulting in 379,859 acres burned across the counties of Fresno and Madera after 111 days ablaze. The most probable cause of this fire is lightning, meaning that this fire was naturally caused. The Creek Fire burned in one of the peak “fire seasons” of California’s history, and burned in an area in the Sierra Nevada mountains that has not been widely studied. The Creek Fire meets the qualifications of a “megafire” and had varying burn severity levels from unburned to high intensity burns, making this fire an optimal burn to further research.

2.9 Digital Soil Mapping

Wildfire has been observed to alter soil health, yet little research has been done to control the other soil forming factors to isolate the effects of wildfire. Soil forming factors consist of climate, organisms, relief/topography, parent material and time (CLORPT) (Jenny, 1946). In locations where soil forming factors are the similar, soils will also be similar (Jenny, 1946). The Creek Fire burn area is a relatively massive burn scar, with extreme variance in soil forming factors (Dahlgren et al., 1997; Stavros et al., 2016). Digital soil mapping utilizes remote sensing data to account for the variability of CLORPT in a region. Some studies suggest that digital soil mapping can be a practical solution for refining the spatial variability of soil in large areas, and can both improve fire prediction models and inform management decisions (Levi et al., 2018). Of the limited studies which incorporate remotely sensed CLORPT factors into models (Levi & Bestelmeyer, 2016; Stefanidis et al., 2022; Tolorza et al., 2022), few have been conducted in the Sierra Nevada. With the increase of megafires in the Sierra Nevada, it is critical to account for ecosystem variability to gain a more accurate understanding of the effect wildfire has on forest soil health.

2.10 Conclusion

The Creek Fire provides an optimal base to explore the effects of megafires on soils, mainly at different severities. Current studies have not measured across burn severity levels (Pellegrini et al., 2022; Raison et al., 1990; Santín et al., 2016). While there are several studies found which account for burn severity, they often are not based in California (Hernández et al., 1997; Pereira et al., 2014; Raiesi & Pejman, 2021; Turner et al., 2007; Xue et al., 2014). Studies that do take into account burn severity levels either do not test a wide enough range of soil health indicators (Adkins et al., 2019; Hamman et al., 2008; Murphy et al., 2006; Santos et al., 2016), or soils were tested in lab, not fully able to take into account physical and landscape features of the fire (Abney & Berhe, 2018; Galang et al., 2010). For these reasons, we believe that there is a knowledge gap in the understanding of fire severity on soil health in the California region. Additionally, we think that studying the soils from the Creek Fire will provide us with the information to fill this knowledge gap. The creek fire burn area has a diverse climate and topographical features, as well as a range of organisms (Adkins et al., 2019; Manley et al., 2000; Murphy et al., 2006). When we control for environmental variables in this study, we can isolate the effects of fire, and ideally further strengthening the relationship between soil burn severity and soil health. By studying the soils health indicators of pH, C, POXC, MinC, N, extractable cations, CEC, P, we believe we will be able to paint a picture of soil health post-fire in this region.

CHAPTER 3 METHODOLOGY

3.1 Site Description

The Creek Fire burn is an area of 379,895 acres located in the southern Sierra Navada in the Sierra National Forest, and burned in late 2020 (September- December) (Cal Fire, n.d.). This area experiences average temperatures of 57°, with high temperatures in the 80°s and lower temperatures in the 20°s. The average annual precipitation over the area is 45 inches, with a mix of rain or snow depending on elevation. The region experiences a xeric soil moisture regime and a thermic to frigid soil temperature regime, depending upon elevation, with cool, wet winters and hot dry summers (U.S. Climate Data, 2023). The dominant soil orders consist of Alfisols, Entisols, and Inceptisols (UC Davis, n.d.) Dominant parent material is granitic residuum and colluvium (Segall et al., 1983). The primary vegetation at lower elevations includes grasslands (*Agropyron desertorum, Eriogonum latens, Sisyrinchium halophilum, Sporobolus airoides*), chaparral (*Adenostoma fasciculatum*, *Arctostaphylos spp.)* and woodlands (*Pinus sabiniana, Quercus douglasii, Quercus wislizenii)*. Higher elevations include the Sierran mixed conifer zone (*Pinus ponderosa, Pinus jeffreyi, Pinus lambertiana, Quercus kelloggii, Pinus lambertiana, Arctostaphylos manzanita)*(Manley et al., 2000). Trees become shorter and more scattered with increasing elevations (Manley et al., 2000). Fire frequency of the area is about every 20 years with a mix of fire effects (California, 2022). Additionally, tree mortality in the area is high due to drought, bark beetle infestation, and forest compaction (Asner et al., 2016; Stephens et al., 2018; Young et al., 2017).

3.2 Sampling Mission and Field Data

Immediately postfire, the US Forest burned area emergency response team (BAER) team conducted field work to identify the degree of thermals damage to soils, and classify soils in to burn severity classes, known as the soil burn severity (SBS) for the purpose of generating a SBS map. These maps identify areas that burned at high severity, and therefore were at high risk for postfire flooding and catastrophic debris flows, with subsequent risk to human life, property, and cultural or natural resources (Parson et al., 2010).The BAER teams utilize satellite images indicating pre- and postfire vegetation conditions, referred to as the Burn Area Reflectance Classification (BARC) (Hudak et al., 2004). Satellite imagery is derived from Normalized Burn Ratio (NBR), which combines near-infrared (NIR) and shortwave-infrared (SWIR) to provide a measure of burn severity (U.S. Geological Survey, n.d.). Healthy vegetation will reflect NIR while rock and bare soil will reflect SWIR. In this region, pre-fire there is high NIR and low SWIR values, postfire the signal is reversed, due to vegetation destruction from fire. The difference between pre and postfire NBR is known as the differentiated Normalized Burn Ratio (dNBR) which determines the damage to vegetation following fire. From both the dNBR data and field validation, an SBS map was constructed (fig. 1).

We returned to the spatial location of BAER field observations two years post fire (June 2022) to collect soil samples, as close to the initial BAER observation as possible (within two 30 m pixels). A total of 117 locations were sampled (out of a total of 172 SBS points for the Creek fire). Surface duff, ash and leaf litter was removed, and mineral soil was sampled at two depths 0-5 cm and 5-10cm, for a total of 234 samples. Samples were air dried, sieved to 8 mm or 2 mm, before analysis. All data are reported on an air-dried soil basis. It is important to note that the collection of samples was limited by the accessibility to the BAER team’s previously mapped location due to dangerous conditions, and for the scope of this study all analysis was conducted on surface depth soils (0-5 cm).

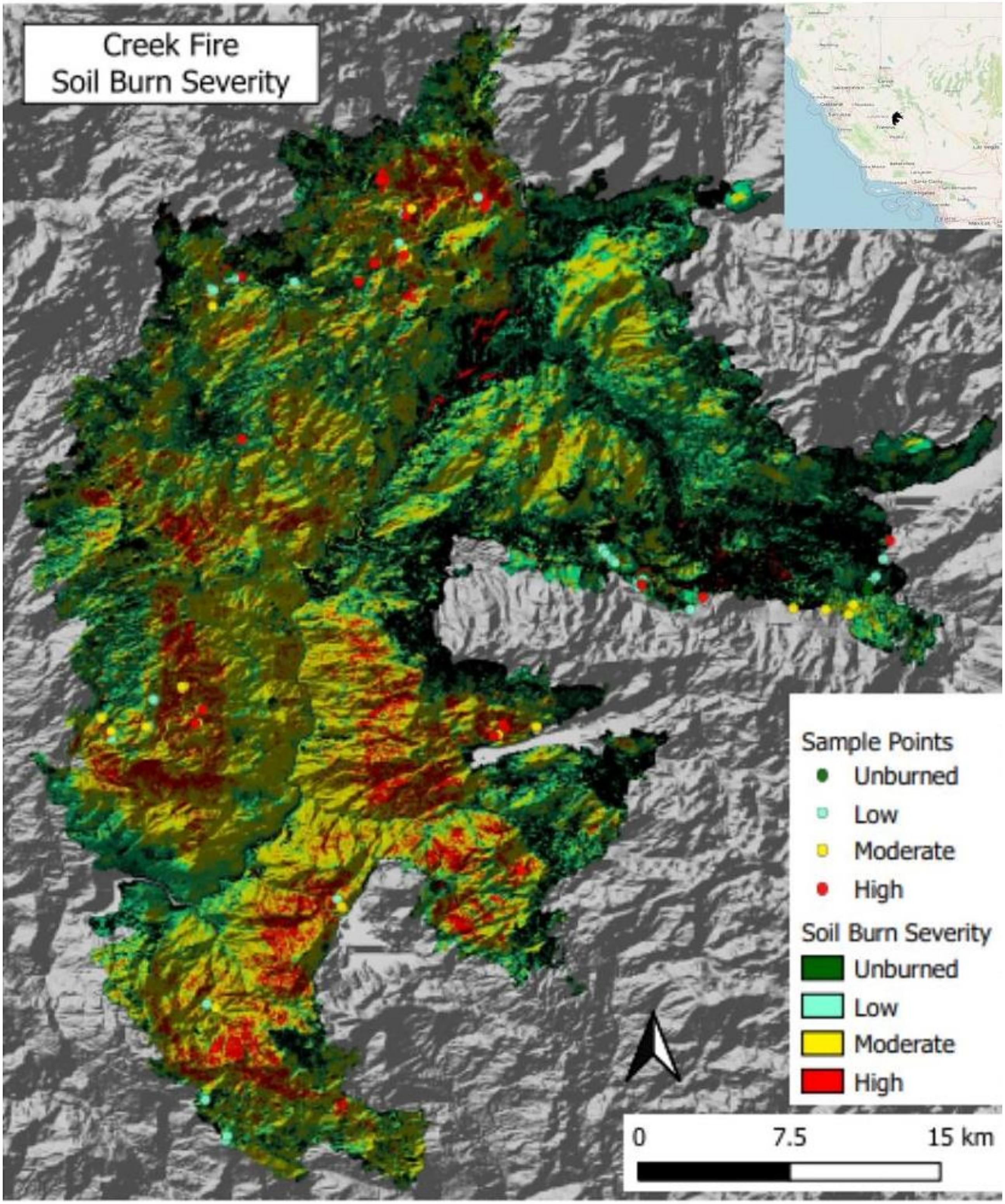

FIGURE 1: CREEK FIRE SOIL BURN SEVERITY MAP
This map shows both sample point location and their corresponding field SBS.

