AN ABSTRACT OF THE THESIS OF

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Title: Douglas-fir Reforestation After Wildfire in the Pacific Northwest of the U.S.A.

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The main goal of this study is to improve our understanding of the challenges associated with reforestation after a fire. This includes assessing the performance of Douglas-fir seedlings and early-seral vegetation dynamics under unburned and burned stands, and the impact of stand age/structure on post-fire conditions. The specific objectives included: (1) determine the effect of wildfire on soil physical and chemical attributes; (2) determine the effect of wildfire microclimate conditions; (3) determine the effect of pre-wildfire stand age/structure on post-fire early-seral vegetation community dynamics; (4) determine the interactive effect of pre-wildfire stand age/structure and Forest Vegetation Management (VM) on soil moisture dynamics; and (5) determine the effect of delayed planting in the effectiveness of VM and Douglas-fir growth and survival.

The study site consisted of four stands with different pre-fire stand structures; (UB) unburned (harvested in fall 2020); (B1y) burned when the stand was 1 year-old; (B12y) burned when the stand was 12 years-old; (B55y) burned when the stand was >55 years-old. At each site, six treatments with different combinations of planting year (right after fire or delayed 1 year) and VM regime (pre and post planting herbicide treatments) were applied using a complete randomized block design with

four replications of each treatment. In each plot, 6 rows of 6 seedlings were planted at 10x10ft spacing. Within each plot, 5 vegetation survey points were installed to monitor vegetation abundance and species richness. In addition, soil moisture measurements were taken 5 times during the 2021 and 2022 growing seasons at each vegetation survey point. A weather station was installed in three of the selected stands (UB, B1y and B12y) to record air temperature, relative humidity, rainfall, wind speed and solar radiation.

Soil organic matter was reduced in all burned stands, which reduced the water holding capacity of the soil. Furthermore, there was also a decrease in cation exchange capacity, calcium, and magnesium at all burned stands. However, nitrate and phosphorus levels were higher in all burned stands relative to the UB stand. Additionally, the presence of standing dead trees created an unfavorable microclimate condition impeding the cooling effect of the wind. The lower wind speed resulted in higher temperature and evaporative demand in the B12y stand.

When applied during the first year after the wildfire (2021), there was no effect of spring release treatment on early-seral vegetation cover at all burned sites, having only effect on the UB stand. During the first year after the fire (2021), shrub cover in the UB site was higher than in all burned stands, while ferns cover in the B12y was higher than in UB. In addition, on burned stands, there was no effect of spring release treatment effect on soil moisture during the first year (2021) after fire.

During the second year after the wildfire (2022), there were significant site and VM treatment effects on the cover of early-seral vegetation at all sites (burned and unburned). There was a positive effect of VM treatments on soil moisture dynamics during the second year after wildfire.

In terms of delayed planting effect on the effectiveness of VM and Douglas-fir performance, seedlings planted during the second winter after the fire (2022) were bigger than seedlings planted during winter immediately after fire. This response may be the effect of the heat dome episode of June 2021. There was a stronger effect of VM treatments on plots where planting was delayed (higher reduction on vegetation cover and higher soil moisture availability).

We found a positive correlation between stem volume and nitrate and phosphorus in soil for 1-year-old Douglas-fir seedlings. On the other hand, there was a strong negative correlation between woody vegetation cover and survival of 1-year-old seedlings at untreated plots. For every 10% increase in woody vegetation cover, there was a decrease of 39 TPA in survival.

Overall, wildfire represents a challenge to reforestation. Our findings highlight the complex interactions between wildlife, soil attributes, early-seral vegetation dynamics, and reforestation practices. They emphasize the importance of considering site-specific factors and management strategies when undertaking reforestation efforts in burned sites.

Douglas-fir Reforestation After Wildfire in the Pacific Northwest of the U.S.A

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Sefa Karabas, Author

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1 Literature Review: Evaluating the Impact of Human-induced Climate Change on Forest Wildfires

1.1 Introduction

Wildfires are a common disturbance in dry conifer forests (Walstad et al., 1990). The western region of the United States has seen a rise in forest fires in recent decades due to various factors, including past fire suppression practices, climate variability, and the effects of human-caused climate change (Abatzoglou & Williams, 2016). The year 2020 was particularly devastating for wildfires in the western United States as over 2.5 million hectares of land were burned, causing widespread damage to property and infrastructure and loss of life. In California, the fires were particularly severe, with over 1.5 million hectares burned. These fires were among the largest ever recorded in the state's history and had a significant impact on communities and ecosystems (Abatzouglu, 2020). Massive wildfires can sweep through forests of diverse ages, structures, and management histories, leaving behind a landscape with diverse post-fire conditions. The differing post-fire conditions and the extensive burn areas present challenges for forest managers in determining and carrying out reforestation priorities. Forest vegetation management is a key tool for reforestation, as it increases the availability of site resources for newly established seedlings, enhancing seedlings' growth and survival. This chapter will examine the impacts of climate change on forestry and the impact of wildfires on plant communities and forest soil.

1.2 Literature Review

1.2.1 Climate Change and Forestry in the Pacific Northwest of the USA

The Pacific Northwest (PNW) of the United States (U.S.) is the western North American region that stretches from California (U.S.) to British Columbia (Canada) and includes the states of Oregon and Washington. The dominant vegetation type in the PNW is coniferous forests, including Douglas-fir, western hemlock, and western red cedar. These forests provide multiple purposes, including timber production, recreation, and wildlife habitat (Waring & Franklin, 1979). Oregon contains a large forested area, approximately half of the state, divided between eastern and western

Oregon. This land is largely used for the production of natural resources. Ponderosa Pine dominates the eastern Oregon forests, whereas Douglas-fir dominates the western Oregon forests. Oregon forest land ownership is distinct from the national average, where 36% of forest land is privately held, whereas the federal government holds a substantial amount of forest area. In addition, Oregon has the second-highest annual net increase of timberland in the United States, at 2,217 billion cubic feet, preceded only by Georgia. Forestry in Oregon contributes significantly to the state's economy, accounting for 4.7% of state output, 3% of state employment, and 3.7% of state gross domestic product. Over 18 billion dollars in output, 71,000 employment, and $8 billion in state gross domestic product are generated by Oregon's forestry industry (Kuusela et al., 2019). Given the importance of forestry to Oregon's economy, major disturbances in forest health and productivity, such as those derived by climate change, might have a significant effect on the state's timber industry. Understanding ongoing and future changes to Oregon's forests is crucial to strengthen their resilience and reduce potential negative consequences. This could involve initiatives to adapt forests to a changing climate and to promote practices that strengthen the resilience of forests.

In the future decades, climate change is anticipated to have a considerable impact on forest ecosystems, affecting the delivery of ecosystem goods and services. It will alter the distribution and abundance of plant species and communities and how terrestrial ecosystems process carbon, water, and nutrients. These changes will have ramifications for wildlife habitat, biodiversity, water availability, disturbance regimes, and the ability of ecosystems to absorb carbon from the atmosphere (Peterson et al., 2014).

There is substantial evidence that the Earth's atmosphere and climate are changing. The average global temperature has risen over the past century, and researchers are convinced that human activities contribute to this trend (Peterson et al., 2014). Since the previous century, the concentration of carbon dioxide (CO_2) in the atmosphere has continuously increased, from around 280 parts per million (ppm) in 1750 to approximately 415 ppm in 2022. So according to studies, this increase in CO_2 has occurred at an exponential rate, with the timeframe for anthropogenic CO_2

doubling (Hofmann et al., 2009). According to the Special Report on Emissions Scenarios (SRES) of the Intergovernmental Panel on Climate Change (IPCC), the lowest emissions scenario may result in global mean CO2 concentrations of 450 ppm by the year 2100 (Intergovernmental Panel on Climate Change, 2007). Scenarios with greater emissions would lead to much greater concentrations, with some forecasts reaching 875 ppm or higher by the end of the century (Clarke, 2007). These forecasts underline the significance of reducing greenhouse gas emissions to reduce climate change's effects. The projected climate models indicate that the global mean surface air temperature will rise during the next century, driven by increased radiative forcing owing to human-caused CO2 emissions, resulting in higher atmospheric CO2 concentrations (Meehl & Tebaldi, 2004).

In accordance with a study by (Mote & Salathé, 2010), climate projections for PNW based on 21 global climate models suggest that there will be an average increase in mean surface air temperature by the late 21st century of 2.5 oC under a low emission scenario and 3.4 oC under the high-emission scenario. These forecasts exceed the expected global mean temperature rises. In addition, forecasts for changes in annual precipitation in the PNW based on climate models are variable, with some models predicting less than a 5% shift and others showing a range from -10% to +20% by the end of the century (Mote & Salathé, 2010). By the end of the century, most models forecast a drop in summer precipitation and a rise in winter precipitation, with average changes of -14% and +8%, respectively.

Climate changes can affect plants in several ways, including through their influence on atmospheric evaporative demand and soil water availability. Changes in precipitation, including its amount, intensity, seasonality, surface and subsurface water transport, evaporation demand, and vegetation pattern, can influence soil water availability. Temperature and humidity variations can also affect evapotranspiration (Peterson et al., 2014). Employing historical climate data and a hydrological model, (Hamlet et al., 2007) examined patterns in evapotranspiration in the PNW over the past century. The model revealed a favorable trend in spring evapotranspiration due to earlier snowmelt, which led to earlier soil water recharging, plant activity, and evaporative losses. Summer evapotranspiration became more dependent on summer precipitation as soil water from melting snowpack decreased. These results imply that alterations in the timing and availability of water can influence evapotranspiration, which can have additional effects on plants.

Climate influences plant growth by altering several variables, including temperature, soil moisture, light, and nutrients. These parameters can influence the rates of photosynthesis, respiration, water and nutrient intake, and biomass production and allocation in plants. The moisture and temperature of the soil can also affect the soil ecosystem, including the decomposition of organic matter, nutrient cycling, and plant resource exchange (Peterson et al., 2014).

1.2.2 Effects of Wildfires on Plant Ecosystems

The occurrence and nature of wildfires are substantially influenced by climate. The likelihood, intensity, and extent of wildfires can be affected by drought, temperature, and the availability of fuel. For instance, drought can increase fuel flammability and the likelihood of fire spread, whereas high temperatures can prolong the fire season and promote more intense fire behavior. These variables can fluctuate over periods ranging from annual to decadal, and their effects on fires depend on the type of vegetation and the fire regime. In the forests of western North America, for instance, drought years are frequently connected with large fires (Peterson et al., 2014). In contrast, in the arid grasslands of the western United States, large fires frequently follow wet years that increase plant productivity and fuel production. The frequency and extent of wildfires are ultimately determined by a combination of climate and fuel availability (Peterson et al., 2014).

(Flannigan et al., 2009) examined the correlation between climate change and wildfire activity. Changes in temperature, precipitation, and wind patterns as a result of climate change are likely to increase the frequency, intensity, and magnitude of wildfires in many regions around the world, including the western U.S. They noticed that the effects of climate change and historical land-use changes had produced optimal circumstances for the occurrence of massive and intense wildfires. The same researchers reported that a warming climate could lengthen the fire season and increase the likelihood of intense fire weather.

Wildfires substantially impact plant ecosystems, the magnitude of which relies on heat fluxes and exposure duration (Bär et al., 2019). Furthermore, the moisture levels, structure, and amount of fuel influence the fire's behavior and the degree of vegetation destruction (Bond & Keeley, 2005). High-intensity crown fires are severe forest fires that consume living and dead foliage and various fuels such as branches and trunks. These fires kill plants instantly because they consume all vegetation and meristems (growth buds) in the three crowns, although they may be able to regenerate from heat-resistant organs (Clarke et al., 2012). However, low to moderate-intensity flames are less likely to kill mature trees immediately. Instead, these fires can cause a variety of damage to the tree's functionality, which could ultimately result in its mortality (Bär et al., 2019).

There are two sorts of fire effects on trees: first-order and second-order effects. First-order effects are the initial consequences of heat transport on plant tissues. These impacts include direct mortality caused by the consumption of foliage and branches and physical damage induced by the heating of tissues. The indirect and long-term consequences of fire on trees are second-order effects. These can include variations in competition, light and water availability, nutrient cycle, disease, and pest susceptibility (Bär et al., 2019). Two major concepts have been offered to explain how fire damage can influence the second-order functionality and mortality of trees. The first hypothesis assumes that fire-induced cambium and phloem tissue necrosis restricts carbohydrate transfer, lowering the tree's energy reserves and triggering carbon starvation. This can hinder the tree's ability to recover from scars, resist pathogens and pests, and ultimately lead to tree mortality (Bär et al., 2019; Michaletz, 2018). However, the hydraulic dysfunction hypothesis assumes that the heat of forest fires can cause harm to a tree's xylem tissue, reducing its ability to transport water and nutrients. This can reduce the tree's ability to transpire and photosynthesize, resulting in water stress and hydraulic failure (Bär et al., 2019; Michaletz, 2018).

1.2.3 Effects of Wildfires on Soil

Wildfires can drastically alter the quantity and distribution of soil organic matter. Low-intensity fires may only consume a percentage of the organic matter, but high-intensity fires may destroy most of them. Fire can have both positive and negative effects on soil organic matter in the ecosystem. On the one hand, fire can accelerate the decomposition of organic materials, returning nutrients to the soil and encouraging plant development. In contrast, the loss of organic matter can diminish soil fertility and stability and alter the local microclimate and hydrology (Neary et al., 2005).

By removing organic matter in the soil, research indicates that fire can significantly reduce the Cation Exchange Capacity (CEC), which is a measure of a soil's ability to retain positively-charged ions that are essential for plant growth and can affect plant development and soil health (Neary et al., 2005). Wildfires can impact cation exchange capacity, which is mostly caused by the decomposition of humus compounds. At approximately 212 oF, fire initiates decomposition in the organic and humic matter, which it can destroy at 932 oF (Neary et al., 2005). Clay soils have a greater cation exchange capacity than sandy soils. When the fire burns the humus, the soil loses its ability to retain nutrients, resulting in significant leaching losses of soluble nutrients (DeBano et al., 1998; Neary et al., 2005). (Soto & Diaz-Fierros, 1993) showed that CEC can decrease from 28.4 to 6.9 meq/100g at temperatures ranging from 338 to 716 oF.

During a fire, the combustion of organic matter can temporarily raise the soil's pH by releasing basic cations. However, the magnitude and length of the pH increase rely on variables such as the initial pH of the soil, the quantity and composition of ash created by the fire, and the climate's humidity. The ash-bed effect happens when a substantial number of nutrients are deposited on the surface of the soil as a result of a fire and can also alter the pH of the soil (Neary et al., 2005).

Specific chemical reactions that occur after a fire can vary the amount of nutrients in the soil, subsequently altering its availability. In the case of nitrogen, due to the fire's combustion, its levels in the soil are often reduced. While nitrogen loss happens at high temperatures, the amount of available nitrogen (NH4) often increases in the first few years following a fire, thus promoting plant development. This increase in nitrogen availability after a fire may show that total nitrogen is present, but it is often temporary and will be rapidly used by plants (Neary et al., 2005). The losses of nitrogen during a fire are temperature-dependent. Nitrogen can be entirely lost at temperatures above 932 ^{o}F, but not below 382 ^{o}F (Neary et al., 2005). Phosphorus is lost at greater temperatures than nitrogen when organic matter is burned, and around 60 percent of the total phosphorus is lost. However, burning organic matter frequently leaves a substantial amount of widely available phosphorus in the surface ash of the soil immediately after a fire (Neary et al., 2005). Sulfur is lost by volatilization at moderate temperatures, and 20-40% of the sulfur in aboveground biomass has been observed to be lost during fires (Neary et al., 2005).

1.2.4 Impact of wildfire on vegetation

The changing climate and altered fire regimes will likely impact the growth and development of forest ecosystems. Shorter intervals between fires can limit the time available for plants to mature and produce seeds, potentially altering post-fire regeneration. Some plant species that rely on sprouting may decline, while others that require seed reproduction may be lost. These changes can have significant effects on the structure and composition of forested landscapes (Halofsky et al., 2020). In addition, the amount of seed resources available to regenerate the burned areas will become limited. The recovery process of species that do not have fire-resistant cones may take longer as it will rely on seed dispersal from far away, especially in larger, high-severity fire areas (Little et al., 1994; Donato et al., 2009; Downing et al., 2019). Furthermore, increased forest drought stress caused by a warming climate could lead to a decline in seedling survival after fires, as warmer and drier conditions following fire events will make it harder for new trees to establish. This, in turn, could impact forest ecosystems' structural and compositional trajectories (Dodson & Root, 2013).

In PNW forest ecosystems, a warming climate and varying disturbance regimes likely lead to plant species composition and structure changes. Overall, increased fire frequency will favor plant species with life history traits that allow for survival with more frequent fires (Chmura et al., 2011). These species can be characterized by the following features including 1) shrubs with coated seeds that remain inactive in the soil until a fire occurs. While some seeds are destroyed by fire, others have their coatings broken, enabling them to absorb water and grow in a favorable environment with plenty of nutrients and competition-free environment such as snowbrush; 2) woody species with adaptations that allow them to complete their life cycle quickly which enables them to produce seeds in the case of consecutive fires close in time such as Bishop pine; 3) woody species with fire-resistant bark: the ability to resist fires such as Douglas-fir, and Ponderosa pine with thick bark, 4) serotinous cones allow for seed dispersal and regeneration of the species after a fire such as Lodgepole pine (Agee, 1996).

Invasive species may have more chances to establish in the forest understory due to increased fire frequency and the scope of fires (Hellmann et al., 2008). The likelihood of species that can survive fires (sprouters) and those that exist in seed banks (evaders) is expected to rise with increased fire frequency. For instance, sprouting shrubs and hardwoods are abundant in southwest Oregon after fires. However, intense fires can destroy seeds stored in the upper soil layers and kill shallow plant roots and repeated fires at close intervals can deplete seed reserves and underground plant resources (Halofsky et al., 2020).

Increased fire frequency is likely to reduce the abundance of avoider species, including species with thin bark and shade-tolerant that slowly invades after fire (Chmura et al., 2011).

Forests dominated by fire-vulnerable evader species, such as western hemlock, will experience higher mortality rates during a fire than forests with more fire-resistant species, like Douglas-fir and western larch. If these fire-sensitive species are unable to grow back in the burned area due to close fire intervals, competition, or unfavorable conditions for seedling growth, they could be completely lost from the site (Stevens-Rumann et al., 2017). As a result of more frequent fires, species that can tolerate or regenerate after fires are likely to become more dominant. In contrast, species that are vulnerable to fire will likely decrease. For example, in southwest

Oregon, an increase in fire frequency could lead to an increase in the abundance of shrubs and hardwoods and a decrease in conifer in some areas (Tepley et al., 2017).

The changes in disturbance regimes will have a significant impact on the structure of forests, not only within individual stands, but also at a larger, landscape level (Reilly et al., 2018). In dry forests, the frequency of fires is likely to lead to a decrease in tree density and a possible increase in the area of open savannas. The understories of these forests may transform from being dominated by duff or forbs to being dominated by shrubs or grass (Agee, 1996).

