POST-CZU LIGHTNING COMPLEX REGENERATION IN THE AÑO NUEVO MONTEREY PINE STAND

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ABSTRACT

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Climate induced increases in fire frequency and severity along with years of fire suppression and drought are predicted for California. The recent CZU Lightning Complex, a high severity wildfire, burned in the Santa Cruz Mountains on California’s central coast and affected an assortment of vegetation, including the survival and recovery of the native Año Nuevo Monterey pine (*Pinus radiata* (D. Don)) stand. This stand is partially located in Cal Poly’s Swanton Pacific Ranch and has been monitored for over 20 years for the presence of pitch canker (*Fusarium circinatum*) fungal disease. This study characterized the survival and initial recovery of vegetation approximately nine months after the fire. To better understand the influence of the CZU Lightning Complex on the vegetation within the 14 measured plots, we used a photo log to collect the vegetation surface area, height, and volume. In this work, we examined the vegetation density and its relationship with burn severity and slope. The regeneration variables and vegetation were observed in maps in ArcGIS Pro. The relationship between burn severity and regeneration dynamics suggested a moderate but negative correlation ($r = -0.68$) where short-term regeneration was less dense when severity was higher with a significant relationship ($p = 0.008$). The results from burn severity suggested better correlation to vegetation than those implied from the correlation to slope ($r = 0.26$), suggesting no significant relationship between slope and vegetation ($p = 0.37$). Observed recovery of understory density was modest in some plots and robust in others but included the reemergence of Monterey pines seedlings. Further studies should focus on continuing to monitor the post-fire vegetation regeneration in the plots as a longitudinal study of the Monterey pines.
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CHAPTER 1. INTRODUCTION

1.1. Background

Climate driven increases in fire frequency and severity are predicted for Mediterranean climatic zones, including the Pacific coast of California (Peñuelas & Sardans, 2021; Rundel et al., 2018). The central coast of California is the native home to Monterey pines (Pinus radiata D. Don) which is of particular interest due to their iconic characteristics as a fire evader, their importance in the ecosystem, their value as a commercial forest species and their significant and growing threats (Reynolds et al., 2019; Rogers, 2002; Stephens et al., 2004; Walker, 2021). One of the primary threats to Monterey Pines is Fusarium circinatum (pitch canker), a plant pathogenic fungus, which leads to death in branches and roots, declines the tree’s overall health and likely leads to mortality (Ferchaw et al., 2013; Piirto & Valkonen, 2005; Storer et al., 2002). Some native stands have demonstrated evidence of pitch canker resistance through natural selection (Aegerter & Gordon, 2006; Estades et al., 2012; Storer et al., 2002). Evidence of the pitch canker resistance has generated research to understand the resistance influenced mortality within the native Monterey pine tree stands. Post-fire natural regeneration allows a wider expansion of genetic diversity in the new host populations which natural selection in favor of resistant genotypes can be imposed by the pathogen (Reynolds et al., 2019; Storer et al., 1999).

Monterey pines have been classified as a fire evader primarily because of its canopy stored seed bank that is released after high severity fires (Stephens et al., 2004). Fire is a major influence affecting the extent and makeup of Monterey pine stands (Dvořák et al., 2017; Stephens et al., 2004). Without fire, cones generally remain attached to trees for many years and do not release all their seeds (Roy, 1966; Storer et al., 2001). However, high severity fires can also create gaps allowing invasive species or competition to take advantage (Lazzeri-Aerts & Russell, 2014; Mahdizadeh & Russell, 2021). Understory competition encourages water stress and water loss through evapotranspiration in Monterey pine stands. Therefore, greater seedling growth is achieved with less competition from understory species (O’Brien et al., 2007).
The current study is focusing on the native Año Nuevo Monterey pine stand on California Polytechnic (Cal Poly’s) University’s Swanton Pacific Ranch (SPR), which has been studied since 2001 for the effectiveness of treatment and genetics on resistance to pitch canker (Ferchaw et al., 2013; Norville, 2017; Piirto & Valkonen, 2005; Stephens et al., 2004; Wise, 2004). In August 2020, the CZU Lightning Complex fire burned 86,509 acres in Santa Cruz and San Mateo counties, including through the native Año Nuevo stand at SPR (CAL FIRE CZU Complex Report, 2020). In March 2021, in collaboration with foresters at CAL FIRE, all the previously researched Monterey pine plots were deemed mortal due to the fire. This fire is considered the worst stand killing fire event for the Santa Cruz Mountains (Mahdizadeh & Russell, 2021). This fire demonstrated the impact of years of fire suppression and prolonged drought on the Monterey pine ecosystems (Mahdizadeh & Russell, 2021; Stephens et al., 2004) but centuries of these conditions coupled with the effects of climate change have led to record fires in recent years (Stephens et al., 2018). To best continue the long-term research on the previously studied native Año Nuevo Monterey pines, the goal of my research was to analyze the vegetation regeneration following the high-severity CZU Lightning Complex fires and to explore the critical influences on the vegetation recovery, particularly topographic characteristics and burn severity.

1.2. Problem Statement

The urgency to protect the iconic native Monterey pines continues to heighten due to threats from urbanization, fire suppression, and pathogens, specifically pitch canker (Blank et al., 2019; Ferchaw et al., 2013; Henry, 2005; Loe, 2010; Millar, 1998). The native ecosystems are an important genetic resource for the P. radiata industry in other parts of the world (Storer et al., 2001; Wu et al., 2021). Monterey pine plantations are favored across the world for their aesthetic value and timber production (Blank et al., 2019). Genetic conservation is also important to contribute to the longevity of the population and maintaining the range of variation found within the species (Moran et al., 1988; Rogers et al., 2006). The native stands are already of concern because there are few remaining, and they are diminishing (Henry, 2005).
Conservation of Monterey pines also includes post-fire vegetation growth evaluation and management. Monterey pine’s history indicate growth correlating with charcoal abundances in the sediment implying that fire plays an important role in the spread of Monterey pines (Millar, 1998; Stephens et al., 2004). Fire may even be considered as a management tool in the future to produce large populations of Monterey pines and the trees resistant to pitch canker fungus will have a greater chance of survival (van Wilgen, 2009). Understanding forest stand development following a major disturbance is important for predicting ecosystem responses and forecasting long-term trajectories in diverse fire-prone forests (Piirto & Valkonen, 2005; Roy, 1966; Storer et al., 2002). Climate change and altered disturbance regimes affect forest ecosystems indirectly through wildfire (Davis et al., 2020). Furthermore, the majority of previous Monterey pine research emphasizes the epidemiology of pitch canker with little focus on Monterey pine ecology and native stand dynamics (O’Brien et al., 2007; Stephens et al., 2004). Increasing fire frequency and severity throughout the Western United States could lead to undesirable ecosystem changes (Mahdizadeh & Russell, 2021), thus, measuring post-fire vegetation recovery for differing burn severities can provide useful information (Bright et al., 2019).

While the CZU fire was catastrophic to the Monterey pine stand in the Año Nuevo Forest, studying vegetation in post-fire areas adds to the existing research and could allow for a better understanding of managing native Monterey pine forests. There is also a need for increased understanding of the regeneration process is needed to promote seedling establishment (O’Brien et al., 2007). Fires in the native Monterey pine regions are not typical with only one recent fire in the native stands in 2009 (Niebrugge, 2012). The CZU Lightning Complex is relatively recent with little research on the regeneration from this specific fire (Mahdizadeh & Russell, 2021). This research holds significance in understanding of the CZU Lightning Complex’s effect on the ecosystem as an entirety. Although fire severity has readily measurable parameters, both on the ground and with remote sensing, ecosystem responses require a much more descriptive and process-based approach (Keeley, 2009). The current research continues a long-term study at Cal Poly’s Swanton Pacific Ranch in the Año Nuevo Monterey pine stand.
1.3. Purpose and Objectives

A long-term (approximately 20 years) experimental field study examined how pitch canker affects Monterey pine seedlings in treated and untreated tree plots within the native Año Nuevo stand. That research evaluated survival and growth of Monterey pine seedlings under different silvicultural treatments on SPR located in native Año Nuevo Monterey pine stand (Loe, 2010; Norville, 2017; Piirto & Valkonen, 2005; Pinkerton, 2006; Wise, 2004; Yun, 2011). Prior research in the Año Nuevo native tree stand at SPR found seedlings and saplings were less infected (46% infected) with pitch canker than large trees (90% infected) which could be due to their susceptibility as older trees (Piirto & Valkonen, 2005).

This research was intended to continue until the CZU Lightning Complex fire damaged SPR causing Monterey pine tree stand mortality. Fire is a natural component of Monterey pine ecosystems, opening cones to release seeds which would otherwise be released at a slow rate. In March 2021, we documented post-fire Monterey pine seedling germination in the Año Nuevo Monterey stand.