3.3 Soil Analyses

Total C and N were analyzed by combustion, using the Vario Max CNS elemental analyzer (Elementar, Langenselbold, Hesse, Germany). Labile C (POXC) was analyzed by permanganate oxidation (Weil et al., 2003). Mineralizable C (MinC) was analyzed by rehydrating 10g of airdried soil samples to 50% water-holding capacity with deionized water and measuring CO_2 concentration (mg CO2-C kg-1 soil hr-1) with a Li-COR Li-850 CO_2/H_2O gas analyzer (Lincoln, NE, USA) after a 48-hour incubation (NRCS, 2019). Soil pH was tested in 1:1 water. Plant available phosphorus (P) was by Olsen and Bray P with ICP (Bray & Kurtz, 1945; Olsen, 1954). Extractable K (K_{ex}), Ca (Ca_{ex}) , and Mg (Mg_{ex}) were tested by the Ammonium Acetate Method (Haby et al., 1990). Cation exchange capacity (CEC) was found by sum of cations (Chapman, 1965). Soil nitrates (NO_3N) was by KCL extraction with analysis by colorimetry (Keeney, 1983).

3.3 Statistical Analysis

To analyze the effect of field SBS (observed 2020) on each variable, we conducted analysis of variance (ANOVA), with soil health properties as the response variable (i.e. Total C) and the field observed SBS class (high, moderate, low, and unburned) for that location as the predictor variable. If ANOVA was significant, means separation followed by Tukey's HSD at $p<0.05$. Normality of residuals was assessed by the Shapiro-Wilks test and homogeneity of variance was assessed using Levene's test. Where variables did not meet assumptions, the response variable was transformed via log transformations to meet the assumptions of the ANOVA. All analysis were conducted in R-Studio utilizing linear regression modeling and Tukey means separation analysis (Keselman & Rogan, 1977). Next, multiple linear regression analysis

(MLR) was conducted to control for variability associated with other environmental factors know to influence soil properties in the region to illuminate the effect of SBS on soil properties across the large fire footprint, as well as to investigate which environmental covariates were associated with each soil health response.

3.4 Spatial analysis

All environmental covariates for analysis were chosen to represent soil forming factors, aside from time which is not within the scope of this project (McBratney et al., 2003). A variety of rasters corresponding to the SCORPAN factors, such as terrain, climate, and topographic variables were gathered and stacked in R (Table 1). The SCORPAN model shares the same variables as the CLORPT model, which includes climate, organisms, relief, parent material, and time/age. The additional variables include intrinsic properties of the soil, and the spatial coordinates of a sample or the location relative to another geographical phenomenon (Minasny et al., 2013).

Terrain data was derived from USGS national elevation data set at 10m resolution and analyzed in the “terra” package in R (R Spatial, n.d.) Terrain variables include topographic position index (TPI), elevation (DEM), slope, aspect, and curvature. Vegetation variables included enhanced vegetation index (EVI) and normalized difference vegetation index (NDVI) from Landsat 8. Climate data was derived through PRISM Climate Group at Oregon State University at 800m resolution. Climate variables include average temperature (PRISMtmean) and average rainfall (PRISMppt). The SBS was obtained from the USFS BAER postfire assessment team. With the data gathered to meet SCORPAN factors, all raster layers were resampled to

30m resolution, cropped to the same extent, stacked, and layered with the fire perimeter and point locations (fig. 2). This was done in R through the “resample” function of the R “terra” package through bilinear interpolation (R Spatial, n.d.). The spatial locations of each soil sample location were used to extract data from the raster stack to generate a MLR model with the soil health variable as the response variable, with field SBS and the data from the raster layers of environmental covariates as the predictor variables (Minasny et al., 2013). Environmental covariates are included to account for the drastic landscape variability within the 375,000 acer burn scar, which allows for the effects of SBS on the soil health variables to be highlighted.

TABLE 1: DESCRIPTIONS OF ENVIRONMENTAL COVARIATES

Environmental Covariates	Description
TERRAIN	
DEM	Digital Elevation Model (Elevation)
Slope	Identifies the gradient or steepness
Aspect	Identifies compass direction of downhill slope
TPI	Topographic Position Index
Curvature	Identifies curvature of a raster surface, including profile
VEGITATION	
EVI	Enhanced Vegetation Index, accounts for cloud cover
NDVI	Normalized Difference Vegetation Index
CLIMATE	
PRISM(ppt)	Mean Precipitation
PRISM(tmean)	Mean Temperature
FIRE	
SBS	Soil Burn Severity

CHAPTER 4 RESULTS

4.1 Effects of Burn Severity on Soil C

Soil C ($p=0.02$) and C/N ratio ($p=0.09$) were marginally affected by field SBS classification. Total C in high SBS soils (*mean = 3.6*) was statistically different than moderate (*mean = 2.8*) and unburned (*mean = 2.7*) areas (Fig. 3a). Additionally, low SBS classified soils (*mean = 3.3*) were statistically different than moderate (*mean* = 2.8) and unburned ($m = 2.7$) soils. The C/N ratio in high SBS soils (*mean = 3.2*) was statistically different than from both low SBS soils (*mean = 3.1*) and unburned SBS soils (*mean = 3.1*) (Fig. 3b). Both POXC and MinC were not affected by SBS ($p > 0.05$) (Fig. 3 c,d).

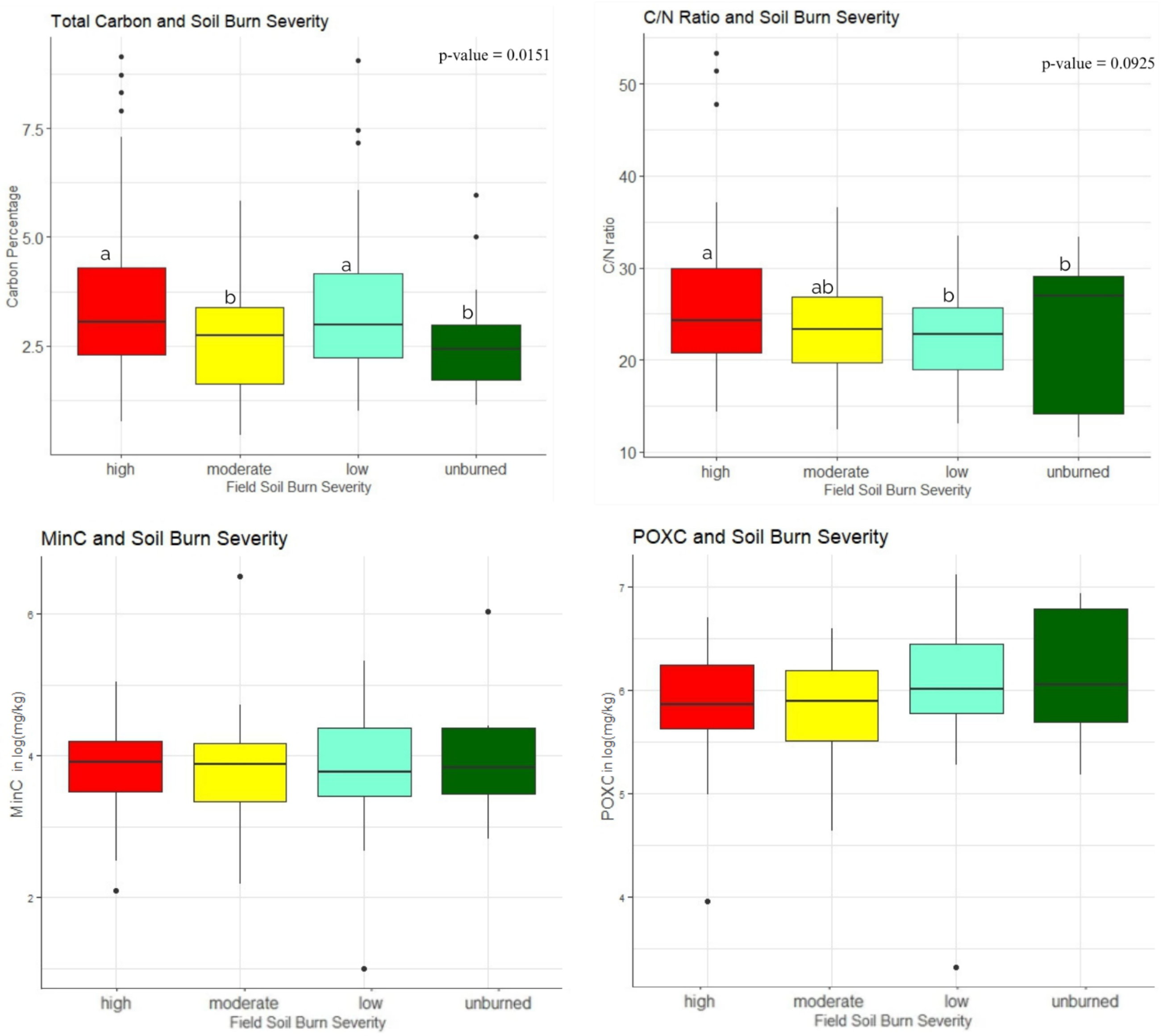

Figure 2 a-d Carbon, C/N Ratio, MINC and POXC and Field soil Burn Severity

4.2 Effects of Burn Severity on Soil Chemical Properties

Both pH (fig. 4) and total N (fig. 5) were not statistically affected by SBS class (*$p>0.05$)*. However, NO_3N (*$p=0.003$)* (Fig. 6) and K_{ex} (*$p=0.03$)* (Fig.7) were significant. For NO_3N, across severities, moderate SBS soils had the highest level of NO_3N (*mean = 2.2*) and were higher than both high SBS soils (*mean = 1.9*) and unburned soils (*mean = 1.0*) . For K_{ex} (Fig 7), moderate SBS soils (*m = 167.8*) had significantly more K_{ex} than unburned soils (*mean = 102.7*). Other extractions such as CEC, Olsen P, Ca_{ex}, and Mg_{ex} were not statistically affected by SBS ($p > 0.05$) (Fig. 8-11).