In the Klamath-Siskiyou region of southern Oregon and northern California, a study by (Tepley et al., 2017) showed that low soil moisture following fires negatively impacted forest regeneration. In areas with less soil moisture, a greater number of propagules (smaller patches with more living seed trees) was necessary to achieve a certain level of regeneration. This implies that even small patches with high fire severity are vulnerable to low regeneration of conifer trees in regions with high water scarcity. Continual fires could also decrease the number of conifer seed sources, thus promoting the growth of shrubs and hardwood trees. In a study by (Donato et al., 2016), a decrease in Douglas-fir regeneration was observed 24 years after a fire in areas with lower elevation and dryer conditions compared to regions with higher elevation and more moisture. The regeneration was found to decline with increased fire severity and was nearly non-existent beyond 100 to 200 meters from the seed source.

1.2.5 Reforestation Challenges

The National Forest Management Act guides the reforestation activities, and this legislation requires that forest lands that have been cleared must be reforested and harvested areas must be replanted within five years of the harvest (NFMA, 1976 [Section 6 Eii]). The reforestation process typically involves several steps, such as; salvage logging, site preparation, planting of seedlings, competition control to promote seedling survival and growth, and pre-commercial or commercial thinning (North et al., 2019; Schubert & Adams, 1971). In many arid western U.S. forests, the increasing frequency and severity of fires and droughts have significantly impacted the capacity and success of reforestation activities. Reforestation efforts face challenges such as high costs and safety issues associated with replanting vast areas of standing dead trees, as well as significant seedling and sapling mortality caused by factors such as water stress, competition from other vegetation, and recurrent fires that destroy young plantations (North et al., 2019).

1.2.6 Forest Vegetation Management in the Pacific Northwest of the USA

In newly planted or naturally regenerated forests, competition between diverse plant species, including trees and natural early-seral vegetation, plays a vital role in determining the establishment and growth of trees. This competition can take various forms, including competition for light, water, and nutrients (Balandier et al., 2005). Forest Vegetation Management (FVM) temporarily reduces the presence of undesirable plant species to enhance the number of resources and growing space available to desired tree species. In other words, FVM is the management of the plant composition of a forest ecosystem to promote the growth and establishment of desired species over undesirable ones (Eyles et al., 2012; Gonzalez-Benecke & Dinger, 2018). Manual slashing, physical removal, prescribed burning, and herbicide sprays are examples of FVM treatments. In the PNW, chemical treatments are commonly employed to develop new tree plantations in FVM (Newton & Preest, 1988). These treatments involve using herbicides to eliminate undesirable vegetation and enhance the growth of desired tree species. The most typical approach is to spray herbicides before planting new seedlings; this is known as "fall site preparation," followed by one or more years of herbicide treatment in the spring following tree planting; this is known as "spring release". During the first year of the new plantation's growth, this chemical treatment method is used to maintain a clear area for planting new trees and to continue suppressing competing species.

For optimal growth and development of conifer species, the FVM strategy must be implemented during the first years of tree seedling establishment. This approach accelerates the growth of the tree and shortens its rotation period. Several studies on conifer reforestation in the PNW have been performed, including (Newton & Preest, 1988; Dinger & Rose, 2010; Dinger & Rose, 2009; Maguire et al., 2009). According to these studies, chemical FVM treatments on Douglas-fir stands can long-term impact growth and biomass accumulation. However, the magnitude of the response will depend on site conditions such as climate, soil type, and vegetation community. Nevertheless, in the PNW, this growth enhancement is mostly due to increased soil moisture availability, lessening planted trees' water stress.

The Vegetation Management Research Cooperative (VMRC) at Oregon State University (OSU) has conducted multiple studies to determine the effects of different herbicide-based vegetation control regimes on Douglas-fir seedling growth, physiology, and soil moisture availability (Dinger & Rose, 2010; Dinger & Rose, 2009; Gonzalez-Benecke & Dinger, 2018; Wightman et al., 2018). The results from these studies highlight that providing Douglas-fir seedlings with greater available soil moisture until late summer using FVM regimes is critical for maintaining appropriate growth and survival.

1.3 Research Objectives and Hypothesis

This study aims to improve understanding of reforestation challenges after the fire, including Douglas-fir seedling performance and early-seral vegetation dynamics under unburned and non-merchantable and merchantable burned stands, and the impact of stand age/structure on post-fire conditions. The specific objectives are:

1. To determine the effect of wildfire on soil physical and chemical attributes.
2. To determine the effect of wildfire microclimate conditions.
3. To determine the effect of pre-wildfire stand age/structure on post-fire early seral vegetation community dynamics.
4. To determine the effect of pre-wildfire stand age/structure and FVM on soil moisture dynamics.
5. To determine the effect of pre-wildfire stand age/structure and FVM on Douglas-fir seedling performance.
6. To determine the effect of delayed planting on the effectiveness of FVM and Douglas-fir growth and survival.

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2 Wildfire and Reforestation Treatments Effects on Soil and Microclimate

2.1 Introduction

Wildfires are a common disturbance in dry conifer forests (Walstad et al., 1990). To comprehend the dynamics of wildfires, it is crucial to understand the concept of the fire behaviors triangle, where the three essential elements required for a wildfire to ignite and persist: weather, fuel, and topography (Agee, 1996), where wind (speed and direction), temperature, humidity and rainfall, together with the slope and aspect of the land interacts with the amount, arrangement and moisture of fuel. The effects of wildfires on the environment are determined by the severity of the fire, which is shaped by several environmental factors that influence the combustion process (Jiménez-Morillo et al., 2020). The impact of wildfires on soil properties can be short-term, long-term or permanent depending on the type of property, severity and frequency of wildfires, and post-fire climatic conditions (Certini, 2005).

Wildfires have the potential to result in significant loss of soil organic matter (SOM) through factors such as physical loss, combustion, accelerated decay, and changes in the quantity and quality of organic matter inputs (Pellegrini et al., 2021). SOM is important for maintaining soil structure, retaining water, and promoting nutrient cycling. When burned at a temperature of 450 ^{o}C wildfires can consume 99% of SOM in just two hours.

The combustion of SOM decreases cation exchange capacity (CEC), which is a measure of a soil's ability to retain positively-charged ions that are essential for plant growth and can effect plant development and soil health (Neary et al., 2005). Wildfires can impact cation exchange capacity, which is mostly caused by the decomposition of humus compounds. At approximately 100 ^{o}C fire initiates decomposition in organic and humic matter, which it can totally destroy at 500 ^{o}C. Clay soils have a greater CEC than sandy soils. When the fire burns the humus, the soil loses its ability to retain nutrients, resulting in significant leaching losses of soluble nutrients (DeBano et al., 1998; Neary et al., 2005). (Soto & Diaz-Fierros, 1993) showed that CEC can decrease from 28.4 to 6.9 meq/100g at temperatures ranging from 170 to 380 ^{o}C.

During a fire, the combustion of SOM can temporarily raise the soil's pH by releasing basic cations. However, the magnitude and length of the pH increase rely on variables such as the initial pH of the soil, the quantity and composition of ash created by the fire, and the climate's humidity. The ash-bed effect happens when a substantial number of nutrients are deposited on the surface of the soil as a result of a fire and can also alter the pH of the soil (Neary et al., 2005).

Specific chemical reactions that occur after a fire can vary the amount of nutrients in the soil, subsequently altering its availability. In the case of nitrogen, as a result of the fire's combustion, its levels in the soil are often reduced. While nitrogen loss happens at high temperatures, the amount of available nitrogen (NH_4) often increases in the first few years following a fire, thus promoting plant development. This increase in nitrogen availability after a fire may provide the appearance that total nitrogen is present, but it is often temporary and will be rapidly used by plants (Neary et al., 2005). The losses of nitrogen during a fire are temperature dependent. Nitrogen can be lost completely at temperatures above 500 ^{o}C, but not at temperatures below 200 ^{o}C (Neary et al., 2005). Phosphorus is lost at greater temperatures than nitrogen when organic matter is burned, and around 60 percent of the total phosphorus is lost. However, the burning of organic matter frequently leaves a substantial amount of widely available phosphorus in the surface ash of the soil immediately after a fire (Neary et al., 2005). Sulfur is lost by volatilization at moderate temperatures, and 20-40% of the sulfur in aboveground biomass has been observed to be lost during fires (Neary et al., 2005).

As more of the landscape is impacted by wildfires, there is a danger of forests transitioning into non-forest habitats (Abatzoglou & Williams, 2016; Guiterman et al., 2017). The change in land cover as a result of large wildfires has a major effect on the local climate by affecting the boundary layer, as well as the balance of surface energy through changes to net radiation (Rother & De Sales, 2020). The ash and charcoal left behind after a fire decreases the reflective capacity of the land and increases the amount of heat released into air (Dintwe et al., 2017). The effect of a wildfire on albedo is largely determined by the severity of the fire and typically does not last long as the ash is quickly scattered by wind and rain (Veraverbeke et al., 2012; Rother et al., 2022). On the other hand, the loss of vegetation often results in a decrease in the release of latent heat, an increase in sensible heat, as the reduced ability to cool through evaporation leads to a rise in land surface temperature. The temperature of the land surface can increase by as much as 8 ^{o}C after a wildfire, with the length of time that this change lasts depending on the type of vegetation (Amiro et al., 2006; Bremer & Ham, 1999, Wendt et al., 2007). With ongoing climate change causing rising temperatures and increased aridity, local micro-climates that can put seedlings under stress and exceed their physiological thresholds (Bell et al., 2013; Dobrowski et al., 2015). Higher temperatures lead to an increase in leaf level vapor pressure deficit (VPD) which can result in catastrophic xylem embolism or carbon starvation, especially in water-stressed seedlings that are more sensitive to extreme conditions than mature trees (Will et al., 2013). Reducing incoming solar radiation can help regulate extreme temperatures in the local climate while increasing relative humidity, which decreases the demand on plant water uptake and transport (Davis et al., 2018).

The Archie Creek Fire occurred in southwest Oregon in September of 2020. The region impacted by the Archie Creek Fire is noted for getting heavy precipitation, predominantly in the form of rain and snow fronts. The majority of this precipitation falls between October and April, with yearly averages ranging from 1,000 to 2,500 mm. The blaze quickly grew and combined with another Star Mountain Fire, burning 109 homes and causing nine additional structures to suffer damage over the course of two days. The fire impacted diverse types of terrain, including national forest, BLM land, private timberland, and a number of villages along the North Umpqua River and Rouge-Umpqua Scenic Highway. The fire burned 64,676 acres of private land, 39 acres of state territory, and 66,881 acres of federal land (Rasmussen et al.).

2.2 Methods

2.2.1 Study Area

This study was carried out in four stands located in a common physiographic area affected by the Archie Creek Fire in Southwest Oregon, near Roseburg. Site conditions across the selected stands, including soil type, site index, aspect, slope, and burning severity, were maintained as homogeneous as possible. The sites consisted of four different pre-fire stand ages/structures: 1) unburned (UB), 2) burned when stand was planted recently (1 year old; B1y)), 3) burned when stand was mid-rotation and non-merchantable (12 year-old; B12y), and 4) burned when stand was mature and merchantable and salvage harvest was carried out (55 year-old; B55y). Sites coordinates and elevation are shown in Table 2.1.

Table 1: Sites location and elevation

Study ID	Burning Condition	Stand Age at Burning	Latitude (N)	Longitude (W)	Altitude (ft)
UB	Unburned	N.A.	43°24`12	123°06`05	1250
B1y	Burned	1 year	43° 22`54	123°04`43	2650
B12y	Burned	12 years	43° 22`55	123°04`45	2750
B55y	Burned	55 years	43° 23`40	123°05`27	1750

The study area has a climate with warm and dry summers and cold and wet winters with a mean annual temperature of 10 ^{o}C and total annual rainfall of 986 mm. Most precipitation occur between October and April. Elevation within the study area ranges from 1250 to 2750 ft.

The locations of the study sites showed in Figure 8 Distance between the most distant sites (UB and B12y) was 2 miles.

2.2.2 Study Design

Within each stand, a randomized complete block design with six reforestation treatments and four blocks was used (24 plots per site). The six reforestation treatments consisted on a combination of VM and delayed planting. The VM treatments consisted of the no-action control (C), fall-site preparation (FSP) and spring-release (SR) applications. The delayed planting consisted on planting either the winter after fire ended (2021) or the winter of the following year (2022). The delaying planting option is a common alternative given logistics constrains due to lack of seedlings and manpower, but it also gives the opportunity to improve efficacy of herbicide treatments by increasing foliage cover for FSP treatments.

Table 2: Description of treatments applied at each study site.

Treatment	Planting Year	FSP	SR
C	2021	0	0
SR	2021	0	1
C.D	2022	0	0
SR.D	2022	0	1
SFP.D	2022	1	0
SFP.D+SR.D	2022	1	1

C: no-action control; FSP: fall-site preparation; SR: spring-release; D: delayed planting

In February 2021, 24 plots consisting of 36 seedlings planted at 10x10 ft spacing were installed within each stand (60 x 60 ft plots). In the B12y site, the DBH of all standing dead trees was measured, and block assignment was implemented based on the Basal Area of each plot, after ranking all plots based on BA. In March 2021, two plots within each block were planted (C and SR treatments). In April 2021, post-planting herbicide application was carried out for the SR plot. In September 2021, pre-planting herbicide application was carried out to each plot assigned for delayed planting (FSP.D). In February 2022, the four plots assigned for delayed planting within each block were planted with the same stock type used the previous year. In April 2022, post-planting herbicide application was carried out for the SR.D plots. Information regarding the rates and dates of herbicide applications is provided in Chapter 3 (Table 3.3).

2.2.3 Soil Moisture and Weather

During both years 2021 and 2022, soil volumetric water content (VWC, cm^3 cm^{-3}) was measured monthly (May to November) on the same five points used for vegetation survey within each plot using a handheld TDR sensor (Hydrosense II, Campbell Scientific). A datalogger (CR300, Campbell Scientific) was placed in the center of the UB, B1y and B12y stands to record weather measurements, which were taken every 5 minutes and averaged every 30 minutes. At each datalogger, a weather station connected to it to measure global solar radiation (CS301, Apogee Instruments), rainfall (TE525MM, Texas Electronics), air temperature, and relative humidity (HMP60, Vaisala) and wind speed (P2546D, Windsensor Inc.). Weather measurements were taken every 30 seconds and averaged every 30 minutes on the datalogger.

2.2.4 Soil Sampling

During March 2021, before VM treatments were applied, a soil sample of the top 20 cm soil was extracted at the center of each plot using a 5 cm diameter pvc core. Soil samples were placed in a cooler with ice packs to prevent them from drying until processing. The samples' initial weight was measured and then placed in the oven for at least 72 hours at 5 ^{o}C. Air-dried soil samples were sieved to 2 mm and separated finite roots and more than 2 mm rocks. The final weight of soil without roots was measured and bulk density was determined for each sample. Finally, all sieved samples were sent to A&L Western Laboratories (Modesto, CA) to determine texture, organic matter, and macronutrients content.

2.2.5 Statistical Analysis

Statistical Analysis Software version 9.4 (SAS Institute Inc. Cary, NC) was used for all statistical analysis. Analysis of variance, including Tukey adjustments, was used to test the effects of site and reforestation treatment on soil properties (PROC MIXED). Repeated measures analysis was used to analyze the time series of soil volumetric water content. Several covariance structures were tested, and the one with the lowest BIC model was chosen. All figures were produced using SigmaPlot version 14 (Systat Software, Inc. San Jose, CA).

2.3 Results

2.3.1 Weather

The daily average temperature, VPD, and daily rainfall were recorded from April 2021 to December 2022. In late June 2021, an extreme temperature was recorded, leading to an increase of about 4 kPa in the mean VPD. The rainy season began in mid-September 2021, with a total rainfall of 134.3 mm observed during the preceding dry season from July to mid-September. However, in 2022, the rainy season did not start until late October, and during the preceding dry season from July to Late October, the total rainfall was only 99.8 mm (Figure 1).

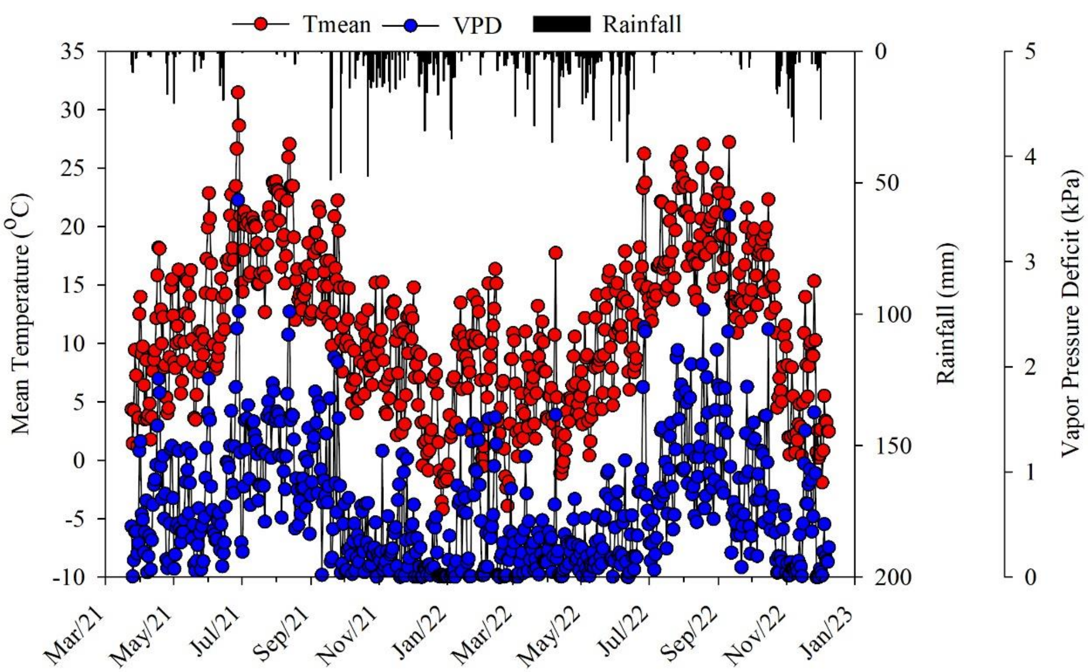


Figure 1: Daily mean temperature (represented by red dots), daily mean vapor pressure deficit (VPD, represented by blue dots), and total rainfall (represented by black bars) during the first two years at B1y.

The B12y site (stand with standing trees) had usually higher maximum daily VPD, temperature and wind speed than the B1y site (Figure 2). When compared with the B12y site, monthly average relative humidity was higher at the B1y stand (P <

0.0001) and monthly average maximum wind speed was higher at the B1y stand (P < 0.0001) (Figure 2).

