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Influence of Prescribed Burning on Coast Redwood (Sequoia Sempervirens) Forest Wildfire Resistance, Resilience, and Risk of Repeated Fire

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INFLUENCE OF PRESCRIBED BURNING ON COAST REDWOOD (*SEQUOIA SEMPERVIRENS*) FOREST WILDFIRE RESISTANCE, RESILIENCE, AND RISK OF REPEATED FIRE

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ABSTRACT

INFLUENCE OF PRESCRIBED BURNING ON COAST REDWOOD (*SEQUOIA SEMPERVIRENS*) FOREST WILDFIRE RESISTANCE, RESILIENCE, AND RISK OF REPEATED FIRE

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Land managers need tools to mitigate wildfire proliferation in the western United States. Prescribed burning may be an effective tool to reduce wildfire severity and promote healthy landscapes. In 2020, a large-scale wildfire provided a rare opportunity to compare early post-fire data between areas with and without a history of prescribed burning. Field data were collected approximately one year after a wildfire in fifty 20-meter plots from sites treated with prescribed fire in either 1999, 2007, or 2011 and fifty from sites without a history of prescribed fire. The influence of prescribed burning on forest resistance, resilience, and risk of repeated wildfire were assessed using generalized linear mixed effects models (GLMMs), in the coast redwood (*Sequoia sempervirens)* forests' southern range, where tanoak (*Notholithocarpus densiflorus*), and Douglas-fir (*Pseudotsuga menziesii*) were the co-dominant tree species. Areas with a history of prescribed burning were found to have greater wildfire resistance, shown by higher percent canopy retention and percent of trees that were living. Increased wildfire resilience was indicated by higher counts of early post-fire coast redwood seedlings. In addition, reduced repeated wildfire risk was demonstrated by fewer total tanoaks, standing dead trees, and a lower stand density compared to sites without prescribed fire. Results indicate that prescribed fire may improve redwood forest stand resistance and resilience, and that these benefits may maintain after a wildfire event.

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LIST OF ABBREVIATIONS

BBRSP – Big Basin Redwoods State Park

CWF – coarse woody fuel (1000+ hr fuel particles, >7.6 cm diameter)

CZU – 2020 CZU Lightning Complex Fire

DBH – diameter at breast height (measured at 1.4 m above soil surface on the uphill side)

GPS – global positioning system

GLMM – generalized linear mixed effect model

FWF – fine woody fuel (1, 10, and 100 – hr fuel particles, <7.6 cm diameter)

JREB – Johansen Road (east) prescribed burn project (burned)

Mg ha−1 – metric tons per hectare

NERU – North escape road sample site (unburned)

OVSB – Ocean view summit prescribed burn project (burned)

SUNU – Sunset trail sample site (unburned)

S2SB – Skyline to the sea prescribed burn project (burned) S2SU: Skyline to the sea prescribed burn project (unburned)

TEK – Traditional ecological knowledge

WUI – wildland-urban interface

1. Introduction

1.1. Motivation and Scope

Proliferation of severe wildfire in the western United States carries major risks to landscape health and ecosystem balance. Climate change and post-colonial fire suppression disrupt historic fire regimes and increase wildfire likelihood and intensity (Greenlee and Langenheim, 1990; Westerling and Bryant, 2008). Prescribed fire is one of the primary tools used by land managers to respond to intensified wildfire frequency and severity. Presence of prescribed fire can reduce fuel load, stabilize fire return intervals to historic levels, and restore ecosystem balance in favor of species with succession or survival advantages in the presence of fire (North et al., 2012; Ryan et al., 2013).

Along with benefits, there are costs associated with prescribed fire. Prescribed fire can displace or kill wildlife (K. R. Russell et al., 1999; R. E. Russell et al., 2009), impact localized air quality, release sequestered CO2, and carry economic costs associated with monitoring, and equipment (Ravi et al., 2019; Florec et al., 2020). While there have been studies that model and predict the effectiveness of prescribed fire in mitigating wildfire (Fernandes and Botelho, 2003; Furlaud et al., 2018), little research exists analyzing the effects of prescribed fire on coast redwood (*Sequoia sempervirens*; hereafter “redwood”) resistance and regeneration after a widespread wildfire. This project investigated the efficacy of prescribed fire to protect redwood forests from large-scale wildfires. Prescribed fire effectiveness was assessed in terms of forest resistance, resilience, and the risk of repeated wildfire.

The primary goals of prescribed fire can be broken into two general categories: (1) to protect sensitive species by re-establishing historic fire regimes that restore ecosystem succession patterns, and (2) to reduce the fuel load to dampen the severity of wildfires and protect developed areas. These goals are inextricably linked, especially with continued expansion into fire prone wildland-urban interface (WUI) areas, defined as areas where houses are in or near wildland vegetation (Radeloff et al., 2018).

Big Basin Redwoods State Park (BBRSP), in the Santa Cruz Mountains of California, is home to the largest stand of old-growth coast redwoods south of San Francisco. This park has utilized prescribed fire as a management tool dating back to 1978. Burns have varied in size and frequency but have mostly been low intensity understory burns. Objectives of managed fire reintroduction have been to protect park resources, maintain natural vegetation balance, preserve old-growth redwoods, and protect nearby private land (California Department of Parks and Recreation, 2003). Cowman and Russell (2021) provided empirical evidence of decreased fuel load and a stand composition which favored redwoods compared to associated species in areas with prescribed fire treatment history within BBRSP. These results indicate that sites treated with prescribed fire are expected to be more resistant to wildfire. Reassessment of these sites after a wildfire helps to verify whether prescribed fire application is achieving the agency's goal of protecting redwood ecosystem health from large-scale wildfire events.

The 2020 CZU Lightning Complex Fire (CZU) started in August 2020, burned 86,509 acres of redwood dominant forest, and destroyed 1,490 structures over 37 days (California Department of Forestry and Fire Protection, 2020). The scale and devastation of this fire was uncharacteristic for the region. However, studying the CZU provides an important opportunity to prepare for mitigation and response to the predicted escalation of fire severity in the state (Goss et al., 2020; McEvoy et al., 2020).

Collecting data following the CZU fires made it possible to test indicators of redwood forest health as well as risk of repeated wildfire between sites with and without a history of prescribed fire treatment. This project contributes important information to increase understanding of the influence of prescribed fire in redwood forest systems, and to help develop best practices for prescribed fire use to ensure the ecological and societal benefits of management actions justify the costs.

1.2. Literature Review

1.2.1. Wildfire proliferation

Climate change and suppressive fire management strategies have altered fire regimes across the globe. Fire seasons have been increasing in length and frequency over the past 40 years (Jolly et al., 2015). Within the United States, wildfires are predicted to increase with longer fire seasons, elevated temperatures, and prolonged droughts (Y. Liu et al., 2013). The incidence of large-scale wildfire events has been increasing in the western US since the 1980's (Dennison et al., 2014). The increases in wildfire frequency, duration, and season periods have resulted in part from higher temperatures and earlier season snowmelt (Westerling et al., 2006). The human and environmental risks from extreme wildfires are anticipated to increase in wildfire-prone areas such as the western United States, where a 20-50% rise in days conducive to extreme wildfire events is predicted (Bowman et al., 2017).

Wildfires pose an elevated risk in California where the annual burned area has increased fivefold between 1972-2018. This stark increase is due to higher temperatures, an atmospheric vapor pressure deficit, fall wind events, and later onsets of winter precipitation (Williams et al., 2019). California's recent history of prolonged periodic drought and continuing trends of dry conditions heightens the likelihood of extreme wildfire events (McEvoy et al., 2020). The State has experienced an intensification of wildfires in fall months resulting from offshore winds and dry vegetation, exacerbated by an uncharacteristic late onset of winter precipitation. Over the four decades prior to 2020, California experienced an approximate 1° C increase in fall temperatures and a 30% reduction in fall precipitation, which has led to fall fire frequency more than doubling since the early 1980's with increasing frequency predicted for the region (Goss et al., 2020).

In addition to climatic changes, wildfire risk in California has also been influenced by a history of fire suppression. Fire suppression has altered fire regimes in systems where human-ignited fires have been an integral part of the landscape for thousands of years (Greenlee and Langenheim, 1990; Keeley, 2002). Fire suppression has led to increased accumulation of fuel across many fire-prone landscapes, which can heighten the risk of high fire severity in those contexts (Syphard et al., 2007; Bowman et al., 2011; Steel et al., 2015). In the absence of regular burning that historically helped clear fuels off lands, accumulated surface fuels now increase the likelihood of severe wildfires in California (Keeley and Syphard, 2019).

Fire suppression is linked to an upsurge of development into fire-prone areas that are part of the WUI (Radeloff et al., 2018). Reducing structure loss has been a contributing factor motivating wildfire suppression (Mell et al., 2010). Across the western United States, the WUI has grown in terms of land area by 33%, with the number of houses increasing by 41% from 1990 to 2010 (Radeloff et al., 2018). Within the United States, California has the highest concentration of WUI housing, with over 5 million housing units (Radeloff et al., 2005). This puts Californians at a difficult crossroads as a balance needs to be found between protecting homes and human lives while also protecting ecosystems and understanding the effects of fire suppression and development on fire regimes and biodiversity conservation (Gill and Stephens, 2009).

1.2.2. *Wildfire associated risks*

The increase in wildfire frequency and severity carries risks to many of the humans and ecosystems of California. Wildfire effects will likely be widespread throughout the State; however, certain environments and demographics may be more acutely impacted due to their exposure or sensitivity. For example, air quality is a primary concern. Satellite modeling between 1997-2016 shows trends in air quality degradation associated with fire events in California, increasing risks of related health impacts including respiratory and cardiovascular diseases, miscarriages and pregnancy complications, and asthma across the most vulnerable populations (Tao et al., 2020). Smoke from wildfire increases the risk of respiratory and cardiovascular diseases. These effects disproportionately place children, the elderly, and those with chronic diseases at risk (J. C. Liu et al., 2015). Doubleday et al. (2020) even linked days with heavy wildfire smoke to increased risk of non-traumatic mortality in Washington State. Mitigation of health impacts from wildfire smoke is important as huge regions can be aversely affected and major costs to well-being and health care costs are incurred.

Wildfire events can also negatively impact water quality, which can have human health implications as well as effects on natural ecosystems. Hydrologic impacts resulting from wildfire include runoff of sediment and nutrients, particularly during storm events (Caldwell et al., 2020), which can degrade downstream habitats and access to clean water. Sankey et al. (2017) found that wildfires increased soil erosion and sedimentation in the United States and that more than one third of watersheds within the western United States will experience an increase of greater than 100% post-fire sedimentation in the next 30 years. California's environments are at an elevated risk as wildfire seasons become longer and are increasingly followed by heavy winter storm events that can trigger post-fire debris flows, flooding, and fatal landslides (Murphy et al., 2015; Williams et al., 2019).

1.2.3. Wildfire management tools

1.2.3.1. Prescribed fire

Broadscale reasons for implementation of prescribed fire include fuel reduction, disposal of logging debris, improving wildlife habitat, stand rehabilitation, managing competing vegetation, controlling insects and disease, improving forage for grazing, and promoting fire-dependent species (Wade, 1990; Weber and Taylor, 1992). Within forest systems, the goals for applying fire as a management tool are more specified. One primary goal is to protect sensitive flora and fauna by re-establishing historic fire regimes that restore ecosystem succession patterns; another is to reduce fuel load to dampen the severity of wildfires. Management actions, such as these, help prevent widespread destruction of natural landscapes and protect adjacent developed areas (Scherer et al., 2016; Knapp et al., 2017; North et al., 2012). While prescribed fire is not the only tool shown to be capable of reaching many of these forest health and management objectives, it is considered to be one of the more cost effective (Penman et al., 2020), and desirable because of its ability to mimic historic natural processes as far as nutrient cycling, and biomass distribution on landscapes (Ryan et al., 2013).