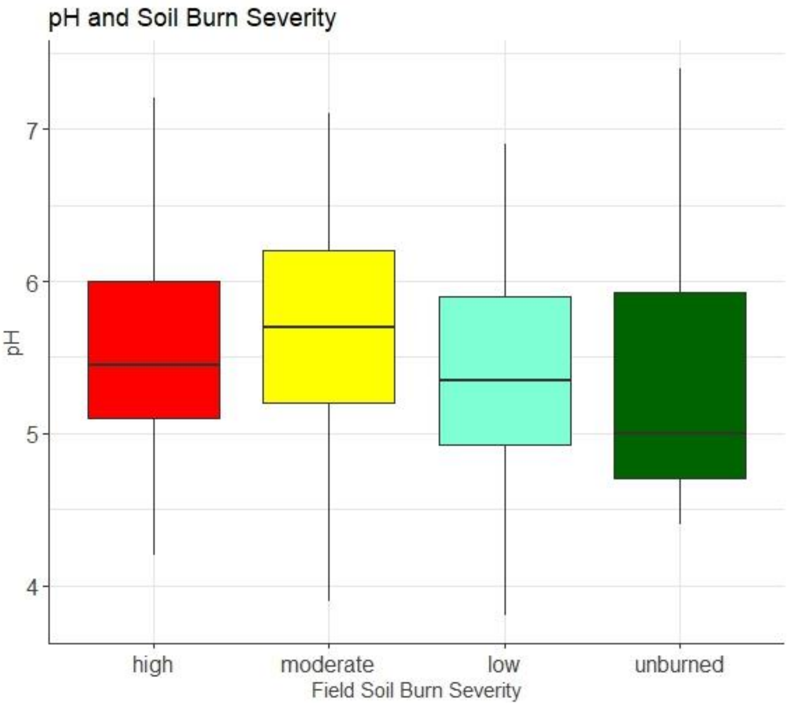

FIGURE 4: PH AND FIELD SOIL BURN SEVERITY

The pH level categorized by field burn severity. The p-value was statistically insignificant ($p>0.05$)

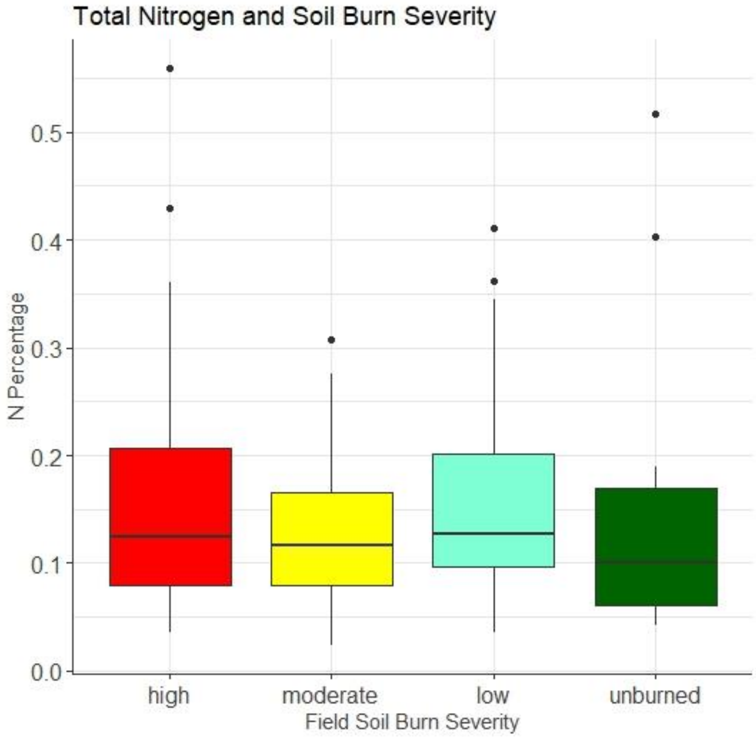

FIGURE 3: N AND FIELD SOIL BURN SEVERITY

The N% categorized by field soil burn severity. The p-value was statistically insignificant ($p>0.05$).

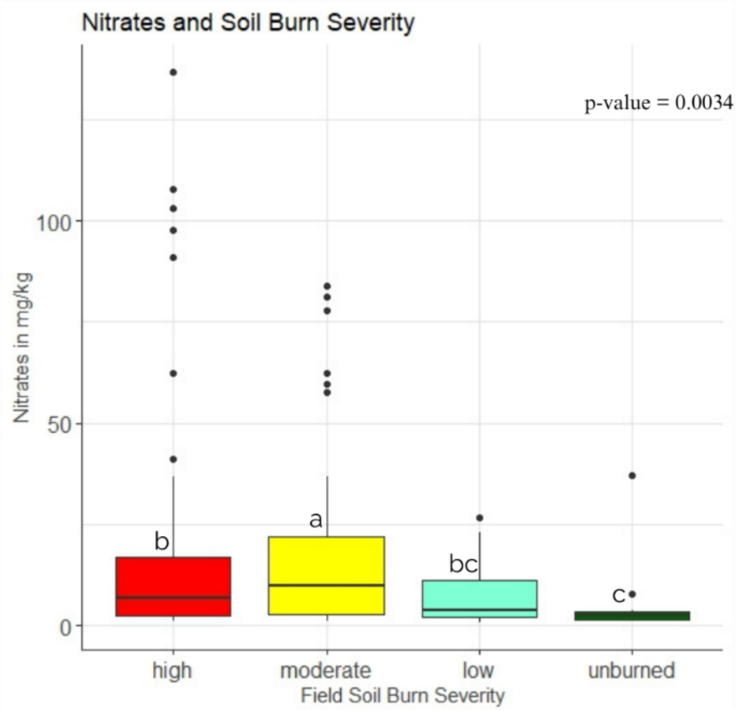

FIGURE 5: NO$_3$N AND FIELD SOIL BURN SEVERITY

Nitrates in mg/kg categorized by field soil burn severity. The p-value was statistically insignificant (p>0.05)

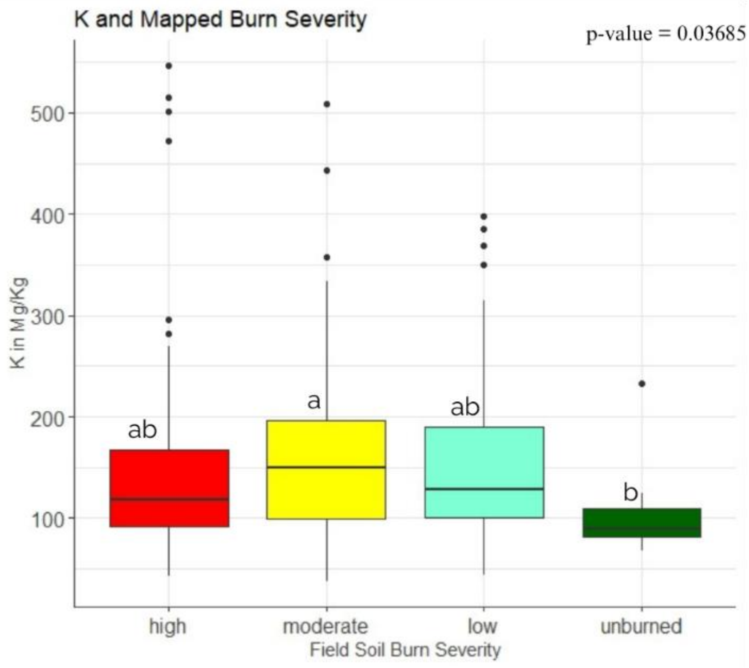

FIGURE 6: K$_{EX}$ AND FIELD SOIL BURN SEVERITY

K_{ex} in mg/Kg categorized by field soil burn severity. The p-value was statistically insignificant (p>0.05)

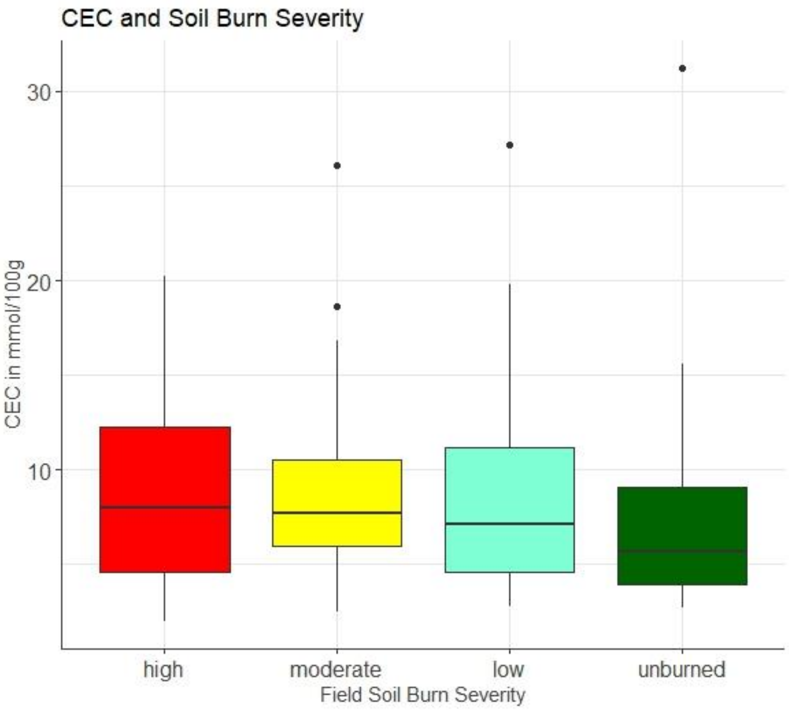

FIGURE 7: CEC AND FIELD SOIL BURN SEVERITY
CEC categorized by field soil burn severity. The p-value was statistically insignificant (p>0.05)