Discovering Monterey pine saplings approximately eight to nine months post-fire, creates a need to update the conditions of the original plots for future research on the Año Nuevo native stand and an understanding of the CZU Lightning Complex. After years of fire suppression, it is critical to monitor the ecosystem’s response to fire. Although lightning storms in the dry season, like those that preceded the CZU Lightning Complex, are historically rare, similar wildfires may be increasing in the area as a result of human induced climate change (Mahdizadeh & Russell, 2021; Torn & Fried, 1992). Furthermore, burn severity indicators and potential ecosystem recovery could provide useful information to post-fire planners. The conditions monitored for this project include the burn severity, slope, and understory vegetation density by plot. The factors which define the vegetation regeneration rate after a wildfire are multiple but burn severity levels and topography (elevation, slope, and orientation) have been previously revealed to be main drivers of post-fire vegetation regeneration (Riaño et al., 2002; Röder et al., 2008; Viana-Soto et al., 2017). An insight into short-term vegetation reproduction based on burn severity and

This project uses a photo log as a system to measure the understory vegetation and make comparisons on a plot-to-plot basis. As fire sizes and frequency increase, time becomes a constraining factor, traditional methods to assess post-fire impact on vegetation have become costly and labor-intensive (Gitas et al., 2012) making way for digital methods for measuring vegetation. In addition, the photo log shows updated conditions of the plots used as preliminary data for future research. In this study, we hypothesize that environmental variables, specifically burn severity and slope, influence post-fire vegetation growth. The principal question addressed is whether these environmental factors (burn severity and slope) influence short-term (nine months after the fire) vegetation regeneration based on the conditions of each plot. With this information, further estimates can be predicted on Monterey pine sapling survival per plot allowing Año Nuevo Monterey pine research to continue.
CHAPTER 2. LITERATURE REVIEW

This chapter reviewed past research on Monterey pines and their characteristics, habitat, and threats including pathogens, fire suppression and drought. This chapter then discusses fire as an integral part of the Monterey pine’s lifecycle, the CZU Lightning Complex and post-fire regeneration and understory competition.

2.1. Monterey pines

Monterey pine (*Pinus radiata*) are a closed-cone coniferous species that are native to North America (Loe, 2010; Wu et al., 2021). Mature trees range from 12 – 31 m (40 – 100 ft) and ~ 0.15 - 0.30 m (6 – 11 in) in diameter depending on age and soil condition (Estades et al., 2012; Roy, 1966). Monterey pines are short-lived with an average lifespan lasting 80 or 90 years with rare occurrences of up to 150 years (Piirto & Valkonen, 2005; Roy, 1966). In comparison, Ponderosa pines (*Pinus ponderosa*) range from 27 m - 40 m (90 - 130 ft) and reach ages of 300 to 600 years (Oliver & Ryker, 1990). The needles appear in bright green clusters of three (on the California stands) (Matheson et al., 2006). They are slender and range from 8 – 15 cm. long with a blunt tip. The cones are 7 – 17 cm long and are brown, egg-shaped and usually are attached at an oblique angle on a branch (Jepson & Hickman, 1993; Wu et al., 2021). The cones remain closed until opened by the heat of a forest fire, which abundantly discharges seeds to regenerate onto burned forest floors (Fonda, 2001; Loe, 2010).

Monterey pines grow optimally in medium to coarse textured soils and medium fertility with a pH range of 4.5 – 5.2 (Loe, 2010). The best sites have soils of at least 3 – 4 feet deep and contain sandy loams in texture and are well-drained (Loe, 2010; MacDonald, 1957). The soils in the three California native stands vary, yet still are deep loams derived from marine sediment (Ferchaw et al., 2013; Roy, 1966). At SPR, the rocks are shales and marine sandstones of the Miocene age, but the Año Nuevo Monterey pines grow mainly on the steep and shallow phases of a clay loam (Ferchaw et al., 2013; Roy, 1966).
Monterey pine is found generally on gentle to moderate slopes and has a sea level with a maximum elevation of 1,000 feet and are located up to six miles inland (Roy, 1966). Monterey pines grow in a humid climate maintained by fogs and a seasonal annual precipitation varying from 15 – 35 inches (Roy, 1966; Walker, 2021).

2.1.1. Distribution & Importance

The natural range of Monterey pine (Figure 1) is limited being confined to three locations on the central California coast (the Monterey, Cambria, and Año Nuevo tree stands) and two on Guadalupe and Cedros islands off of Baja California, Mexico (Reynolds et al., 2019; Storer et al., 2001). Although, the native stands are limited, *Pinus radiata* is one of the most widely planted and imported pines in the world due to their economic value (Critchfield & Little, 1927; Estades et al., 2012). Other than the native stands, Monterey pines are planted primarily in Australia, New Zealand, Chile and Europe with over 4 million ha planted (Estades et al., 2012). They are valued economically for their rapid growth and lumber and pulp qualities (Estades et al., 2012).
Figure 1. The Natural Range of *Pinus radiata*
The native stands are considered a valuable and unique asset to the ecosystem rather than a resource. The trees play a role in condensing fog which is central to the greater ecology of the native areas especially the two island stands (Moran et al., 1988; Rogers et al., 2006). The natural and planted populations in California are also home to approximately one hundred species of arthropods with communities in the tree crowns (Ohmart & Voigt, 1981), including the iconic migratory monarch butterflies which have been recorded clustering on native Monterey pines (Griffiths & Villablanca, 2015). The largest native stand is the Monterey stand with 3,500 – 4,500 hectares on and adjacent to the Monterey Peninsula. The Cambria stand is the second largest and it is approximately 1,000 hectares surrounding the town of Cambria. The Año Nuevo stand is the smallest in California and the northern-most stand with less than 600 ha mainly in the Swanton creek area (Loe, 2010; Roy, 1966). Estimates of total stand area are constantly evolving with little information on the two island locations (de Jesus-Reyes et al., 2020; Wise, 2004).

*P. radiata* plays a keystone ecological role in closed-cone California forests (Millar, 1998). The cone morphology of the California closed-cone pines varies, yet all are encountered more frequently in areas where fires are more common, seeing that fire is a part of these pine’s life cycle (Linhart, 1978; Stephens et al., 2004). Monterey pines are essential to closed-cone forests in California, where fires are typical, as a genetic addition to cone variation (Millar, 1986). Another markable quality of *Pinus radiata* may be its extreme variation of tree types (Matheson et al., 2006; Thomson, 1950). Monterey pines genetically have complex patterns of variation within and among the different native populations (Millar, 1986; Rogers, 2004). For example, the cones on the island pines are smaller than those from the mainland population (Rogers, 2004). The genetic diversity within the five populations are considered somewhat unique for an outcrossing coniferous species (Cuautitlán & Izcalli, n.d.; Rogers, 2002). Other than the trees’ ecological importance, the Monterey pine is an icon of the central California coast framing beaches and creating beautiful coastal scenery (Millar, 1986).
2.2. Threats

Monterey pine stands are disturbed by urbanization, fire suppression, and exotic pathogens, such as Western gall rust and pitch canker (Henry, 2005; Loe, 2010). Urbanization in the native stands has made fire suppression an essential management goal, due to the potential threat to human life, structures, and air quality which results in less prescribed fires (Storer et al., 2001). Pathogens cause a small-scale disturbance over short periods of time with the ability to become a disruption at a larger scale as time passes (Gordon et al., 2001). In contrast, fire acts as a large-scale disturbance over a short period of time (Storer et al., 2001). Major changes to prehistoric Monterey pine forests have also been a result of fire regimes, though absence of stand replacing fires allows the tree stands to become overmature and die (Storer et al., 2001; Whitmore, 1989).

2.2.1. Pitch Canker Fungal Disease

Pitch canker (*Fusarium circinatum*) is currently one of the most important pathogens in all five native Monterey pine stands (Blank et al., 2019; Gordon et al., 2001). These stands are the principal host of the fungus in California, yet other pines species and Douglas-fir are also affected (Blank et al., 2019; Wise, 2004). The pathogen was first noted in 1946 in the southeastern United States and later was discovered in Santa Cruz County, California in 1986 (McCain et al., 1987; Wise, 2004). It is unknown whether the pathogen was of long residence in California and only became severe enough to receive notice because of a prolonged drought period that prompted susceptible hosts or if the 1980’s discovery was a recent introduction (Gordon, 2006). Natural occurring regeneration provided by fire often will be sufficient to result in a forest that is less affected by the pitch canker pathogen (A. J. Storer et al., 2001). The California native Monterey pine stands are overmature, largely because of fire suppression and drought, yet are in an area with fire as an ecological function (A. J. Storer et al., 2001; Sugihara et al., 2006). With the absence of fire in Monterey pines, regeneration is decreased, thus, the evolution of a Monterey pine forest in dynamic equilibrium with pitch canker will be delayed (Blank et al., 2019; A. J. Storer et al., 1999, 2001).
The pitch canker pathogen only infects wounds, which can be caused by insect feeding, weather-related damage, or silviculture practices (Gordon, 2006). The pitch canker fungus has been isolated in several beetle species that are commonly associated with Monterey pines, most namely the twig and the cone beetles which carry the pathogen to branch tips and cone whorls, and the engraver beetles which transmit the fungus to the bole of the tree (A. Storer et al., 1994). Many beetles generally feed on weak or fallen trees, but also attack trees suffering stress due to fire, droughts, or diseases (Bezos et al., 2017).