Prescribed fire has been found to promote healthy forest systems and can help replicate natural disturbance regimes (Scherer et al., 2016). Researchers have found greater tree spacing, age, and size heterogeneity in species composition following prescribed fire (Scherer et al., 2016; Knapp et al., 2017). Scherer et al. (2016) found that prescribed fire treatments lead to lower overstory densities, reduced stand basal area, and greater diameters of trees as compared to sites not treated with prescribed fire. Within fire-adapted forest ecosystems, prescribed fire is being increasingly adopted as a necessary tool for mitigating the environmental consequences resulting from fire suppression.

The scale at which prescribed fire should be implemented is a question that has been debated by researchers and land managers. Current prescribed fire implementation goals and policies have largely been a reaction to years of fire suppression and are a triage to address fuel build-up rather than a proactive approach to utilize prescribed fire as a tool to achieve ecologically beneficial outcomes (North et al., 2012). While evidence exists to support more active fire management in the form of fire re-introduction due to the demonstrated benefits of prescribed fire in many ecosystems, implementation of prescribed fire may not be appropriate for all ecosystem types. Pastro et al. (2011) found that while prescribed fire can be an effective tool to create fuel breaks to protect habitats of fire-sensitive species from large-scale wildfire effects, prescribed fire has little utility for biodiversity conservation due to unpredictable taxon-specific response. Connell et al. (2019) found that fire management strategies dependent on high intensity prescribed fire are likely to negatively impact threatened species due to low species population size and restricted ranges. However, with consistent evaluation, adaptive management to experiment and tailor techniques to specific local contexts, and flexibility, fire introduction may benefit individual ecosystems (Connell et al., 2019).

1.2.3.2. Prescribed fire in redwoods

While humans have intentionally ignited fires in California for thousands of years, the past 150 years has been characterized by fire suppression in California, historic fire return interval lengths are not clearly understood. Previous research in the redwood forest has provided estimates with a potentially concerning amount of variability, with fire return intervals ranging from 6 to 135 years (Finney and Martin, 1989, 1992; Greenlee and Langenheim, 1990; P. M. Brown and Swetnam, 1994; P. M. Brown and Baxter, 2003; Stephens and Fry, 2005; Jones and Russell, 2015), which makes replicating historic fire regimes through fire re-introduction difficult because we are lacking a clear baseline for historic fire frequency. While ignitions by Indigenous peoples have not been clearly confirmed in redwood forests of the Santa Cruz Mountains, evidence of millennia-old anthropogenic ignitions have been established in neighboring grasslands and oak savannas. It is likely that human ignited fires from these adjacent landscapes spread to redwood systems (Stephens and Fry, 2005). Due to its resource abundance and temperate climate, the

Monterey Bay region was densely populated prior to colonial contact. However, redwood forests were not frequently inhabited due to a relative lack of resources and the presence of grizzly bears as compared to the coast, oak woodlands, and prairies (Greenlee and Langenheim, 1990). Greenlee and Langenheim (1990) found that prairies, coastal sage, and oak woodlands all had fire-return intervals of between just one and two years in the pre-colonial period of human habitation. It is this high frequency of fires ignited in adjacent lands to redwood forests that could explain higher redwood forest fire frequencies compared to time periods without established human settlement nearby (Greenlee, 1983; Greenlee and Langenheim, 1990; Stephens and Fry, 2005; Jones and Russell, 2015).

While fires may have historically spread to redwood forests from adjacent landscapes, the ability to mimic this trend by reintroducing fire to adjacent landscapes and allowing it to burn into redwood forests is challenged by human development that now fragments these habitats. This expanding WUI has broken continuous landscapes and makes prescribed fire application from adjacent habitats dangerous due to risk of escape (Black et al., 2020). While intentionally starting fires in adjacent habitats is no longer feasible in the Santa Cruz Mountains, carefully managed low intensity prescribed fire has been a component of fire management dating back to 1978 (California Department of Parks and Recreation, 2003). Low intensity prescribed fires can be used in the understory of redwood forests to reduce fuel load, increase relative dominance of redwoods, and reduce stand density. Prescribed fire has little effect on redwood overstory structure and therefore can be used to reduce densities of competitors and understory structure without high risk of mortality to redwoods (Ramage et al., 2010; Engber et al., 2017). Prescribed fire can be a useful tool for land managers to address imbalances in redwood forests resulting from fire suppression and to shift the competitive dynamics back in favor of redwoods (Ramage et al., 2010).

1.2.3.3. Mechanical thinning

While prescribed fire is an important tool to reduce fuel load and re-introduce fire to systems where fire has been suppressed, there are also alternative land management tools that managers can consider when looking to restore and/or reduce wildfire risk on the landscapes they steward. Mechanical forest thinning is the process of selective removal of trees and fuels that can involve harvesting and removing materials from a site, lop and scatter, mastication, chipping, and other forms of leaving all or part of the "removed" materials on site. Thinning is sometimes promoted as a lower risk surrogate for fire (due to the fear of prescribed fires escaping), or as a tool in combination with prescribed fire due to fuel loads on landscapes fostering undesired fire severity outcomes if prescribed fire is applied without changing the fuel composition or structure (Pollet and Omi, 2002; Harrod et al., 2009; Knapp et al., 2017).

A wide body of literature exists comparing the effects of mechanical thinning and prescribed fire on forest systems. Harrod et al. (2009) compared thinning and prescribed fire treatments, alone and in combination, in a conifer forest and found that thinning had a greater effect on reducing canopy density, while prescribed fire did not affect canopy fuel loads. Thinning was more effective at altering stand structure; however, burns were conducted in spring which likely dampened fire intensity and any desirable tree mortality that could have assisted in modifying the stand structure. In a similar study on mixed-conifer forest, Knapp et al. (2017) found that prescribed fire was less effective in producing stand thinning management outcomes than mechanical thinning alone. However, thinning paired with prescribed fire most effectively created conditions similar to historic forest structures. Agee and Lolley (2006) also found that prescribed fire had a lower effectiveness at reducing canopy density when compared to thinning in a conifer forest. Within conifer forests, thinning appears more effective in reducing canopy fuel loads than prescribed fire, which can help to mitigate crown fires (i.e., fires that move through tree canopies and are associated with high tree mortality). However, thinning is shown to be most effective across several ecosystem types when used in conjunction with prescribed fire rather than in its place (Agee and Lolley, 2006, Ritchie et al., 2007; Harrod et al., 2009; Knapp et al., 2017).

The choice of fire risk management practice greatly depends on the individual landscape. Prescribed fire tends to better replicate natural patterns of pre-suppression fire events. Yet negative immediate effects to human populations, such as air quality degradation and potential for escape, means that thinning may be a more socially acceptable and appropriate wildfire risk reduction tool close to developed areas, whereas prescribed fire may be better suited to remote and wildland areas (Bright et al., 2007).

1.2.3.4. Redwood forest ecology

Natural coast redwood (*S. sempervirens*) forests are found exclusively on the west coast of the United States ranging from California's central coast up to southern Oregon (Lorimer et al., 2009). These long-lived conifers are the tallest tree species in the world and are typically either the dominant or co-dominant tree species within their range (Griffith, 1992). Redwoods can be found at elevations between sea level and about 900 meters. Redwoods occur in a maritime Mediterranean climate characterized by cool winters and dry summers with heavy fog, which allows redwoods to persist through drier months. While reliant on coastal fog, redwoods are not tolerant of ocean winds or salinity and occur in areas with at least some buffer from the ocean. The preferred soil types for redwoods are those formed from sandstone, limestone, slate, chert, and schist (University of California Agriculture and Natural Resources, 2016).

While there is not a consensus on a definition of “old-growth” (Wirth et al., 2009), a common definition is based on whether a stand has any history of being harvested (Sawyer et al., 2000). Only about 200,000 acres of the approximately 2.2 million acres of redwood habitat is composed of old-growth stands. Most of the remaining old-growth habitat exists in the northern range of the species (University of California Agriculture and Natural Resources, 2016). BBRSP is home to the largest continuous stand of old-growth redwoods in the southern range of the species (California State Parks, 2021).

1.2.3.5. Associated species

Common herbaceous understory species associated with southern range old-growth redwood forests include Hooker’s fairy bells (*Prosartes hookeri* Torr.), Pacific starflower (*Lysimachia latifolia* [Hook.]), Pacific trillium (*Trillium ovatum* Pursh), redwood violet (*Viola sempervirens* Greene), redwood sorrel (*Oxalis oregana* Nutt.), wild ginger (*Asarum caudatum* Lindl.), false solomon seal (*Maianthemum racemosum* [L.] Link), fetid adder’s tongue (*Scoliopus bigelovii* Torr.), western sword fern (*Polystichum munitum* [Kaulf.] C. Presl) and huckleberry (*Vaccinium ovatum* Pursh). These species can be used as indicators of forest health regeneration after disturbance events and act as early indicators of redwood succession following fire events (Sinclair, 2013; W. Russell et al., 2014; Hanover and Russell, 2018; W. Russell, 2020).

Overstory and subcanopy tree species that occur in redwood forests in the Santa Cruz Mountains include, Pacific madrone (*Arbutus menziesii* Pursh), tanoak (*Notholithocarpus densiflorus* [Hook. & Arn.]), oak species (*Quercus spp.*), Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco), and California bay (*Umbellularia californica* [Hook. & Arn.] Nutt.) (Martin, 1998; Lazzeri-Aerts and Russell, 2014). In a study comparing redwood, Pacific madrone, oak species, tanoak, and Douglas-fir between three sites that experienced wildfire in the Santa Cruz Mountains, Lazzeri-Aerts and Russell (2014) found that in the presence of wildfire, Douglas-fir exhibited the highest rate of mortality, followed by tanoak, oak, and Pacific madrone, with redwoods having the lowest mortality rate.

1.2.3.6. Redwood fire ecology

Redwoods are not reliant on fire events for reproduction, but they are adapted to be resilient through fire events and can regenerate through gap phase succession that can result from disturbances (Busing and Fujimori, 2002). Redwoods are capable of reproducing both from seed and through basal and epicormic sprouting. Vigorous sprouting after disturbances makes redwood remarkably resilient species to fire, which allows the species to outcompete other species and take advantage of disturbance events (Douhovnikoff et al., 2004; Ramage et al., 2010; Lazzeri-Aerts and Russell, 2014; O'Hara et al., 2017). For example, Lazzeri-Aerts and Russell (2014) found that redwoods demonstrated higher levels of seed dispersal, sprouting rates, canopy retention, and tree survivorship following wildfires as compared to co-occurring tree species.

Regardless of fire severity, redwoods demonstrate a relative resilience and advantage compared to co-occurring species (Ramage et al., 2010). While fire suppression has allowed some fire-sensitive species to outcompete and displace redwoods, vegetative sprouting and seed establishment from disturbances other than fire, such as falling logs and slope failure, allow redwoods to regenerate in the absence of fire events (Lorimer et al., 2009).