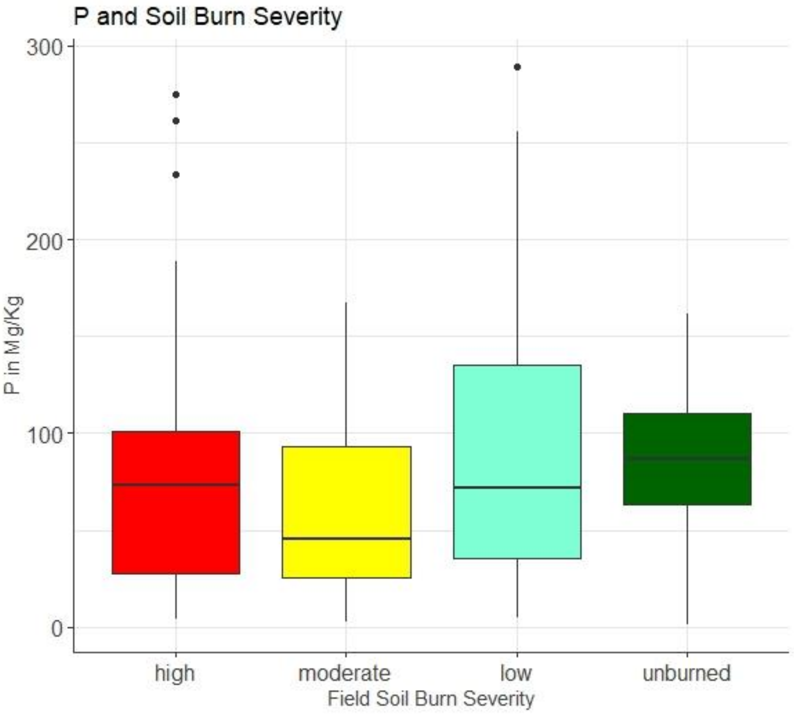

FIGURE 8: P AND FIELD SOIL BURN SEVERITY
P in mg/kg and field soil burn severity. The p-value was statistically insignificant (p>0.05)

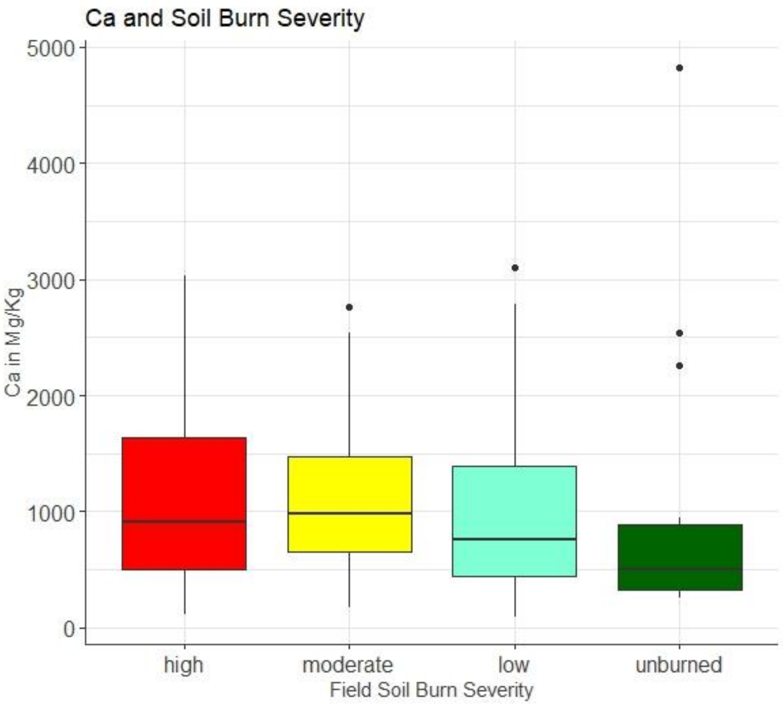

FIGURE 10: CA_{EX} AND FIELD SOIL BURN SEVERITY

Ca$_{ex}$ in mg/kg and field soil burn severity. The p-value was statistically insignificant ($p>0.05$)

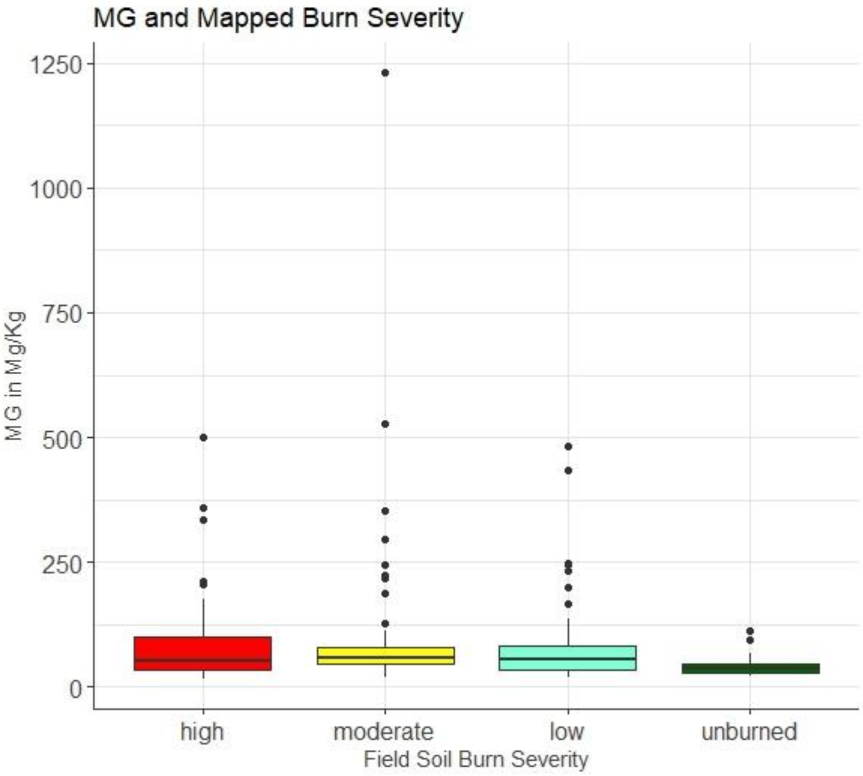

FIGURE 9:MG$_{EX}$ AND FIELD SOIL BURN SEVERITY

Mg$_{ex}$ in mg/kg and field soil burn severity. The p-value was statistically insignificant (p.0.05)

4.3 Influence of Environmental Covariates on Soil Health

To understand potential environmental controls on soil health in postfire landscapes, we examined a suite of environmental covariates and how they are related to measured soil health properties through multiple linear regression modeling (MLR). The intentions were to understand whether other environmental variables known to drive the variability of both soil health and fire behavior, along with the field SBS classification, would help explain the variance in our soil health metrics. Table 2 shows corresponding beta coefficients of each soil health response variable to each of the environmental covariate predictor variables, with asterisks to indicate significance. The beta coefficients represent the degree of change for each model when comparing soil health variables and environmental covariates. For example, for every 1 unit increase in SBS, Total C decreases by 0.83. For every 1 unit increase in temperature, total C will increase by 0.49, with all other variables held constant.

4.4 Effect of SBS on Soil Health Variables: ANOVA versus Mixed Linear Regression Modeling

With ANOVA analysis, SBS was statistically significant with total C ($p<0.05$), NO_3N ($p<0.05$), K_{ex} ($p<0.05$), and was marginally significant with C/N ratio ($p<0.1$). Through MLR modeling, which included environmental covariates (elevation, slope, aspect, TPI, curvature, EVI, NDVI, mean precipitation and mean temperature), the statistical significance of C/N ratio ($p<0.05$) and K_{ex} ($p<0.001$) increased. Additionally, POXC showed statistical significance ($p<0.05$), and pH ($p<0.1$), P ($p<0.1$) and Mg_{ex} ($p<0.1$) showed marginal significance. With MLR, total C now showed a significant decrease (ß= -0.25) as opposed to the increase shown in the ANOVA model(ß= 0.01) (Table 3). A significant decrease with an increase in SBS was also shown in POXC (ß= -0.06), C/N ratio (ß= -0.25), and P (ß= -0.27). Increase with and increase in SBS was observed in pH (ß= 0.02), and Mg_{ex} (ß= 0.43).

4.5 Environmental Covariates Effects on Soil Health Variables

Several of the raster environmental covariates significantly explained variance in soil health variables (Table 2). The model for total C was significant *(R^2= -0.25, p-value<0.001)* and suggested that SBS, slope, TPI, curvature, and average annual temperature significantly explained the variance in Total C (Table 2). Total C increased with an increase in TPI (ß= 5.36) and temperature (ß= 0.49) and decreased with an increase in SBS (ß= -0.83), slope (ß= -0.23), and curvature (ß= -5.2). The model for POXC was significant (*R^2= 0.39, p-value<0.001)* and suggested that SBS and average precipitation explained the variance in POXC. Increases in both SBS (ß= -0.06) and average precipitation (ß= -0.32) decreased POXC. The MinC model was significant (*R^2= 0.01, p-value<0.001)*, with curvature, EVI, and NDVI, significantly explaining MinC variance. MinC significantly decreased with an increase in curvature (ß= -0.47) and EVI (ß= -0.24) and increased with an increase in NDVI (ß= 0.31). The C/N ratio model was significant (*R^2= 0.24, p-value<0.001)* with SBS, elevation, slope, EVI, mean precipitation, and mean temperature explaining the variance. The C/N ratio had a significant increase with an increase in slope (ß= 0.16), EVI (ß= 0.25), and precipitation (ß= 0.39) and significant decrease with and increase in average temperature (ß= -0.5), SBS (ß= -1.92), elevation (ß= -0.31).

The MLR model for pH was significant *(R^2=0.18, p-value < 0.001)* and suggested that pH significantly decreased with increase in elevation (ß= -0.42) and significantly increased with an increase in slope (ß= 0.2) and NDVI (ß= 0.45). Additionally, pH was marginally significant with

SBS (ß= 0.02). The model for total N was significant *(R^2=0.16, p-value < 0.001)* had a significant increase with elevation (ß= 0.41), EVI (ß= 0.17), and average temperature (ß= 0.50) and significant decreases with an increase in slope (ß= -0.22) and curvature (ß= -6.77). The model for NO_3N was significant (*R^2=0.15, p-value < 0.001)* with variance significantly explained by SBS and elevation. The amount of NO_3N significantly increased with an increase in SBS (ß= 0.38) and decreased with a rise in elevation (ß= -0.25). The model for K_{ex} was significant *(R^2=0.38, p-value < 0.001)* with SBS, elevation, slope, curvature, EVI, mean precipitation, and mean temperature significantly explaining the variance. The amount of K_{ex} significantly decreased with an increase in EVI (ß= -0.05) and increased with an increase in SBS (ß= 0.09), elevation (ß= 0.3), slope (ß= 0.06), precipitation (ß= 0.2), and average temperature (ß= 0.09). The model for P was significant (*R^2=0.06, p<0.05)* significantly decreased with SBS (ß= -0.27) and precipitation (ß= -0.36). The CEC model was significant (*R^2=0.22, p<0.001)* with variance explained by elevation, slope, aspect, and EVI. The CEC significantly decreased with an increase in elevation (ß= -0.51) and increased with an increase in slope (ß= 0.12) and EVI (ß= 0.09). The model for Ca_{ex} was significant (*R^2=0.04, p<0.001),* with variance explained by elevation, aspect, EVI and slope. The amount of Ca_{ex} significantly decreased with an increase in elevation (ß= -0.58), aspect (ß= -0.07), and EVI (ß= -0.14) and increased with an increase in slope (ß= 0.18). The model for Mg_{ex} was statistically significant (*R^2=0.26, p<0.001)* with variance explained by SBS, elevation, EVI, and mean temperature. The amount of Mg_{ex} significantly increased with an increase in SBS (ß= 0.43), elevation (ß= 0.32), EVI (ß= 0.35) and average temperature (ß= 0.53).