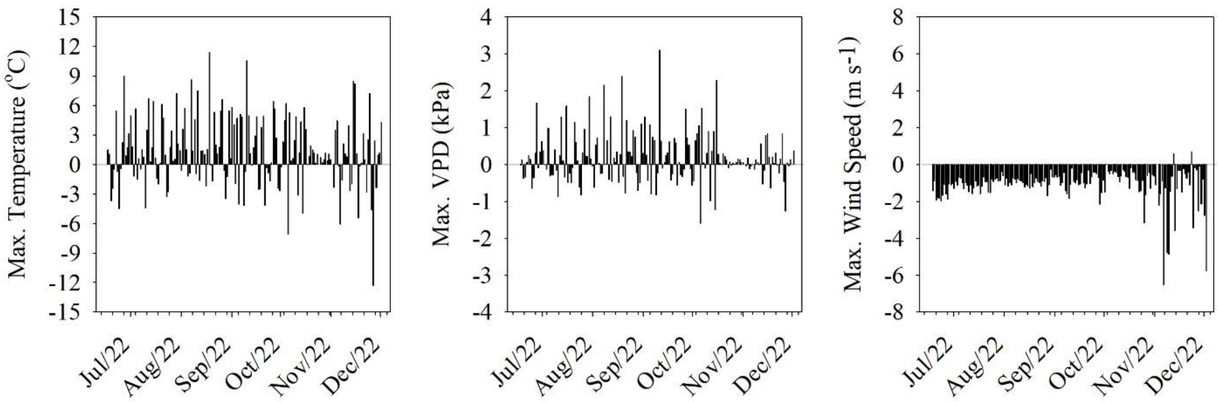


Figure 2: Difference in daily maximum temperature (Max. Temperature (°C), on the left), vapor pressure deficit (Max. VPD (kPa), in the center), and wind speed (Max. Wind Speed (m s^{-1}), on the right) between the B12y and B1y.

2.3.2 Intercepted Photosynthetic Active Radiation by Standing Burned Trees

In the B12 site, there was a weak, but significant relationship between standing dead trees basal area (m^2 ha^{-1}) and fractional intercepted photosynthetically active radiation (Figure 3).

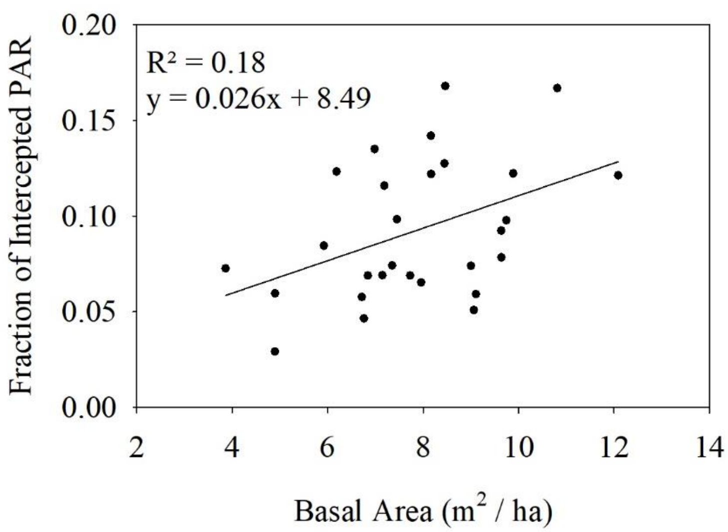


Figure 3: Fraction of intercepted photosynthetic active radiation (fPAR) and basal area (m^2 ha^{-1}) for each plot (black dots) at B12y site.

At that site, fractional intercepted photosynthetically active radiation by standing trees ranged between 0.3 and 0.17., and the basal area of those trees ranged between 4 to 12 m^2 ha^{-1}. Overall, for every 1 m^2 ha^{-1} decrease in basal area, there was a 2.6% increase in intercepted PAR (Figure 3).

2.3.3 Soil physical and chemical attributes.

At the UB site, the percentage of sand was 2% lower than both B1y and B55y sites, respectively, but it was 9% higher than B12y ($P>0.288$). The percentage of silt at UB site was 1.5% higher than B1y and 3% lower than B12y, but it was similar to B55y ($P>0.832$). The clay percent at UB was 2% higher than B55y, but it was 4% less than B12y ($P>0.09$) (Figure 4).

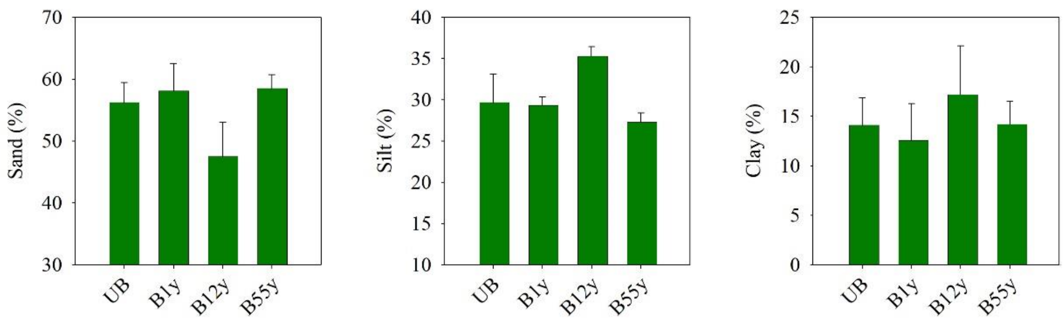


Figure 4: Percentage of sand (on the left), the percentage of silt (in the center), and the percentage of Clay (on the right) in top 20 cm soil at UB, B1y, B12y, and B55y sites.

Table 3 shows the soil physical and chemical properties of each site, including UB, B1y, B12y, and B55y. The table provides information on bulk density (BD), soil organic matter (SOM), pH, cation exchange capacity (CEC), nitrogen (N), phosphorus (P), potassium (K), magnesium (Mg), sulfur (S), and calcium (Ca). Each site was subjected to a different stand age/structure: UB was unburned, B1y was burned when the stand was 1-year-old, B12y was burned when the stand was 12-year-old, B55y was burned when the stand was 55y-year-old. Nitrogen was measured as NO_3 , phosphorus as Bray procedure, and sulfur as SO_4. The values in the table are represented in different units, and the letters represent significant differences across the sites.

Table 3: The table summarizes the physical and chemical properties of the soil in each site, including bulk density (BD), soil organic matter (SOM), pH, cation exchange capacity (CEC), and concentrations of nitrogen (N), phosphorus (P), potassium (K), magnesium (Mg), sulfur (S), and calcium (Ca). The sites include UB (Unburned), B1y (Burned when stand was 1-year-old), B12y (Burned when stand was 12-year-old), B55y (Burned when stand was 55-year-old). The values presented in the table represent means of the four sites, and different lowercase letters indicate significant differences (P<0.05) across sites for each trait.

	Site			
Trait	UB	B1y	B12y	B55y
BD (g cm^{-3})	0.98 b	0.92 b	0.95 b	1.11 a
SOM (%)	9.4 a	7.0 b	7.1 b	5.4 b
pH	5.2 a	5.0 a	5.2 a	4.9 a
CEC (meq/100g)	33.4 a	16.5 b	20.3 b	7.4 c
N (ppm)	2.0 b	12.8 a	8.8 a	7.3 a
P (ppm)	6.4 b	17.6 a	10.8 a	18.6 a
K (ppm)	244.1 a	281.6 a	363.3 a	184.8 b
Mg (ppm)	1,027.4 a	399.8 b	450.9 b	145.8 c
S (ppm)	9.6 a	9.8 a	9.0 a	8.8 a
Ca (ppm)	4,728.9 a	2,390.8 b	3,024.1 b	1,047.6 c

UB: Unburned; B1y: Burned when stand was 1 year-old; B12y: Burned when stand was 12-year-old; B55y: Burned when stand was 55-year-old; N: NO_3; P: Bray procedure; S: SO_4.

Soil bulk density of the B55y site (1.11 g cm^3) was larger than the other stands (P<0.0001), which averaged 0.95 g cm^3. All burned sites showed reduced soil OM (P <0.0001), which ranged between 25 to 43% reduction. The B55y site showed lower pH than the other sites (P = 0.0055). The UB site showed 2 to 4 times larger CEC than the burned stands (P < 0.0001). Burned sites showed 4 to 6 times larger NO_3 and 1.7 to 3 times larger P than the UB site (P < 0.0001). B12y and B1y sites showed larger K than UB and B55y sites (P < 0.0001), which showed no differences among them. The UB site showed 2 to 7 times larger Mg than the burned stands (P < 0.0001). There were no differences in S across burned and UB sites (P = 0.83). The UB site showed 2 to 4.5 times larger Ca than the burned sites (P < 0.0001) (Table 3).

2.3.4 Soil Volumetric Water Content

Table 4 provides a summary of the analysis of variance (ANOVA) for the volumetric water content (VWC) time series during the growing season of 2021 and 2022. The ANOVA was conducted on four factors: Site, VM treatment, Date, and the interaction between Site x VM treatment x Date. The table presents the p-values for each factor, which are indicators of the statistical significance of the differences observed.

Table 4: Summary of ANOVA for VWC time series in 2021 and 2022 at different sites and vegetation management (VM) treatments. The table shows the p-values for the main effects of Site, VM treatments, and Date as well as the interaction between Site x VM treatment x Date. P-values indicate statistical significance at a significance level of 0.05.

Growing Season	Factor	Site	VM Treatment	Date	Site x VM Treatment	Date x Treatment	Date x Site	Site x VM Treatment x Date
2021	VWC	0.0007	0.0237	<.0001	0.6260	0.0033	0.0002	0.8426
2022	VWC	<.0001	<.0001	<.0001	<.0001	<.0001	<.0001	0.0134

During 2021, there were significant differences in VWC levels among the different sites, VM treatments, and Dates (P<0.023). However, there were no significant interactions among these factors (Site x VM treatment x Dates, P=0.842) (Table 4).

During 2022, there were significant differences in VWC levels among the different sites, VM treatment, and Date (P<0.001). In addition, there was a significant interaction between Site x VM treatment x Date factors (P=0.013, Table 4).

The study found that during the winter season in 2021 and 2022, the Volumetric Water Content (VWC) at or above the field capacity level (>0.4 cm3 cm-3) was maintained at UB and B1y sites due to consistent rainfall. The UB had a higher capacity to retain water compared to B55y, with a VWC about 25% higher (0.42 vs 0.33) during both growing seasons. As the dry season progressed, the differences among VM treatments started to become clearer during both growing seasons (Figure 5 and Figure 6).

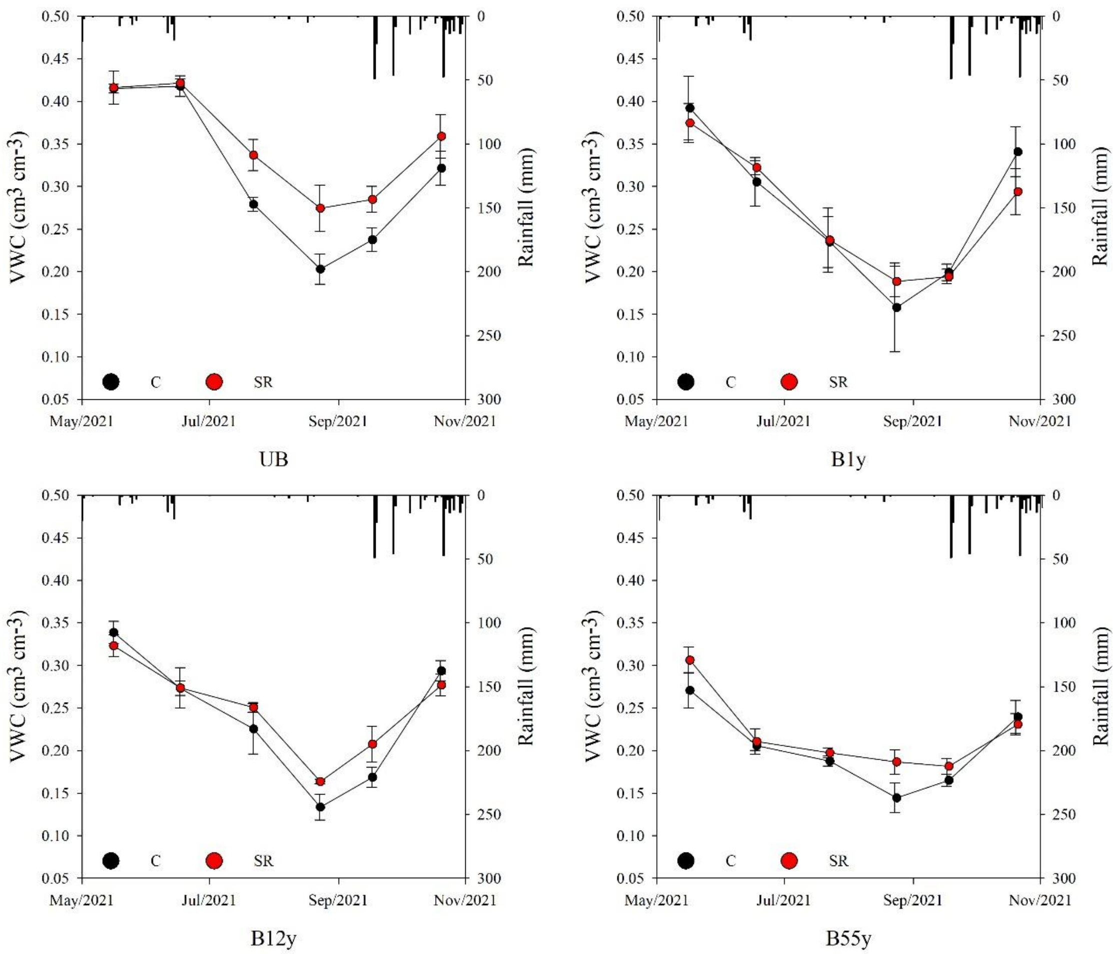
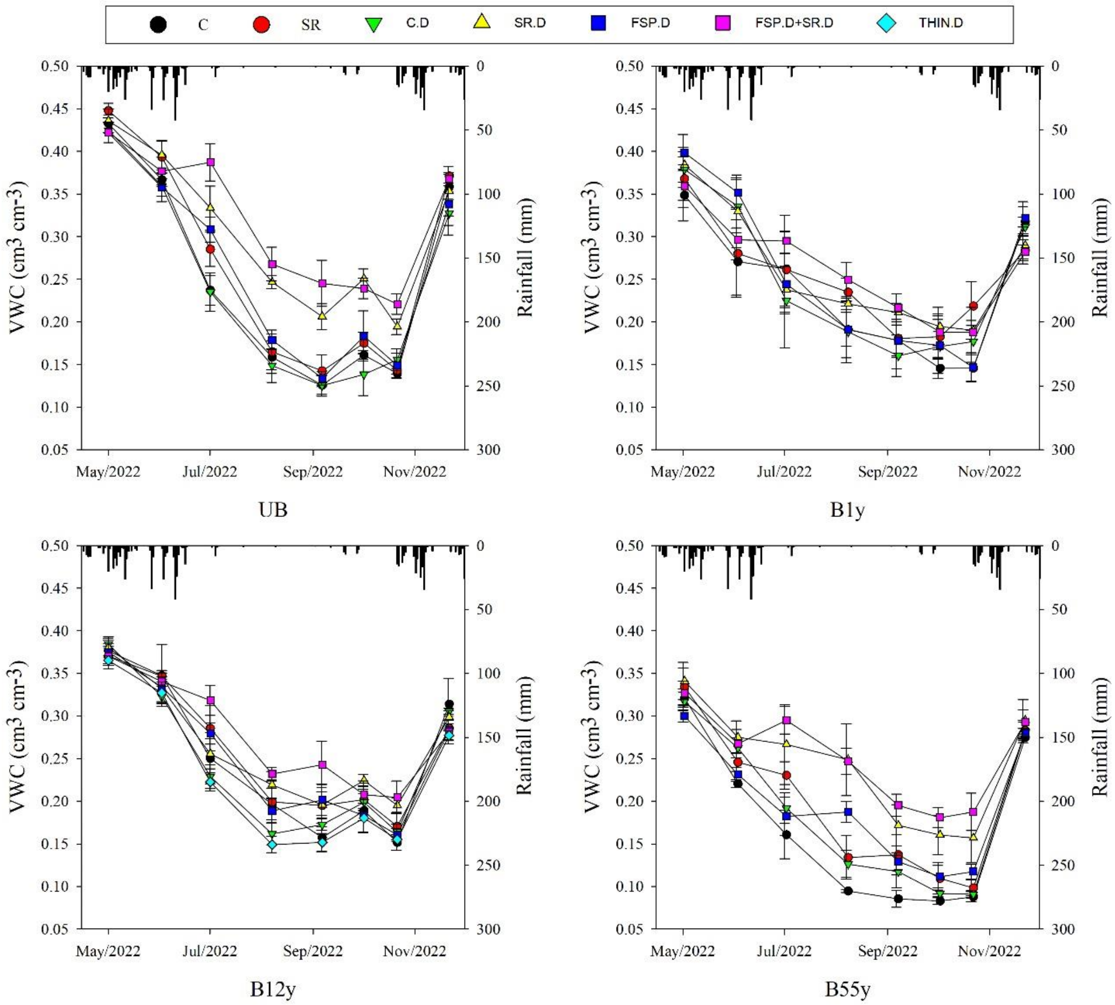


Figure 5: Seasonal dynamics of soil volumetric water content (VWC, cm^3 cm^{-3}) during year 2021 for plots that received VM treatments at the unburned (UB, upper left panel), burned at age 1 year (B1y, upper right panel), burned at age 12 years (B12y, lower left panel), and burned at age 55 years (B55y, lower right panel) sites. Bars on top of each panel depicts daily rainfall (mm). Site and VM regimes information is provided in Tables 1 and 2, respectively. Error bar represents standard error.

The VWC dynamics exhibited two primary responses to the various VM treatments. Firstly, the VM treatments could be categorized into two distinct groups, with plots that did not received herbicide during spring (C and C.D) exhibiting lower VWC levels compared to those that received herbicide during spring and/or fall (SR, SR.D, and FSP.D+SR.D). Although the differences within these groups were minimal, significant differences were observed between the groups. This trend was observed at both UB and B55y sites during the 2022 growing season. Secondly, a unique VWC pattern was observed for all six VM treatments throughout the 2022



growing season, as well as the 2021 growing season (Figure 5, Figure 6).

Figure 6: Seasonal dynamics of soil volumetric water content (VWC, cm^3 cm^{-3}) during year 2022 for plots that received VM treatments at the unburned (UB, upper left panel), burned at age 1-year (B1y, upper right panel), burned at age 12 years (B12y, lower left panel), and burned at age 55 years (B55y, lower right panel) sites. Bars on top of each panel depict daily rainfall (mm). Site and VM regimes information is provided in Tables 1 and 2, respectively. Error bar represents standard error.

During the growing season 2021 and 2022, the lowest of VWC was observed at C and C.D treated plots. Both treated plots followed a similar pattern of decreasing

VWC. The lowest point during the study (from March 2021 to December 2022) was observed in 2022 during the growing season for both C and C.D treatments. The lowest values of VWC were recorded at UB, B1y, B12y, and B55y sites, reaching 0.12, 0.15, 0.16 and 0.08 $cm^3\ cm^{-3}$, respectively (Figure 5 and Figure 6).