1.2.3.7. Redwood forest fire regimes

The history of fires in southern range of redwood forests remains relatively ambiguous despite numerous studies investigating fire return intervals and history of fire presence within redwood forests. Greenlee (1983) determined from modeling anthropological and historical records that prior to human presence (starting in 11,000 BP in the northeastern Santa Cruz Mountains), lightning ignited fires occurred approximately every 100 to 150 years. Through examination of historic records, fire scar dating, and modeling of fire behavior, fire frequency was found to be higher after human habitation of the area, ranging between a mean fire return of 17 to 82 years prior to Spanish colonization, 82 years during Spanish settlement of the region, 20 to 50 years during early Anglo settlement, and 130 years since fire suppression became prevalent in the 20^{th} century (Greenlee and Langenheim, 1990). Through analysis of fire scars, Jones and Russell (2015) found that fire-return intervals were longer than Greenlee and Langenheim (1990) found in the same region. However, they also found a high variation between samples which added ambiguity to the findings and suggests that managing fire based on measurements of historic fire-return intervals may not be appropriate in southern range redwood forests (Jones and Russell, 2015). Inconsistent data across the region indicates cultural burning practices have experienced major spatial and temporal fluctuations (Jones, 2014).

Estimations of fire frequency vary greatly in southern range redwood forests. However, there is consensus that the relative high frequency of fires prior to implementation of fire suppression in a region with such low sources of natural ignition suggests a history of anthropogenically ignited fires (Greenlee, 1983; Greenlee and Langenheim, 1990; Keeley, 2002; Stephens and Fry, 2005; Jones, 2014). It is also clear that fire intervals have increased in the period since fire suppression became prevalent (Greenlee and Langenheim, 1990; Jones, 2014).

1.2.3.8. Indigenous fire use in the Santa Cruz mountains

The central and northern coasts of California are characterized by heavy fog and low occurrences of lightning events (Stephens and Fry, 2005; Lorimer et al., 2009). Frequency of lightning ignited fire for this region is one of the lowest in North America. Not only is natural fire ignition low, but the region also had one of the highest densities of pre-colonial populations in North America, which suggests anthropogenically ignited fires were likely (Keeley, 2002). Because of the lack of lightning induced fires, Native Americans would purposefully start fires to improve food gathering efficiency, exterminate insects that ate acorns, attract game to rich grasslands, encourage production of materials to construct high quality cordage, and to clear ways for travel (Stephens and Fry, 2005).

The effects of fire suppression and removal of intentional human ignited fires are being experienced in the form of proliferating wildfires. As a result, land managers are beginning to work alongside tribal communities to re-establish Indigenous fire knowledge and application (Eriksen and Hankins, 2014). In addition to the benefits of prescribed fire to promote target landscapes and habitats as well as to reduce fuel loads to mitigate wildfire severity, Indigenous fire stewardship and cultural burning can increase desired harvest vegetation to sustain knowledge and traditional ceremonial practices, livelihoods, and economies for fire-dependent cultures (Lake and Christianson, 2019).

Indigenous knowledge and practices are often passed down experientially. In much of California, and specifically the area that encompasses the southern range of redwoods, Indigenous communities were stripped from their ancestral lands and removed as active stewards (Rizzo-Martinez, 2022). This presents a challenge to communities who now are attempting to revitalize cultural burning practices (Nikolakis and Roberts, 2020). Land management agencies are now beginning to work alongside contemporary Indigenous groups to combine Western and Indigenous knowledge and resources to reconstruct not only historic landscapes, but historic knowledge as well. Traditional Ecological Knowledge (TEK) can be integrated into environmental management strategies implemented by agencies to gain a better place-based understanding of ecosystems and their needs and to create common ground between researchers and Indigenous communities through a shared commitment to land restoration and stewardship (Hill et al., 2012; Johnson and Larsen, 2013).

The Amah Mutsun Tribal Band are a contemporary Indigenous community with an active connection to the Santa Cruz Mountains and adjacent coastal, woodland and savannah areas. Their ancestors have engaged in fire management practices for thousands of years (Cuthrell et al., 2012; Lightfoot and Lopez, 2013). Amah Mutsun traditions of how to use fire were lost when their people were forcefully moved to Spanish missions. Following the mission period, ancestors of the Amah Mutsun were forced into slave labor and could not practice cultural traditions or pass on traditional knowledge (Lopez, 2018). During the Euro-American period, TEK was further severed through an active effort of genocide and extermination of California Indians (Cuthrell et al., 2012; Lightfoot and Lopez, 2013; Lopez, 2018; Amah Mutsun Tribal Band, 2021).

The Amah Mutsun retain their connection to ancestral lands and the management of those lands. However, they are faced with the challenge of lost knowledge of traditional land management practices and the fact that they do not have ownership of their ancestral homelands (Lopez, 2018). The Amah Mutsun have partnered with California State Parks, National Parks, the Bureau of Land Management, landowners, and conservation organizations to gain access to their lands. Additionally, they are working with universities to help restore lost knowledge, restore landscapes, and bring back traditional native plants. The tribal community has been able to reconstruct knowledge of which plants should be restored through archaeological records and collaborations with researchers. This has allowed the Amah Mutsun Tribal Band to restore their culture and regain access to plants and resources important to their traditional practices (Lightfoot and Lopez, 2013; Lopez, 2018; Amah Mutsun Tribal Band, 2021). Collaborations have taken the form of burn practices combining the knowledge of researchers, ecologists, archaeologists, and tribal members to restore traditional habitats and traditions attached to those lands (Lightfoot and Lopez, 2013; Lopez, 2018). The Amah Mutsun, through the Native Stewardship Corps, continue to engage tribal members in Indigenous stewardship through cultural burn projects.

1.2.3.9. Fire literacy, and communications

Successful implementation of fuel management activities across socially and ecologically stratified landscapes often necessitates (1) collaboration and/or cooperation across landownership types or across stake holder groups, and/or (2) garnering public support for agency-led land management activities. A key element to encouraging public engagement with and participation in landscape and natural resource management is through wildfire outreach (Paveglio et al., 2021). Creating fire resilient landscapes often requires an aggregate of actions across diverse communities and stakeholders with different experiences with and understandings of wildfire and wildfire risk (Paveglio et al., 2019). Through a survey of residents of the WUI, Bright et al. (2007) found that support or opposition to fire management strategies was most greatly influenced by the current condition of a forest. In a study of communities throughout the Lake Tahoe Basin in the Californian Sierras. Ascher et al. (2013) found that residents' positive perceptions of the benefits and effectiveness of prescribed burning positively influenced levels of support for prescribed burning in the region. This suggests that focusing public relations and communication efforts on the forest health benefits and reduced future fire risks associated with prescribed fire can be an effective way to build community support for the practice (Ascher et al., 2013).

Fire science literacy can take the form of education and research or be applied through the work of policymakers and land managers. Creating wildfire literacy can be enhanced through experiments that test environmental factors and report data and results on fuels, fire behavior and forest health for education, to further academic research, and to influence application by policymakers and land managers (McGranahan and Wonkka, 2018). In this way, those engaged in promoting fire literacy rely on the findings of academic research to build community support for landscape health wildfire preparedness policies and practices.

1.3. Problem statement

Fire suppression and climate change exacerbate wildfire risks in the western United States, including California (Westerling and Bryant, 2008; Williams et al., 2019). Wildfire threatens the safety and health of nearly all California residents and puts many natural ecosystems at risk (McEvoy et al., 2020). Redwood forests are a system where fire plays an important role in succession. While redwoods show a high fire resilience (Ramage et al., 2010), extreme wildfire may adversely impact aspects of redwood forest health. The CZU fire is a recent example of a large-scale wildfire event. The CZU burned 86,509 acres over 37 days in the Santa Cruz Mountains (California Department of Forestry and Fire Protection, 2020). The area impacted by this fire contains the largest continuous stand of old-growth redwoods south of San Francisco (California State Parks, 2021). This makes this a prime sample area to understand the effects of large-scale wildfires on old-growth redwood forests.

Active land management strategies are needed to reduce wildfire risks. Prescribed fire can be an effective method to reduce fuel load and restore historic ecosystem fire intervals (Stephens et al., 2009; Stephens et al., 2012). However, negative outcomes of prescribed fire may result, including reduced air quality, wildlife and human displacement, or mortality of wildlife, as well as potential for fire to escape and burn more acres or property than intended (R. E. Russell et al., 2009; Ravi et al., 2019; Florec et al., 2020). Land managers need information on the effectiveness of prescribed fire projects to determine best practices and to make informed decisions about when prescribed fire treatment is appropriate to protect property and natural landscapes.

Anthropogenic fire has influenced redwood systems in the Santa Cruz Mountains for millennia. Heavy fog and low lightning frequency in the area indicate that fires may have been ignited by humans historically (Greenlee and Langenheim, 1990; P. M. Brown et al., 1999; Keeley, 2002; Stephens and Fry, 2005; Lorimer et al., 2009; Jones, 2014). Redwoods are comparatively more resilient to fire than associated species (Ramage et al., 2010; Lazzeri-Aerts and Russell, 2014). This poses a challenge for land managers responsible for protecting redwoods as other species may encroach on and outcompete redwoods in systems without fire playing its historical ecological disturbance role. Under the right conditions, proper application of prescribed fire may restore balance to these systems by reducing density of other species encroaching into redwood habitat in the absence of fire.

While numerous studies have modeled and predicted prescribed fire effectiveness in reducing wildfire hazards (Fernandes and Botelho, 2003; Waring et al., 2016; Furlaud et al., 2018), little research exists directly measuring post-wildfire recovery comparing areas with and without a history of prescribed fire treatment. This project collected data on post-wildfire redwood forest vegetation recovery and fuel consumption in sites without a history of prescribed fire and sites that experienced prescribed fire in different years. All sample sites were burned in the CZU fires and have similar biodiversity compositions (Cowman and Russell, 2021). This consistency in biodiversity composition between sites permits data comparisons and allows us to gain a better understanding of the effectiveness of prescribed fire in reducing wildfire intensity and protecting redwood stands.

The Santa Cruz Mountains represent a highly populated WUI with residents living near agency managed natural landscapes. Additionally, the area serves a large population of visitors from the San Francisco Bay Area and across the globe for activities such as mountain biking, hiking, foraging, camping, etc. There is great interest among residents, conservationists, recreationists, and tourism-driven businesses to protect the area's remaining old-growth redwood forests. The results of this study will inform California State Parks and other land managers on the effectiveness of prescribed fire to protect redwood systems and highlight the circumstances under which prescribed fire may be useful for fostering more resilient redwood systems under a changing climate likely to have severe wildfire events. Findings from this study can also be integrated into public communications and education programs to promote and restore fire management literacy to the region. This study comes at an important time for an area that has been recently affected by a devastating wildfire event. The CZU catalyzed community interest in wildfire resilience and increased active management, but both managers and community members need ecosystem and context-specific information to guide their actions to better prepare for wildfires, mitigate risks, and prevent wide-scale destruction similar to what was experienced in the CZU fire.

1.4. Research objectives

The objective of this study was to better understand the influence of prescribed burning on forest resistance, resilience, and risk of repeated wildfire in redwood forests based on data collected after a wildfire in an area with temporally variable prescribed fire history. This study is intended to inform land managers on the effectiveness of prescribed fire implementation as a tool to mitigate wildfire severity and promote resistance and resilience of coast redwood systems to prepare for large-scale wildfire events.