Table 2: Environmental Covariates and Soil Health Variables

Soil health variables and environmental covariates with correlated beta coefficients. An asterisk (*) represents statistical significance at **.** $p<0.1$, *$p<0.05$, **$p<0.01$, ***$p<0.001$.

	SBS	Elev.	Slope	Aspect	TPI	Curv.	EVI	NDVI	Mean Precip.	Mean Temp.	_Model R^2_
C	-0.25 *	0.5	-0.23 **	-0.15	5.36 *	-5.2 *	-0.03	-0.05	-0.15	0.49 **	0.12 ***
POXC	-0.06 *	0.39	0.23	-0.08	-0.97	0.97	-0.34	-0.04	-0.32 *	0.57	0.39 ***
MinC	-0.06	-0.31	0.16	0.04	0.53	-0.47 ***	-0.24 *	0.31 *	0.39	-0.5	0.01 ***
C/N	-0.25 *	-0.31 ***	0.16 *	0.04	0.53	-0.47	-0.25 ***	0.31	0.39 **	-0.5 .	0.24 ***
pH	0.02 .	-0.42 **	0.2 ***	-0.07	1.16	-1.15	-0.51	0.45 **	-0.12	-0.41	0.18 ***
N	0.06	0.41 *	-0.22 **	-0.14	6.83	-6.77 **	0.17 *	-0.22	-0.29	0.5 *	0.16 ***
NO_3N	0.38 ***	-0.25 ***	-0.06	0.07	-0.64	0.59	0.43	-0.28	-0.02	0.19	0.15 ***
K	0.09 **	0.3 ***	0.06 ***	0.03	1.68	-1.7	-0.05 .	0.16	0.2 *	0.09 **	0.38 ***
P	-0.27 .	0.4	0.24	-0.03	-1.55	1.54	-0.27	-0.13	-0.36 *	0.59	0.06 *
CEC	0.32	-0.51 ***	0.12 **	-0.05	4.51	-4.49	0.09 ***	0.12	-0.09	-0.15	0.22 ***
Ca	0.22	-0.58 ***	0.18 ***	-0.07 .	3.44	-3.42	-0.14 ***	0.27	-0.11	-0.32	0.04 ***
Mg	0.43 .	0.32 .	-0.01	-0.08	3.15	-3.15	0.35 ***	-0.21	-0.21	0.53 *	0.26 ***

Table 3: Comparison of SBS Significance on Soil Health Variables Between ANOVA and MLR Models

Soil health variables and SBS with correlated beta coefficients. SBS_1 shows beta coefficients from ANOVA models., while SBS_2 shows beta coefficients from MLR models. The R^2 values are from MLR models. An asterisk (*) represents statistical significance at **.** *p<0.1, *p<0.05, **p<0.01, ***p<0.001.*

	C	POXC	MinC	C/N	pH	N	NO_3N	K	P	CEC	Ca	Mg
SBS_1	0.01 *	-0.01	-0.03	-0.09 .	0.16	0.05	0.38 ***	0.26 *	-0.19	0.22	0.12	0.31
SBS_2	-0.25 *	-0.06 *	-0.02	-0.25 *	0.02 .	0.06	0.38 ***	0.09 **	-0.27 .	0.32	0.22	0.43 .
R^2	*0.12*	*0.39*	*0.01*	*0.24*	*0.18*	*0.16*	*0.15*	*0.38*	*0.06*	*0.22*	*0.04*	*0.26*

CHAPTER 5 DISCUSSION

We evaluated the relationship between SBS and soil properties both alone, then in concert with environmental predictor variables to examine the relationship between SBS, environmental predictors variables and soil health two years post fire. Results from analysis of variance were not as expected, with results not conforming to our original hypothesis, however, after inclusion of environmental predictor variables that are known to explain variance in soil properties (i.e. the SCORPAN/CLORPT factors), burn severity affected soil health variables in a way more consistent with the literature.

5.1 Impacts of Soil Burn Severity on Soil Health Variables

Results from ANOVA analysis with only SBS as the predictor variable suggest that SBS did not directly impact soil health variables as strongly as hypothesized. Total C marginally increased due to SBS, which was not what we anticipated, as other studies have found that total C decrease with increase in SBS, with the largest decrease at higher severities (Huerta et al., 2020). Research has shown severe fires tend to reduce thickness of the organic soil horizon and therefore, total C (Neill et al., 2007). However, other reports have found lower severity fire can increase soil organic carbon (SOC) content through the deposition of dry leaves and plant material (Chandler et al., 1983; González-Pérez et al., 2004). These conflicting studies indicate that the relationship between SBS and SOC may be dependent on other sources of variability outside of fire. This theory aligns with the findings of Huerta et al. who found when SBS was analyzed in conjunction with vegetation type, it strengthened the effect of SBS on total C, as fire affects the chemical composition of SOC, as well as rates of decomposition, and these changes are conditioned by severity, but ultimately controlled by vegetation (Huerta et al., 2020).

We had hypothesized that the K_{ex}, NO_3N and C/N ratio would either increase or remain unchanged. Both K_{ex} and NO_3N showed significant increase with increase SBS, which aligns with other research and our initial hypothesis (Hernández et al., 1997; Turner et al., 2007). The ANOVA analysis for K_{ex} is consistent with the findings in other studies, where overall increase was observed with a small decrease only at highest burn severities (Franklin et al., 2003; Huerta et al., 2020; Neary et al., 2005). In our analysis, increase of K_{ex} at low and moderate severities is most likely due to the mineralization of soil and vegetation organic matter, which mobilizes cations into a soluble fraction, resulting in ash deposited during fire containing high concentrations of cations (DeBano et al., 1998). Amounts of NO_3N followed a similar pattern to K_{ex}, with increase at low and moderate severities, and slight decrease at high severity. Increase of NO_3N post-wildfire has been recorded in previous studies of fire in forested ecosystems (Hernández et al., 1997); accrediting the increase in NO_3N to higher nitrification potentials of burnt pine forests (Kutiel & Shaviv, 1992). Additionally, Dunn et al. 1979, found that NO_3N levels in burn soils remained higher at least 1 year post fire and suggested that heterotrophic nitrification occurred in burned soils (Dunn et al., 1979). The C/N ratio marginally decreased with an increase in SBS, which may be attributable to lower volatilization temperature for C than N, leading to an overall decrease in C/N ratio (Araya et al., 2017; Pereira et al., 2014; Santos et al., 2016). Although, some studies have shown that C/N ratio is often only affected in more severe wildfires (Johnson et al., 2008), indicating that similar to total C there may be site specific factors affecting C/N ratios. While SBS did not have a significant effect on other soil health variables, this is most likely due to the effects of SBS being masked by other soil forming factors (SCORPAN/CLORPT) which can be explained in part through the environmental covariate multiple linear regression model.

5.2 Soil Burn Severity impact on Soil Health Variables with Inclusion of Environmental Covariates through Multiple Linear Regression Modeling

5.2.1 Soil Carbon and Nitrogen Variables

When other environmental covariates that were expected to drive soil variability were included through MLR, the effect of SBS on soil health became more significant, where eight of the twelve soil health properties were now statistically significant with SBS. For total carbon, SBS, slope, TPI, curvature, and mean temperature all significantly impacted total carbon. Interestingly, total C now showed an overall decrease with increase in SBS, where it had shown a slight increase in the ANOVA analysis without environmental predictor variables, which is more inline with what has been reported in prior studies and with our hypothesized decrease of mineral SOC with increased SBS. With inclusion of environmental predictors, SBS was significant at explaining the variability to total C, along with terrain attributes and mean annual temperature. Terrain and climate have been widely reported to affect soil total C (Adkins et al., 2019; Dahlgren et al., 1997; Devine et al., 2020; Haney et al., 2008). For example, Dahlgren et al. 1997, found that microbial activity in higher elevations was limited by temperature, slowing decomposition, and leading to more SOC. Additionally, slope impacts total C content, as erosion increases at steeper slopes, increasing soil acidity and decreasing total C content (Hamid et al., 2021). Labile C showed significant decrease with SBS, which aligns with the hypothesized decrease, as other research has shown labile C pools in burned soils are often less than or equal to amounts in unburned soils (Adkins et al., 2019; Fernández et al., 1997). In our model, average precipitation significantly increased POXC. Increase in precipitation leads to an increase in C, as rainfall can increase microbial activity and promote the decomposition of soil organic content (Wang et al., 2022). By accounting for the variables that control landscape variability within the burn area, such as elevation, slope, EVI and precipitation, we can explain the increased significance of SBS on total C and POXC.

The C/N ratio significantly decreased with increased SBS, and this effect increased with inclusion of the environmental variables. Wildfire results in decreased in fresh litter material for decomposition, as well as slowed decomposition due to a severe soil moisture deficit, (Holden et al., 2015), leading to a decrease in total C, thus a decrease in C/N ratio. Research also suggests that soil N in the post-fire landscape correspond heavily to post-fire conditions (Johnson et al., 2012; Santos et al., 2016; St. John et al., 1976). In research done by St. John et al.,1976, it was found that burned plots immediately post-fire had decreased total N. Yet, with the onset of spring and rainfall, total N was higher in burned plots than controlled, indicating nitrogen fixation post-fire is heavily impacted by both vegetation and precipitation (St. John et al., 1976). This process would increase total N, thus decreasing overall C/N ratio. This explains why the inclusion of environmental covariates, especially terrain variables, mean precipitation, and EVI, increased the overall significance of C/N ratio with SBS in our study.