When evaluating the most contrasting VM treatments at each site, the largest differences in VWC were observed. For example, SR.D and FSP.D+SR.D treatments and C and C.D treatments exhibited the greatest contrast during the driest months of growing season of 2022 at the sites. This difference at UB site lasted 13 weeks (early July to early October; $P<0.001$), and for B55y site, it lasted 9 weeks (early July to early September; $P < 0.005$). In addition, At the B55y, the only significant difference was seen between C.D and FSP.D, which was lasted for no more than four weeks (from early August to early September; $P<0.001$).There was no other significant difference at other sites for both growing seasons (Table 4).

The reduction of soil organic matter (SOM) caused by fire had a significant impact on both the available soil water holding capacity (AWHC) and the upper limit of VWC (Table 3 and Figure 7). Specifically, for every 1% decrease in SOM, there was a decline of 1.67 cm3 cm-3 in AWHC and a decrease of 2.37 cm3 cm-3 un the upper limit of VWC. In addition, there was a strong correlation observed between AWHC and the upper limit of VWC. Specifically, for every 1 cm3 cm-3 increase in the upper limit of VWC, there was a corresponding increase of 0.68 cm3 cm-3 in AWHC. Further, the lower limit of VWC was found to be correlated with the percentage of clay content in the soil, with an increase of 0.4 cm3 cm-3 in the lower limit of VWC observed for every 1% increase in clay content.

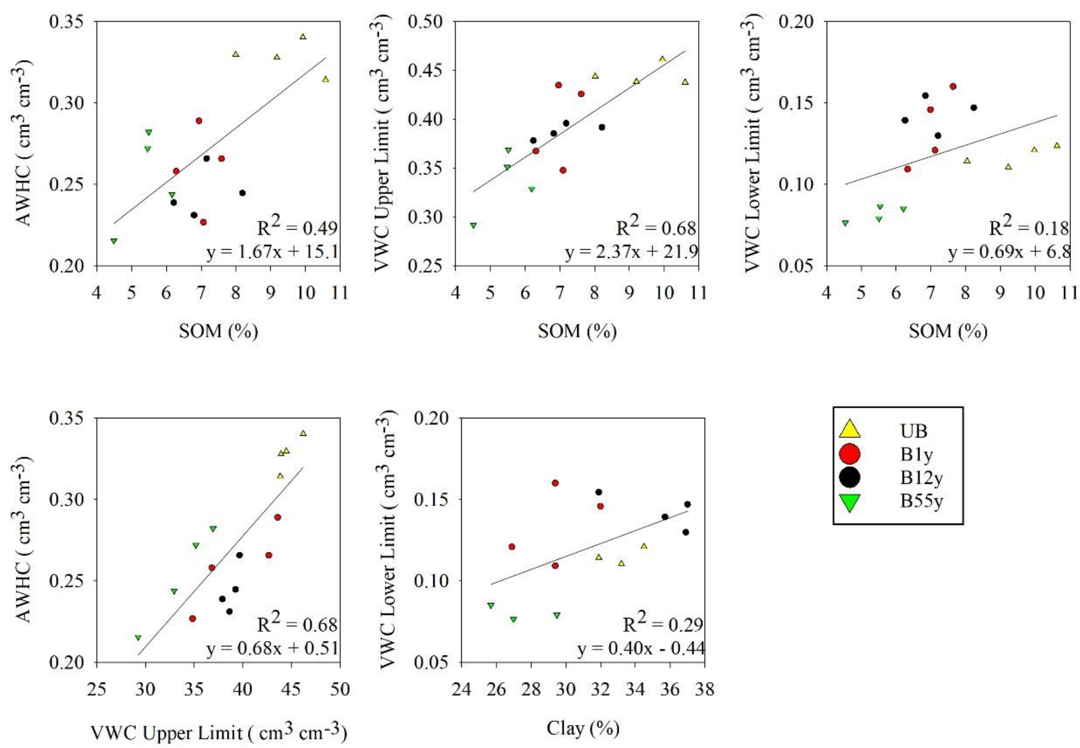


Figure 7: Correlation analysis between soil organic matter (SOM, %) and soil available water holding capacity (AWHC, cm^3 cm^{-3}, top-left), upper limit of water holding capacity (WHC-UL, cm^3 cm^{-3}, mid-top) and lower limit of water holding capacity (WHC-LL, cm^3 cm^{-3}, top-right), correlation between WHC-UL and AWHC (bottom-left), and correlation between Clay content (%) and WHC-LL (bottom-right) at the unburned (UB, yellow triangle), burned at age 1 year (B1y, red circle), burned at age 12 years (B12y, black circle), and burned at age 55 years (B55y, green triangle) sites. Sites information is provided in Table 1.

2.4 Discussion

The presence of standing trees decreases the amount of PAR that reaches the soil surface, which reduces the intensity of heating in the topsoil and ultimately creates more favorable conditions for seedling growth (Urretevizkaya et al., 2019). Furthermore, the standing trees provides shade protection, which not only decreased the amount of PAR and daytime topsoil temperatures, but also mitigated the impact of low nighttime temperatures, which leads to the creation of a more favorable microclimate for the growth of newly planted seedlings (Urretevizkaya et al., 2019; Castro et al., 2011). Our results demonstrated a reduction in the amount of PAR within the B12y site when standing trees were present, compared to outside without standing dead trees. This is consistent with the findings of Urretevizkaya et al. (2019) and Castro et al. (2011). Nonetheless, our findings indicated that there was a distinct microclimate present within the B12y site, which was characterized by higher maximum daily temperature and VPD, and maximum wind speed compared to outside without standing dead trees. Interestingly, the standing dead trees appear to create an unfavorable microclimate for seedlings, despite the increase in shading observed. This may be due to standing dead trees impeding the cooling effect of the wind within the B12y site, which could cause an increase in temperature and, hence, in VPD. Increased VPD will result in reduced stomatal conductance and reduced xylem water potential, especially in low soil moisture conditions. This reduction in stomatal conductance and xylem water potential, can lead to excessive xylem embolism that may lead to hydraulic failure and/or carbon starvation, especially in water-stressed seedlings (Kramer and Boyer, 1995; Will et al., 2013).

Considering the non-fire conditions, biological activity in soil enhances nutrient cycling and synthesizing cleating agents, and decomposing organic matter, but also storing the nutrients and releasing it slowly. However, fire can rapidly increase the decomposition of organic matter rate (DeBano et al., 1998). The level of organic matter decreased when burned and was also reduced when heated at temperatures of 300 ^{o}C and higher (Stoof et al., 2010). During a fire, nitrogen and sulfur are easily lost from the soil due to their low volatilization temperatures (200-300 ^{o}C). On the other hand, metal cations such as calcium, magnesium, potassium, and sodium are much more resistant to volatilization and will only burn off at much higher temperatures. Phosphorus can be released into air at temperatures between 550 and 750 (Alauzis et al., 2004). Our results indicated that all burned sites showed reduced SOM, CEC, magnesium, and calcium, similar sulfur, and higher nitrate, phosphorus and potassium levels compared to the UB site.

SOM has a significant impact on water retention and is considered a key component in determining the soil's water holding capacity (Kramer and Boyer, 1995; Libohova et al., 2018). Multiple studies have indicated that soil that has been burned holds less water compared to unburned (Kitzberger et al., 2005; Silva et al., 2006). The results of this study indicate that the reduction of SOM caused by fire negatively affected the AWHC by reducing the upper limit of soil water retention (WHC-UL and AWHC at the UB site were higher than at the burned sites). We found that a 1% decrease in SOM causes a decrease of 1.67 $cm^3\ cm^{-3}$ in WHC-UL. This result highlights the importance of maintaining healthy levels of SOM in the soil. Furthermore, the observed correlation between AWHC and the WHC-UL suggests that they are interdependent variables, and managing one variable can positively impact the other. Lastly, the correlation between WHC-LL and content of clay content in the soil suggest that soil texture is also an important factor to consider when evaluating WHC. Implications of changes in soil nutrient availability on seedling growth in Chapter 4.

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3 Wildfire and Reforestation Treatments Effects on Early-Seral Vegetation

3.1 Introduction

Understanding how plant species respond to fire is crucial for predicting the impacts of wildfires on plant communities. The response of individual plant species to fire depends on their ability to regenerate from seeds or through resprouting (Pyke et al., 2010). Some plant species can evade fire mortality by being dormant during the regular fire season or by having perennating buds positioned to protect them from lethal temperatures. Others can regenerate from seeds that have dispersed and found safe sites unaffected by the lethal temperatures caused by fires (Ditomaso et al., 2006).

Plants that can withstand fires and rapidly regrow in burned areas often gain advantages from post-fire environments. The changes in soil nutrients after a fire depend on the intensity of the fire. High-intensity fires can cause the loss of nutrients through volatilization or runoff and soil erosion, while lower-intensity fires may increase the availability of nutrients for plant growth (Neary et al., 1999). Furthermore, low-intensity fires may produce a surge of soil nitrate and ammonium that becomes readily available for plants to uptake (Wan et al., 2001; Stubbs & Pyke, 2005).

Plants that survive fires, particularly dormant ones with protected buds, tend to face less competition in the fire's immediate aftermath. This is because fire-sensitive species die off, and many surviving species are reduced in size. Therefore, fewer plants compete for resources after the fire, allowing the surviving plants to grow and thrive with less interference from other plants (Pyke et al., 2010). Fires may cause the death of most individuals within a species, and the species itself may be able to re-establish quickly through seed germination (Pyke et al., 2010). However, species populations with seeds that have a short lifespan of two years or less are more vulnerable to experiencing a decline in population numbers due to fires than species with long-lived seeds. Populations of species with short-lived seeds can be decreased by fires or herbicide treatments, especially if adult plants are killed before they can reproduce (DiTomaso et al., 1999; D'Antonio et al., 2001).

The vulnerability of annual plants to fires typically relies on the season and intensity of the fire. Most herbaceous perennials, woody plants, and annual plants complete their life cycle well before the natural occurrence of fires. As such, fires usually occur during these species' dormant seasons, making annual plants more susceptible to fire damage (Pyke et al., 2010). The pre-existing plant composition before a fire can significantly impact the subsequent plant succession after the fire. However, this is only one of the many crucial variables, and all those factors can interact. These variables include fire severity, pre and post-fire weather, post-fire disturbance, ecological site resilience, and resistance to invasives (Miller et al., 2013).

The objective of this research project is to investigate the effect of pre-wildfire stand age/structure on post-fire early seral vegetation community dynamics. By studying the early seral vegetation community dynamics, this research project can provide valuable insights into the effects of pre-wildfire stand age/structure on the post-fire plant community's composition and structure.

3.2 Methods

3.2.1 Study Area

The present study was carried out in a common physiographic area affected by the Archie Creek Fire in Southwest Oregon, near Roseburg. Four stands were selected for this study. The study aimed to ensure that the site conditions across these stands, including soil type, site index, aspect, slope, and burning severity, remained as homogeneous as possible. The selected stands comprised different pre-fire stand ages/structures: unburned (UB), burned when the stand was recently planted (1-year-old; B1y), burned when the stand was mid-rotation and non-merchantable (12-year-old; B12y), and burned when the stand was mature and merchantable, and salvage harvest was carried out (55-year-old; B55y). The coordinates and elevation of the study sites are presented in Table 5.

Table 5: Sites location and elevation

Sites	Burning Condition	Stand Age at Burning	Latitude (N)	Longitude (W)	Altitude (ft)
UB	Unburned	N.A.	$43^{o}24`12$	$123^{o}06`05$	1250
B1y	Burned	1 year	43^{o} 22`54	$123^{o}04`43$	2650
B12y	Burned	12 years	43^{o} 22`55	$123^{o}04`45$	2750
B55y	Burned	55 years	43^{o} 23`40	$123^{o}05`27$	1750

The study was conducted in a region characterized by a Mediterranean climate with warm and dry summers and cold and wet winters, resulting in a mean annual temperature of 10 ^{o}C and total annual rainfall of 986 mm. The majority of precipitation occurs between October and April. The study sites are situated within an elevation range of 1250 to 2750 ft. The soil texture is consistent across sites, consisting of silty-clay-loam, with no significant differences observed in particle size distribution ($P>0.009$; data not shown). The locations of the study sites are depicted in Figure 8, and the furthest distance between the most distant sites (UB and B12y) was 2 miles.

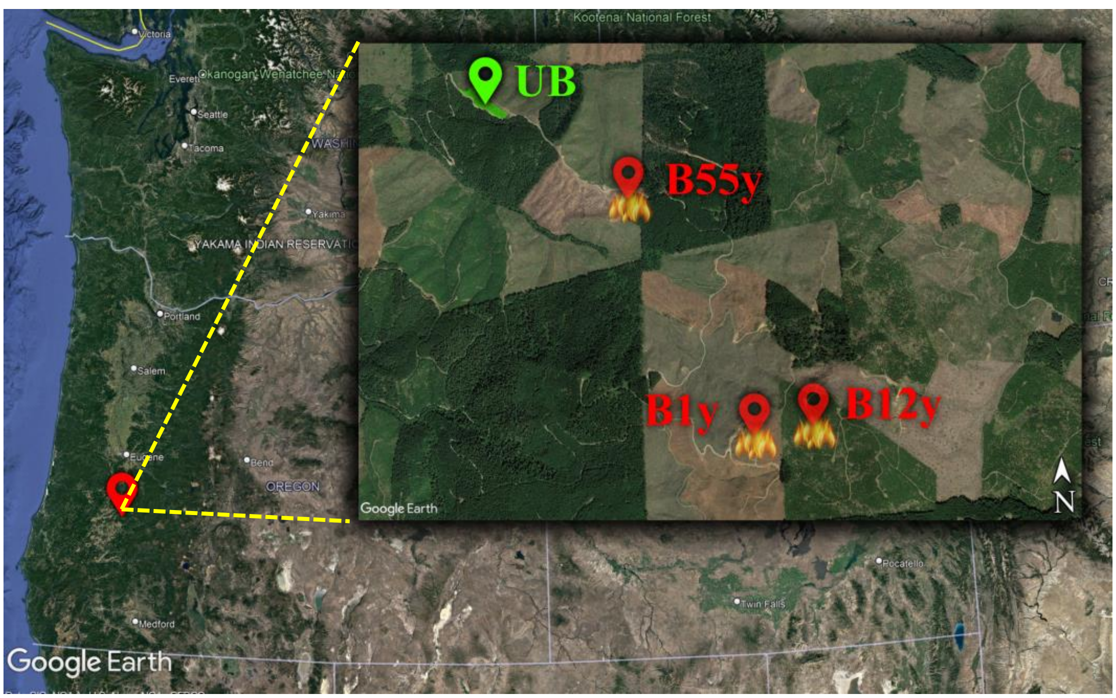

Figure 8: Location of the four sites selected for this study: Unburned (UB), burned when the stand was 1 year old (B1y), burned when the stand was 12 years old (B12y), burned when the stand was 55 years old (B55y) (Google Earth Pro 6.2.1.6014 (beta)).

3.2.2 Study Design

In February 2021, a total of 24 plots, each comprising 36 seedlings planted at 10x10 ft spacing, were installed across the four selected stands, which measured 60 x 60 ft per plot. A randomized complete block design was employed within each stand, consisting of six reforestation treatments and four blocks, totaling 24 plots per site. Notably, at the B12y site, the diameter at the breast height (DBH) of all standing dead trees was recorded, and the block assignment was carried out based on the Basal Area (BA) of each plot after ranking all plots based on BA.

The six reforestation treatments were composed of a combination of vegetation management (VM) and delayed planting. The VM treatments included the control (no action), fall-site preparation (FSP), and spring-release (SR) applications. Meanwhile, the delayed planting comprised of two options - planting either the winter following the fire's cessation (2021) or the winter of the subsequent year (2022). It is worth mentioning that the delayed planting option is commonly used due to logistical constraints such as the need for seedlings and workforce. Nonetheless, it also presents an opportunity to enhance the effectiveness of herbicide treatments by increasing foliage cover for FSP treatments.

Table 6: Description of treatments applied at each study site.

Treatment	Planting Year	FSP	SR
C	2021	0	0
SR	2021	0	1
C.D	2022	0	0
SR.D	2022	0	1
FSP.D	2022	1	0
FSP.D+SR.D	2022	1	1

C: no-action control; FSP: fall-site preparation; SR: spring-release; D: delayed planting

In March of 2021, two plots within each block were designated for planting, with the C and SR treatments being implemented. Following this, post-planting herbicide application was conducted for the SR plot in April 2021. In September of the same year, pre-planting herbicide was applied to each plot assigned for delayed planting (FSP.D). In February 2022, the four plots assigned for delayed planting within each block were planted using the same stock type utilized the previous year. Finally, post-planting herbicide application was carried out for the SR.D plots in April 2022. The rates and dates of herbicide applications are detailed in Table 7.

Table 7: Herbicide tank mixes for the vegetation management treatments applied (all doses are per acre basis). Total application amount (chemical + water): 6.5 gal/acre.

Treatment ID	Date	Products
SR	4/27/2021	1.5 lbs Velpar
FSP.D	8/30/2021	7 oz Esplanade
SR.D	4/7/2022	80 oz Velossa 10 oz Transline

3.2.3 Vegetation Surveys

In order to evaluate the early-seral vegetation, measurements were taken at five permanent survey points per measurement plot in July of both 2021 and 2022. The assessments included the measurement of height (cm) and visual estimates of cover (%) by species. Each species of vegetation was classified based on its growth form, which included forbs, ferns, graminoids, shrubs, trees, and brambles (Rubus species). Furthermore, all species from all growth forms were combined to determine the total (native and introduced) and native species richness, which represents the total number of species in all five survey points per plot, and abundance, which is the average total cover of all individuals across all five survey points per plot.

3.2.4 Statistical Analysis

Statistical Analysis Software version 9.4 (SAS Institute Inc. Cary, NC) was used for all statistical analysis. Analysis of variance, including Tukey adjustments, was used to test the effects of site and reforestation treatment on vegetation community cover and species richness (PROC MIXED). All figures were produced using SigmaPlot version 14 (Systat Software, Inc. San Jose, CA).

3.2.5 Results

3.2.6 Cover (%) of Total (Native and Introduced) Early Seral Vegetation

Table 8 presents the ANOVA results for early-seral vegetation cover and species richness, with the mean abundance and species richness of native and introduced early-seral vegetation categorized by growth form. The table reports the findings from the first (2021) and second (2022) growing seasons after planting for various VM treatments, including the no-action control (C), spring release (SR), control delayed (CD), spring release delayed (SR.D), fall site preparation delayed (FSP.D), and the combination of fall site preparation delayed and spring release delayed (FSP.D+SR.D) plots, across different sites.

Table 8: P-values of the effects of Site, VM Treatment and their interaction on abundance (cover, %) and species richness of total (native + introduced) and only native early-seral vegetation by growth habit during first (2021) and second (2022) growing season after planting.