1.5. Research questions

RQ1: Does prescribed fire increase post-wildfire canopy retention in a redwood forest?

RQ2: Does prescribed fire increase redwood forest regeneration, measured in terms of number of seedlings, sprouts, and percent groundcover?

RQ3: Does prescribed fire decrease repeated wildfire risk in a redwood forest, measured in terms of surface fuels, tree density, and number of standing dead trees?

2. Journal Article

2.1. Abstract

Land managers need tools to mitigate wildfire proliferation in the western United States, especially with regard to high severity fires. Prescribed burning may be an effective tool for reducing wildfire severity and promoting healthy landscapes. In 2020, a large-scale wildfire burned through an area with mixed land management application, providing a rare opportunity to compare early post-fire data between areas with and without a history of prescribed burning. Field data were collected approximately one year after a wildfire in fifty 20-meter plots from sites treated with prescribed fire in either 1999, 2007, or 2011 and fifty from sites without a history of prescribed fire. The influence of prescribed burning on forest resistance, resilience, and risk of repeated wildfire were assessed using generalized linear mixed effects models (GLMMs), in the coast redwood (*Sequoia sempervirens)* forests' southern range, where tanoak (*Notholithocarpus densiflorus*), and Douglas-fir (*Pseudotsuga menziesii*) were the co-dominant tree species. Areas with a history of prescribed burning were found to have greater wildfire resistance, shown by higher percent canopy cover (79% without prescribed fire and 85% with prescribed fire) and a higher percent of total trees that were living (22% without prescribed fire and 42% with prescribed fire). Increased wildfire resilience in areas with prescribed fire was indicated by higher counts of early post-fire coast redwood seedlings (1 per hectare without prescribed fire and 108 per hectare with prescribed fire). In addition, reduced repeated wildfire risk was demonstrated by fewer total tanoaks (193 per hectare without prescribed fire and 94 per hectare with prescribed fire), fewer total standing dead trees (278 per hectare without prescribed fire and 155 per hectare with

prescribed fire), and a lower total stand density (352 per hectare without prescribed fire and 255 per hectare with prescribed fire) compared to sites without a history of prescribed burning. Results indicate that prescribed fire may improve redwood forest stand resistance and resilience, and that these benefits may maintain after a wildfire event.

2.2. Introduction

Forests in the western United States have long been shaped by fire (Marlon et al., 2012). Yet, recent history has seen a dramatic increase in both the frequency and severity of wildfires due to over a century of fire-suppression, poor timber harvest practices, and climate change (Westerling, 2016; Steel et al., 2015). Fire suppression became the dominant wildfire management strategy following European settlement of the western United States. While this may have protected human life and property from wildfire risk in the short term, it led to live and dead fuel accumulation on landscapes historically cleared by fire (Hagmann et al., 2021). In addition, increased temperatures and prolonged drought brought on by climate change (Williams et al., 2019) paired with expansion of the Wildland-Urban Interface (WUI) complicates the wildfire problem (Radeloff et al., 2018).

Coast redwood (*Sequoia sempervirens*; hereafter “redwood”) forests face many of the same risks threatening all forest types across the western United States. With just five percent of old-growth redwood forests remaining (Thornburgh et al., 2000) and areas in and surrounding redwood forests under pressure from rapid residential development, these forests, though known for their resistance and resilience to wildfire, are far from immune from the wildfire problem. Redwood forests are only found within a narrow band starting in California's central coast and extending into southern Oregon (Lorimer et al., 2009).

Redwoods are found at elevations between sea level and 900 meters and reside in a Mediterranean climate characterized by cool winters and dry summers with heavy fog. The preferred soil types for redwoods are sandstone, limestone, slate, chert, and schist (University of California Agriculture and Natural Resources, 2016). Within the California central coast range of redwood forests, the most common codominant tree species are Douglas-fir (*Pseudostuga menziesii*; hereafter “Douglas-fir”) and tanoak (*Notholithocarpus densiflorus*; hereafter “tanoak”). While each species has unique adaptations to coexist with fire, redwoods have been found to have the lowest fire induced mortality rates among these codominant species (Ramage et al., 2010).

Redwood forests are host to a number of fire adaptations. As the tallest trees in the world, high canopies help redwoods resist most low to moderate intensity surface fires. Thick, tannin rich bark also helps redwoods to resist fire. Uncommon among conifers, redwoods can resprout both basally and epicormically after disturbance (O’Hara et al., 2017). Redwoods have also been shown to regenerate via seedlings after wildfire events, though it is a less common post-fire regeneration strategy compared to sprouting (Douhovnikoff et al., 2004). These combined adaptations make redwoods both resistant and resilient to fire. The most common associate tree species of redwood along California’s central coast is tanoak (Sawyer et al., 2000). When exposed to fire, tanoaks topkill easily, yet resprout vigorously (Donato et al., 2009). For this reason, tanoaks do not exhibit a high resistance to fire, but are very fire resilient. With impressive height, thick bark and a high crown, mature Douglas-fir trees exhibits some of the same fire resistant characteristics as redwoods, even if to a lesser extent (Hermann and Lavender 1990). However, Douglas-fir does not have the ability to resprout basally or epicormically after fire, and thus total loss of foliage ensures individual tree mortality. Since seeds are stored in the crown and regeneration can only occur through seedling dispersal, Douglas-fir exhibits some post-fire resilience under low and moderate surface fire conditions, but struggles to regenerate when faced with high intensity or crown fire (Hood et al., 2007).

Within redwood forests, humans have shaped landscapes with fire for thousands of years. Indigenous burning in and adjacent to central California redwood forests was intended to improve food gathering efficiency, exterminate insects that ate acorns, attract game to rich grasslands, encourage production of materials to construct high quality cordage, and to clear ways for travel (Stephens and Fry, 2005). European colonization and removal of Indigenous people from their lands disrupted millennia old practices of intentional burning. The removal of Indigenous burning from the region has resulted in longer fire return intervals and less frequent, yet more intense wildfires (Keeley, 2002).

The importance of anthropogenic fire in California's forests has become more widely acknowledged within the wildfire management field in the past several decades, prompting an increased application of prescribed fire as a fuel management tool (Ryan et al., 2013). Low intensity prescribed fires can be used in redwood forests to reduce fuel load, increase relative dominance of redwoods, and reduce stand density (Teraoka and Keyes, 2011). Prescribed fire has little effect on redwood overstory structure, and therefore can be used to reduce densities of competing species without high risk of mortality to redwoods (Ramage et al., 2010).

This study expands on a field of research on the impacts of prescribed fire in redwood forests (Finney and Martin, 1993; Ramage et al., 2010; Engber et al., 2017) and specifically examines a forest with mixed land management history that recently burned in a large-scale wildfire. In this study, I sampled the same areas as Cowman and Russell (2021), but three years later and after the 2020 CZU Lightning Complex (CZU) fire. I assessed the influence of prescribed burning, or the lack of it, on redwood forest post-wildfire:

- Resistance, in terms of canopy retention and percent of total trees that were living.
- Resilience, indicated by understory groundcover and tree species basal sprouting and seedling establishment.
- Risk of repeated wildfire, measured by surface fuel loads, aerial fuels, and stand density.

2.3. Methods

2.3.1. Study area

Big Basin Redwoods State Park (BBRSP) is located in the Santa Cruz Mountains of California and is home to the largest stand of old-growth redwoods south of San Francisco (California Department of Parks and Recreation, 2003). BBRSP is within the southern end of the Marine West Coast Climatic Zone (Martin, 1998). Due to proximity to the coast and maritime influence, BBRSP experiences minimal seasonal variation with high relative humidity and consistent temperatures throughout much of the year (Martin, 1998). Elevation ranges from sea level to about 600 meters; however, the areas with old-growth *S. sempervirens* range from approximately 300 to 600 meters in elevation. Prescribed fire has been used as a management tool in BBRSP since 1978. Burns varied in size and frequency, but were mostly low intensity understory burns (California Department of Parks and Recreation, 2003).

I sampled sites with and without prescribed fire treatments that all subsequently burned during the CZU fire (Cowman and Russell, 2021; Fig. 1; Table 1). Environmental and biological variations between sites were limited. All sites had similar overstory and understory plant composition, topographic position, and seasonal moisture levels. All sites contained soil types predominantly composed of loam or sandy loam (Cowman and Russell, 2021). The prescribed fire sites burned in 1999, 2007, and 2011. All prescribed burn plans required daily conditions of air temperatures between 45 and 75 degrees Fahrenheit, relative humidity between 25 and 89 percent, wind speeds of less than 16 kilometers per hour, and 10-hour dead fuel moistures between 8 and 14 percent (Cowman and Russell, 2021).

The CZU burned 86,509 acres over 37 days in the Santa Cruz Mountains including 97% of BBRSP (California Department of Forestry and Fire Protection, 2020). On August 16th, 2020, a thunderstorm that brought down over 300 lightning strikes in Santa Cruz and San Mateo Counties ignited fires across the region. A strong wind event on the night of August 18th converged three of these fires into one fire system known as the CZU (County of Santa Cruz, 2021). This wildfire was unusually intense for the Santa Cruz Mountains with 43% of the burn area classified as moderate or high burn severity according to a watershed emergency response team (Crockett, 2022). While the fire was declared 100% contained 37 days after ignition by CalFire, smoldering fires were found well into 2021 (Harrell, 2021). The CZU burned all the research sites on August 18th, 2020.