5.2.2 Soil Chemical Variables

We report and increase in K_{ex} with increased SBS, as has been reported for low severity fire (Franklin et al., 2003; Huerta et al., 2020). With the inclusion of the environmental covariates, the significance of SBS on K_{ex} increased. DeBano et al., 1998 found that the increase of K_{ex} from wildfire is due to the mineralization of soil and vegetation organic matter, which mobilizes cations into a soluble fraction, resulting in ash deposited during fire containing high concentrations of cations (DeBano et al., 1998). Other studies indicate that affects to K_{ex} depends on both SBS level and ecosystem type (Franklin et al., 2003; Kutiel & Shaviv, 1992). Research shows that K_{ex} decrease with elevation increase (Ohdo & Takahashi, 2020; Smith et al., 2002; Unger et al., 2010), which was not found in our study as K_{ex} showed slight increase with increased elevation and slope. Some studies have shown that cations can increase with increased elevation, as increase may be accredited to the vegetation shift through an elevational gradient, not the elevational properties exclusively (Reese & Moorhead, 1996). This indicates that accounting for vegetation and terrain attributes are critical in isolating the effect of SBS and explains why inclusion of for EVI, elevation, and slope in the MLR model increased the significance of SBS.

Total pH, P and Mg_{ex} also showed significance with SBS in the MLR, where pH and Mg_{ex} both increased with increasing SBS, while P decreased with increasing SBS. Our initial hypothesis stated that an increase in SBS would increase pH, which was what was found in the MLR model and is consistent with previous studies (Boerner et al., 2009; Wells, 1979). Fire often leads to an increase in pH in soils due to release of base cations from combustion, deposition of ashes, and loss of hydroxyl groups from clays (Badía & Martí, 2003; Certini, 2005). Although conflicting research in the Sierra Nevada region has indicated that there is no significant effect of burn severity on pH (Murphy et al., 2006). Additionally, some studies have found that there is potential for pH decrease at higher elevations in fire effected soils (Badía & Martí, 2003; Huerta et al., 2020). Increased elevation and slope lead to slower decomposition rates due to extreme temperature and erosion, thus an accumulation of weakly degraded organic matter which creates more acidic soils. Due to this process, soils at higher elevations and at steeper slopes decreases soil pH (Hamid et al., 2021). Controlling elevation and slope in this model allowed for the isolated effects of SBS and explains the significance in the MLR model.

With respect to P, there is conflicting data on the effects of SBS on soil P. Older studies have reported that P increases at lower soil burn severities (Adams et al., 1994; McKee, 1982). Yet, other research has found that concentrations of P were higher in runoff from the burned areas after wildfire (W. W. Miller et al., 2013), which could account for the marginal loss seen in our study, as sampling was done two years post fire. Therefore, it would make sense that the inclusion of environmental covariates such as slope and average precipitation would reveal significance of SBS with P. Extractable Mg increase with SBS was not what research has typically shown, where Mg_{ex} is not volatilized at lower burn temperatures (Cerda, 2009). Some studies have shown that vegetation has a strong effect on long-term soil nutrient levels in post-fire landscape when compared to unburned sites due to recycling by post-fire vegetation (Johnson et al., 2012), which explains the increased significance in SBS with the inclusion of EVI on Mg_{ex}.

5.3 Catastrophic Megafire has Significant Effects on Soil Properties, even Two Years Post-Fire

The MLR model has shown that severe wildfire leaves an imprint on soil health two years after burning. Within the Creek Fire Burn Scar, total Carbon, POXC, C/N ratio and P all show to have decreased significantly with increases in SBS. This is an indication that carbon, both total and labile, as well as the C/N ratio and total P are not recovering to pre-fire levels in the two-year time scale when accounting for other environmental covariates. Comparatively, there is marginal increase in pH and Mg_{ex}, with strong increases in NO3N and K_{ex} in the post-fire landscape. Indicating that there is a decrease in soil acidity and an increase in soil nutrient levels. This data indicates that two years post-fire, soils in this region have not recovered to healthy levels. While there are signs that it is trending towards recovery in this region, in an area heavily prone to reburn, the soil may not have enough time between fires to recover to former levels (Dove et al., 2020; Liang et al., 2017).

5.4 Study Limitations

This study was limited mainly by accessibility. Several of the soil sample locations were not reachable by the means of the sampling team, which in part led to a skewed number of samples, mainly a lack of unburned samples proportional to the other soil burn severities. Additionally, if time allowed, this study would benefit from another year of analysis, to see changes that SBS had on soil health from two years post-fire to three years post-fire. I believe that this timescale would allow for clarity on how SBS is interacting with soil health, as well as the role of the environmental covariates. The burn area had many microtopographic features, and field data on terrain variables for each sample may also help better understand what is happening to soils, as the 10m resolution of the DEM model may not be providing enough information.

CHAPTER 6 CONCLUSION

This study aimed to understand both how field SBS and soil forming factors, represented through environmental covariates, affect soil health. A total of 117 samples at a depth of 0-5cm were collected, then processed for sampling. Twelve soil health indicators (Total C, POXC, MinC, C/N ratio, pH, Total N, NO_3N, K, P, CEC, Ca, and Mg) were selected and samples were tested in the lab for each indicator. Field-validated SBS levels were assigned to each sample, and through ANOVA testing in R, four variables were significant with SBS (Total C, C/N Ratio, NO_3N, and K_{ex}). Multiple linear regression modeling was then used to observe the effects of SBS when accounting for environmental covariates on the samples. Environmental covariates were chosen to represent various soil forming factors including terrain, climate, and organisms. With the inclusion of the other environmental covariates, the effect of SBS on the soil health variables increased in significance, with total C, POXC, C/N ratio, K_{ex}, NO_3N, pH, Mg_{ex}, and P now all showing statistical significance. This information is critical in that soils are the result of not one, but several soil forming factors and processes. With the data from the MLR, it is evident that the soils in the Creek Fire Burn scar have not returned to their pre-fire soil health levels, and that two years is not enough time for soils to recover from wildfires of this caliber. With this information, we recommend that when doing soil analysis post wildfire, other environmental variables must be considered to provide a compete soil story.

REFERENCES

Abney, R. B., & Berhe, A. A. (2018). Pyrogenic Carbon Erosion: Implications for Stock and Persistence of Pyrogenic Carbon in Soil. *Frontiers in Earth Science*, *6*. https://www.frontiersin.org/articles/10.3389/feart.2018.00026

Adams, M. A., Iser, J., Keleher, A. D., & Cheal, D. C. (1994). Nitrogen and Phosphorus Availability and the Role of Fire in Heathlands at Wilsons Promontory. *Australian Journal of Botany*, *42*(3), 269–281. https://doi.org/10.1071/bt9940269

Adkins, J., Sanderman, J., & Miesel, J. (2019). Soil carbon pools and fluxes vary across a burn severity gradient three years after wildfire in Sierra Nevada mixed-conifer forest. *Geoderma*, *333*, 10–22. https://doi.org/10.1016/j.geoderma.2018.07.009

Agbeshie, A., Abugre, S., Atta-Darkwa, T., & Awuah, R. (2022). A review of the effects of forest fire on soil properties. *Journal of Forestry Research*, *33*, 1–23. https://doi.org/10.1007/s11676-022-01475-4

Araya, S. N., Fogel, M. L., & Berhe, A. A. (2017). Thermal alteration of soil organic matter properties: A systematic study to infer response of Sierra Nevada climosequence soils to forest fires. *SOIL*, *3*(1), 31–44. https://doi.org/10.5194/soil-3-31-2017

Asner, G. P., Brodrick, P. G., Anderson, C. B., Vaughn, N., Knapp, D. E., & Martin, R. E. (2016). Progressive forest canopy water loss during the 2012–2015 California drought. *Proceedings of the National Academy of Sciences*, *113*(2), E249–E255. https://doi.org/10.1073/pnas.1523397113

Badía, D., & Martí, C. (2003). Plant Ash and Heat Intensity Effects on Chemicaland Physical Properties of Two Contrasting Soils. *Arid Land Research and Management*, *17*(1), 23–41. https://doi.org/10.1080/15324980301595

Binkley, D., Richter, D., David, M. B., & Caldwell, B. (1992). Soil Chemistry in a Loblolly/Longleaf Pine Forest with Interval Burning. *Ecological Applications*, *2*(2), 157–164. https://doi.org/10.2307/1941772

Boerner, R. E. J., Huang, J., & Hart, S. C. (2009). Impacts of Fire and Fire Surrogate treatments on forest soil properties: A meta-analytical approach. *Ecological Applications*, *19*(2), 338–358. https://doi.org/10.1890/07-1767.1

Bray, R. H., & Kurtz, L. T. (1945). DETERMINATION OF TOTAL, ORGANIC, AND AVAILABLE FORMS OF PHOSPHORUS IN SOILS. *Soil Science*, *59*(1), 39.

Bruun, T. B., Egay, K., Mertz, O., & Magid, J. (2013). Improved sampling methods document decline in soil organic carbon stocks and concentrations of permanganate oxidizable carbon after transition from swidden to oil palm cultivation. *Agriculture, Ecosystems & Environment*, *178*, 127–134. https://doi.org/10.1016/j.agee.2013.06.018

Buragienė, S., Šarauskis, E., Romaneckas, K., Adamavičienė, A., Kriaučiūnienė, Z., Avižienytė, D., Marozas, V., & Naujokienė, V. (2019). Relationship between CO2 emissions and soil properties of differently tilled soils. *Science of The Total Environment*, *662*, 786–795. https://doi.org/10.1016/j.scitotenv.2019.01.236

California, S. of. (2022, January 24). *2021: Another historic Sierra Nevada fire season*. Sierra Nevada Conservancy. https://sierranevada.ca.gov/2021-another-historic-sierra-nevada-fire-season/

Cerda, A. (2009). *Fire Effects on Soils and Restoration Strategies*. CRC Press.