The pathogen can also be passed from parent tree to offspring (Matheson et al., 2006). Furthermore, parent tree resistance is assumed to be conferred to offspring increasing the chance of survival and repopulating native stands (Aegerter & Gordon, 2006; Norville, 2017). An absence of pitch canker symptoms on previously severely affected Monterey pines have been observed, implying that the individual trees had developed a higher resistance to the disease (Aegerter & Gordon, 2006). Through experimentation, initial resistance ratings (susceptible to resistant) are shown to be key factors in survival, with resistant genotypes more likely to survive over time (Norville, 2017).

Within the Año Nuevo native stand, gap plot treatments and site procedures executed at SPR identified predictors of pitch canker infection and the growth of Monterey pine seedlings (Loe, 2010; Norville, 2017; Piirto & Valkonen, 2005; Pinkerton, 2006; Wise, 2004; Yun, 2011). Genetics of the mother trees was found to be a significant predictor to determine pathogen resistance (Norville, 2017). Pile and burn site treatments had higher seedling survival rates than lop and scatter site treatments (Loe, 2010) potentially due to fire’s ecological benefits which are later discussed (Bright et al., 2019). In 2007, survival was driven by site treatment and parent tree genetics. By 2015, stands with higher density strongly influenced a higher survival (Ferchaw et al., 2013; Norville, 2017). The influences on survival imply the importance of studying regeneration after a fire for both native stands and the nonnative commercial industry (Nolan et al., 2021).

2.2.2. Fire Suppression and Drought
For over a century, California has engaged in systematic fire suppression. This, combined with prolonged drought, has affected, and may continue to affect the local vegetation (Mahdizadeh & Russell, 2021). Since the early 20th century, Monterey pine management has included fire suppression, but before that, Native Americans burned portions of the stands much more frequently, usually about every ten years (Storer et al., 2001). This fire suppression resulted in overmature trees, excessive understory vegetation, and dead trees and needles (Storer et al., 2001). Prescribed fires or other similar small-scale disturbances in the Monterey pine life cycle typically provides temporary relief from vegetation competition (Storer et al., 2001). Otherwise, fire and drought combined as a disturbance agent may drive pine forests to permanently shift to shrublands when damage to trees and seedbanks are fatal or replacement resprouting tree species are absent (Connor et al., 2021).

Drought and excessively high temperatures play a critical role in the success of post-fire regeneration, especially in the early stages of seedling establishment, seeing that their survival is very vulnerable to both these factors (Kemp et al., 2019). Moisture gradient varies the resilience of pine forests to stand-replacing fires and characteristically, dry sites are more susceptible to fires (Dodson & Root, 2013). Low root temperatures and soil drying both considerably reduce growth within the trees (Kaufmann, 1977; Nantongo et al., 2021). Monterey pines are more drought tolerant under climatic conditions in their native areas. However, prolonged drought is common to the California coast subjecting *P. radiata* to more severe soil moisture stress (Hankin et al., 2019; Heth & Kramer, 1975).

### 2.3. Fire Regimes in the Monterey Pine Lifecycle

Fire is an integral part of Monterey pine’s ecosystems (Sugihara et al., 2006). These closed-cone pines require fires to reproduce (Piirto & Valkonen, 2005). The high temperatures melt the cone’s waxy seals to release their seeds giving them a selective advantage over other trees (Piirto & Valkonen, 2005; Williamson & Black, 1981). Without fire, cones generally remain attached to trees for many years and usually do not release all their seeds during the first year after maturing.
Cones open when their moisture content is reduced to less than 20 percent but cone opening without fire generally requires blocking of the water supply (Roy, 1966). Seeds are then mainly dispersed by wind and gravity (García-Morote et al., 2017; Stefferud, 1948). Fire opens the cones so that all available seeds are shed on weed-free seedbeds and standing dead trees provide a shade pattern (Perry et al., 2011; Roy, 1966). Monterey pines usually endure both surface fire and crown fires seeing that these fires typically occur in mixed evergreen ecosystems (Sugihara et al., 2006). Pine species can adapt to general patterns of fire by having characteristics, such as thicker bark type, that make them competitive in the presence of recurring fire, yet extremes of fires are not predictable (Pausas, 2015; Sugihara et al., 2006). Fire patterns interact with biotic communities and depend on them to provide fuel, ultimately basing the dynamics of ecosystems on fire regimes (Sugihara et al., 2006).

Both macro- and microevolutionary studies suggest that pines provide the most convincing evidence of species shaped by fire (Pausas, 2015). The California Monterey pines fire regime contains a mixture of low, moderate, and high severity fires (Stephens et al., 2004). High severity fires (e.g., abundant sections of the CZU Lightning Complex) typically kill most trees in the area, allowing for episodic regeneration of the forest (Stephens et al., 2004). Moderate fires cause some overstory tree mortality, but substantial numbers of trees in the larger size-classes survive these events (Stephens et al., 2004) due to their thick bark protecting the inside of the trunk and their nutrients (Mullen, 2017). Monterey pine is a shade intolerant plant (Bustamante et al., 2003; Stephens et al., 2004), and likely will have higher growth in the canopy openings from these fires (Kremer et al., 2021). The clustering of seedling growth from gaps in the canopy, which predictably has an increased seedling mortality from greater fire intensities beneath dense canopies, will be eliminated through moderate fires deducting overstory competition (Curtin et al., 2020). Low severity fires consume ground and surface fuels but cause little overstory mortality in most forests (Stephens et al., 2004). In the absence of overstory mortality, shade-intolerant trees that become established beneath a closed canopy may not
persist and would be absent from the understory vegetation regeneration (Curtin et al., 2020). Despite this, excessive sunlight exposure allows the shade-intolerant understory vegetation to develop, preventing Monterey pine seedling germination and establishment (Loe, 2010).

Monterey pine has been classified as a fire evader primarily because of its canopy stored seed bank that is released after high severity fires (Stephens et al., 2004). In addition, Monterey pine could also be classified as a fire tolerator because it can succeed when exposed to diverse fire severities (Stephens et al., 2004). However, Monterey pines in the past were found to have a reduced germination process with high concentrations of ash (Reyes & Casal, 1998; Stephens et al., 2004). Subsequently, fifty-one percent of the Monterey pine within the Año Nuevo Monterey pine tree stand regenerated within five years of three mixed severity fires when average mean fire return intervals were 11.2 – 20.1 years (Stephens et al., 2004).

2.3.1. CZU Lightning Complex

The CZU Lightning Complex fires began in August 2020 in San Mateo and Santa Cruz counties, CA and were considered high severity fires (Mahdizadeh & Russell, 2021). The complex was caused by lightning strikes and burned 86,509 acres and lasted 38 days, affecting nonnative, invasive, and native species including the Año Nuevo Monterey pine tree stand (CAL FIRE CZU Complex Report, 2020). A northwest wind reaching up to 74 miles an hour and decades of fire suppression led to an increase in highly flammable fuels (“CZU Lightning Complex and Community Science,” 2021). The lightning strikes initially started separate fires near Davenport, CA, the southern area of the fire, and Waddell Creek, which would become the north edge of the fire. A change in wind conditions caused these northern fires to rapidly expand. The CZU fires burned extremely hot in some areas and moderately hot in others (CAL FIRE CZU San Mateo-Santa Cruz., 2021). The occurrence of lightning storms in the dry season, as those that advanced the CZU Lightning Complex Fire, are historically rare but can have a significant impact when they occur (Mahdizadeh & Russell, 2021).
2.3.2. Post-Fire Regeneration and Understory Competition

Post-fire pine recruitment has shown to be significantly higher following lower fire severity as opposed to high fire severity, which could be due to less damage to the seed bank and forest floor conditions (Dodson & Root, 2013; Viana-Soto et al., 2017). With that being said, the same was true for understory competition (Maia et al., 2012). Many communities will remain resilient to changing fire regimes in the short term. However, longer-term changes to vegetation structure, demography and species composition are likely (Nolan et al., 2021). Post-fire ecosystem recovery is dependent upon abundant tree generation which in turn is largely directed by mechanisms linked to seed production, seed dispersal, and seedling establishment and survival (Kemp et al., 2019).

The post-fire natural *Pinus* seedling germination generally has other vegetation competing to colonize a recently burnt area. Stands with high vegetation density demonstrate long-term persistence and may guarantee re-establishment, despite sapling mortality due to competition with shrub canopy (Curtin et al., 2020). In some pine studies in mixed conifer forests, topographical factors influenced post-fire regeneration, such as higher seedling success rates at the higher elevation sites (Dodson & Root, 2013; Kemp et al., 2019). In areas with a fire of high severity, more heterogeneous vegetation was established initially and then decreased over time suggesting high severity fires could lead to vegetation conversion (Mahdizadeh & Russell, 2021; Moradizadeh et al., 2020; Perry et al., 2011).