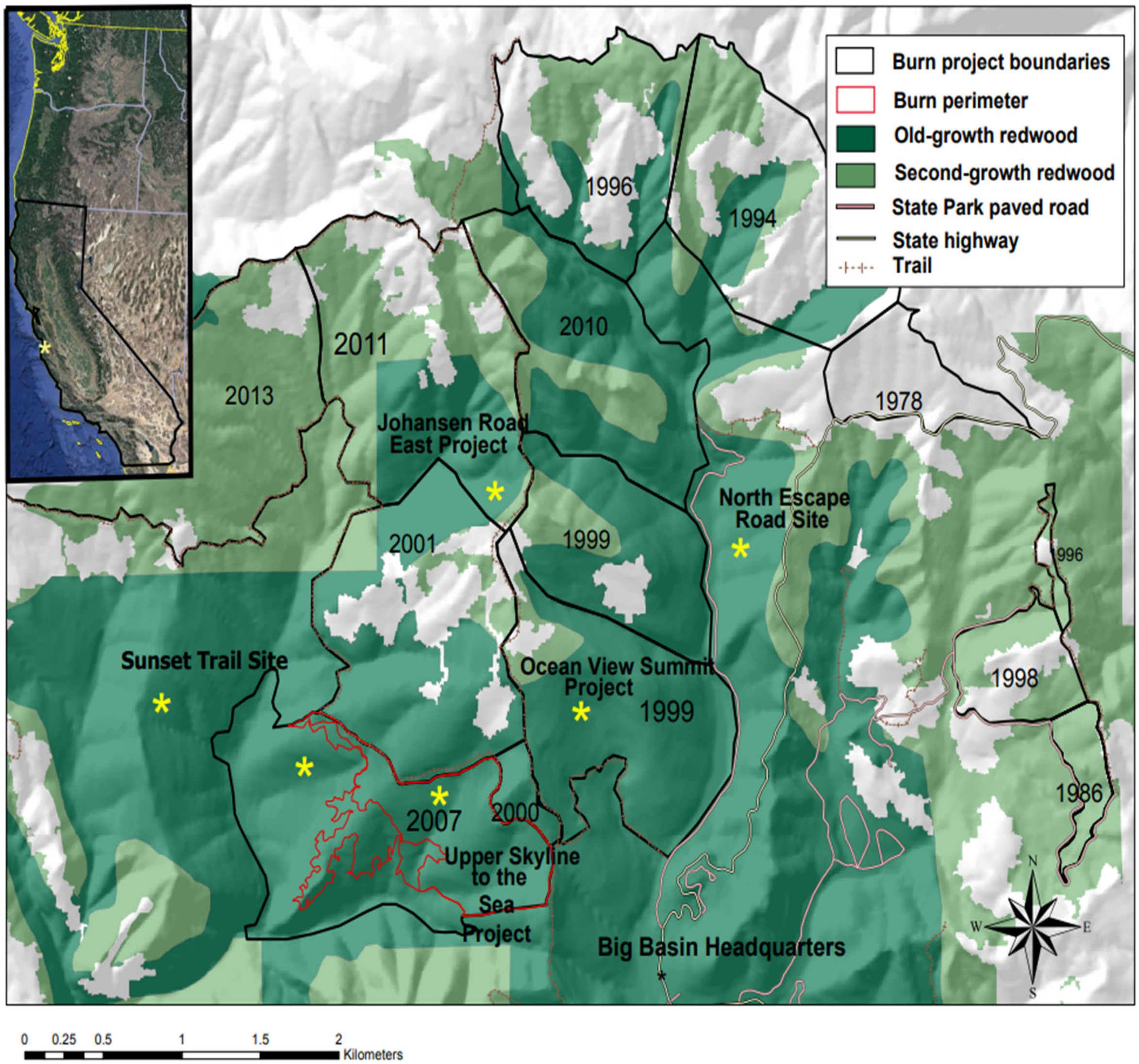

Fig. 1. Research region in Big Basin Redwoods State Park, Boulder Creek, California, USA. Yellow stars approximate sample site locations. Black lines indicate prescribed fire boundaries. Red lines indicate the actual area burned within the project site. Listed years and names represent previous prescribed fire sites. Figure replicated from Cowman and Russell (2021) with permission.

Table 1
Management history and site conditions for all study sites, including year that a prescribed fire occurred and the area burned by prescribed fire if applicable, common soil texture, evidence of logging (Cowman and Russell, 2021), and average heat load across the management unit (McCune and Keon, 2002).

Study Sites	**Prescribed burn sample sites**			**No prescribed fire sample sites**		
	OVSB	S2SB	JREB	SUNU	S2SU	NERU
Prescribed Burn Year	1999	2007	2011	NA	NA	NA
Area Burned (ha)	~109	146.2	< 206.7	NA	NA	NA
Harvest Evidence	None	None	Limited	None	None	Limited
Heat Load Average (unitless)	0.812	0.915	0.927	0.847	0.880	0.881

2.3.2. Study Design

Within the six sites (three with prescribed fire history, and three without), sample plots were randomly chosen. Plot locations were based on the random plot locations from Cowman and Russell (2021). However, these plots were not precisely landmarked, so my study plots represent an approximate resampling. Circular 20 m diameter plots were located at least 10 m from adjacent plots and 200 m from any paved roads to avoid edge effects.

Measurements were not taken in plots where the aspect slope from the plot center was greater than 40 degrees as conditions to take measurements were unsafe. Plots with evidence of more recent second burns from flare-ups following CZU fire containment were not sampled to maintain consistency between wildfire conditions across plots.

2.3.3. Data Collection

I sampled all sites between July and November of 2021, roughly one year after the CZU. Slope, location coordinates, and aspect were recorded from each plot center point. Slope was measured using a Nikon Forestry Pro II Laser Hypsometer (Nikon Ltd., Tokyo, Japan). GPS coordinates were recorded with a Garmin GPSMAP 64SX (Garmin Ltd., Olathe, Kansas, USA). Heat load (McCune and Keon, 2002; Appendix B) was calculated based on field measurements above.

To determine percent canopy cover, a concave spherical densiometer was used. Canopy cover measurements were taken from the plot center in all four cardinal directions to increase accuracy (W. Russell and Michels, 2010). All trees with a diameter at breast height (DBH) of 10 cm or greater with 50% or more basal area within the defined plots were measured and recorded (Busing and Fujimori, 2002). Trees were recorded as top killed if the only green foliage was basal and no epicormic sprouting was observed. Tree regeneration was measured both in terms of basal sprouts and number of seedlings. Basal sprouts were counted and totaled for each plot since basal sprouts' association with specific trees were not always possible to determine (Douhovnikoff et al., 2004; Ramage et al., 2010). Living seedlings, regardless of height, were recorded by tree species.

Understory herbaceous and shrub cover were ocularly estimated. Plant identification was based on the Jepson Manual (Baldwin et al., 2012). Percent cover was estimated for each species using the Braun-Blanquet (1932) method. To improve estimations, each plot was broken into four quadrants. Fuel load data was collected following a modified version (University of California at Santa Cruz, 2017) of J. K. Brown's (1974) fuel sampling methods (Appendix A).

There was a pre-wildfire study in our sampling units, however I sampled unique plots and used different statistical methods (Cowman and Russell, 2021). Therefore, I analyzed some similar variables when possible, including canopy cover, fuel measurements, total living and dead trees, and living and dead trees for Douglas fir, redwood, and tanoak, to affirm similar pre and post-fire trends between studies.

2.3.4. Statistical analyses

I performed statistical analyses in R software version 4.2.2 (R Core Team, 2022). For each plot level response variable, I fit a Generalized Linear Mixed Effect Models (GLMM) with a gamma family distribution with "glmmTMB" (Brooks et al., 2017). I assessed model fit with the "DHARMa" package (Hartig, 2022). Models were checked for overdispersion using a nonparametric dispersion test of the standard deviation of fitted vs. simulated residuals based on a 95% confidence interval. The models predict the statistical significance and effect size of prescribed fire presence or absence for each response variable. Models included heat load (Appendix B) and prescribed fire presence/absence as fixed effects, and site and plot number as random effects. To confirm model validity, I ran Mann-Whitney U tests and all statistically significant GLMM results passed.

2.4. Results

I completed 20 analyses based on the data (Table 2). Only six response variables were statistically significant for the presence/absence of prescribed fire. Results and figures for output variables with a statistically significant relationship with prescribed fire are provided in more depth in the following results sections. The only statistically significant result for heat load was for percent groundcover (P-value= 0.02) where percent groundcover was predicted to be 1.6% (±0.72) when heat load was 0.6 and 5.7 (±1.6) when heat load was 1. Means and standard error are provided for prescribed fire presence and absence for each response variable (Table 3).

Table 2

Results for all response variable analyses sorted by P-value for prescribed fire. All P-values < 0.05 for prescribed fire were considered significant and are highlighted in gray. Predicted effect size and standard error are provided for prescribed fire presence and absence for each response variable.

Response Variable	Prescribed Fire P-value	No Prescribed Fire Effect Size	Prescribed Fire Effect Size
Living and dead tree density (stems ha^{-1})	0.00	352.37 (±20.64)	255.49 (±16.56)
Living *Sequoia sempervirens* (redwood) seedling density (ha^{-1})	0.00	0.06 (±0.07)	107.78 (±198.44)
Standing dead tree density (stems ha^{-1})	0.00	278.21 (±40.01)	154.56 (±22.23)
Living and dead *Notholithocarpus densiflorus* (tanoak) trees density (stems ha^{-1})	0.02	192.98 (±45.53)	94.29 (±20.78)
Percent canopy cover	0.03	79.06 (±1.76)	84.48 (±1.76)
Percent living trees	0.03	22.09 (±4.63)	41.63 (±8.73)
Relative percent dominance of living and dead redwood	0.13	30.05 (±7.97)	53.45 (±14.17)
Coarse woody fuels (Mg ha^{-1})	0.16	52.72 (±17.56)	27.14 (±9.04)
Living and dead *Pseudostuga menziessi* (Douglas-fir) trees density (stems ha^{-1})	0.22	45.53 (±19.67)	21.37 (±9.24)
Fuel depth (cm)	0.17	5.88 (±2.33)	2.62 (±1.17)
Relative percent dominance of living and dead tanoak	0.20	53.81 (±11.43)	37.56 (±7.76)
Tanoak sprouts density (ha^{-1})	0.26	10,569.76 (±1,591.12)	8,307 (±1,250.58)
Percent understory cover	0.31	4.9 (±1.57)	3.1 (±1.0)
Litter depth (cm)	0.33	5.2 (±0.91)	6.62 (±1.15)
Relative percent dominance of living and dead Douglas-fir	0.40	12.93 (±5.40)	7.89 (±3.29)
Douglas-fir seedling density (stems ha^{-1})	0.47	476.98 (±498.30)	152.08 (±173.82)
Living and dead redwoods trees density (stems ha^{-1})	0.51	110.53 (±30.46)	139.72 (±38.5)
Number of redwood sprouts density (stems ha^{-1})	0.55	10,673.51 (±2939.29)	13,419.34 (±3,695.44)
Duff depth (cm)	0.84	>0.01 (±0.00)	>0.01 (±0.00)
Fine woody fuels (Mg ha^{-1})	0.85	2.23 (±0.73)	2.42 (±0.74)

Table 3

Means and standard error for prescribed fire presence and absence for each response variable.

Response Variable	No Prescribed Fire Sample Means (SE)	Prescribed Fire Sample Means (SE)
Living and dead tree density (stems ha^{-1})	356.46 (±18.64)	260.34 (±15.10)
Living *Sequoia sempervirens* (redwood) seedling density (ha^{-1})	119.02 (±50.11)	6,443.66 (±3,405.51)
Standing dead tree density (stems ha^{-1})	276.23 (±17.64)	156.57 (±13.36)
Living and dead *Notholithocarpus densiflorus* (tanoak) trees density (stems ha^{-1})	192.22 (±16.09)	96.74 (±12.40)
Percent canopy cover	79.22 (±1.26)	84.31 (±1.09)
Percent living trees	3.7 (±2.90)	41.33 (±3.85)
Relative percent dominance of living and dead redwood	31.25 (±3.88)	53.51 (±4.23)
Coarse woody fuels (Mg ha^{-1})	52.28 (±9.39)	26.67 (±6.41)
Living and dead *Pseudostuga menziessi* (Douglas-fir) tree density (stems ha^{-1})	45.83 (±7.23)	21.64 (±5.10)
Fuel depth (cm)	5.89 (±1.63)	2.74 (±0.75)
Relative percent dominance of living and dead tanoak	54.06 (±3.69)	37.78 (±4.08)
Tanoak sprouts density (ha^{-1})	10,567.37 (±794.16)	8,309.8 (±893.20)
Percent understory cover	5.6 (±1.38)	3.96 (±0.97)
Litter depth (cm)	5.49 (±0.37)	6.72 (±0.44)
Douglas-fir seedling density (stems ha^{-1})	1,616.64 (±751.63)	449.99 (±332.48)
Relative percent dominance of Douglas-fir	13.35 (±1.96)	8.06 (±2.01)
Living and dead redwoods trees density (stems ha^{-1})	113.93 (±15.65)	138.75 (±14.17)
Number of redwood sprouts density (stems ha^{-1})	10,684.48 (±1764.54)	13,406.68 (±1,679.14)
Duff depth (cm)	0.0 (±0.0)	0.07 (±0.04)
Fine woody fuels (Mg ha^{-1})	2.49 (±0.39)	2.75 (±0.37)

2.4.1. Forest resistance to wildfire

Prescribed fire was predicted to increase canopy cover following wildfire compared to areas without prescribed fire (P=0.03, RM=0.094, RC=0.159, Fig. 2). Following wildfire, there was a 6% difference in canopy cover between treatments. Canopy cover was predicted to be 79% without prescribed fire and 85% with prescribed fire (95% confidence intervals: 76-83% and 81-88% respectively).