Growth Form	Variable	Type of Vegetation	Year	Site	VM Treatment	Site x VM Treatment
Forb	Cover	Introduced + Native	2021	0.080	0.028	0.408
			2022	0.005	<0.001	0.148
		Native	2021	0.034	0.049	0.480
			2022	0.002	<0.001	0.139
	Species Richness	Introduced + Native	2021	0.020	0.076	0.304
			2022	0.006	<0.001	0.390
		Native	2021	0.026	0.032	0.247
			2022	0.002	<0.001	0.480
Fern	Cover	Introduced + Native	2021	0.010	0.028	0.408
			2022	0.022	0.020	0.180
		Native	2021	0.010	0.124	0.453
			2022	0.022	0.020	0.180
	Species Richness	Introduced + Native	2021	0.034	1.000	0.304
			2022	<0.001	0.112	0.662
		Native	2021	0.034	1.000	0.817
			2022	<0.001	0.112	0.662
Graminoid	Cover	Introduced + Native	2021	0.784	0.122	0.546
			2022	0.004	<0.001	0.116

		Native	2021	0.023	0.249	0.110
			2022	0.879	0.079	0.624
	Species Richness	Introduced + Native	2021	0.110	0.046	0.103
			2022	0.212	0.001	0.815
		Native	2021	<0.001	0.016	0.288
			2022	0.307	0.121	0.977
Brambles	Cover	Introduced + Native	2021	0.435	0.637	0.243
			2022	0.001	<0.001	0.036
		Native	2021	0.330	0.667	0.219
			2022	<0.001	<0.001	0.007
	Species Richness	Introduced + Native	2021	0.015	0.786	0.697
			2022	0.001	<0.001	0.004
		Native	2021	<0.001	0.698	0.922
			2022	0.006	<0.001	<0.001
Shrub	Cover	Introduced + Native	2021	0.001	0.094	0.244
			2022	<0.001	0.004	<0.001
		Native	2021	0.001	0.094	0.244
			2022	<0.001	0.005	<0.001
	Species Richness	Introduced + Native	2021	<0.001	0.016	0.288
			2022	<0.001	0.021	<0.001
		Native	2021	<0.001	0.016	0.288
			2022	<0.001	0.021	<0.001
Total	Cover	Introduced + Native	2021	0.0041	0.009	0.4352
			2022	0.0002	<0.001	0.0130
		Native	2021	0.0007	0.0152	0.1131
			2022	<0.0001	<0.001	<0.0001
	Species Richness	Introduced + Native	2021	0.0045	0.0219	0.2025
			2022	0.0080	<0.001	0.5356
		Native	2021	0.0003	0.001	0.0494
			2022	0.0005	<0.001	0.2903

During the first year after the fire (2021), the early-seral vegetation cover, including both total (native) and total (native + introduced species), was significantly affected by site and VM treatments ($P<0.015$), with no significant interaction between these factors ($P>0.113$). Furthermore, the cover of forb species (both native and native + introduced) and native fern species were also significantly influenced by site and VM treatments (P<0.049), without any significant interaction between these effects (P>0.244) (Table 8).

During the second year after the fire (2022), the analysis showed significant effects of both site and VM treatments on the cover of total (native) and total (native + introduced) early-seral vegetation cover (P<0.001), and the interaction of these effects was significant (P<0.013). Additionally, there was a significant site and VM treatments effect on the cover of forb (native and native + introduced), fern (native and native + introduced), and graminoid (native + introduced) species (P<0.022). However, no significant interaction was observed (P>0.116). Moreover, significant site and VM treatment effects on the cover of brambles (native and native + introduced) and shrub (native and native + introduced) species were observed (P<0.036), and the interaction of these effects was significant (P<0.036) (Table 8).

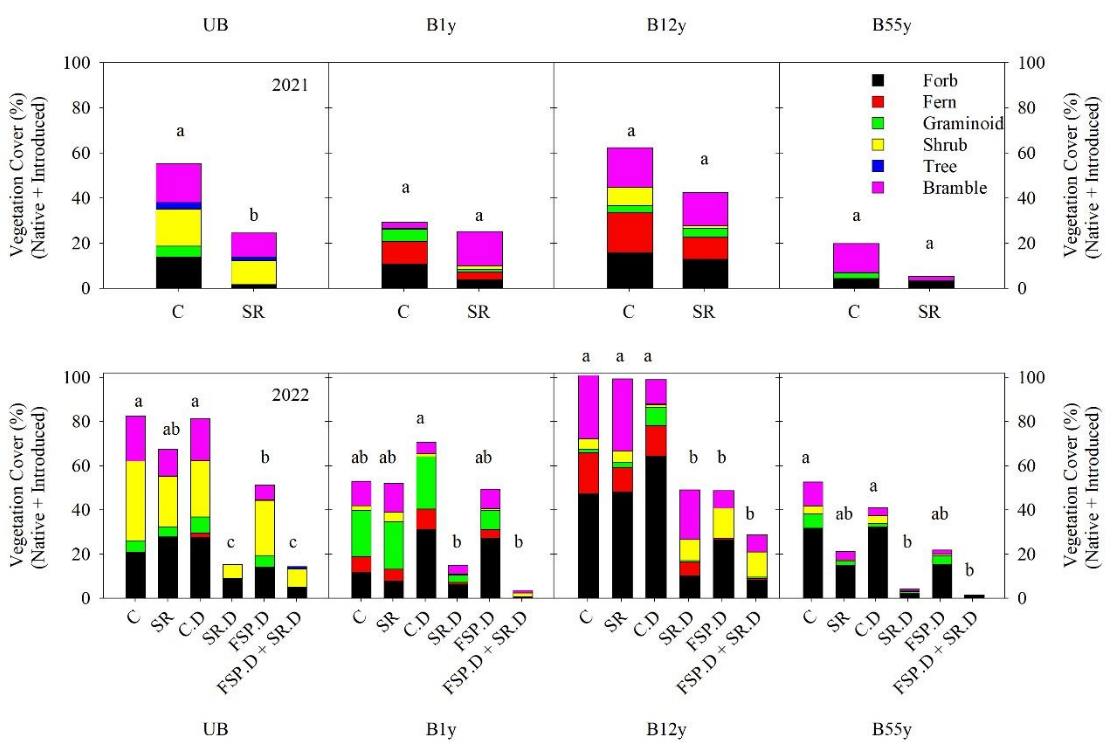

Figure 9: Total (native and introduced) vegetation cover (%) under different VM treatments at UB, B1y, B12y, and B55y sites in years 2021 (upper panel) and 2022 (bottom panel). Letters within a panel (year) represent statistical differences at α=0.05 across treatments within each site.

During the first year following the fire (2021), the cover of total (native + introduced) early-seral vegetation varied between 5.3% and 62.4%. Among the VM treatments, plots treated with C at UB, B1y, B12y, and B55y showed a greater cover of total (native + introduced) early-seral vegetation than those treated with SR. Specifically, the percentage of total (native + introduced) early-seral vegetation cover for C treated plots were 43.5%, 15.5%, 49.8%, and 11.6%, respectively (Figure 9). However, there was no significant difference in total (native + introduced) early-seral vegetation cover between C and SR treated plots at all sites ($P>0.213$) (Table 10).

During the second year after the fire (2022), the cover of total (native + introduced) early seral vegetation varied from 1.4% to 100.8%. The C.D treated plots at UB, B1y, B12y, and B55y had a larger cover of (total + introduced) early seral vegetation than SR.D, FSP.D, and FSP.D+SR.D treated plots. Specifically, the percentage of the total (native + introduced) early seral vegetation cover for C.D treated plots was 81.2%, 70.6%, 98.8%, and 40.9%, respectively (Figure 9). Moreover, a significant difference was observed between SR.D, FSP.D+SR.D, and C.D treated plots at all sites ($P<0.001$) (Appendix 7).

3.2.7 Cover (%) of Total (Native) Early Seral Vegetation

During the first year following the fire (2021), the cover of total (native) early-seral vegetation ranged between 2.5% and 49.8%. Notably, the C treated plots at UB, B1y, B12y, and B55y exhibited a greater cover of total (native + introduced) early seral vegetation than the SR treated plots. Specifically, the percentage of the total (native + introduced) early seral vegetation cover for C treated plots was 55.3%, 29.3%, 62.4%, and 19.7%, respectively (Figure 10). There was no significant difference in total (native) early seral vegetation cover between C, and SR treated plots at all sites ($P> 0.213$) (Appendix 8).

During the second year of after fire (2022), cover of total (native) early seral vegetation ranged between 0.13% and 77.8%. Compared to SR.D, FSP.D, and FSP.D+SR.D treated plots, the C.D. treated plots at UB, B1y, B12y. and B55y had larger cover of total (native) early seral vegetation. Specifically, the percentage of total (native) early seral vegetation cover for C.D. treated plots were 68.0%, 28.3%, 57.6% and 10.8%, respectively (Figure 10). A significant difference between SR.D and C.D treated plots was only observed at UB (P<0.001) while cover of total (native) early seral vegetation on FSP.D+SR.D treated plots differed from C.D. treated plots at UB and B12y (P<0.009) (Appendix 8).

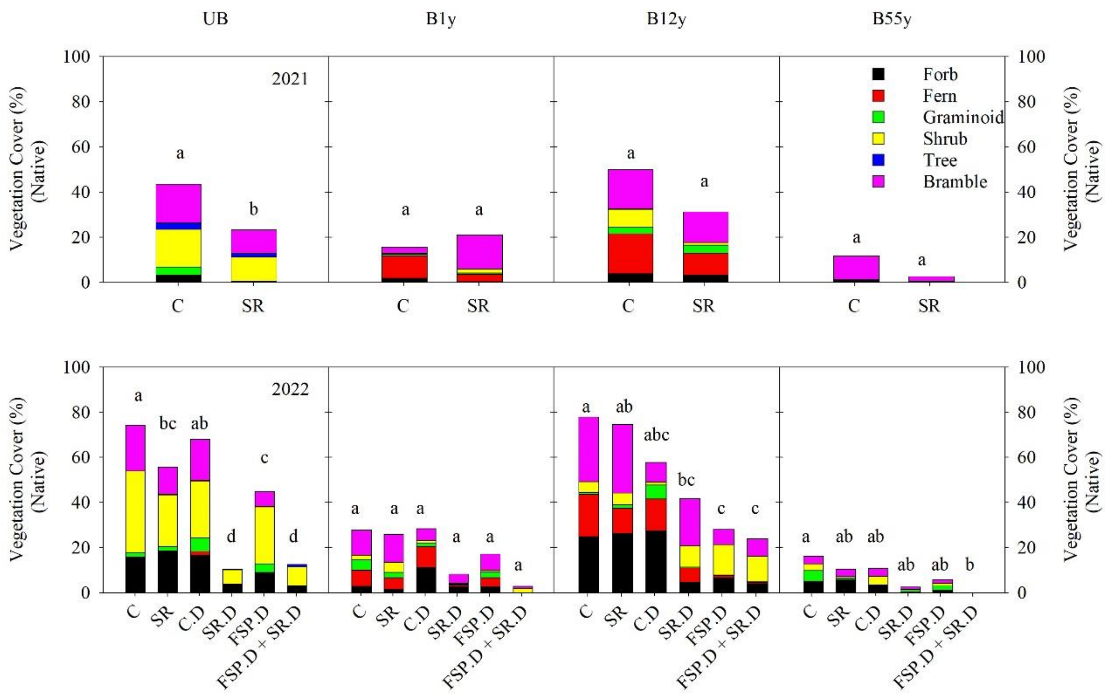

Figure 10: Native vegetation cover (%) under different VM treatments at UB, B1y, B12y, and B55y sites in years 2021 (upper panel) and 2022 (bottom panel). Letters within a panel (year) represent statistical differences at α=0.05 across sites and treatments.

3.2.8 Species Richness of Total (Native and Introduced) Early Seral Vegetation

During the first year after fire (2021), there was significant site and VM treatments effect on species richness of total (native and native + introduced) (P<0.021) early-seral vegetation (Table 8). The interaction of these effects on species richness of total (native + introduced) was not significant (P=0.202) while the interaction of these effects on species richness of total (native) was significant (P=0.049). In addition, significant site and VM treatments effect was observed on species richness of forb (native), graminoid (native), and shrub (native and native +

introduced) (P<0.032), but this interaction was non-significant (P>0.247) (Appendix 7 and 11).

During the first year following the fire (2021), the species richness of total (native and native + introduced) early-seral vegetation was significantly affected by both site and VM treatments (P<0.021), as reported in Table 8. The interaction between these effects on the species richness of total (native + introduced) was not significant (P=0.202), but it was significant on the species richness of total (native) (P=0.049) (Table 8). Furthermore, significant site and VM treatment effects were also observed on the species richness of forb (native), graminoid (native), and shrub (native and native + introduced) (P<0.032), although the interaction between these effects was not significant (P>0.247) (Appendix 7 and 11).

During the first year after the fire (2021), the species richness of total (native and native + introduced) early-seral vegetation exhibited significant variations across the study sites, ranging between 7 and 24. Notably, the C. treated plots at UB, B1y, B12y, and B55y demonstrated a greater number of total (native + introduced) species richness compared to the SR. treated plots. Specifically, the number of total (native + introduced) early-seral vegetation species richness for C. treated plots were 24, 10, 22, and 8, respectively (Figure 11). A significant site and VM treatments effect was observed on species richness of total (native and native + introduced) early-seral vegetation (P<0.021) (Table 8). Moreover, while the interaction of these effects on species richness of total (native + introduced) was not significant (P=0.202), the interaction of these effects on species richness of total (native) was significant (P=0.049). Additionally, there was a significant site and VM treatments effect on species richness of forb (native), graminoid (native), and shrub (native and native + introduced) (P<0.032). However, this interaction was not significant (P>0.247) (Appendix 7 and 11).

During the second year after the fire (2022), the species richness of total (native + introduced) early seral vegetation ranged from 4 to 21. The C.D treated plots at UB, B1y, B12y, and B55y had a significantly greater number of total (native + introduced) species richness compared to the SR.D treated plots. Specifically, the

C.D treated plots had 23, 13, 21, and 19 total (native + introduced) early seral vegetation species richness, respectively (Figure 11).

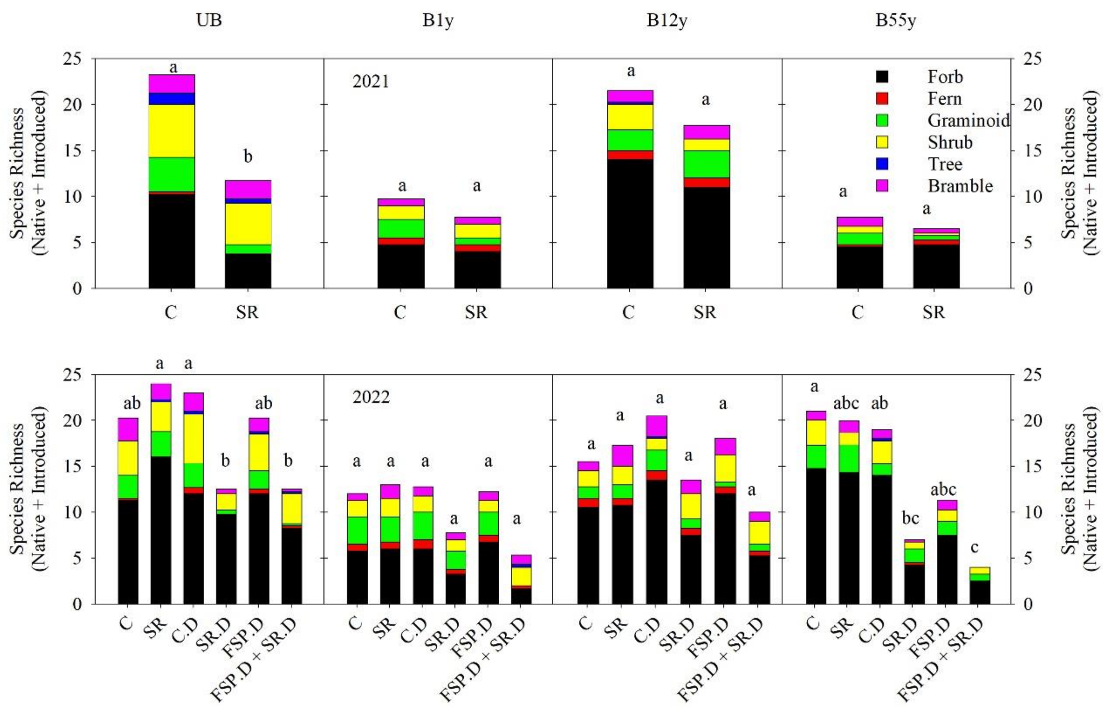

Figure 11: Total (native and introduced) vegetation species richness under different VM treatments at UB, B1y, B12y, and B55y sites in years 2021 (upper panel) and 2022 (bottom panel). Letters within a panel (year) represent statistical differences at α=0.05 across sites and treatments.

3.2.9 Species Richness of Total (Native) Early Seral Vegetation

During the first year after fire (2021), species richness of total (native) early seral vegetation ranged between 3 and 16. Compared to SR. treated plots, the C. treated plots at UB, B1y, B12y, and B55y had a greater number of total (native) species richness. Specifically, the number of total (native) early seral vegetation species richness for C. treated plots were 16, 6, 12 and 4, respectively (Figure 12). A significant difference on total (native) early seral vegetation species richness was observed between C. and SR treated plots at UB (P=0.009). There was no other significant difference on total (total + introduced) early seral vegetation species richness between C. and SR. treated plots at B1y, B12y and B55y (P> 0.636) (Appendix 8).