Understory vegetation plays many critical roles in forest ecosystems (Campbell et al., 2018). Prior research on overstory and understory vegetation effects has found that sites with less understory competition provided more available growing space for Monterey pine seedlings (O’Brien et al., 2007). Shrub cover has a greater effect on seedling occurrence than canopy cover. Additionally, high percentages of shrub cover reduce the probability of seedling presence (O’Brien et al., 2007). Nevertheless, canopy cover still plays an important role for shade intolerant species because they require sunlight to succeed (Bustamante et al., 2003). Within the limited research
on the recent CZU Lightning Complex, understory species recovery in the affected area of the Coast Redwood Forest was observed to be low and was strongly correlated to overstory conditions (Mahdizadeh & Russell, 2021). In addition, competition from understory vegetation for water and nutrients can limit productivity of post-fire young tree growth (Nambiar & Sands, 1993; Wang et al., 2021). It is argued that the key component to pine regeneration is the absence of competition rather than fire (Dawe et al., 2020).
3.1. Study Site

The study site lies in Cal Poly’s SPR at the southern end of the indigenous Año Nuevo stand (Figure 2). This is located on the Central Coast of California in Santa Cruz County. At the Western end of the unit where the ground is moderately level with only a few slopes ranging from 0% to 30+%, Monterey pine grows in domination (Loe, 2010; Norville, 2017; Piirto & Valkonen, 2005). Elsewhere at SPR, Monterey pines grow among coast redwood (*Sequoia sempervirens*, Douglas-fir (*Pseudostuga menziesii*), California nutmeg (*Torreya californica*), and a variety of broadleaf species, most notably coast live oak (*Quercus agrifolia*) and Shreve oak (*Quercus parvula var shrevei*) (Loe, 2010; Piirto & Valkonen, 2005).

Characterized by a Mediterranean climate that is slightly more mesic than the interior of California because of its immediate proximity to the Pacific Ocean, the site receives regular summertime fog, and more winter moisture than California’s central valley. Temperature is mild throughout the year with lows rarely below 4°C/40°F and highs rarely above 26°C/80°F. The wet season lasts from November to April, although like much of California, this area is subject to drought conditions, which have been increasing over the last 20 years (Wuebbles et al., 2017).
Figure 2. Año Nuevo Monterey Pine Stand with Studied Plots
3.1.1. Site Background

In the Fall of 2001, eighteen circular plots (0.05 ha, 0.1 ha and 0.2 ha: six replicates of each) were harvested within the study area (Figure 3). Monterey pine, coast live oak, and Shreve oak were harvested using group selection, and the plots were cleared of brush after harvest (Loe, 2010; Norville, 2017; Wise, 2004). Following the harvest and site preparation, nine plots received a pile and burn slash treatment and nine received a lop-and-scatter slash treatment. Nine untreated plots were established as controls to determine the amount of regeneration that would occur if a “hands off” management approach was continued within the stand; no trees were cut and the site was not prepared for planting (Wise, 2004).

Seedlings were re-visited four times over twenty-one years in 2003, 2004, 2007, 2015, and 2021. In years 2003, 2004, 2007, and 2015 survival, height, and diameter of each seedling were recorded.

Figure 3 represents the locations of the plots studied at SPR. This figure represents the treatment in each plot and the plot size. The field design entails three replications of nine plots which represent three gap-sizes, two site treatments and one control (Wise, 2004).
Figure 3. Monterey Pine plots studied for Management at SPR (Loe, 2010; Norville, 2017)
3.2. Data Collection

This researcher’s original intentions were to revisit the plots and follow through with the inventory collected previously. The CZU Lightning Complex fires burned all the managed plots. Survivability evaluations were carried out in coordination with Kim Corella, Christopher Lee, and Tori Norville of CALFIRE in March 2021. To determine the survival of each tree the following were observed: crown and bole scorch, needle color and shedding, presence of bark beetles, scorch height, assessment of cambium, and bark shedding.

Despite some of the trees still standing, their mortality deemed a pitch canker inventory collection unproductive, and it was decided to switch the study to a post-fire evaluation project.

The creation of a post-fire evaluation conditions assessment was carried out by photographing each plot in all four directions. Dividing the plot into quadrants at the four cardinal directions (e.g., North, South, East, and West) is an analysis tactic derived from Loe (2010). Uniform photos were taken in May 2021, nine months after the fire. Each plot has four photos, and each photo contains the tree closest to the plot’s edge labeled with the direction, the tree’s id, and the plot name. The trees were relabeled in May 2021 as well because many of the prior labels were absent or unclear after the fire. The purpose of the label is for photo clarification as well as visually determining the plot’s edge. This photo log is used to visually assess the differences on a plot-to-plot basis as well as over time. A few plots are absent in this study due to safety concerns making these plots’ locations inaccessible (L-8-2, B-8-1, and the two southernmost plots, B-4-3 and L-8-3). In addition, the control plots from previous research were not in this study because they are not detectable post-fire as they had not been marked in the original study. Thus, this research used the remaining 14 plots. Six were 0.20 ha (B-2-1, B-2-2, B-2-3, L-2-1, L-2-2, L-2-3), five were 0.10 ha (B-4-1, B-4-2, L-4-1, L-4-2, L-4-3) and three were 0.05 ha (B-8-2, B-8-3, L-8-1). The 14 plots with four quadrants each equated to 56 total quadrants. Please note, all the photos used for this research were taken in May 2021, but the photos were taken in two parts on separate days, May 16th, 2021, and May 30th, 2021.
3.3. Data Analysis

Using the photo log, the understory density levels were visually assessed on a plot-to-plot basis. The determination of the understory density was like the methods used in Wise (2004) on the same plots. Each plot has four photos that divided the plots into four quadrants with the outermost tree labeled for reference of the plot edge. Specific vegetation species were not assessed because the purpose of this research was to determine the vegetation density collectively. The data analysis included determining vegetation surface area, vegetation height, and vegetation volume for each plot. The vegetation surface area, height and volume were converted into percentages for a consistent comparison between plots considering that not all the plots are of the same measurements.

The vegetation surface levels were visually assessed in the quadrant photos using grids overlapping the photos. To keep the data uniform, each photo received forty-five grids. This method was derived from Campbell et al., 2018 and is used to analyze an overall vegetative cover for a picture by judging how many boxes (grids), or what portions of each box are covered by vegetation. The grids covered the ground area to the plot edge. In each photo the overlapping grids that contained vegetation rather than leaf and needle litter were determined. If a single grid only showed vegetation in half the grid, this grid was only counted as half and so on for different amounts of vegetation. Then to determine the surface area covered, the number of grids with vegetation (or portions of vegetation) in the photo were added up and then divided by the total amount of grids (45). This resulted in a fraction which was then used to determine the surface area as a percent value. The following table (Table 1) states the total surface areas of each quadrant per ha size. The surface area percentage of the total equaled the surface area for each quadrant or plot as a value ($m^2$).
Table 1. Total Surface Area of Each Plot and Plot Quadrant

<table>
<thead>
<tr>
<th>Plot Description</th>
<th>Surface Area</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total 0.05 ha (Eighth Acre)</td>
<td>505.86 m²</td>
</tr>
<tr>
<td>Total 0.10 ha (Forth Acre)</td>
<td>1011.71 m²</td>
</tr>
<tr>
<td>Total 0.20 ha (Half Acre)</td>
<td>505.86 m²</td>
</tr>
<tr>
<td>0.05 Hectare Quadrant</td>
<td>126.46 m²</td>
</tr>
<tr>
<td>0.10 Hectare Quadrant</td>
<td>252.93 m²</td>
</tr>
<tr>
<td>0.20 Hectare Quadrant</td>
<td>505.86 m²</td>
</tr>
</tbody>
</table>

To find the total vegetation surface area for each plot, all four of the quadrant vegetation surface areas were added together. The surface area percentage signifies the percentage of ground covered by vegetation. This information was calculated using standard functions in an ongoing dataset in Microsoft Excel version 16.54.