Certini, G. (2005). Effects of fire on properties of forest soils: A review. *Oecologia*, *143*(1), 1–10. https://doi.org/10.1007/s00442-004-1788-8

Chandler, C., Cheney, P., Thomas, P., Trabaud, L., & Williams, D. (1983). Fire in forestry. Volume 1. Forest fire behavior and effects. Volume 2. Forest fire management and organization. *Fire in Forestry.*

Volume 1. Forest Fire Behavior and Effects. Volume 2. Forest Fire Management and Organization. https://www.cabdirect.org/cabdirect/abstract/19850699124

Chapman, H. d. (1965). Cation-Exchange Capacity. In *Methods of Soil Analysis* (pp. 891–901). John Wiley & Sons, Ltd. https://doi.org/10.2134/agronmonogr9.2.c6

Chapter 2: Land–Climate interactions — Special Report on Climate Change and Land. (n.d.). Retrieved June 6, 2023, from https://www.ipcc.ch/srccl/chapter/chapter-2/

Creek Fire Incident Report. (n.d.). Retrieved November 4, 2022, from https://www.fire.ca.gov/incidents/2020/9/4/creek-fire/#incident-damages-losses

Dahlgren, R. A., Boettinger, J. L., Huntington, G. L., & Amundson, R. G. (1997). Soil development along an elevational transect in the western Sierra Nevada, California. *Geoderma*, *78*(3), 207–236. https://doi.org/10.1016/S0016-7061(97)00034-7

DeBano, L. F., Neary, D. G., & Ffolliott, P. F. (1998). *Fire Effects on Ecosystems*. John Wiley & Sons.

Devine, S. M., O'Geen, A. T., Liu, H., Jin, Y., Dahlke, H. E., Larsen, R. E., & Dahlgren, R. A. (2020). Terrain attributes and forage productivity predict catchment-scale soil organic carbon stocks. *Geoderma*, *368*, 114286. https://doi.org/10.1016/j.geoderma.2020.114286

Díaz, S., Hector, A., & Wardle, D. A. (2009). Biodiversity in forest carbon sequestration initiatives: Not just a side benefit. *Current Opinion in Environmental Sustainability*, *1*(1), 55–60. https://doi.org/10.1016/j.cosust.2009.08.001

Dove, N. C., Safford, H. D., Bohlman, G. N., Estes, B. L., & Hart, S. C. (2020). High-severity wildfire leads to multi-decadal impacts on soil biogeochemistry in mixed-conifer forests. *Ecological Applications*, *30*(4), e02072. https://doi.org/10.1002/eap.2072

Drought, Tree Mortality, and Wildfire in Forests Adapted to Frequent Fire | BioScience | Oxford Academic. (n.d.). Retrieved November 29, 2023, from https://academic.oup.com/bioscience/article/68/2/77/4797261?login=false

Dunn, P. H., DeBano, L. F., & Eberlein, G. E. (1979). Effects of Burning on Chaparral Soils: II. Soil Microbes and Nitrogen Mineralization. *Soil Science Society of America Journal*, *43*(3), 509–514. https://doi.org/10.2136/sssaj1979.03615995004300030016x

Fernández, I., Cabaneiro, A., & Carballas, T. (1997). Organic matter changes immediately after a wildfire in an atlantic forest soil and comparison with laboratory soil heating. *Soil Biology and Biochemistry*, *29*(1), 1–11. https://doi.org/10.1016/S0038-0717(96)00289-1

Forests as carbon sinks—Benefits and consequences | Tree Physiology | Oxford Academic. (n.d.). Retrieved October 31, 2022, from https://academic.oup.com/treephys/article/31/9/893/1676008

Franklin, S. B., Robertson, P. A., & Fralish, J. S. (2003). Prescribed burning effects on upland Quercus forest structure and function. *Forest Ecology and Management*, *184*(1), 315–335. https://doi.org/10.1016/S0378-1127(03)00153-1

Galang, M. A., Markewitz, D., & Morris, L. A. (2010). Soil phosphorus transformations under forest burning and laboratory heat treatments. *Geoderma*, *155*(3), 401–408. https://doi.org/10.1016/j.geoderma.2009.12.026

Giovannini, C., Lucchesi, S., & Giachetti, M. (1990). EFFECTS OF HEATING ON SOME CHEMICAL PARAMETERS RELATED TO SOIL FERTILITY AND PLANT GROWTH. *Soil Science*, *149*(6), 344–350.

González-Pérez, J. A., González-Vila, F. J., Almendros, G., & Knicker, H. (2004). The effect of fire on soil organic matter—A review. *Environment International*, *30*(6), 855–870. https://doi.org/10.1016/j.envint.2004.02.003

Goss, M., Swain, D. L., Abatzoglou, J. T., Sarhadi, A., Kolden, C. A., Williams, A. P., & Diffenbaugh, N. S. (2020). Climate change is increasing the likelihood of extreme autumn wildfire conditions across California. *Environmental Research Letters*, *15*(9), 094016. https://doi.org/10.1088/1748-9326/ab83a7

Haby, V. A., Russelle, M. P., & Skogley, E. O. (1990). Testing Soils for Potassium, Calcium, and Magnesium. In *Soil Testing and Plant Analysis* (pp. 181–227). John Wiley & Sons, Ltd. https://doi.org/10.2136/sssabookser3.3ed.c8

Hamid, M., Khuroo, A. A., Malik, A. H., Ahmad, R., & Singh, C. P. (2021). Elevation and aspect determine the differences in soil properties and plant species diversity on Himalayan mountain summits. *Ecological Research*, *36*(2), 340–352. https://doi.org/10.1111/1440-1703.12202

Hamman, S. T., Burke, I. C., & Knapp, E. E. (2008). Soil nutrients and microbial activity after early and late season prescribed burns in a Sierra Nevada mixed conifer forest. *Forest Ecology and Management*, *256*(3), 367–374. https://doi.org/10.1016/j.foreco.2008.04.030

Haney, R. L., Brinton, W. H., & Evans, E. (2008). Estimating Soil Carbon, Nitrogen, and Phosphorus Mineralization from Short-Term Carbon Dioxide Respiration. *Communications in Soil Science and Plant Analysis*, *39*(17–18), 2706–2720. https://doi.org/10.1080/00103620802358862

Hatten, J., Zabowski, D., Scherer, G., & Dolan, E. (2005). A comparison of soil properties after contemporary wildfire and fire suppression. *Forest Ecology and Management*, *220*(1), 227–241. https://doi.org/10.1016/j.foreco.2005.08.014

Hernández, T., García, C., & Reinhardt, I. (1997). Short-term effect of wildfire on the chemical, biochemical and microbiological properties of Mediterranean pine forest soils. *Biology and Fertility of Soils*, *25*(2), 109–116. https://doi.org/10.1007/s003740050289

Holden, S. R., Berhe, A. A., & Treseder, K. K. (2015). Decreases in soil moisture and organic matter quality suppress microbial decomposition following a boreal forest fire. *Soil Biology and Biochemistry*, *87*, 1–9. https://doi.org/10.1016/j.soilbio.2015.04.005

Hudak, A., Robichaud, P., Evans, J., Clark, J., Lannom, K., Morgan, P., & Stone, C. (2004). Field validation of Burned Area Reflectance Classification (BARC) products for post fire assessment. *USDA Forest Service / UNL Faculty Publications*. https://digitalcommons.unl.edu/usdafsfacpub/220

Huerta, S., Fernández-García, V., Calvo, L., & Marcos, E. (2020). Soil Resistance to Burn Severity in Different Forest Ecosystems in the Framework of a Wildfire. *Forests*, *11*(7), Article 7. https://doi.org/10.3390/f11070773

Jenny, H. (1946). ARRANGEMENT OF SOIL SERIES AND TYPES ACCORDING TO FUNCTIONS OF SOIL-FORMING FACTORS. *Soil Science*, *61*(5), 375.

Johnson, D. W., Walker, R. F., McNulty, M., Rau, B. M., & Miller, W. W. (2012). The Long-Term Effects of Wildfire and Post-Fire Vegetation on Sierra Nevada Forest Soils. *Forests*, *3*(2), Article 2. https://doi.org/10.3390/f3020398

Johnson, Dale. W., Fenn, Mark. E., Miller, W. W., & Hunsaker, C. F. (2008). Chapter 18 Fire Effects on Carbon and Nitrogen Cycling in Forests of The Sierra Nevada. In A. Bytnerowicz, M. J. Arbaugh, A. R. Riebau, & C. Andersen (Eds.), *Developments in Environmental Science* (Vol. 8, pp. 405–423). Elsevier. https://doi.org/10.1016/S1474-8177(08)00018-1

Karlen, D. L., Mausbach, M. J., Doran, J. W., Cline, R. G., Harris, R. F., & Schuman, G. E. (1997). Soil Quality: A Concept, Definition, and Framework for Evaluation (A Guest Editorial). *Soil Science Society of America Journal*, *61*(1), 4–10. https://doi.org/10.2136/sssaj1997.03615995006100010001x

Keeley, J. E., Syphard, A. D., Keeley, J. E., & Syphard, A. D. (2018). Historical patterns of wildfire ignition sources in California ecosystems. *International Journal of Wildland Fire*, *27*(12), 781–799. https://doi.org/10.1071/WF18026

Keeney, D. R. (1983). Nitrogen—Availability Indices. In *Methods of Soil Analysis* (pp. 711–733). John Wiley & Sons, Ltd. https://doi.org/10.2134/agronmonogr9.2.2ed.c35

Keselman, H. J., & Rogan, J. C. (1977). The Tukey multiple comparison test: 1953–1976. *Psychological Bulletin*, *84*(5), 1050–1056. https://doi.org/10.1037/0033-2909.84.5.1050

Khorshidi, M. S., Dennison, P. E., Nikoo, M. R., AghaKouchak, A., Luce, C. H., & Sadegh, M. (2020). Increasing concurrence of wildfire drivers tripled megafire critical danger days in Southern California between1982 and 2018. *Environmental Research Letters*, *15*(10), 104002. https://doi.org/10.1088/1748-9326/abae9e

Kutiel, P., & Shaviv, A. (1992). Effects of soil type, plant composition and leaching on soil nutrients following a simulated forest fire. *Forest Ecology and Management*, *53*(1), 329–343. https://doi.org/10.1016/0378-1127(92)90051-A

Lal, R. (2004). Soil Carbon Sequestration Impacts on Global Climate Change and Food Security. *Science*, *304*(5677), 1623–1627. https://doi.org/10.1126/science.1097396

Landsat Normalized Burn Ratio | U.S. Geological Survey. (n.d.). Retrieved November 29, 2023, from https://www.usgs.gov/landsat-missions/landsat-normalized-burn-ratio

Lehmann, J., Bossio, D. A., Kögel-Knabner, I., & Rillig, M. C. (2020). The concept and future prospects of soil health. *Nature Reviews Earth & Environment*, *1*(10), Article 10. https://doi.org/10.1038/s43017-020-0080-8