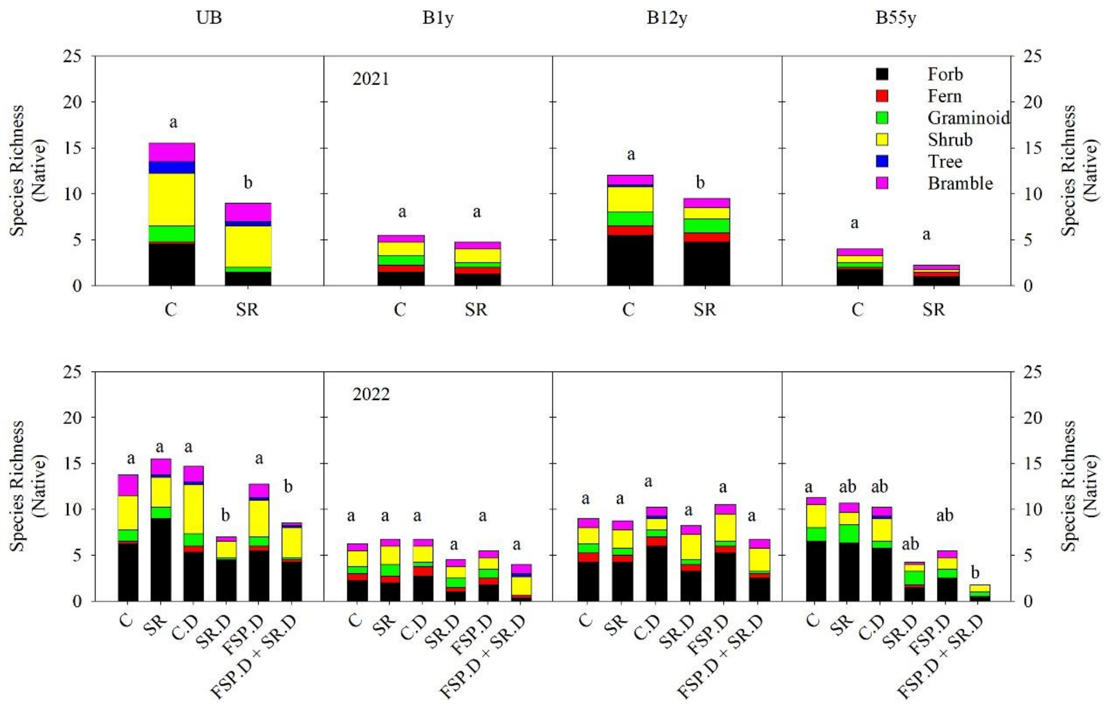

Figure 12: Native vegetation species richness under different VM treatments at UB, B1y, B12y, and B55y sites in years 2021 (upper panel) and 2022 (bottom panel). Letters within a panel (year) represent statistical differences at α=0.05 across sites and treatments.

During the second year after fire (2022), species richness of total (native) early seral vegetation ranged between 2 and 16. Compared to SR.D treated plots, the C.D treated plots at UB, B1y, B12y, and B55y had a greater number of total (native) species richness. Specifically, the number of total (native) early seral vegetation species richness for C.D treated plots were 15, 7, 11 and 11, respectively (Figure 12). There was no significant difference on total (total) early seral vegetation species richness between C.D and SR.D, FSP.D and FSP.D + SR.D treated plots at UB, B1y, B12y and B55y (P> 0.057) (Appendix 8).

3.2.10 Total vegetation (Native + Introduced) by Growth Form

Table 9 shows mean values of cover and species richness for total (native + introduced) and native early-seral vegetation categorized by growth form, in the no-action control (C. and C.D) plots across all sites during first two years after planting. This table provides an overview of the species richness and abundance of the habit-specific vegetation community at each site. The P-values for SR in both 2021 and 2022 are based on a comparison with C. plots. On the other hand, the P-values for SR.D, FSP.D and FSP.D+SR.D are based on a comparison with C.D plots.

Table 9: Mean abundance (cover, %) and species richness of total (native + introduced) and native early-seral vegetation by growth habit at the no-action control (C and C.D) plots for each site during first growing season after planting. UB: Unburned; B1y: Burned at 1-year-old; B1y: Burned at 12 years-old; B55y: Burned at 55 years-old; C: No herbicide, planted in winter after fire (2021); C.D: No herbicide, delayed planted in winter of following year after fire (2022). Letters represent statistical differences at α=0.05 across sites.

Habit Origin	Trait	Site	Forbs		Graminoids		Ferns		Brambles		Shrubs + Tree	
			2021	2022	2021	2022	2021	2022	2021	2022	2021	2022
Introduced + Native	Cover	UB	13.9 a	23.6 ab	4.7 a	6.2 ab	0.1 b	0.9 a	17.2 a	19.8 a	19.5 a	31.6 a
		B1y	10.9 a	21.4 ab	5.4 a	22.3 a	10.0 ab	8.2 a	2.8 a	8.2 a	0.3 b	1.8 b
		B12y	15.9 a	55.7 a	3.0 a	4.9 b	17.8 a	16.3 a	17.6 a	19.9 a	8.2 ab	3.0 b
		B55y	4.2 a	32.0 b	2.4 a	3.9 b	0.1 b	0 a	12.8 a	7.2 a	0.2 b	3.7 b
	Species Richness	UB	10.3 a	11.6 a	3.8 a	2.6 a	0.3 a	0.4 bc	2.0 a	2.3 a	7.1 a	4.5 a
		B1y	4.8 a	5.9 a	2.0 a	3.0 a	0.8 a	0.9 ab	0.8 a	0.9 b	1.5 c	1.8 b
		B12y	14 a	12.0 a	2.3 a	1.8 a	1.0 a	1.0 a	1.3 a	1.6 ab	3.1 b	1.6 b
		B55y	4.5 a	14.4 a	1.3 a	1.9 a	0.3 a	0.0 c	1.0 a	1.0 b	0.8 c	2.7 b
Native	Cover	UB	3.3 a	16 a	3.5 a	3.7 a	0.1 b	0.9 a	17.2 a	19.3 a	19.5 a	31.6 a
		B1y	1.8 a	7.0 ab	0.7 a	3.0 a	10.0 ab	8.2 a	2.8 a	8.1 a	0.3 b	1.8 b
		B12y	3.8 a	26.3 b	2.7 a	3.5 a	17.8 a	16.3 a	17.4 a	18.6 a	8.2 ab	3.0 b
		B55y	0.7 a	4.0 b	0.1 a	2.7 a	0.1 b	0 a	10.6 a	3.4a	0.2 b	3.4 b
	Species Richness	UB	4.5 a	5.9 a	1.8 a	1.3 a	0.3 a	0.4 bc	2.0 a	2.0 a	7.1 a	4.5 a
		B1y	1.5 a	2.5 a	1.0 a	0.6 a	0.8 a	0.9 ab	0.8 b	0.8 b	1.5 c	1.8 b
		B12y	5.5 a	5.1 a	1.5 a	0.9 a	1.0 a	1.0 a	1.0 ab	1.0 ab	3.1 b	1.6 b
		B55y	1.8 a	6.1 a	0.5 a	1.1 a	0.3 a	0 c	0.8 b	0.9 ab	0.8 c	2.6 b

3.3 Early Seral Vegetation Abundance (Cover) by Growth Forms

3.3.1 Forbs (Native + Introduced) Vegetation Abundance

During the first year after the 2021 fire, the total (native + introduced) forb cover on the no-action C plots ranged from 4.2 to 15.9%. At the UB site, the SR treatment resulted in lower cover compared to the C treatment (P=0.245), while no significant difference was observed between C and SR treatment at all other burned sites (P>0.815) (Appendix 7). Furthermore, no significant differences were observed in total forb cover between UB (13.9%), B1y (10.9%), B12y (15.9%), and B55y (4.2%) (P>0.445) (Table 9).

During the second year after the fire (2022), total (native + introduced) forb cover on the no-action CD plots ranged from 21.4 to 55.7%. Notably, total forb cover on CD plots at the B12 site was significantly higher than that of UB (P=0.043) and B1y (P=0.028). At B12y, the SR.D treatment resulted in lower cover compared to the CD treatment (P<0.001), while there was no significant difference in forb cover between CD and SR.D treatment at the UB site (P=0.995). Furthermore, the FSP.D. treatment led to less vegetation cover compared to CD treatment at the B12y site (P<0.077), while there was no significant difference in forb cover between FSP.D. and CD treatment at UB (23.6%), B1y (21.4%), and B55y (32.0%) (P>0.992) (Table 9 and 10).

3.3.2 Graminoids (Native + Introduced) Vegetation Abundance

During the first year after the fire (2021), total (native + introduced) graminoid cover on the no-action C. plots ranged from 2.4% to 5.4%. No differences were observed between UB (4.7%), B1y (5.4%), B12y (3.0%), and B55y (2.4%) (P>0.912). During the second year after the fire (2022), the total (native + introduced) graminoids cover on the no-action CD plots ranged from 1.3% to 23.8% (Table 9). SR.D treatment resulted in lower cover than the CD treatment at B1y (P=0.019), while no difference in graminoid cover was observed between CD and SR.D treatment at the UB (7.4%), B12y (8.4%), and B55y (1.3%) (P>0.995). In addition, FSP.D and FSP.D+SR.D treatment showed lower cover than CD treatment only at B1y (P<0.077) (Table 9 and Appendix 7).

3.3.3 Ferns (Native + Introduced) Vegetation Abundance

During the first year after the fire (2021), the total (native + introduced) ferns cover ranged from 0.1% to 17.8%. There was a difference between UB (0.1%) and B12y (17.8%) (P=0.023). In addition, the total (native + introduced) ferns cover at B12y differed from B55y (0.1%) (P=0.022). During the second year after the fire (2022), total (native + introduced) ferns cover on the no-action CD plots ranged from 0% to 16.3%. There was no difference between VM-treated plots and CD plots at all sites (P>0.251) (Table 9 and 10) .

3.3.4 Brambles (Native + Introduced) Vegetation Abundance

During the first year after the fire (2021), total (native + introduced) brambles cover on the no-action C. plots ranged from 2.8% to 17.6%. No difference was observed between C. and SR. treatments at all sites (P>0.738). There was no difference in brambles cover observed between UB (17.6%) and B1y (2.8%), B12y (17.6%), and B55y (7.2%) (P>0.541). During the second year after the fire (2022), total (native + introduced) brambles cover on the no-action C. plots ranged from 7.2% to 19.8%. No differences were observed in brambles cover between CD and VM treatments at the B1y (5.2%), B12y (11.1%), and B55y (3.5%) (P>0.226) (Table 9 and Appendix 7).

3.3.5 Shrubs (Native + Introduced) Vegetation Abundance

During the first year after the fire (2021), total (native + introduced) shrubs cover on the no-action C. plots ranged from 1.8% to 31.6%. No difference was observed between C. and SR. treatments at all sites (P>0.445). The UB site showed higher shrubs cover (19.5%) than B1y (0.3%), B12y (8.2%), and B55y (0.2%) (P<0.001). During the second year after the fire (2022), the total (native + introduced) shrubs cover on the no-action CD plots ranged from 1.8% to 31.6%. SR.D. and FSP.D+SR.D treatments resulted in lower shrub cover than CD treatment at UB (P<0.037), while no differences were observed in shrub cover than between CD and SR.D and FSP.D+SR.D treatments at the B1y (1.8%), B12y (3.0%) and B55y (3.7%) (P>0.286) (Table 9 and Appendix 7).

3.4 Early Seral Vegetation Species Richness by Growth Forms

3.4.1 Forbs (Native + Introduced) Species Richness

During the first year after the fire (2021), the number of total (native + introduced) forb species on the no-action C. plots ranged from 5 to 11. No differences were observed between UB (11) and B1y (5), B12y (14), and B55y (5) (P>0.059). During the second year after the fire (2022), the number of total (native + introduced) forb species richness on the no-action CD plots ranged from 6 to 15. FSP.D+SR.D treatment resulted in lower species richness than CD treatment at B55y (P=0.032) (Table 9 and Appendix 7).

3.4.2 Graminoids (Native + Introduced) Species Richness

During the first year after the fire (2021), the number of total (native + introduced) graminoid species on the no-action C. plots ranged from 3 to 6. No significant differences were observed in the graminoid species richness between UB (5) and B1y (6), B12y (3), and B55y (3) (P>0.124). During the second year after the fire (2022), total (native + introduced) graminoid species richness on the no-action CD plots ranged from 2 to 3. No significant differences were observed between CD and VM treatments at all sites (P>0.565) (Table 9 and Appendix 7).

3.4.3 Ferns (Native + Introduced) Species Richness

During the first year after the fire (2021), the number of total (native + introduced) fern species were observed on average on the no-action C. plots. No significant differences were observed in the fern species richness between UB and B1y, B12y, and B55y (P>0.190). During the second year after the fire (2022), total (native + introduced) fern species richness was 1 on average on CD plots. There was no significant difference between CD and VM treated plots (P>0.800) (Table 9 and Appendix 7).

3.4.4 Brambles (Native + Introduced) Species Richness

During the first year after the fire (2021), the number of total (native + introduced) brambles species ranged from 1 to 2 on the no-action C. plots. No significant differences were observed in the bramble species richness between UB and B1y, B12y, and B55y (P>0.178). During the second year after the fire (2022), total (native + introduced) bramble species richness ranged from 1 to 3 on CD plots. FSP.D+SR.D and SR.D treated plots differed from CD plots at UB (P<0.069). There was no other significant difference between CD and VM treated plots at burned sites (P>0.209) (Table 9 and Appendix 7).

3.4.5 Shrubs (Native + Introduced) Species Richness

During the first year after the fire (2021), the number of total (native + introduced) shrub species ranged from 1 to 8 on the no-action C. plots. Significant differences in shrub species richness were observed between UB (8) and B1y (2), B12y (4), and B55y (1) (P<0.002). In addition, the number of shrub species in B12y was different from B1y and B55y (P<0.037). During the second year after the fire (2022), total (native + introduced) shrub species richness ranged from 2 to 5 on C.D plots. SR.D treated plots differed from C.D plots only at UB (P<0.001). There was no other significant difference between C.D and VM-treated plots at burned sites (P>0.209) (Table 9 and Appendix 7).

3.5 Early Seral Native Vegetation Abundance (Cover) by Growth Forms

3.5.1 Forbs (Native) Vegetation Abundance

During 2021, after a fire, the percentage of native forbs covering the no-action C plots varied between 0.7% and 3.8%, with no significant differences observed between UB, B1y, B12y, and B55y. However, during 2022, the total native forb cover on the C.D. plots ranged between 4.0% and 26.3%. There were differences in the native forb cover between the B12 site and B1y and B55y at the C.D. plots (P<0.035), but no significant differences were observed between UB, B1y, B12y, and B55y (P>0.239) (Table 9 and Appendix 8).

3.5.2 Graminoids (Native) Vegetation Abundance

During the first year after the fire (2021), total (native) graminoid cover on the no-action C. plots ranged from 0.1% to 3.5%. No differences were observed between UB (3.5%), B1y (0.7%), B12y (2.7%), and B55y (0.1%) (P>0.253). During the second year after the fire (2022), total (native) graminoids cover on the no-action

C.D. plots ranged from 2.7% to 3.5%. No differences were observed in graminoid cover between C.D and VM treatments at the UB (3.7%), B1y (3.0%), B12y (3.5%), and B55y (2.5%) (P>0.802) (Table 9 and Appendix 8).

3.5.3 Ferns (Native) Vegetation Abundance

During 2021, following a fire, the total native fern cover on the C.D plots ranged from 0.1% to 17.8%. The native fern cover at UB (0.1%) was significantly different from B12y (17.8%) (P=0.022). Moreover, there were significant differences in the total native fern cover between B12y and B55y (0.1%) (P=0.023). During 2022, during the second year after the fire, the total native fern cover on the no-action C.D plots varied from 0% to 16.3%. No significant differences were observed between VM-treated plots and C.D plots at all sites (P>0.251) (Table 9 and Appendix 8).

3.5.4 Brambles (Native) Vegetation Abundance

During the first year after the fire (2021), the total native bramble cover on the no-action C plots ranged from 2.8% to 17.4%. No significant differences were observed in the native bramble cover between UB (17.2%) and B1y (2.8%), B12y (17.4%), and B55y (10.6%) (P>0.456). In 2022, during the second year after the fire, the total native bramble cover on the no-action C plots varied from 3.4% to 19.3%. No significant differences were observed in the bramble cover between the C.D. and VM treatments at UB (19.3%), B1y (8.1%), B12y (18.6%), and B55y (3.4%) (P>0.159) (Table 9 and Appendix 8).

3.5.5 Shrubs (Native) Vegetation Abundance

During the first year after the fire (2021), the total native shrub cover on the no-action C plots ranged from 0.2% to 19.5%. The total native shrub cover on C plots at UB (19.5%) was different from B1y (0.3%) and B55y (0.1%) (P<0.007). During the second year after the fire (2022), the total native shrub cover on the no-action C.D plots ranged from 3.0% to 31.6%. The total native shrub cover at UB differed from that at B1y, B12y, and B55y (P<0.001). The SR.D and FSP.D+SR.D treatments at UB resulted in lower shrub cover than the C.D treatment (P<0.037). However, no significant differences were observed in shrub cover between the C.D and SR.D and FSP.D+SR.D treatments at B1y (1.8%), B12y (3.0%), and B55y (3.4%) (P>0.281) (Table 9 and Appendix 8).

3.6 Early Seral Native Vegetation Species Richness by Growth Forms

3.6.1 Forbs (Native) Species Richness

During the first year after the fire (2021), the number of total (native) forb species on the no-action C. plots ranged from 2 to 6. No differences were observed between UB (5) and B1y (2), B12y (6), and B55y (2) (P>0.159). During the second year after the fire (2022), the number of total (native) forb species richness on the no-action C.D plots ranged from 3 to 7. No differences were observed in brambles cover between C.D and VM treatments at the UB (6), B1y (3), B12y (6), and B55y (7) (P>0.251) (Table 9 and Appendix 8).

3.6.2 Graminoids (Native) Species Richness

During the first year after the fire (2021), the number of total (native) graminoid species on the no-action C. plots ranged from 1 to 4. No significant differences were observed in the graminoid species richness between UB (4) and B1y (1), B12y (3), and B55y (1) (P>0.079). During the second year after the fire (2022), total (native) graminoid species richness on the no-action C.D. plots ranged from 3 to 4. No significant differences were observed between C.D and VM treatments at all sites (P>0.998) (Table 9 and Appendix 8).

3.6.3 Ferns (Native) Species Richness

During the first year after the fire (2021), the number of total (native) fern species on average on the no-action C. plots ranged from 0 to 1. No differences were observed in the fern species richness between UB and B1y, B12y, and B55y (P>0.190). During the second year after the fire (2022), total (native + introduced) fern species richness was 1 on average on C.D plots. There was no significant difference between C.D and VM-treated plots (P>0.800) (Table 9 and Appendix 8).

3.6.4 Brambles (Native) Species Richness

During the first year after the fire (2021), the number of total (native) brambles species ranged from 1 to 2 on the no-action C. plots and UB differed from the B1y, B12y, and B55y ($P<0.077$). During the second year after the fire (2022), total (native) bramble species richness ranged from 1 to 2 on C.D plots. FSP.D+SR.D and SR.D treated plots differed from C.D plots at UB ($P<0.062$). There was no other significant difference between C.D and VM-treated plots at burned sites ($P>0.126$) (Table 9 and Appendix 8).

3.6.5 Shrubs (Native) Species Richness

During the first year after the fire (2021), the number of total (native) shrub species ranged from 1 to 8 on the no-action C. plots. Native shrub species richness at UB (8) differed from B1y (2), B12y (4), and B55y (1) ($P<0.001$). In addition, the number of shrub species in B12y differed from B1y and B55y ($P<0.037$). During the second year after the fire (2022), total (native) shrub species richness ranged from 2 to 5 on C.D plots. SR.D treated plots differed from C.D plots only at UB ($P<0.001$). There was no other significant difference between C.D, and VM-treated plots at all sites ($P>0.241$) (Table 9 and Appendix 8).

3.7 Discussion

The results of this study provide valuable insights into the dynamics of early-seral vegetation communities in the aftermath of wildfires and the potential effects of different VM strategies. Overall, the results suggest that the early-seral vegetation (native + introduced) species abundance and richness can vary widely depending on a range of factors, including site characteristics, management practices, and time elapsed since the fire event.