The average vegetation height was approximated by observing the height in comparison to the labels on the trees, seeing that the labels on the trees were marked around 1.9812 meters (6.5 feet). This height was chosen to keep uniformity with prior studies (Norville, 2017) and because the tallest person of the research team consistently marked this height. It was also greater than or equal to the average height of the vegetation recorded. The grid overlapping the photo was used to assist in estimating the plants’ heights. The average heights were an approximation using the photos. The percentage of height was also determined by evaluating the average height divided by the maximum height of 1.9812 meters, the label on the tree. For purposes of determining the percentage, the maximum height was necessary to analyze.
the other heights. Although, unlike comparing surface area, the different plot sizes did not require comparisons of height to be in percent form for best accuracy. The percentages were added to the Excel dataset for analysis and comparison. After the average heights and surface cover were determined for each quadrant, the volume of the vegetation was determined using the following formula:

\[
\text{Vegetation Volume (m}^3\text{)} = \text{Surface Area of Vegetative Cover (m}^2\text{)} \times \text{Average Height (m)}
\]

This Vegetation Volume (VV) formula has been used as a method by USGS to measure the amount of three-dimensional vegetative structure at a site without consideration of vegetation type, species, or quality (Wood et al., 2015). USGS defines the VV in a 2-meter by 2-meter station to yield a total volume and an overall site average. Due to limited resources and time, this research only did data collection of the plot’s entirety. Thus, the VV was determined based on an average from the photos. To account for irregular plot sizes, the VV was also converted into a percentage for a more accurate comparison. Since the plots range from three different sizes, the percentage of VV is more precise when comparing plots of unequal sizes rather than using the VV value. The VV as a value was used when analyzing plots of equal sizes, but overall examining the VV as a percentage of plots of unequal sizes was deemed the most accurate. To convert the VV to a percentage, the sum of the estimated volumes was divided by the total available volume (area sampled multiplied by total height) to produce a percentage estimate (Threlfall et al., 2016). The total surface area, height and volume and their percentages was recorded in the dataset for comparisons on a plot-to-plot basis. The information from this data set was added to the ArcGIS plot data from the original studies. The plots absent in this study were not included in the maps.

With this information, the plots’ VV were then compared visually using maps in ArcGIS Pro with the CZU Lightning complex burn severity. The CZU Lightning Complex burn severity data was collected and mapped by CAL FIRE. The understory density levels of the plots were compared with the slope as well. The SPR slope is Cal Poly’s data. Streams, boundaries, and roads in SPR were added to the maps for representation of the area in proximity to the plots. It should be noted that the plots still hold their original names.
Using the information found in ArcGIS Pro, correlation and significance tests were conducted to see if post-fire vegetation growth had a relationship with burn severity and with slope on a plot level. Both the correlation tests and the visual observations found from the maps were used to determine relationships between vegetation density, burn severity and slope.
CHAPTER 4. RESULTS

The following section outlines the main results of this study, which aimed to address the post-fire vegetation based on the CZU Lightning Complex Burn Severity and characteristics of SPR. Within the maps presented, the vegetation volume is in a percentage form for the most precise measurements.

4.1. Vegetation Reproduction

At all the plots, vegetation was identified in every quadrant. Although the individual vegetation species were not specifically collected, a few of the most notable species were California blackberry (*Rubus ursinus*), Western Poison Oak (*Toxicodendron diversilobrum*), milk thistle (*Silybum marianum*), and bull thistle (*Cirsium vulgare*). These species were also noted in prior research (Loe, 2010). There were also multiple Monterey pine seedlings identified. Figure 4 represent seedlings found at L-2-2 in March 2021 and again in May 2021.

![Figure 4. Monterey Pine Seedling found in L-2-2 in March 2021](image-url)
The surface area, height and volume are represented with the measurements in meters as well as in percent form. The following tables display the plot names, the plot size, the calculations for each of the four quadrants per plot and for the total plot. The quadrants are named based on their directions: North (N), South (S), East (E), West (W).

4.1.1. **Vegetation Surface Area**

Table 2 represents the average surface area vegetation covered per quadrant with the total surface area in meters\(^2\). The quadrant percentages are a percentage of a fourth of the plot and the total percentage is a percentage of the entire plot. The total average of all the plots is 933.47 m\(^2\) with an average percentage of 68.45%. All the plots except for L-4-2 and B-8-2 had a total surface area coverage of over 50%. The plots with the highest surface area coverage were B-2-3, L-8-1, and B-8-3; all are over 90%. In the ANOVA test on the averages from each quadrant, \(p = 0.91\), suggesting no significance between the means of the four quadrants, implying the post-fire vegetation surface area did not seem to favor a quadrant direction. The South quadrants had the lowest average surface area of 64.13% and the North quadrants has the highest average of 72.09%. The East quadrants had an average of 70.06% and the West quadrants had an average of 64.13%. The lowest surface level percentage of all the quadrants is found in B-8-2’s South quadrant with 8% coverage. Although none of the plots’ total percentages were 100%, ten different quadrants had a surface area percentage of 100% full coverage. In addition, 24 quadrants were greater than 80%, meaning, over half of the quadrants had over 80% vegetation surface area coverage.
Table 2. Surface Area Covered per Quadrant (m²)

<table>
<thead>
<tr>
<th>Plot</th>
<th>Plot Size</th>
<th>N</th>
<th>E</th>
<th>S</th>
<th>W</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>B-2-1</td>
<td>0.20 ha</td>
<td>452.0</td>
<td>89.35%</td>
<td>308.7</td>
<td>61.03%</td>
<td>227.1</td>
</tr>
<tr>
<td>B-2-2</td>
<td>0.20 ha</td>
<td>505.9</td>
<td>100%</td>
<td>315.1</td>
<td>62.29%</td>
<td>486.9</td>
</tr>
<tr>
<td>B-2-3</td>
<td>0.20 ha</td>
<td>456.9</td>
<td>90.31%</td>
<td>505.9</td>
<td>100%</td>
<td>396.3</td>
</tr>
<tr>
<td>B-4-1</td>
<td>0.10 ha</td>
<td>98.6</td>
<td>39.00%</td>
<td>185.0</td>
<td>73.13%</td>
<td>114.3</td>
</tr>
<tr>
<td>B-4-2</td>
<td>0.10 ha</td>
<td>252.9</td>
<td>100%</td>
<td>252.9</td>
<td>100%</td>
<td>252.9</td>
</tr>
<tr>
<td>B-8-2</td>
<td>0.05 ha</td>
<td>40.6</td>
<td>32.11%</td>
<td>34.8</td>
<td>27.50%</td>
<td>10.1</td>
</tr>
<tr>
<td>B-8-3</td>
<td>0.05 ha</td>
<td>126.5</td>
<td>100%</td>
<td>126.5</td>
<td>100%</td>
<td>126.5</td>
</tr>
<tr>
<td>L-2-1</td>
<td>0.20 ha</td>
<td>440.5</td>
<td>87.08%</td>
<td>275.1</td>
<td>54.38%</td>
<td>246.6</td>
</tr>
<tr>
<td>L-2-2</td>
<td>0.20 ha</td>
<td>111.3</td>
<td>22.00%</td>
<td>354.1</td>
<td>70.00%</td>
<td>480.6</td>
</tr>
<tr>
<td>L-2-3</td>
<td>0.20 ha</td>
<td>470.4</td>
<td>93.00%</td>
<td>303.5</td>
<td>60.00%</td>
<td>373.1</td>
</tr>
<tr>
<td>L-4-1</td>
<td>0.10 ha</td>
<td>188.3</td>
<td>74.44%</td>
<td>224.0</td>
<td>88.57%</td>
<td>191.8</td>
</tr>
<tr>
<td>L-4-2</td>
<td>0.10 ha</td>
<td>40.5</td>
<td>16.00%</td>
<td>60.7</td>
<td>24.00%</td>
<td>55.2</td>
</tr>
<tr>
<td>L-4-3</td>
<td>0.10 ha</td>
<td>192.2</td>
<td>76.00%</td>
<td>151.8</td>
<td>60.00%</td>
<td>50.6</td>
</tr>
<tr>
<td>L-8-1</td>
<td>0.05 ha</td>
<td>113.8</td>
<td>90.00%</td>
<td>126.5</td>
<td>100%</td>
<td>113.8</td>
</tr>
</tbody>
</table>
4.1.2. Vegetation Height

Though most plots had a high percentage of surface area coverage, only half of the quadrants had vegetation growth over 0.5 meters (1.64 feet). Table 3 represents the average vegetation height in each quadrant and in each plot. The average height for all the plots was approximately 0.68 m (2.24 ft). While the South quadrants had the lowest surface level average (68.71%), these plots had the highest height average of 0.73 m. The South quadrant of B-8-3 was noted to have the tallest average of vegetation growth and the only quadrant with 100% height. For clarification, a few individual plants were above the maximum height of 1.9812 meters. However, not enough plants were tall enough to change the average. The average height for the North quadrants is 0.71 m and the average for the West quadrants is 0.65 m. The East quadrants have the lowest average height at 0.64 m. The ANOVA test on the height averages across quadrants suggests no significance ($p = 0.96$) across four different quadrants.