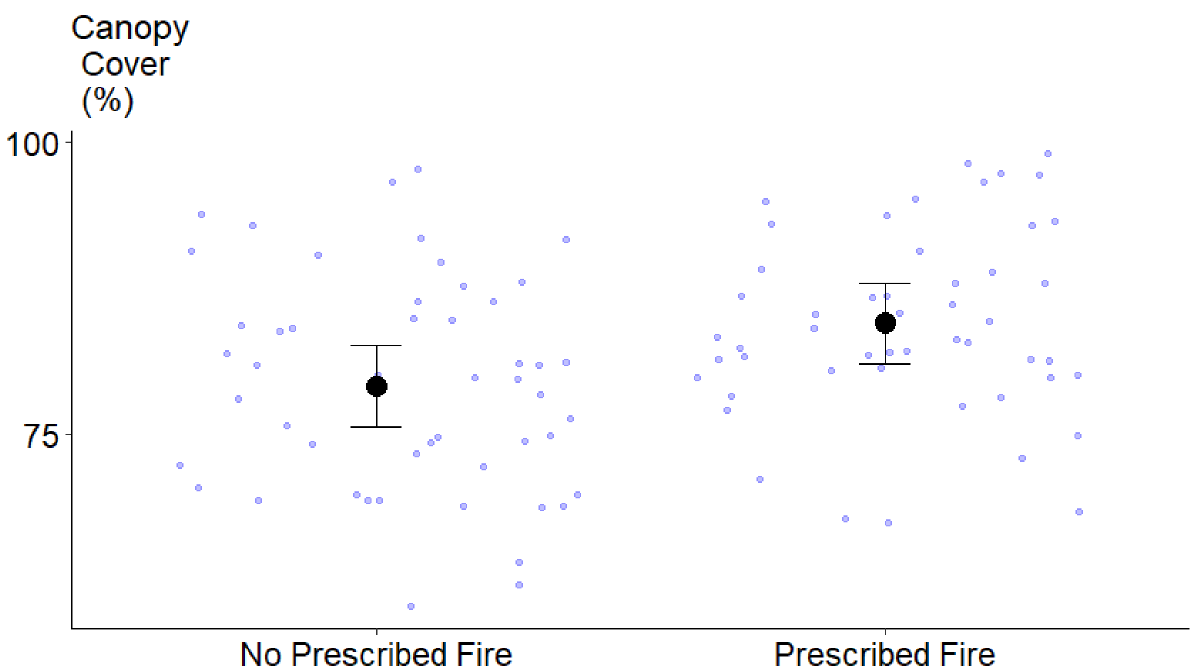

Fig. 2. Post-wildfire canopy cover response in areas treated with and without prescribed fire. Black dots indicate the model estimates, whiskers indicate 95% confidence interval, and blue dots are field observations.

Prescribed fire was predicted to increase the percent of total trees that were living following wildfire compared to areas without prescribed fire (P=0.03, RM=0.046, RC=0.046, Fig. 3). The percent of trees that were living was predicted to be 22% without prescribed fire and 42% with prescribed fire (95% confidence intervals: 15-33% and 27-63% respectively).

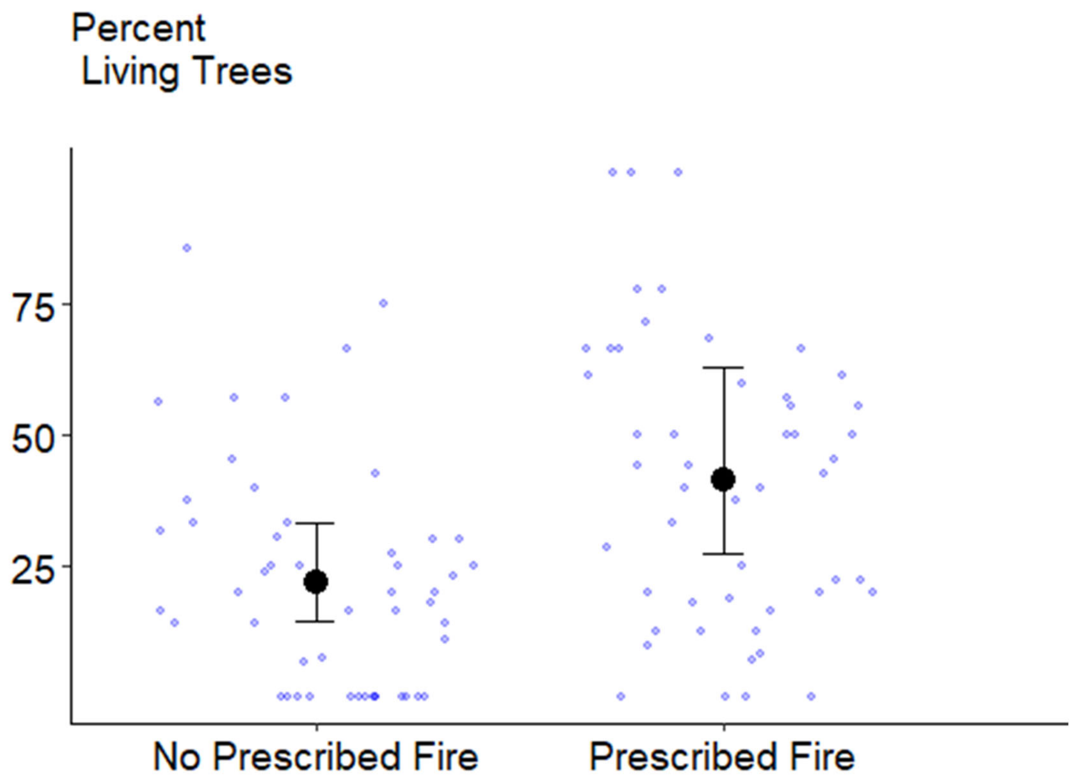

Fig. 3. Post-wildfire percent of total trees that were living in areas treated with and without prescribed fire. Black dots indicate the model estimates, whiskers indicate 95% confidence interval, and blue dots are the field observations.

2.4.2. Forest resilience to wildfire

After wildfire, the number of redwood seedlings was statistically greater in prescribed fire areas compared to untreated areas (P=0.004, RC=0.2428, RM=0.888, Fig. 4). Post-wildfire redwood seedling counts were low in most sample plots regardless of prescribed fire presence or absence. However, several plots in one prescribed burn site, had notably higher counts with the highest field counts being 4,505; 2,595; and 1,692. Areas without prescribed fire were predicted to have less than 1 redwood seedling per hectare, and prescribed fire plots were predicted to have 108 redwood seedlings per hectare (95% confidence intervals: 0-1 and 3-4,164 respectively).

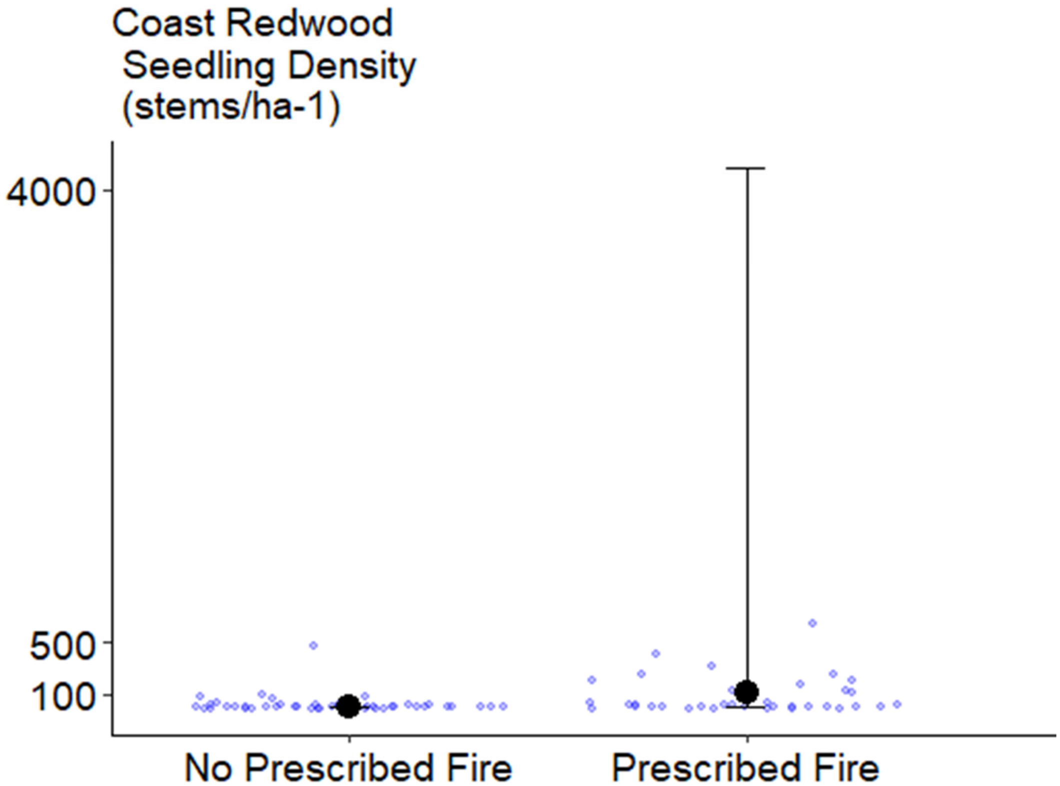

Fig. 4. Post-wildfire coast redwood seedling density per hectare in areas treated with and without prescribed fire. Black dots indicate the model estimates, whiskers indicate 95% confidence interval, and blue dots are the field observations.

2.4.3. Risk of repeated wildfire

Post-wildfire, fewer living and dead tanoak trees were predicted in prescribed fire areas compared to areas without prescribed fire (P=0.022, RM=0.052, RC=0.052, Fig. 5). Areas without prescribed fire were predicted to have 193 tanoak trees per hectare while prescribed fire areas were predicted to have 94 tanoak trees per hectare (95% confidence interval: 125-299 and 60-146 respectively).