One interesting finding is that during the first year after the fire, there were no significant differences in native forb cover between C and FSP plots. However, during the second year, there was a significant increase in native forb cover on the no-action C.D plots, suggesting that forb species are strongly recovering from pre-planting herbicide application without active VM. The abundance of forbs is closely linked to the amount of moisture available. The abundance of these plants varies significantly from year to year due to changes in precipitation and moisture availability across different environmental gradients (Miller et al., 2013). Studies that have examined the change in perennial forb cover or biomass in the short-term following a fire have produced mixed results (Miller et al., 2013). Several studies found that there was no significant change in the biomass or cover of forbs during the first year after a fire when compared to their levels before the burn or nearby unburned plots (Bates et al., 2011; Rhodes et al., 2010; Beck et al., 2009; West & Yorks, 2002; Fischer et al., 1996).

Nonetheless, Wrobleski & Kauffman (2003) found an increase in forbs abundance in burned plots compared to unburned plots in Southwestern Oregon. During the first year of our study, the forbs cover in C plots at the UB site was similar to B1y and B12y sites. However, during the second year, forbs abundance in C plots at the UB site was remarkably lower than the B12y site, while it was similar to B1y and B55y sites.

The abundance of graminoids typically decreases in the first year after a fire and then gradually recovers to the same level as before the burn in the second or third year (Miller et al., 2013). Our results indicate no differences in cover between C and FSP plots during the first year after the fire. However, during the second year, the application of VM treatments resulted in lower graminoid cover at some sites. Specifically, the SR.D treatment resulted in lower cover at B1y, while the FSP.D and FSP.D+SR.D treatments showed lower cover at B1y compared to the C.D treatment. In contrast, the results for native graminoid cover were more consistent over both years, with no significant differences between treatments. These results suggest that while graminoid (native + introduced) cover can recover relatively quickly after a wildfire, VM treatments can impact its subsequent growth and development. During the first year of this study (immediately after a fire), the graminoid species abundance was less than 5% in C plots at UB and burned sites. However, there was a remarkable increase in the graminoid species cover in C plots during the second year, especially at the B1y site. Several studies found a decrease in graminoid cover during the first year after a fire (Rhodes et al., 2010; Ellsworth & Kauffman, 2010; Davies & Bates, 2008; Seefeldt et al., 2007). In addition, according to West & Yorks (2002), the cover of graminoids was consistently higher on burned and ungrazed plots 5 to 18 years after a fire compared to unburned and ungrazed plots. This was consistent with what we found in this study, especially since the graminoid cover in C plots at the B1y site was remarkably higher than at the UB site during the second year.

In contrast to graminoids, ferns (native + introduced) showed significant differences in cover between sites during the first year after the fire, with the highest cover observed at B12y. During the second year, however, there were no differences between ferns, suggesting that ferns may be more resilient to wildfire and subsequent VM practices than graminoids. In addition, the cover of native ferns varied significantly between the different treatments and sites, with some sites showing a complete absence during the second year after the fire. This underscores the importance of considering site-specific factors when planning post-fire management strategies.

The results for brambles showed that during the first year after the fire, there were no differences in cover between C and FSP plots. However, during the second year, the application of VM treatments resulted in lower bramble cover at some sites. Specifically, SR.D and FSP.D+SR.D treatments resulted in lower cover at UB than C.D treatment. Bramble species, such as *Rubus* species, were observed at UB and burned sites. Such species can resprout rapidly from root crowns after a fire (Bennett, 2006). During the first and second years of study, the abundance of brambles in C plots was similar at UB and burned sites.

After a fire, one of the most significant changes in plant community structure and composition is the immediate reduction of shrubs. The successful re-establishment of shrubs depends on several factors, including severity, intensity, complexity, and size of the fire and the composition of fire-tolerant and fire-intolerant species in the aboveground vegetation and soil seed banks before the fire. The composition and abundance of seed sources after the fire, post-fire weather, and site characteristics, including soil types, are also essential to consider (Miller et al., 2013). Our results for shrubs indicated that during the first year after the fire, there were no significant differences in cover between C and FSP plots. Another important finding is that during the second year, the SR.D and FSP.D+SR.D treatments resulted in lower (native + introduced) shrub cover at UB compared to the C.D treatment. This suggests that active VM may be necessary to prevent shrub encroachment and promote the recovery of desired species.

Overall, the results of this study highlight the complexity and recovery of early-seral native and introduced vegetation abundance and species richness following a wildfire and emphasize the need for site-specific management strategies. In particular, it is essential to consider the unique characteristics of each site, including soil type, vegetation composition, and fire severity, when developing management plans. By doing so, it may be possible to promote the recovery of desired plants in the aftermath of a wildfire.

3.8 References

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4 Wildfire and Reforestation Treatments Effects on Seedling Growth and Survival

4.1 Introduction

Climate change is affecting coniferous forests worldwide, including the PNW region. One of the main effects of climate change on these forests is that they are becoming drier, which makes them more vulnerable to wildfires. These wildfires are becoming more frequent, larger, and more intense due to the aridity and other changes in the environment, exacerbating the damage caused by climate change (Abatzoglou et al., 2021; Breshears et al., 2005; Allen et al., 2015; Halofsky et al., 2020; Stephens et al., 2020). Severe and frequent fires can reduce the availability of seeds by destroying mature vegetation and can also directly harm seedlings' growth by killing off seeds, seedlings, and young trees. These negative impacts are particularly problematic in conditions of high evaporative demand, which makes it even more challenging for new plants to grow and survive after a fire (Turner et al., 2019; Tepley et al., 2017; Johnstone et al., 2016).

In the PNW, soil moisture availability is a significant factor affecting plant growth (Dinger 2010, 2009). During summer, intense competition can further reduce soil moisture availability and increase planted seedlings' water stress, which can inhibit their growth (Dinger & Rose, 2009; Zutter, 1986). Further, the volume growth of Douglas-fir seedlings was found to be significantly impacted by a decrease in soil moisture during mid-August (Gonzalez Benecke & Dinger, 2018). The primary factor influencing tree establishment and growth, and consequently, succession, in newly planted plantations and naturally regenerated forests is competition between desired trees and other plants (Balandier, 2006). To address these challenges, forest managers can employ FVM to control unwanted plant species and increase site resources to encourage the growth and establishment of desired seedlings (Eyles et al., 2012; Gonzalez-Benecke & Dinger, 2018).

In this chapter, this research project aims to determine the effect of fire, pre-wildfire stand age/structure, and FVM on Douglas-fir seedling performance and the effect of delayed planting on Douglas-fir seedling's growth and survival.

4.2 Methods

4.2.1 Study Area

The study was conducted in a region of Southwest Oregon that was affected by the Archie Creek Fire. The study's objective was to examine four specific stands in the area while ensuring that the environmental conditions of the stands, such as soil type, site index, aspect, slope, and burning severity, were as similar as possible. The four stands selected for the study had different pre-fire ages/structures, including unburned stands, stands burned when recently planted (1-year-old), stands burned when mid-rotation and non-merchantable (12-year-old), and stands burned when mature and merchantable, and were subjected to salvage harvesting (55-year-old). Table 12 shows the coordinates and elevation of the study sites.

Table 10: Sites location and elevation

Study ID	Burning Condition	Stand Age at Burning	Latitude (N)	Longitude (W)	Altitude (ft)
UB	Unburned	N.A.	43^{o}24\`12	123^{o}06\`05	1250
B1y	Burned	1 year	43^{o} 22\`54	123^{o}04\`43	2650
B12y	Burned	12 years	43^{o} 22\`55	123^{o}04\`45	2750
B55y	Burned	55 years	43^{o} 23\`40	123^{o}05\`27	1750

The current study was conducted in an area with a Mediterranean climate characterized by warm and dry summers and cold and wet winters. This results in an average annual temperature of 10 ^{o}C and an annual rainfall of 986 mm. The majority of rainfall occurs between October and April. The study sites are situated at elevations ranging from 1250 to 2750 ft. The soil texture is uniform across the sites, consisting of silty-clay-loam, with no significant variation in particle size distribution observed (P>0.009; data not displayed). Figure 8 shows the locations of the study sites, and the farthest distance between the most remote sites (UB and B12y) was 2 miles.

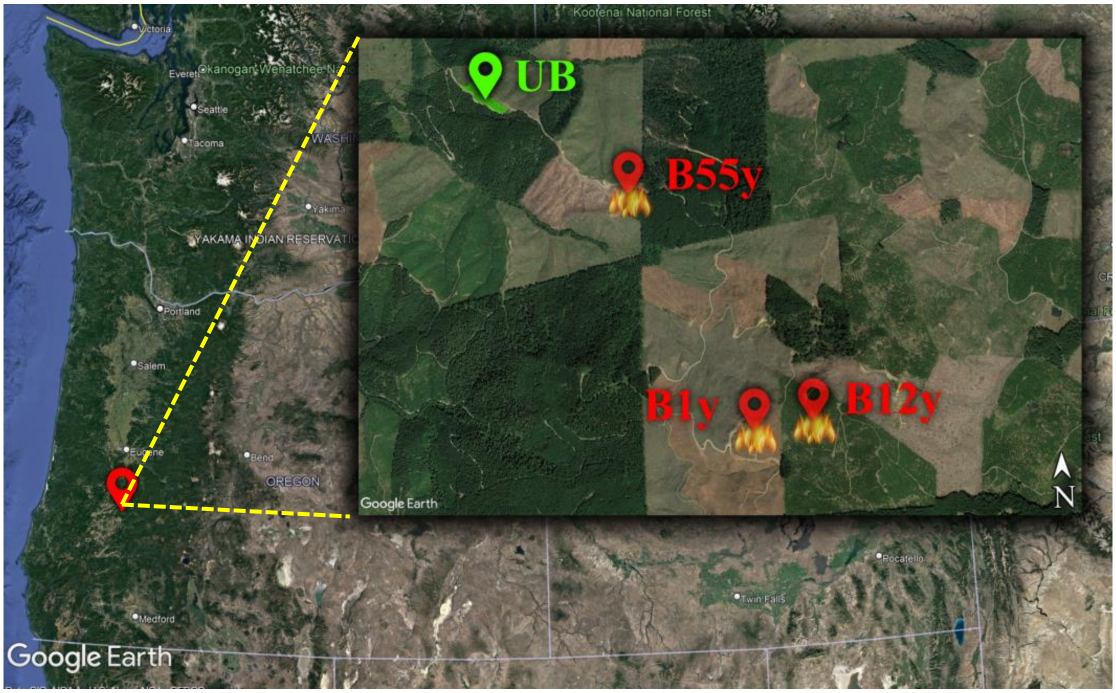

Figure 13: Locations of the four sites selected for this study: Unburned (UB), burned when the stand was 1 year old (B1y), burned when the stand was 12 years old (B12y), burned when the stand was 55 years old (B55y) (Google Earth Pro 6.2.1.6014 (beta)).

4.2.2 Study Design

During February 2021, 24 plots were established across the four selected stands, each containing 36 seedlings planted at 10x10 ft spacing, with the size of each plot being 60 x 60 ft. A randomized complete block design was implemented within each stand, including six reforestation treatments and four blocks, for a total of 24 plots per site. At the B12y site, the DBH of all standing dead trees was recorded, and the block assignment was based on the Basal Area (BA) of each plot after ranking all plots based on BA.

The six reforestation treatments involved vegetation management (VM) and delayed planting. The VM treatments included the control (no action), fall-site preparation (FSP), and spring-release (SR) applications. The delayed planting consisted of two options - planting either the winter following the fire's cessation (2021) or the winter of the subsequent year (2022). The delayed planting option was used due to logistical constraints, such as the need for seedlings and workforce, and to enhance the effectiveness of herbicide treatments by increasing foliage cover for FSP treatments. Two plots within each block were designated for planting in March 2021, implementing the C and SR treatments. A post-planting herbicide application was performed for the SR plot in April 2021. Pre-planting herbicide was applied to each plot assigned for delayed planting (FSP.D) in September 2021. The four plots assigned for delayed planting within each block were then planted using the same stock type used the previous year in February 2022. Finally, post-planting herbicide application was carried out for the SR.D plots in April 2022. Table 13 provides details on dates of herbicide applications.

Table 11: Description of treatments applied at each study site.

Treatment	Planting Year	FSP	SR
C	2021	0	0
SR	2021	0	1
C.D	2022	0	0
SR.D	2022	0	1
SFP.D	2022	1	0
SFP.D+SR.D	2022	1	1

C: no-action control; FSP: fall-site preparation; SR: spring-release; D: delayed planting

4.2.3 Seedling growth and survival

An initial inventory of seedling height (cm) and basal diameter measured at 15 cm from the ground line (D15, mm) was conducted shortly after planting and repeated at the end of the first growing season to assess seedling growth and survival.

4.2.4 Statistical Analysis

Statistical Analysis Software version 9.4 (SAS Institute Inc. Cary, NC) was used for all statistical analysis. Analysis of variance, including Tukey adjustments, was used to test the effects of site and reforestation treatment on seedling growth and survival (PROC MIXED). Percent survival was transformed for ANOVA analysis by taking the arcsin of the square root of percent survival divided by 100. The random effect of the block was included in all tests. In addition, early-seral vegetation data and soil physical and chemical attributes data from Chapter 2 and 3 was used to make correlation analysis with Douglas-fir seedlings survival and growth. All figures were produced using SigmaPlot version 14 (Systat Software, Inc. San Jose, CA).

4.3 Results

Table 14 shows the results of an analysis of variance (ANOVA) for the three growth traits of seedlings (height, diameter, and TPA) during the first growing season across study sites, and VM treatments.

Table 12: P-values of the effects of Site, VM Treatment and their interaction on height, diameter at 15 cm height (D15), and survival (TPA) of Douglas-fir seedlings during first and second growing season.

Trait	Growing Season	Site	VM Treatment	Site x Treatment
Height	1	<0.0001	0.0004	0.0052
	2	0.0005	0.0338	0.4425
D15	1	0.0002	0.0072	0.2054
	2	0.0007	0.0066	0.2628
TPA	1	0.0001	<0.0001	<0.0001
	2	<0.0001	0.1189	0.7305

Across all study sites, there was site and VM treatment effect on height for both growing seasons. The interaction in the first growing season was significant, but not for the second growing season. Also, there was a site and VM treatments effect on diameter, but the interaction of these effects was not significant for both growing seasons. In addition, there was a site effect on survival for both growing seasons, but for the second growing season, there was no VM treatment effect. For the first growing season, the interaction of site and VM treatments was significant (Appendix 8). It is important to note that seedlings in C and SR plots were planted in 2021 (immediately after the fire), and seedlings in C.D, SR.D, FSP.D, and FSP.D+SR.D plots were planted in 2022 (1 year after the fire).

4.3.1 Seedling height

At the end of the first year after the fire (December 2021), Douglas-fir seedlings planted the winter after the fire (March 2021) growing in burned sites were taller than seedlings growing in the UB site ($P<0.001$). Further, a significant effect of SR on seedling height was only observed at the UB site ($P<0.001$). At the end of the second year after the fire (December 2022), no effect of VM treatments was observed at any site, either planted the winter after the fire (winter 2021) or delayed-planted during the second winter after the fire 2022 ($P>0.963$) (Figure 13).

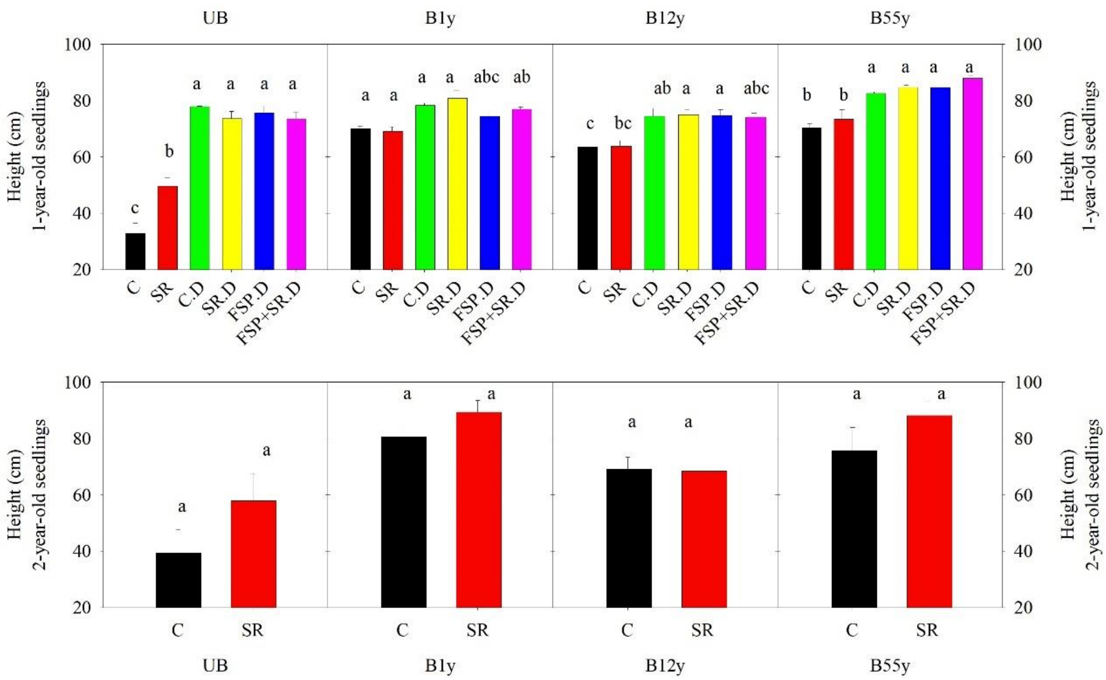

Figure 14: Mean height of Douglas-fir seedlings growing in burned and unburned sites under different vegetation management treatments at the end of the first (upper panel) and second (lower panel) growing seasons. Significance letters on the upper panel indicates the comparison of VM treatments across sites. Significance letters on the lower panel indicates the comparison of VM treatments within each site. Error bar represents standard error.

4.3.2 Seedling diameter

At the end of the first year after the fire (December 2021), Douglas-fir seedlings planted the winter after the fire (March 2021) growing in B55y and B1y were larger than seedlings growing in the UB site (P=0.039). Further, no significant effect of SR on seedling diameter was observed at any sites (P>0.742). At the end of the second year after the fire (December 2022), Douglas-fir seedlings planted the winter after the fire (March 2021) growing in burned sites were larger than seedlings growing in the UB site (P<0.036). In addition, no effect of VM treatments was observed at any site, either planted the winter after the fire (winter 2021) or delayed planted during the second winter after the fire 2022 (P>0.834) (Figure 14).