Some plots had high percentages of both surface area coverage and height. These plots include B-2-1, B-2-3, B-4-2, B-8-3, L-2-3, and L-8-1. Contrarily, other plots had high surface area percentages, yet their average heights are not even over 0.4 m. This can be seen in plots B-2-2, L-2-2, and L-4-2. The correlation test of the relationship between the plots’ surface area and height is $r = 0.626$ and implies a moderately positive relationship, where a larger surface area coverage correlates to a larger average height. Figure 5 demonstrate two 0.20 ha plots with similar surface area coverages but different heights. While the North quadrant of B-2-2 as seen in Figure 5, has full surface area coverage, this plot only had an average height of 0.34 m. Conversely, the North quadrant of B-2-3 has a high surface area as well, but this plot also had an average height of 1.41 m. Thus, differences in density like these imply the importance of comparing vegetation volume by plot.
Figure 5. B-2-2 North Quadrant Demonstrating High Surface Area Coverage and Low Height
Table 3. Average Vegetation Height per Quadrant (m)

<table>
<thead>
<tr>
<th>Plot</th>
<th>Plot Size</th>
<th>N</th>
<th>E</th>
<th>S</th>
<th>W</th>
<th>Total Avg (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>B-2-1</td>
<td>0.20 ha</td>
<td>0.91</td>
<td>46.15%</td>
<td>0.61</td>
<td>30.77%</td>
<td>0.76</td>
</tr>
<tr>
<td>B-2-2</td>
<td>0.20 ha</td>
<td>0.46</td>
<td>23.08%</td>
<td>0.30</td>
<td>15.38%</td>
<td>0.30</td>
</tr>
<tr>
<td>B-2-3</td>
<td>0.20 ha</td>
<td>1.52</td>
<td>76.92%</td>
<td>1.52</td>
<td>76.92%</td>
<td>1.37</td>
</tr>
<tr>
<td>B-4-1</td>
<td>0.10 ha</td>
<td>0.15</td>
<td>7.69%</td>
<td>0.30</td>
<td>15.38%</td>
<td>0.76</td>
</tr>
<tr>
<td>B-4-2</td>
<td>0.10 ha</td>
<td>0.91</td>
<td>46.15%</td>
<td>0.61</td>
<td>30.77%</td>
<td>1.37</td>
</tr>
<tr>
<td>B-8-2</td>
<td>0.05 ha</td>
<td>0.30</td>
<td>15.38%</td>
<td>0.30</td>
<td>15.38%</td>
<td>0.30</td>
</tr>
<tr>
<td>B-8-3</td>
<td>0.05 ha</td>
<td>1.83</td>
<td>92.31%</td>
<td>1.83</td>
<td>92.31%</td>
<td>1.98</td>
</tr>
<tr>
<td>L-2-1</td>
<td>0.20 ha</td>
<td>0.91</td>
<td>46.15%</td>
<td>0.30</td>
<td>15.38%</td>
<td>0.61</td>
</tr>
<tr>
<td>L-2-2</td>
<td>0.20 ha</td>
<td>0.30</td>
<td>15.38%</td>
<td>0.30</td>
<td>15.38%</td>
<td>0.30</td>
</tr>
<tr>
<td>L-2-3</td>
<td>0.20 ha</td>
<td>0.91</td>
<td>46.15%</td>
<td>0.61</td>
<td>30.77%</td>
<td>0.61</td>
</tr>
<tr>
<td>L-4-1</td>
<td>0.10 ha</td>
<td>0.30</td>
<td>15.38%</td>
<td>0.76</td>
<td>38.46%</td>
<td>0.30</td>
</tr>
<tr>
<td>L-4-2</td>
<td>0.10 ha</td>
<td>0.30</td>
<td>15.38%</td>
<td>0.30</td>
<td>15.38%</td>
<td>0.30</td>
</tr>
<tr>
<td>L-4-3</td>
<td>0.10 ha</td>
<td>0.30</td>
<td>15.38%</td>
<td>0.30</td>
<td>15.38%</td>
<td>0.30</td>
</tr>
<tr>
<td>L-8-1</td>
<td>0.05 ha</td>
<td>0.76</td>
<td>38.46%</td>
<td>0.91</td>
<td>46.15%</td>
<td>0.91</td>
</tr>
</tbody>
</table>
4.1.3. Vegetation Volume

Table 4 represents the vegetation volume for each plot and each quadrant within the plot. The average volume across all plots is 692.746 m³ and the average percentage is 26.8%. The North quadrants had the highest VV percentage at 30.49%. The East, South and West quadrants had very similar percentages which are listed as follows: East had 26.76%; South had 27.58%; West had 26.34%. The ANOVA test for the averages across quadrants implies no statistical significance (p = 0.98) across all four quadrants, suggesting that there is not a preferential direction the vegetation favors. For visualization, Figure 6 below compares plots B-8-2 and B-8-3 as an example. Both are in the same proximity yet have a much different vegetation growth. B-8-3 West Quadrant measured a 14.62% vegetation volume while B-8-2 measured an 84% vegetation volume. One important observation is the differences in VV across the three different plot sizes. The plots of 0.05 ha have the largest percentage of vegetative reproduction by a considerable amount at 45.78%. The 0.10 ha plots have a VV percentage of 15.89% and the 0.20 ha plots have a VV percentage of 26.4%. However, these results do not allow for evidence of reproduction preferring smaller plots over larger plots.

Figure 6. B-8-2 West Quadrant and B-8-3 West Quadrant
<table>
<thead>
<tr>
<th>Plot</th>
<th>Plot Size</th>
<th>N</th>
<th>E</th>
<th>S</th>
<th>W</th>
<th>Total</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td>B-2-1</td>
<td>0.20 ha</td>
<td>413.29</td>
<td>41.24%</td>
<td>188.18</td>
<td>173.04</td>
<td>66.23</td>
<td>840.75</td>
</tr>
<tr>
<td>B-2-2</td>
<td>0.20 ha</td>
<td>231.28</td>
<td>23.08%</td>
<td>96.045</td>
<td>148.4</td>
<td>147.18</td>
<td>622.9</td>
</tr>
<tr>
<td>B-2-3</td>
<td>0.20 ha</td>
<td>696.24</td>
<td>69.47%</td>
<td>770.93</td>
<td>543.5</td>
<td>616.74</td>
<td>2627.4</td>
</tr>
<tr>
<td>B-4-1</td>
<td>0.10 ha</td>
<td>15.033</td>
<td>3.00%</td>
<td>56.374</td>
<td>87.131</td>
<td>216.24</td>
<td>374.78</td>
</tr>
<tr>
<td>B-4-2</td>
<td>0.10 ha</td>
<td>231.28</td>
<td>46.15%</td>
<td>154.19</td>
<td>346.92</td>
<td>31.223</td>
<td>763.6</td>
</tr>
<tr>
<td>B-8-2</td>
<td>0.05 ha</td>
<td>12.378</td>
<td>4.94%</td>
<td>10.6</td>
<td>3.0837</td>
<td>36.619</td>
<td>62.681</td>
</tr>
<tr>
<td>B-8-3</td>
<td>0.05 ha</td>
<td>231.28</td>
<td>92.31%</td>
<td>231.28</td>
<td>250.55</td>
<td>100%</td>
<td>923.57</td>
</tr>
<tr>
<td>L-2-1</td>
<td>0.20 ha</td>
<td>402.81</td>
<td>40.19%</td>
<td>83.838</td>
<td>150.33</td>
<td>87.372</td>
<td>724.35</td>
</tr>
<tr>
<td>L-2-2</td>
<td>0.20 ha</td>
<td>33.921</td>
<td>3.38%</td>
<td>107.93</td>
<td>146.48</td>
<td>37.261</td>
<td>325.59</td>
</tr>
<tr>
<td>L-2-3</td>
<td>0.20 ha</td>
<td>430.18</td>
<td>42.92%</td>
<td>185.02</td>
<td>227.42</td>
<td>593.61</td>
<td>1436.2</td>
</tr>
<tr>
<td>L-4-1</td>
<td>0.10 ha</td>
<td>57.391</td>
<td>11.45%</td>
<td>170.71</td>
<td>58.462</td>
<td>59.104</td>
<td>345.66</td>
</tr>
<tr>
<td>L-4-2</td>
<td>0.10 ha</td>
<td>12.335</td>
<td>2.46%</td>
<td>18.502</td>
<td>16.82</td>
<td>10.793</td>
<td>58.45</td>
</tr>
<tr>
<td>L-4-3</td>
<td>0.10 ha</td>
<td>58.59</td>
<td>11.69%</td>
<td>46.256</td>
<td>15.419</td>
<td>69.383</td>
<td>189.65</td>
</tr>
<tr>
<td>L-8-1</td>
<td>0.05 ha</td>
<td>86.729</td>
<td>34.62%</td>
<td>115.64</td>
<td>104.08</td>
<td>96.366</td>
<td>402.81</td>
</tr>
</tbody>
</table>
Lastly, vegetation growth between March 2021 and May 2021 had noticeably increased. The purposes of this research were to observe the post-fire conditions in May 2021, but this is an observation worth emphasizing. The photos taken in March 2021 were not uniform and consistent across plots. Nonetheless, the photos help visually describe a plot’s difference from approximately seven months post-fire to nine months. Figure 7 can be used as an example of the obvious difference with just two months on plot L-4-1. The photo on the left is in the close vicinity of the North quadrant so the two photos are in nearly identical areas. L-4-1’s North quadrant has a low VV percentage at 11.45% (May 2021) while the VV in the left photo (March 2021) is very scarce and almost nonexistent.