Post-wildfire, living and dead tree density was found to be lower in prescribed fire areas compared to areas without prescribed fire (P<0.0001, RM=0.151, RC=0.329, Fig. 6). Sample areas without prescribed fire were predicted to have 352 total trees per hectare while areas

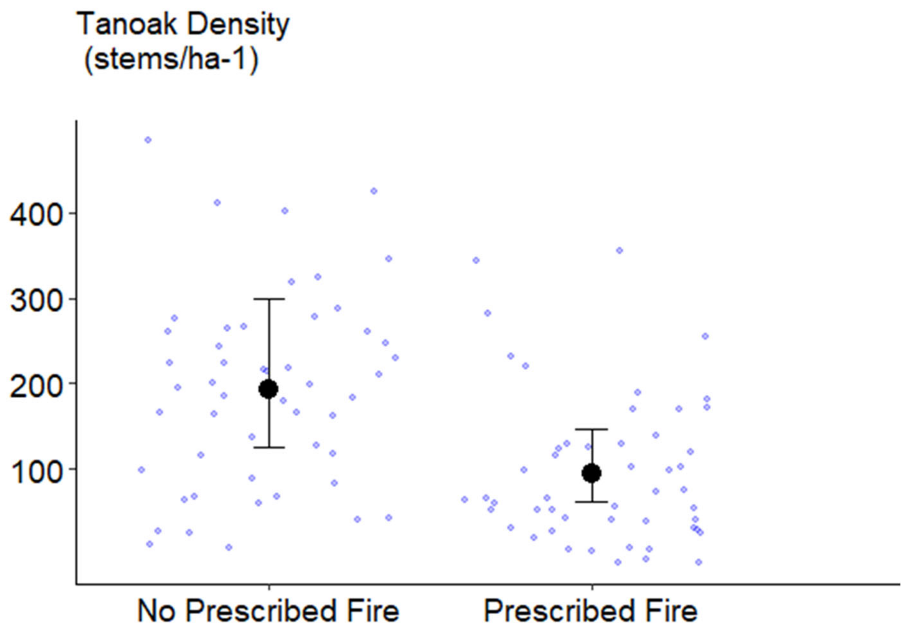

Fig. 5. Influence of prescribed fire on living and dead tanoak tree density per hectare in areas treated with and without prescribed fire. Black dots indicate the model estimates, whiskers indicate 95% confidence interval, and blue dots are the field observations.

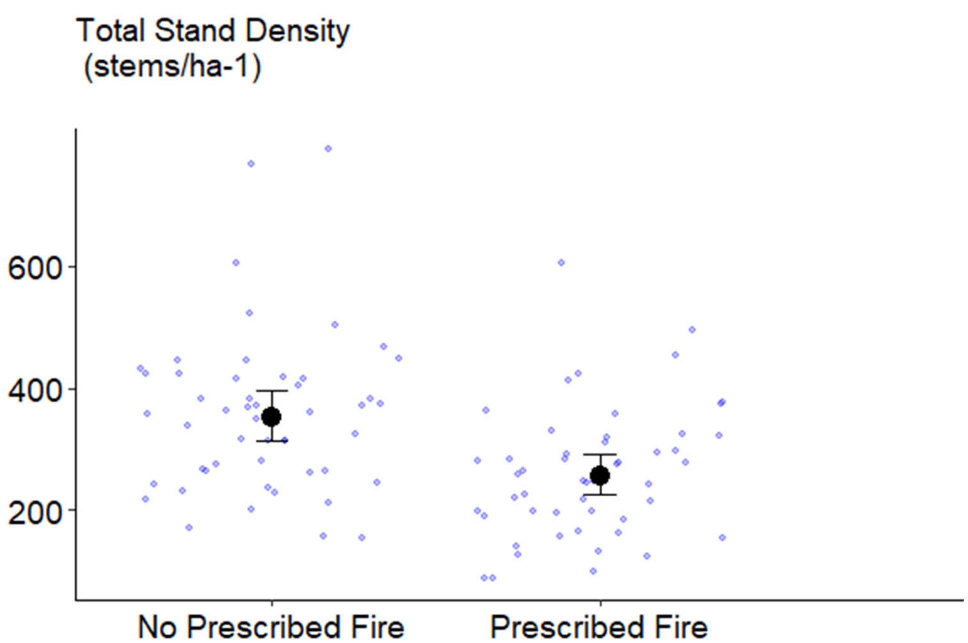

Fig. 6. Influence of prescribed fire on the density of living and dead trees per hectare in areas treated with and without prescribed fire. Black dots indicate the model estimates, whiskers indicate 95% confidence interval, and blue dots are the field observations.

with prescribed fire were predicted to have 255 total trees per hectare (95% confidence intervals: 314-396 and 225 respectively).

Post-wildfire, fewer standing dead trees were predicted in areas with prescribed fire compared to areas without prescribed fire (P=0.004, RM=0.079, RC=0.079, Fig. 7). Sample areas without prescribed fire were predicted to have 278 standing dead trees per hectare while areas with prescribed fire were predicted to have only 155 standing dead trees per hectare (95% confidence intervals: 209-370 and 116-206 respectively).

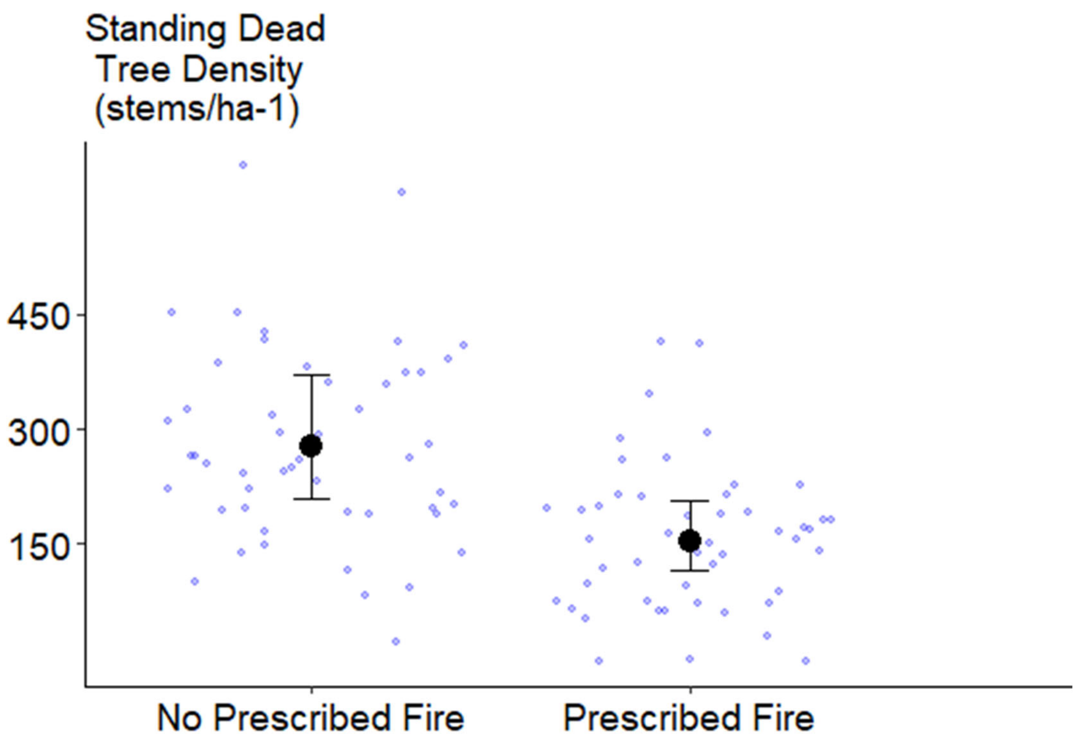

Fig. 7. Influence of prescribed fire on the density of standing dead trees per hectare in areas treated with and without prescribed fire. Black dots indicate the model estimates, whiskers indicate 95% confidence interval, and blue dots are the field observations.

2.5. Discussion

2.5.1. Forest resistance to wildfire

Redwood forests are known for their resistance to wildfire (Ramage et al., 2010; Stephens and Fry, 2005). Redwood forests in the species' northern range have even resisted stand replacement after high severity wildfire (Woodward et al., 2020). However, wildfire proliferation and climate change may create new risks to redwood survival through future wildfires. While the 1.74 million ha burned in California in 2020 was likely similar to pre-Euroamerican conditions, the area burned was more than twice the previous documented record in California (Safford et al., 2022). New records for high severity wildfires may pose greater risks to redwood forests like in other Mediterranean forest types. For example, mixed conifer forests in the Sierra Nevada Mountains, CA shifted from mature conifer forest to non-forested vegetation after record setting droughts and wildfires (Steel et al., 2023).

My results suggest that prescribed fires, even those applied as much as 21 years prior, reduce the impacts of severe wildfire. I found more canopy cover retained in areas with prescribed burning after wildfire than in areas without prescribed burning. This is in contrast to pre-wildfire 2019 canopy cover measurements where areas with prescribed fire had 91% canopy cover and areas without prescribed fire had 94% canopy cover (Cowman and Russell, 2021). Therefore, prescribed fire areas lost 6% of their canopy while areas without prescribed fires lost 15% of their canopy. While this difference could be partially due to sampling distinct plots in each study and distinct analysis methods, there is also a mechanistic reason for this difference. There were significantly different fuels between treatments. Prescribed fire areas likely retained more canopy because they had fewer ground fuels (duff, litter, 1-hr fuels) and ladders fuels (shrub cover and stems < 10 cm DBH and > 1m in height) which likely caused lower wildfire intensities than in areas without prescribed fire (Cowman and Russell, 2021).

My results show that prescribed fire helps to protect old-growth redwood canopy habitat. These findings support previous studies which show that prescribed fire is an effective management tool for mitigating crown mortality and crown fires in other forest types (Pollet and Omi, 2002). Canopy retention not only protects valuable and unique arboreal habitat (Baker et al., 2006; Sillett and Van Pelt, 2007), but may also prevent a shift of understory species away from shade tolerant redwood associate species toward more sun tolerant species. Additionally, retained canopy following wildfire provides more shade, which may mitigate fuels drying and help prevent fire regime shifts toward more frequent and severe wildfires (Ellis et al., 2022).

2.5.2. Forest resilience to wildfire

Redwoods and tanoaks are inherently resilient to fire because they resprout from the base (Ramage et al., 2010). I found prolific tanoak and redwood sprouts across treatments. Sprouts were counted roughly one year after the wildfire. Many of the basal sprouts on both tanoak and redwood are unlikely to survive in following years as they naturally thin themselves out (O'Hara et al., 2017). One limitation of this study is that no distinction was made between basal sprouts size classes. Distinction between size classes in the field would have allowed for more meaningful comparisons since basal sprouts have greater density in their earlier stages (O'Hara et al., 2017).

Redwood seedling establishment is not essential for post-fire regeneration in the short-term given the species' ability to resprout prolifically. However, seedlings do help to ensure genetic diversity that cannot be achieved by resprouting (Douhovnikoff et al., 2004; Douhovnikoff and Dodd, 2015). This is especially important because redwoods have low genetic diversity among stands, particularly in the species' southern range (Brinegar, 2012). Species and population genetic diversity may prove increasingly important as the species faces new obstacles in a changing environment (Schierenbeck, 2017; Razgour et al., 2019). Redwood seedlings had a strong association with prescribed fire. This association was partially driven by redwood seedlings occurring in extremely high numbers in several plots near to one another in one prescribed fire site. Others similarly found high redwood seedling densities concentrated in small areas post-wildfire (Lazzeri-Aerts and Russell, 2014). Given that plots with high seedling densities were found in only one of the three prescribed fire sites, it is possible variables other than prescribed fire such genetic differences and local environmental conditions may have driven seedling recruitment. Further research is needed to understand this phenomena.

Given that seedlings were sampled about one year after the fire, long-term viability of seedlings is uncertain and it is likely many of these seedlings will not survive to become saplings (Lazzeri-Aerts and Russell, 2014). While higher seedling counts indicate stronger redwood resilience, additional sampling once seedlings have had more time to establish is recommended for a better assessment of the impacts of prescribed fire on redwood resilience.