Levi, M. R., & Bestelmeyer, B. T. (2016). Biophysical influences on the spatial distribution of fire in the desert grassland region of the southwestern USA. *Landscape Ecology*, *31*(9), 2079–2095. https://doi.org/10.1007/s10980-016-0383-9

Levi, M. R., & Bestelmeyer, B. T. (2018). Digital soil mapping for fire prediction and management in rangelands. *Fire Ecology*, *14*(2), 11. https://doi.org/10.1186/s42408-018-0018-4

Liang, S., Hurteau, M. D., & Westerling, A. L. (2017). Potential decline in carbon carrying capacity under projected climate-wildfire interactions in the Sierra Nevada. *Scientific Reports*, *7*(1), Article 1. https://doi.org/10.1038/s41598-017-02686-0

Luyssaert, S., Schulze, E.-D., Börner, A., Knohl, A., Hessenmöller, D., Law, B. E., Ciais, P., & Grace, J. (2008). Old-growth forests as global carbon sinks. *Nature*, *455*(7210), Article 7210. https://doi.org/10.1038/nature07276

Manley, P. N., Zielinski, W. J., Stuart, C. M., Keane, J. J., Lind, A. J., Brown, C., Plymale, B. L., & Napper, C. O. (2000). Monitoring Ecosystems in the Sierra Nevada: The Conceptual Model Foundation. *Environmental Monitoring and Assessment*, *64*(1), 139–152. https://doi.org/10.1023/A:1006419624637

McBratney, A. B., Mendonça Santos, M. L., & Minasny, B. (2003). On digital soil mapping. *Geoderma*, *117*(1), 3–52. https://doi.org/10.1016/S0016-7061(03)00223-4

McKee, W. H. (1982). *Changes in Soil Fertility Following Prescribed Burning on Coastal Plain Pine Sites*. U.S. Department of Agriculture, Southeastern Forest Experiment Station, Forest Service.

Merino, A., Jiménez, E., Fernández, C., Fontúrbel, M. T., Campo, J., & Vega, J. A. (2019). Soil organic matter and phosphorus dynamics after low intensity prescribed burning in forests and shrubland. *Journal of Environmental Management*, *234*, 214–225. https://doi.org/10.1016/j.jenvman.2018.12.055

Miller, J. D., Safford, H. D., Crimmins, M., & Thode, A. E. (2009). Quantitative Evidence for Increasing Forest Fire Severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. *Ecosystems*, *12*(1), 16–32. https://doi.org/10.1007/s10021-008-9201-9

Miller, W. W., Johnson, D. W., Gergans, N., Carroll-Moore, E. M., Walker, R. F., Cody, T. L., & Wone, B. (2013). Update on the Effects of a Sierran Wildfire on Surface Runoff Water Quality. *Journal of Environmental Quality*, *42*(4), 1185–1195. https://doi.org/10.2134/jeq2012.0472

Minasny, B., McBratney, A. B., Malone, B. P., & Wheeler, I. (2013). Chapter One—Digital Mapping of Soil Carbon. In D. L. Sparks (Ed.), *Advances in Agronomy* (Vol. 118, pp. 1–47). Academic Press. https://doi.org/10.1016/B978-0-12-405942-9.00001-3

Murphy, J. D., Johnson, D. W., Miller, W. W., Walker, R. F., & Blank, R. R. (2006). PRESCRIBED FIRE EFFECTS ON FOREST FLOOR AND SOIL NUTRIENTS IN A SIERRA NEVADA FOREST. *Soil Science*, *171*(3), 181–199. https://doi.org/10.1097/01.ss.0000193886.35336.d8

Neary, D. G., Ryan, K. C., & DeBano, L. F. (2005). *Wildland fire in ecosystems: Effects of fire on soils and water*. RMRS-GTR-42-V4. https://doi.org/10.2737/RMRS-GTR-42-V4

Neill, C., Patterson, W. A., & Crary, D. W. (2007). Responses of soil carbon, nitrogen and cations to the frequency and seasonality of prescribed burning in a Cape Cod oak-pine forest. *Forest Ecology and Management*, *250*(3), 234–243. https://doi.org/10.1016/j.foreco.2007.05.023

Ohdo, T., & Takahashi, K. (2020). Plant species richness and community assembly along gradients of elevation and soil nitrogen availability. *AoB PLANTS*, *12*(3), plaa014. https://doi.org/10.1093/aobpla/plaa014

Olsen, S. R. (1954). *Estimation of Available Phosphorus in Soils by Extraction with Sodium Bicarbonate*. U.S. Department of Agriculture.

Parson, A., Robichaud, P. R., Lewis, S. A., Napper, C., & Clark, J. T. (2010). Field guide for mapping post-fire soil burn severity. *Gen. Tech. Rep. RMRS-GTR-243. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 49 p.*, *243*. https://doi.org/10.2737/RMRS-GTR-243

Pereira, P., Úbeda, X., Martin, D., Mataix-Solera, J., Cerdà, A., & Burguet, M. (2014). Wildfire effects on extractable elements in ash from a Pinus pinaster forest in Portugal. *Hydrological Processes*, *28*(11), 3681–3690. https://doi.org/10.1002/hyp.9907

Reese, R. E., & Moorhead, K. K. (1996). Spatial Characteristics of Soil Properties along an Elevational Gradient in a Carolina Bay Wetland. *Soil Science Society of America Journal*, *60*(4), 1273–1277. https://doi.org/10.2136/sssaj1996.03615995006000040045x

Santos, F., Russell, D., & Berhe, A. A. (2016). Thermal alteration of water extractable organic matter in climosequence soils from the Sierra Nevada, California. *Journal of Geophysical Research: Biogeosciences*, *121*(11), 2877–2885. https://doi.org/10.1002/2016JG003597

SEGALL, P., & POLLARD, D. D. (1983). Joint formation in granitic rock of the Sierra Nevada. *GSA Bulletin*, *94*(5), 563–575. https://doi.org/10.1130/0016-7606(1983)94<563:JFIGRO>2.0.CO;2

Shi, Y., Zang, S., Matsunaga, T., & Yamaguchi, Y. (2020). A multi-year and high-resolution inventory of biomass burning emissions in tropical continents from 2001–2017 based on satellite observations. *Journal of Cleaner Production*, *270*, 122511. https://doi.org/10.1016/j.jclepro.2020.122511

Smith, J. L., Halvorson, J. J., & Bolton, H. (2002). Soil properties and microbial activity across a 500m elevation gradient in a semi-arid environment. *Soil Biology and Biochemistry*, *34*(11), 1749–1757. https://doi.org/10.1016/S0038-0717(02)00162-1

Soil Data Explorer | California Soil Resource Lab. (n.d.). Retrieved December 1, 2023, from https://casoilresource.lawr.ucdavis.edu/sde/?series=SHAVER#osd

St. John, T. V., & Rundel, P. W. (1976). The role of fire as a mineralizing agent in a Sierran coniferous forest. *Oecologia*, *25*(1), 35–45. https://doi.org/10.1007/BF00345032

Stavros, E. N., Tane, Z., Kane, V. R., Veraverbeke, S., McGaughey, R. J., Lutz, J. A., Ramirez, C., & Schimel, D. (2016). Unprecedented remote sensing data over King and Rim megafires in the Sierra Nevada Mountains of California. *Ecology*, *97*(11), 3244–3244. https://doi.org/10.1002/ecy.1577

Stefanidis, S., Alexandridis, V., Spalevic, V., & Mincato, R. (2022). Wildfire Effects on Soil Erosion Dynamics: The Case of 2021 Megafires in Greece. *The Journal Agriculture and Forestry*, *68*, 49–63. https://doi.org/10.17707/AgricultForest.68.2.04

The terra package—R Spatial. (n.d.). Retrieved December 19, 2023, from https://rspatial.org/pkg/1-introduction.html

Tolorza, V., Poblete-Caballero, D., Banda, D., Little, C., Leal, C., & Galleguillos, M. (2022). An operational method for mapping the composition of post-fire litter. *Remote Sensing Letters*, *13*(5), 511–521. https://doi.org/10.1080/2150704X.2022.2040752

Turner, M. G., Smithwick, E. A. H., Metzger, K. L., Tinker, D. B., & Romme, W. H. (2007). Inorganic nitrogen availability after severe stand-replacing fire in the Greater Yellowstone ecosystem. *Proceedings of the National Academy of Sciences*, *104*(12), 4782–4789. https://doi.org/10.1073/pnas.0700180104

Unger, M., Leuschner, C., & Homeier, J. (2010). Variability of indices of macronutrient availability in soils at different spatial scales along an elevation transect in tropical moist forests (NE Ecuador). *Plant and Soil*, *336*(1), 443–458. https://doi.org/10.1007/s11104-010-0494-z

Wang, H., Wu, J., Li, G., Yan, L., & Wei, X. (2022). Effects of rainfall frequency on soil labile carbon fractions in a wet meadow on the Qinghai-Tibet Plateau. *Journal of Soils and Sediments*, *22*(5), 1489–1499. https://doi.org/10.1007/s11368-022-03170-7

Weather averages Huntington Lake, California. (n.d.). Retrieved April 18, 2023, from https://www.usclimatedata.com/climate/huntington-lake/california/united-states/usca0501

Weil, R. R., Islam, K. R., Stine, M. A., Gruver, J. B., & Samson-Liebig, S. E. (2003). Estimating active carbon for soil quality assessment: A simplified method for laboratory and field use. *American Journal of Alternative Agriculture*, *18*(1), 3–17. https://doi.org/10.1079/AJAA200228

Wells, C. G. (1979). *Effects of Fire on Soil: A State-of-knowledge Review*. Department of Agriculture, Forest Service.

Xue, L., Li, Q., & Chen, H. (2014). Effects of a Wildfire on Selected Physical, Chemical and Biochemical Soil Properties in a Pinus massoniana Forest in South China. *Forests*, *5*(12), Article 12. https://doi.org/10.3390/f5122947

Young, D. J. N., Stevens, J. T., Earles, J. M., Moore, J., Ellis, A., Jirka, A. L., & Latimer, A. M. (2017). Long-term climate and competition explain forest mortality patterns under extreme drought. *Ecology Letters*, *20*(1), 78–86. https://doi.org/10.1111/ele.12711