Figure 7. L-4-1 Seven Months (Left) and Nine Months (Right) Post-Fire
4.2. CZU Lightning Complex

The CZU Lightning Complex in August of 2020 severely affected SPR and the surrounding areas. Figure 8 represents the CZU Lightning Complex’s final burn severity and the location of SPR within the fire. SPR was in the Southern region of the fire.

As seen from Figure 9, Swanton endured high, medium, and low burn severity. For this research, burn severity is broken down into five categories: low, low-moderate, moderate, moderate-high, and high. The CZU Lightning Complex appears to hold a higher burn severity in the center of the fire and a low burn severity around the edges.

Figure 9 represents each plot’s post fire vegetation volume based on their shades of green with a darker green signifying a plot with more dense vegetation and a lighter green or white indicating less dense vegetation. Based off observations from this map, post-fire vegetation is moderately denser in areas with a lower burn severity. This can be seen when comparing plots B-8-2 and B-8-3 which are both 0.05 ha. B-8-2 has a much lower vegetation volume than B-8-3 and B-8-3 is in an area with a lower burn severity than B-8-2. The most notable differences are between B-2-3 and L-2-2 which are the same size (0.20 ha), and both located slightly more South. B-2-3 sits on the outside of the area with the highest burn severity and has a very dense vegetation volume. L-2-2 has a low amount of vegetation volume, and the location of this plot is in an area with a high burn severity. Overall, the relationship between the vegetation volume and the burn severity is moderately negative where \( r = -0.68 \) in a Pearson correlation. This correlation is statistically significant with \( p = 0.008 \), implying a relative correlation between higher vegetation density and lower levels of burn severity. The burn severity of SPR as an entirety is revealed in Figure 10.
Figure 8. CZU Lightning Complex Burn Severity
Figure 9. CZU Lightning Complex Burn Severity and Plots in SPR
Figure 10. Burn Severity of Swanton Pacific Ranch
In comparison to other adjacent areas of the plots, the plots were affected by a high burn severity as demonstrated in Figure 10. In addition, the high levels of burn severity appear in a pattern following the plots. The high burn severity encompasses the entire area of the plots then is lower in the immediate surrounding parts. The burn severity is shown to decrease in regions closer to the ocean. Lastly, Figure 10 shows an area with lower burn severity near a region with a high elevation, yet in the region where the plots are located, this is the opposite.

4.3. Swanton Pacific Ranch Topography Characteristics

The regions near the plots range from elevations of approximately 95 m – 140 m. The plots overall are oriented on north-facing slopes. This area contains slopes of mixed percentages fluctuating from 0% to > 50% which can be seen in the following image (Figure 11).

Figure 11 represents the plot’s vegetation volume as a percentage in relation to the slope of SPR. The plots with the highest slope are L-8-1, L-2-1, B-2-1, L-4-1, L-2-3, and B-4-1. The plot with the single highest slope is L-8-1. Observations from the map do not connect slope with vegetation and the results of a Pearson correlation test suggest little association between post-fire vegetation density and slope with $r = 0.26$, a weak and positive relationship. The correlation is not statistically significant ($p = 0.37$), implying no significance between vegetation and slope. In pre-CZU Lightning Complex research, these plots were originally positioned intentionally in areas with a slope less than 35% for easier access (Loe, 2010; Norville, 2017; Wise, 2004). The data used for defining the relationship between slope and vegetation volume would not show extreme results because the plots are in regions consisting of only gentle to moderate slopes. Furthermore, the slope of all the plots is relatively alike with an average of 16%, and a standard deviation of 1.65, thus, not allowing for much comparison.
Figure 11. Slope of Swanton Pacific Ranch with Plots
The plots overall are located close to branches of the Scotts Creek watershed. These streams are approximately 0.2 kilometers from the southern plots and surround them on either side. Not all the northern plots are surrounded by streams like the southern plots. However, they still are located close to Scott’s creek and L-4-2, a northern plot, is the closest plot to a stream. The streams also divide through the highest percentages of slope. The plots could not be directly adjacent to a stream because there is too extreme of a slope.

The elevation in this region of SPR slightly varies but moderately increases more towards the southern plots. The northern plots are adjoining to the SPR boundary, and the southern plots push toward the center. Although Figure 11 does not demonstrate other parts of SPR, it should be noted, approximately 0.5 km south and inward of the southern plots, the elevation increases more drastically with levels of 230 m – 250 m. Figure 12 represents the slope of SPR and the difference in elevation. As demonstrated in this map, there is less of a slope directly below the plots. Also, a continuous large, sloped area borders Scott’s creek along the plots. Lastly, SPR has portions with major slope percentages above 50%, yet the overall boundary contains a consistent slope.
Figure 12. Slope of Swanton Pacific Ranch
Fire frequency and severity are projected to increase due to human induced climate change in California and in Mediterranean systems resulting in significant shifts in vegetation patterns (Lenihan et al., 2008; Mahdizadeh & Russell, 2021; Moriondo et al., 2006). The Año Nuevo Monterey pines in the past had characteristics of a fire tolerator because it can succeed when exposed to diverse fire severities (Stephens et al., 2004). However, the recent CZU Lightning Complex fires of unprecedented magnitudes resulted in full mortality of Año Nuevo Monterey pines and have presented opportunities to study response of this forest’s regeneration under extreme fire conditions. The results of this study highlight the post-fire vegetative regeneration considering both burn severity and topography.

Nine months had passed between the CZU Lightning Complex and the data collection. During this period, vegetative regeneration occurred. Analysis of vegetation recovery show a moderately negative and significant relationship between burn severity and regeneration levels ($r = -.68$, $p = .008$), which are lower when the burn severity is higher. Only two plots had a vegetation volume over 50% (B-8-3 and B-2-3) and these two plots were in an area of moderate to low burn severity. These two plots were also near each other. The plots that had a vegetation volume of under 10% (B-8-2, L-2-2, L-4-2, and L-4-3) are positioned in a much more burn intensive area of the fire. These findings support previous research that suggests short-term regeneration was slowed down when severity was higher in a Pinus forest of similar climate (Viana-Soto et al., 2017). Native biota in systems characterized by natural disturbances typically possess functional traits of adaptation to regrow and establish following fires, even when the fires create large patches burned with high severity (Donato et al., 2009; Lentile et al., 2007; Viana-Soto et al., 2017).

Burn severity results from this research suggest consequences on the vegetation despite natural recovery abilities and establishment. These results could be because fire recovery rates strongly depend on the predominant type of vegetation (Gouveia et al., 2010). Post-fire vegetation recovery has shown longer recovery times for regions dominated by coniferous forests (Gouveia et al., 2010). Inside the boundaries of SPR, coniferous trees are balanced with deciduous trees, but in the Western part, where the plots are, the coniferous Monterey pines grow in domination (Loe, 2010; Piirto &
Valkonen, 2005). While Monterey pines are a fast-growing tree, the analyzed post-fire vegetation was
dominating, which could likely change in a long-term study. Few major species found in post-CZU
Lightning Complex vegetation were also found in the plots in previous studies (Ferchaw et al., 2013; Loe,

The implications of the correlation between a low vegetation volume and high burn severity could trace
back to research linking burn severity to soil hydraulic properties (Moody et al., 2015). Ashes deposited in
the soil surface after a fire provide nutrients that are available to the plants (Ferreira et al., 2005) and has
previously promoted Monterey pine growth (Jordan & Rodriguez, 2004). High burn severity generally
indicates increased exposure of ash (Lutes et al., 2006; Robichaud et al., 2007), but losses of nutrients by
ash convection increase with fire intensity (Boerner, 2006; Meneses, 2021). The ash in the plots with
moderate or moderate-high burn severity may hold more nutrients than the ash in the plots with high
burn severity, thus explaining the results obtained. In contrast, a previous study supports conifer
stands, such as the studied area of the Año Nuevo Forest, showed ash and charred organic
matter generated by the combustion of vegetation and containing abundant available nutrients were
unevenly distributed over the post-fire landscape (Kokaly et al., 2007). Kokaly et al. (2007) also revealed
scorched conifer trees, which retained dry needles heated by the fire but not fully combusted by the
flames, covered much of the post-fire landscape. An unequal distribution of post-fire nutrients in
addition to Monterey pines’ needles not fully combusted may have effects of the results of this
study, disallowing the relationship between burn severity and post-fire regeneration to have a
stronger correlation ($r = -.68, p = .008$).