More Douglas-fir seedlings were found in areas without prescribed fire, but these areas also had more Douglas-fir trees. Overall, few Douglas-fir seedlings were found across plots, with no significant difference detected between treatments. Greater canopy cover in areas treated with prescribed fire may lead to quicker recovery for Douglas-fir as crowns containing Douglas-fir seeds may be more prevalent; however, this on its own is not enough to infer improved Douglas-fir regeneration. Given the severity of the fire and impact to forest canopy, many Douglas-fir seeds were likely consumed by fire causing regeneration of species to be slow. Douglas-fir trees in nearby areas to sites included in this study that experienced lower severity fire had greater crown retention and seedling establishment (based on field observations made outside of the study sites).

No difference was detected for the effect of prescribed fire on understory cover. However, heat load was found to have a statistical and positive relationship with understory cover. This is a logical relationship as higher heat load indicates greater solar radiation reaching the forest floor, allowing more sunlight to reach understory species and prompt a quicker recovery. Overall, understory cover estimates were low as few species had re-established at the time of sampling. To better assess the influence of prescribed fire on post-wildfire recovery of understory species, sampling once more time has passed after a fire event is recommended.

2.5.3. Risk of repeated wildfire

In the wake of the CZU, BBRSP is still at risk of future wildfires. While many surface fuels were consumed in the wildfire, fuel structure was also reorganized. Weakened trees fell to the ground due to the fire, and now pose additional risk of igniting from surface fires and spreading fire to aerial fuels. A statistical difference in the density of standing dead trees was found between treatment groups following the CZU. Prescribed fire areas had lower total tree density and fewer standing dead trees compared to areas without prescribed fire. Prescribed burn areas had a more open forest structure which may mitigate the rate of wildfire spread in the future as continuous fuels may be disrupted (Stephens et al., 2009). In addition to greater tree spacing, the decrease in standing dead trees in prescribed burn areas indicates a lower risk of repeated wildfire as there are fewer dead and dried out aerial fuels compared to plots without prescribed fire (Goodwin et al., 2021).

Post-wildfire forest structure trends were similar to what Cowman and Russell (2021) found based on data collected in 2019. We both found that total and tanoak stand densities were lower in areas with prescribed fire. One of the most pronounced differences in both studies between treatment groups was the reduced prevalence of tanoak trees in areas previously treated with prescribed fire. Post-wildfire reductions in overall stand density and the number of standing dead trees were likely related to lower tanoak stand density found prior to the CZU (Cowman and Russell, 2021). Unlike redwood and Douglas-fir, tanoak topkills from fire in most cases (Ramage et al., 2010). Given that the predicted number of tanoaks in this study was nearly double the number in areas without prescribed fire, it reasons that areas with greater tanoak density were also predicted to have more standing dead trees after wildfire. Prescribed fire benefits in terms of reduced total and tanoak stand density in BBRSP found by Cowman and Russell (2021) were maintained following wildfire. This is an important finding as it suggests that prescribed fire benefits to stand structure may persist even after wildfire events.

In terms of surface fuel loads, Cowman and Russell (2021) found that the average for coarse woody fuels (CWF) in areas without prescribed fire was 123 Mg ha^{-1} and only 72 Mg ha^{-1} in prescribed fire areas. Following the CZU, I found considerably lower levels of CWF, but a similar trend for averages with 52 Mg ha^{-1} in areas without prescribed fire and only 27 Mg ha^{-1} in prescribed fire areas. Similar to Cowman and Russell's (2021) findings, measurements for litter depth, fuel depth, and fine woody fuel (FWF) loads were similar between treatment groups. Cowman and Russell (2021) found that before the CZU, average duff depth was 7.5 cm in areas without prescribed fire compared to 3.8 cm in areas with prescribed fire. While I measured duff, all measures were close to zero as nearly all duff was consumed across all sample sites in the wildfire and not enough moisture had accumulated or time passed for duff to occur at the time of sampling. While it is possible that surface fuels in areas with and without prescribed fire will accumulate at different rates following the CZU, at the time of my sampling no statistical difference was detected between treatment groups for any of the surface fuels (CWFs, FWFs, litter, duff, or fuel depth) measured in my study. It is my recommendation that surface fuels be re-sampled once more time has elapsed since the CZU to better understand whether Cowman and Russell's (2021) findings of reduced surface fuel loads in prescribed fire sites were maintained following wildfire.

2.5.4. Management implications

When the length of time between prescribed burns and subsequent wildfire (EX: between 9 and 21 years) as well as severity of the subsequent wildfire are considered, finding any positive impacts of previous prescribed burns is a strong validation for their use. Prescribed burns protected forest canopy from wildfire, which demonstrates that prescribed fire is an effective tool to mitigate risk of crown loss from high severity crown fires**.** This supports similar findings in other forest types showing prescribed fire as an effective tool to mitigate crown fire potential (Pollet and Omi, 2002). This study further supports that prescribed fire, even 21 years later, can be an effective tool for land managers aiming to reduce total stand density and stand density of tanoak in redwood forests, with beneficial effects extending even after a wildfire.

My finding that post-wildfire redwood seedling establishment increased in areas with prescribed fire is important. However, it should also be considered that redwoods and tanoaks exhibited such strong resilience to fire regardless of prescribed fire history that these species may not need intervention to promote post-wildfire regeneration. Prescribed fire has the potential to reduce overall ecological and social risks associated with wildfire, but an ecological justification on its own for prescribed fire should be given careful consideration. For example, forest goals to protect Douglas-fir or other species that are comparatively less adapted to wildfire may provide stronger ecological justifications for prescribed fire use compared to goals to protect redwoods which are more likely to resist and recover from wildfire events without intervention from land managers.

Aside from ecological value, redwood forests also hold value in terms of anthropogenic use, ranging from residential living to recreation and utilization. Even if forest ecosystems may be able to regenerate after wildfire, the human communities that live and use these forests may not be as resilient. In this way, prescribed fire may fill the niche of permitting fire to play its ecological role in redwood forests while also reducing impacts to human communities and fostering social-ecological resilience to severe wildfire.

Even though wildfire can reduce fuel load, it can also restructure fuels in ways that increase risks for repeated wildfires (Menning and Stephens, 2007). Many living aerial fuels in my study area burned and fell, becoming drier surface fuels in the post-wildfire landscape. Dead trees are drier than living trees and do not have the same natural defenses to fire as living trees (Nolan et al., 2020); they are therefore subject to an elevated risk of burning (Passovoy and Fulé, 2006). Following wildfires, land managers are faced with the question of how to prevent repeated wildfires. While a relationship between prescribed burning and post-wildfire surface fuel loading was not established in this study, relatively high levels of surface fuel loading in all sample sites following wildfire suggest that risk of repeated wildfire in redwood forests is a serious concern regardless of whether fire mitigation strategies were implemented prior to a wildfire event. Further research into the efficacy of implementing prescribed fire after wildfire events to reduce risk of repeated wildfire is recommended.

2.5.5. Conclusion

Recent years have been marked by record breaking wildfires as our landscape continues to reconcile accumulated fuel loads resulting from over a century of fire suppression, which are also accelerated by climate change. Redwood forests are very resistant to low and moderate severity fires and are resilient to severe wildfires. Prescribed fire can mitigate intense wildfire conditions (shown by retained canopy cover) and reduce the risk for repeated wildfire events (shown by reduced density of stand and standing dead trees).

Wildfires may reduce fuel load, but they also reorganize fuels in a way that may increase risk for future fires and begin a regime shift toward more frequent or severe wildfires. Greater canopy cover and a reduction in standing dead trees in areas treated with prescribed fire in this study indicates they will be more resistant to future wildfires. However, it should not be assumed that areas that have been treated with prescribed fire are immune from wildfire risk, and even the risk of repeated wildfire. Benefits of prescribed burns shown in this study add support to the utilization of this land management tool. However, managing high fuel loads resulting from fire suppression with prescribed fire should not be seen as the end-all solution. Our society must also learn to better coexist with more frequent and severe wildfire seasons.

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Appendix A. Fuel Load Methods and Calculations

Fuel Data Collection- To measure dead and downed woody fuels, two transect lines from each plot center were sampled, one along the dominant slope and the other 90 degrees clockwise from the first transect line. To determine duff, litter, and fuel depths, measurements were taken at 1.52 m and 3.05 m from the center of each plot. Fuels were broken down into burn classes: 1-hour (0 - 0.64 cm), 10 hour (6.4 – 2.54 cm), 100-hour (2.54 – 7.62 cm), and 1000+ hour (7.6+ cm). On each transect line, fuels were counted for 1-hour fuels between 0 and 1.52m, 10-hour and 100-hour fuels between 0 and 3.05m, and 1000-hour fuels between 0 and 11.34m (J. K. Brown, 1974; University of California at Santa Cruz, 2017). 1000-hour fuels were categorized as either “sound” or “rotten” (Cowman and Russell, 2021).

Fuel Calculations- Fuel load was broken into fine woody fuels (FWF) and coarse woody fuels (CWF) based on calculations first described in J. K. Brown (1974) to calculate mean fuel load. Mean fuel load was first calculated in tons per acre and then converted to metric tons per hectare (Mg ha−1). Fuel load calculated as FWF and CWF for each transect line was averaged to have one (Mg ha−1) measurement for each plot.

Fine woody fuels in (1-hour, 10-hour, and 100-hour fuels) were calculated using Equation A.1, based on J. K. Brown (1974) where *k* is a constant of 11.64, *n* represents the number of FWF individual intersections counted in each size class, *d*2 is the square of the quadratic mean diameter of each size class, *s*oc is the composite specific gravity, *a* is the composite angle correction factor for each size class, *c* is the average transect slope correction factor, and *Nl* is the transect length. The slope correction factor (*c*) calculated based on the percent slope of each transect line using Equation A.2 (*d*2, *a,* and *s*oc), were constant values for non-slash, conifer, and Western species found in J. K. Brown (1974).

$$Equation\ (\mathrm{A.1}) \qquad Fine\ Woody\ Fuel\ Load\ (\mathrm{Mg\ ha^{-1}}) = \frac{sknd^2 s_{oc} ac}{Nl}$$

$$Equation\ (\mathrm{A.2}) \qquad c\sqrt{1 + \left(\frac{percent\ slope}{100}\right)^2}$$

Calculations for CWF (≥1000-hour fuels) were conducted separately for sound and rotten thousand-hour fuels (Cowman and Russell, 2021; Glebocki, 2015). CWF calculations were made using Equation A.3 from J. K. Brown (1974). ∑d2 is the sum of diameters of particles in each decomposition class squared, and all other terms are the same as defined for Equation A.2.

$$Equation\ (\mathrm{A.3}) \qquad Coarse\ Woody\ Fuel\ Load\ (\mathrm{Mg\ ha^{-1}}) = \frac{k(\sum d^2)\, s_{oc} ac}{Nl}$$

Appendix B. Heat Load Index Calculations

Calculations for heat load were based on methods outlined by McCune and Keon (2002) which rely on tables of incident radiation using slope, aspect, and latitude. Aspect was "folded" on the Northeast-Southwest line so Northeast equals zero degrees and southwest becomes 180 degrees using Equation B.1.

Equation (B.1) Folded aspect = $| 180 - |aspect - 225| |$

Latitude, slope, and folded aspect were all converted into radians and entered into a formula provided by McCune and Keon (2002) using Microsoft Excel. Estimates are made by least-squares multiple regression using trigonometric functions of latitude, slope, and aspect (McCune, 2007). Results for heat load are considered unitless as there is not a basis for converting the results into a measure of temperature (McCune, 2007).