With regards to the influence of slope, a positive but weak relationship between regeneration and slope
of an area was found ($r = .25$) with no significance ($p = 0.37$). As previously mentioned, none of the plots’
establishments were in an area with a slope of more than thirty-five percent (Loe, 2010; Norville, 2017;
Wise, 2004). Within plant establishment, slope is an important physiographic factor (Mataji et al.,
2010). In Mediterranean climates, high slopes present higher regeneration rates due to less
evapotranspiration and higher humidity content (Viana-Soto et al., 2017). The studied plots
were established for assessment of Monterey pines (Ferchaw et al., 2013; Loe, 2010; Wise, 2004) and Monterey pine’s root strength heavily relies on slope stability (O’loughlin & Watson, 1979). Thus, the slope of this area favors the Monterey pine reestablishment rather than the vegetation establishment as an entirety. Perhaps a stronger correlation could be detected in areas with more variation of slope.

Although, the plots are not located in areas above a thirty-five percent slope, large portions of the areas adjacent to the plots are above a thirty-five percent slope. In addition, this area appears to have suffered a more moderate to low burn severity, especially compared to the burn severity of the plots. The maps allow a visualization of the surrounding areas near the plots, which appear to have areas of high slope and lower burn severity. However, this is only merely an observation and statistical analysis was not conducted to support these observations. Prior research identified fires burning more severely in areas located on flatter slopes supporting the observation that slope and burn severity have a negative relationship (Cocke et al., 2005; Stevens-Rumann et al., 2016). In contrast, elevation appears to be a more significant predictor of burn severity (Stevens-Rumann et al., 2016).

Elevation has previously been used to predict burn severity (Prichard & Kennedy, 2014; Stevens-Rumann et al., 2016). Figure 10 displays the burn severity across SPR with a low burn severity on an area with a higher elevation. Elevation was not used as a variable in this study seeing that the elevation across the plots only fluctuates by approximately 30 m. Similarly to slope, the plot locations were intentionally chosen to keep consistency across elevations (Wise, 2004). With regards to vegetation reproduction, other research found higher elevation decreases vegetation growth (Donato et al., 2009; Haffey et al., 2018; Hankin et al., 2019), although the elevation throughout the area of the Monterey pines plots does not differentiate vastly like that of the previous research. Contrarily to the past studies, the uniformity of the elevation across the plots is likely causing elevation to play a minor role in vegetation reproduction.

Assessment of recovery rates and topographic features allude aspect influenced the abundance of vegetation growth overall. The plots are mostly north-facing (Stephens et al., 2004) and despite the slope and vegetation recovery results containing no statistical significance ($p = 0.37$), the robust recovery rates are inclusive of all plots. Many past studies indicating revegetation recovery is greater in north-
facing slopes (Chen et al., 2019; Evangelides & Nobajas, 2020; Kong et al., 2019; Viana-Soto et al., 2017). This is because North-facing slopes usually have higher moisture contents, hence increasing plant cover, soil organic matter contents, soil structural stability, and resistance against water erosion (del Pino & Ruiz-Gallardo, 2015). Plausibly, the plot orientation contributed to vegetation growth.

Although we were unable to determine significant connections between slope and post-fire vegetation growth, the connections between slope and burn severity in areas adjacent to the plots are noteworthy. The burn severity was higher in the plots compared to the areas directly bordering the plots (Figure 10). Additionally, in the areas adjacent to the plots, the slope is above 35% with large quantities of area near 50% (Figure 12). The original goal of this research was to analyze the relationship between slope and vegetation reproduction, yet the observed overlap between slope and burn severity is compelling.

Our study indicated a relationship between higher levels of burn severity and lower levels of post-fire vegetation in the Año Nuevo Monterey pine stand from the Pearson correlation test results with $r = -0.67$ and $p = 0.008$. Given the evidence of high burn severity levels influencing revegetation, whether the post-fire conditions affecting regrowth include soil hydraulic properties, vegetation species domination, or biomass loss is unknown in this research. Similarly, whether the novel biotic conditions, including tree size and pathogens, known to influence burn severity (Cocke et al., 2005) influenced the vegetative establishment per plot is unknown because the most recent data on pre-CZU Lightning Complex conditions is from 2015 (Norville, 2017). Nonetheless, all the plots contained vegetation.

The post-fire recovery of understory cover had an average vegetation volume was 26.8%, which is somewhat low. Though cover and richness of many plots were reduced to conditions created by the fire, it was encouraging that the plots contained an average surface coverage of 68.45% and an average height of 0.68 m. Despite the scarcity of vegetation at some plots, they still showed post-fire regeneration. The appearance of vegetation at all plots may be due the ecological benefits from forest fires including the removal of accumulated fuels, an increase in water yield, the control of insects and diseases, the preparation of seedbeds, and the release of seeds from serotinous cones (Kozlowski, 2002; Moradizadeh et al., 2020; Stephens et al., 2004).
5.1. Limitations and Future Research

The post-hoc nature of this study unfortunately limits the ability to report the plot’s ecosystems directly before the fire, therefore calculating the loss following this fire is challenging (Mahdizadeh & Russell, 2021). This research only calculated the plot’s vegetation levels nine months after the fire (May 2021). However, photos from March 2021, seven months after the fire, in comparison to the May photos indicated large quantities of growth in both height and surface area (Figure 9). To better assess the regeneration dynamics, a periodic evaluation should advance. Previous research has used similar methods to observe post-fire regeneration with biannual observations over a five-year period (Viana-Soto et al., 2017). A longitudinal study on the vegetation rates in the Monterey pine plots could aid in future research on fire regimes in California Monterey pines (Stephens et al., 2004) and species resistance to fires of this magnitude (Enright et al., 2014). Additionally, observing the plots long-term would provide information on the patterns of the post-fire understory vegetation.

Limitations of this study included time constraints which affected the saturation of the results. A more extensive study should observe the patterns of the post-fire vegetation results over a longer period.

Further research examining the Año Nuevo Monterey pines should use this research as aid for determining success of the pines. Plots that were observed to have higher density of competition species may damage the development of the Monterey pines (Calvo et al., 2008). Potentially destructive in-field measurements should proceed with caution due to the fragile and delicate state of the Monterey pine seedlings (Salas-Aguilar et al., 2017). While the results of this study showed reproduction of Monterey pine seedlings, longer term implications for resprouting species may become threatened (Enright et al., 2014). In addition, changes caused by competitive interactions may lead to transformations in the density of *Pinus radiata* reproduction success. Further studies should also observe pitch canker severity in the regeneration of the Monterey pines to evaluate success. Escalations in climate-change driven fires of high intensity and frequency could eventually inhibit the recovery of even the most resilient species, overall leading to vegetation type conversion (Mahdizadeh & Russell, 2021).
implying importance of continuing this research to further monitor the post-fire ecosystem dynamics of
the CZU Lightning Complex.

It will likely be many years before the influence of the CZU Lightning Complex on the forest community
will be fully understood, and the specter of climate change and continuing drought may stimulate
additional fires that will further influence successional patterns. The nine-month post-fire results of
the vegetation in the Año Nuevo Monterey pine tree stand research suggest the CZU
Lightning Complex burn severity impacts regeneration, where high burn severity reduced growth while
slope is not a prominent factor for the vegetation recovery.

5.2. Conclusion

Climate driven increases in fire frequency and severity are predicted for the coast of
California, emphasizing the importance to study the post-fire recovery patterns and
dynamics. The recent high severity CZU Lightning Complex affected a variety of vegetation types,
including the Año Nuevo Monterey pine (*Pinus radiata*) stand and the vegetation that occupied it.
The purpose of this study is to provide analysis of the short-term impact of the fire’s burn severity
and topographical variables on the vegetation regeneration.

We sampled an area in Cal Poly’s SPR which contained plots, that were previously used to study
pitch canker infections in Monterey pines. The fatal damage from the CZU Lightning Complex
shifted the research to observe the post-fire vegetation rates. The vegetation was measured
digitally using a photo log and the average vegetation surface level, height, and volume was
recorded and added to maps for visual assessments. Correlation and significance tests
conducted for the relationship between vegetation regeneration and burn severity and vegetation
reproduction and slope.

Regeneration measurement for the 14 examined plots has been related to the burn severity with
good results (*r* = -0.68). Burn severity degree measured visually and through correlation tests
indicates that post-fire vegetation is less dense when burn severity was higher (*p* =
0.008). In contrast, the slope of the plots had a low correlation with vegetation density ($r = 0.26$), with results suggesting no significance slope and regeneration ($p = 0.37$).

Observed recovery of the understory vegetation following this extreme fire included evidence of *Pinus radiata* growth suggesting potential future recovery of this stand. Although, in some plots with high levels of vegetation density of other species, the competitive interactions may prevent the further success of the young seedlings. This research aims to understand the dynamic between post-fire regeneration with burn severity and slope, but it also aims to provide more of a context into the effects of the recent CZU Lightning Complex. Moreover, the results obtained are useful in improving knowledge about the factors which determine the post-CZU Lightning Complex regeneration patterns of the Año Nuevo Forest and the native Monterey pine ecosystem. Therefore, these advances could help forest planners in understanding the vegetation patterns after this fire and which areas Monterey pines will be naturally successful and thus require the implementation of restoration programs.


*CAL FIRE CZU San Mateo-Santa Cruz.* (2021, September 27). [Youtube].


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