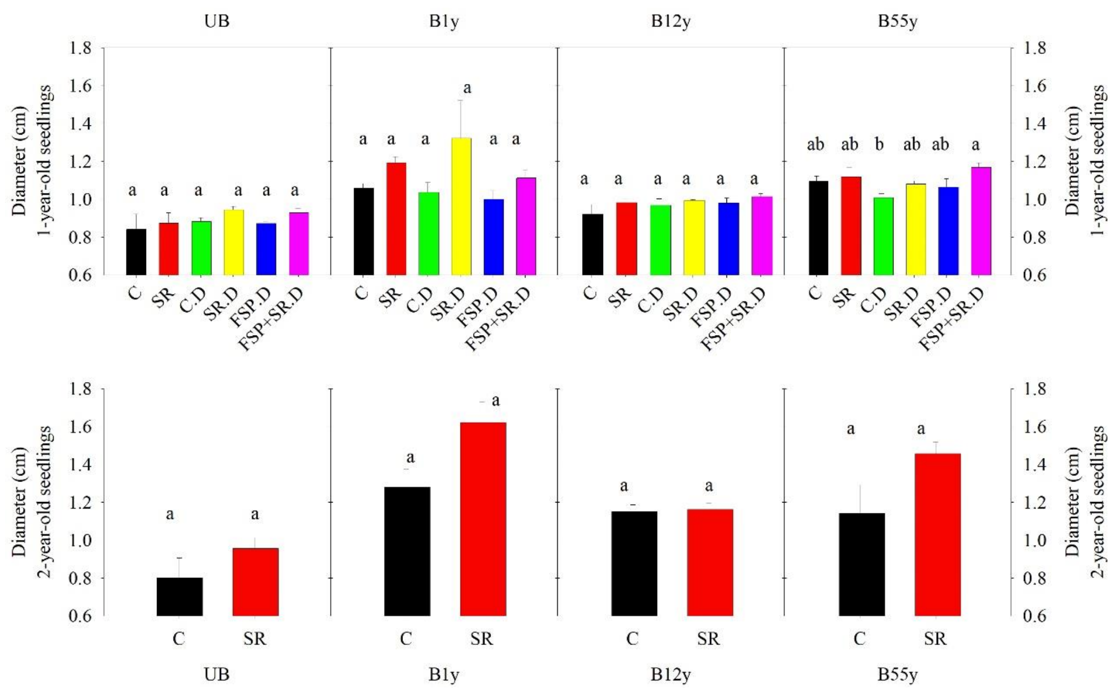

Figure 15: Mean diameter (D15) of Douglas-fir seedlings growing in burned and unburned sites under different vegetation management treatments at the end of the first (upper panel) and second (lower panel) growing seasons. Significance letters on each panel indicates the comparison of the VM treatments within each site. Error bar represents standard error.

4.3.3 Seedling Survival

At the end of the first year after the fire (December 2021), the survival of Douglas-fir seedlings planted the winter after the fire (March 2021) growing in burned sites was higher than seedlings growing in the UB site (P<0.033). Further, at the end of the second year after the fire (December 2022), no effect of VM treatments was observed at any site, either planted the winter after the fire (winter 2021) or delayed-planted during the second winter after fire 2022 (P>0.443).

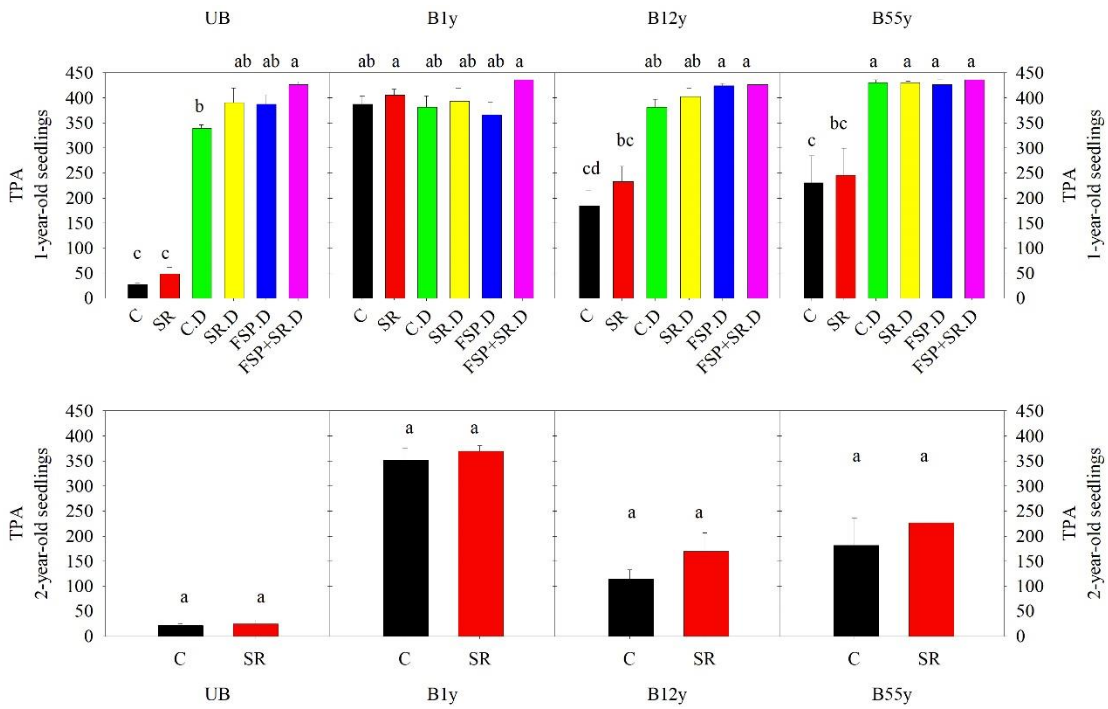

Figure 16: Mean survival (trees per acre, TPA) of Douglas-fir seedlings growing in burned and unburned sites under different vegetation management treatments at the end of the first (upper panel) and second (lower panel) growing seasons. Significance letters on the upper panel indicates the comparison of the VM treatments across sites. Significance letters on the lower panel indicates the comparison of the Sites across VM treatments. Error bar represents standard error.

4.3.4 Relationships between seedling volume and survival, and soil NO_3 and P, and vegetation abundance.

There was a moderate positive correlation between stem volume and soil NO_3 (R^2 = 0.41) and soil P (R^2 = 0.42) for 1-year-old Douglas-fir seedlings growing under C.D treatment. Bigger seedlings were found in plots with larger soil NO_3 and P concentration Specifically, for every 10 ppm increase in soil NO_3, seedlings were 0.063 (dm^3) larger. For every 10 ppm increase in soil P, seedlings were 0.060 (dm^3) larger (Figure 16).

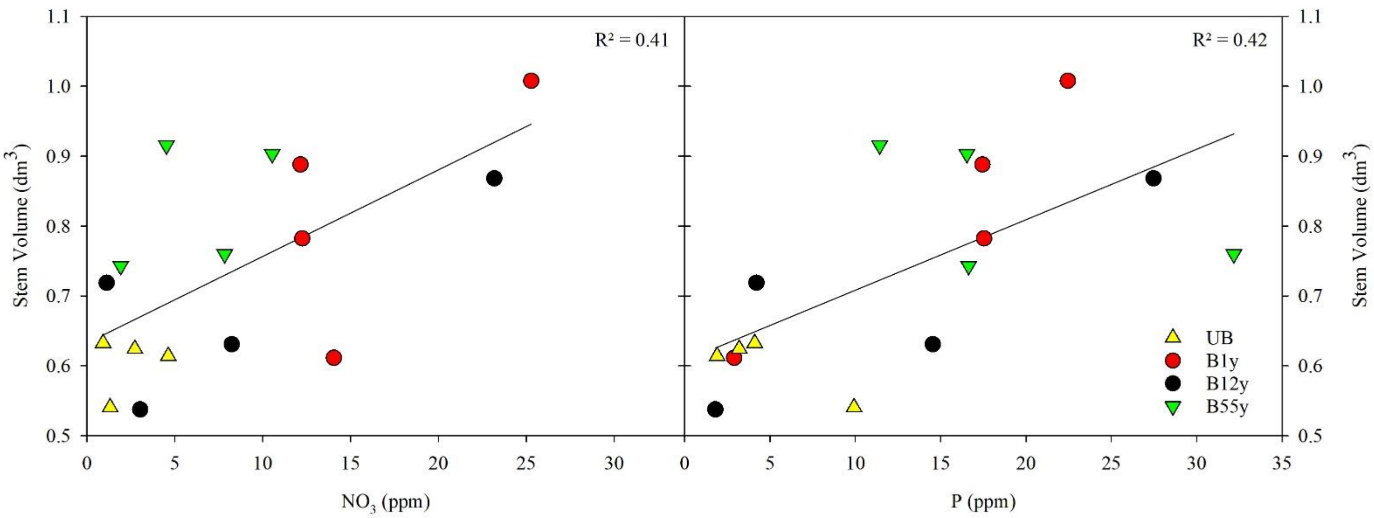

Figure 17: Relationships between a) nitrogen (NO_3, ppm) and b) phosphorus (P, ppm) and stem volume (dm^3) of 1-year-old Douglas-fir seedlings (planted at 2022) at C.D plots, across all sites, unburned and burned. Sites information is provided in Table 1.

There was a strong correlation between woody species cover and survival of 1-year-old Douglas-fir seedlings at C.D plots (R2=0.58, Figure 18). As woody species cover increased, the survival of 1-year-old seedlings decreased. Specifically, for every 10% increase in woody species cover, there was a decrease 39 TPA decrease in survival.

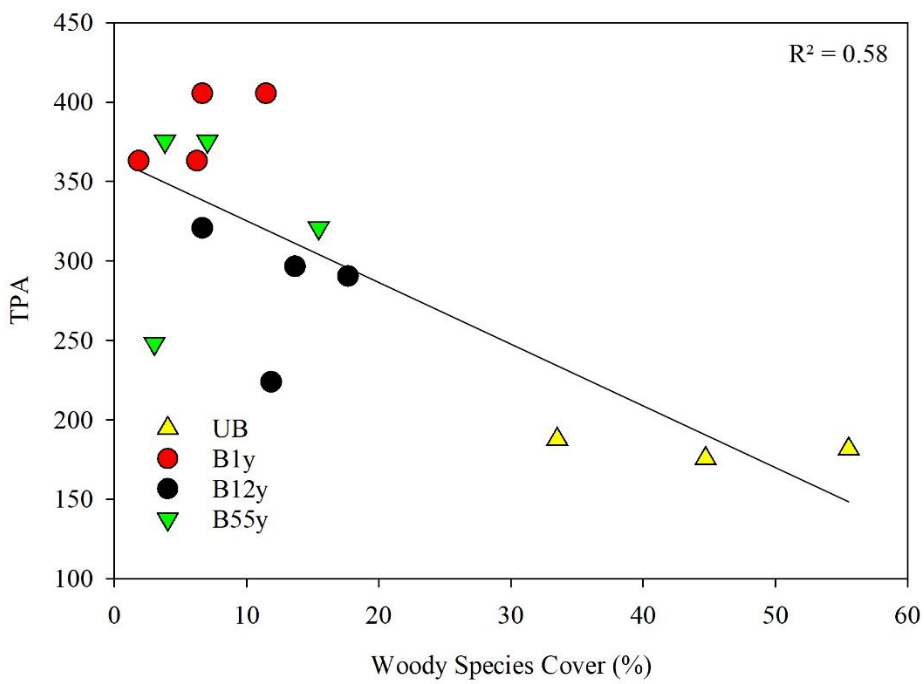

Figure 18: Relationship between woody vegetation cover (sum of shrubs, brambles, and trees) and survival (TPA) of 1-year-old Douglas-fir seedlings (planted at 2022) at C.D plots across all sites, unburned and burned. Sites information is provided in Table 1.

4.4 Discussion

The findings from our study reveal that the effects of VM treatments and delayed planting on the growth and survival of seedlings vary according to the age/structure of the stand before the fire. Our results show that specific VM treatments were more effective than others in promoting seedling growth and survival in some stands. Studies have shown that using FVM methods to manage competing vegetation can lead to increased seedling growth and survival, as well as larger soil moisture availability (Cole & Newton, 2020; Dinger and Rose, 2009, 2010; Gonzalez-Benecke & Dinger, 2018) and reduced seedling water stress (Dinger & Rose, 2009). However, the effects of FVM can depend on factors such as tree species, weather conditions, site quality, resource availability, and timing of application (Balandier et al., 2006; Dinger & Rose, 2010; Flamenco et al., 2019).

Previously conducted studies (Flamenco et al., 2019; Dinger & Rose, 2009; Dimock et al., 1983; Newton et al., 1988; Rose et al., 2006; Maguire et al., 2009) have observed that chemical vegetation control had a positive impact on the growth and survival of Douglas-fir seedlings. The increased performance of seedlings in plots treated with FVM can be attributed to increased soil moisture (Dinger & Rose, 2009, 2010; Gonzalez-Benecke & Dinger, 2018), which was already discussed in Chapters 2 and 3.

At the end of the first year after the fire (2021), FVM had a strong effect on seedling height at UB site: seedlings at the SR plots were larger than those growing at the C plots. Nevertheless, no significant effect of SR treatments was observed at either of the burned sites (B1y, B12y, and B55y). On the other hand, at the end of the second year after the fire (2022), a significant effect of FVM was observed at B1y and B55y sites. In the B1y, the height of 1-year-old seedlings at SR.D treated plots was larger than at C.D treated plots. Also, in B55y, the height of 1-year-old seedlings at SR.D, FSP.D, and FSP.D+SR.D treated plots were larger than seedlings at C.D treated plots.

Furthermore, delayed planting had a positive effect on the height of 1-year-old seedlings. The height of the seedlings in delayed treated plots, including C.D, SR.D, FSP.D, and FSP.D+SR.D, was larger than seedlings planted at C and SR treated plots. Especially in the B55y, seedlings at FSP.D + SR.D treated plots had the largest height growth across sites and treatments. In addition, at the end of the second year after the fire (2022), No effect of SR was observed on the height of 2-year-old seedlings. At UB and B12y, the height of 2-year-old seedlings was shorter than seedlings at B1y and B55y.

At the end of the first year after the fire (2021), no FVM effect was observed on the diameter of 1-year-old seedlings, and those seedlings at the UB site were smaller than other sites. At the end of the second year after the fire (2022), the effect of FVM was observed on seedlings' diameter at delayed VM plots only in B55y. However, for 2-year-old seedlings, there was no effect of VM treatments at any site. At the end of the first year after the fire (2021), the effect of SR treatment was observed on 1-year-old seedling survival at UB, B1y, B12y, and B55y. However, the mortality was remarkably higher in UB at C, and SR treated plots than burned sites (B1y, B12y, B55y). At the end of the second year after the fire (2022), the survival of 1-year-old seedlings at delayed VM-treated plots was remarkably higher than the survival of 1-year-old seedlings planted at 2021. In addition, the effect of VM was observed at UB, B1y, and B12y sites, but no effect of VM treatments was observed at B55y. In addition, SR treatments had no effect on 2-year-old seedling survival, but the mortality of the seedlings was still higher in UB than in B1y, B12y, and B55y. Furthermore, we found a positive correlation between NO3 and P resources and 1-year-old Douglas-fir seedlings' growth at C.D plots (Figure 16). There were more nutrient resources, such as NO3 and P, in burned sites than in UB site (Chapter 1, Table 3). According to findings, it was observed that seedlings had greater availability of site resources for N-fixation at burned sites, particularly at B1y and B55y. This primarily led to lower demand for site resources than UB and promoted seedlings' growth more at burned sites, especially at B1y and B55y than UB.

As the woody species cover increases, the survival of newly planted Douglas-fir seedlings decreases. This finding is consistent with (Cinoglu et al., 2021), who found that cover of early-seral vegetation such as *Ceanothus* can significantly hinder Douglas-fir seedlings' growth and/or survival. The correlation between survival and growth of newly established Douglas-fir seedlings and soil moisture (Figure 17, 18) indicates that the high abundance of woody species was intense at C.D plots (Figure 9, Chapter 3) and negatively impacted newly planted Douglas-fir survival and growth. It is important to note that competition from woody species, particularly shrubs, in UB site can be attributed to the reason why there was less growth and survival of newly established Douglas-fir seedlings (Gray et al., 2005; Oakley et al., 2006).

4.5 References

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5 Conclusions

5.1 Summary of Findings

Chapter 2 examined the effects of wildfire on soil physical and chemical attributes, microclimate conditions, and FVM on soil moisture dynamics. The results showed that a distinct microclimate at the B12y site is characterized by higher maximum temperature, maximum VPD, and maximum wind speed compared to outside without standing trees. Although standing trees reduce the amount of PAR present, they can create an unfavorable microclimate for seedlings by impeding the cooling effect of the wind, leading to increased temperatures and VPD. In addition, the B55y stand had a higher BD and lower pH than the other stands, while all burned sites showed reduced SOM due to fire. Meanwhile, the UB site had a larger CEC, Mg, and Ca content than burned sites but lower NO3 and P levels. The B12y and B1y sites had higher K content than the UB sites. Furthermore, C. and C.D. (control and control delayed – no action) plots had lower AVWC than those that received SR treatments, and there was a reduction in SOM at burned stands, leading to decreased AWHC and upper limit VWC and AWHC.

Chapter 3 examined the effect of pre-wildfire stand/age structure on post-fire early-seral vegetation community dynamics. The results highlighted the potential effects of different VM strategies and other site-specific factors on the abundance and richness of early-seral vegetation communities. The total (native + introduced) forb cover was higher in the second year after the wildfire, and the SR treatment resulted in lower cover than the C. treatment only at one site. The total (native + introduced) graminoids also cover varied among treatments and sites in the second year after the fire, with SR.D. treatment resulting in lower cover than the C.D treatment. The total fern cover showed no difference between the VM treatments at all sites during the second year after the wildfire. The total brambles cover showed no difference between C.D. and VM treatments at burned sites during the second year after the fire. The total shrub cover was higher at the UB site than at all burned sites during the first year after the wildfire, and no difference was observed in C and SR. treated plots at all sites in both years. Overall, the findings suggest that different VM treatments and pre-wildfire stand age/structure may have different effects on the abundance of early-seral vegetation cover after the wildfire.

Chapter 4 examined the effect of pre-wildfire stand age/structure and FVM on Douglas-fir seedling performance and the effect of delayed planting on the effectiveness of FVM on Douglas-fir growth and survival. The results showed that the effect of FVM on the height and diameter of seedlings varied across sites and years. The delayed planting had a positive effect on seedling growth. The study also found a positive correlation between NO3 and P resources and seedling growth. The abundance of woody species negatively impacted the growth and survival of seedlings establishment, especially in UB site.

Overall, the four chapters presented in this study provide insight into the effects of wildfire on soil attributes, vegetation community dynamics, and Douglas-fir seedling performance.

5.2 Management Implications and Future Directions

Wildfires are prevalent in the arid coniferous forests of the western US, with contributing factors such as climate change, climate variability, and suppression making the region more susceptible to fires. The 2020 fire season was especially catastrophic, with the Archie Creek Fire in southwest Oregon scorching over 131,000 acres. The process of reforestation after a fire presents challenges, including seedling production (including availability of seeds and nurseries production capacity), planting (including timing, labor availability and logistics), and competing vegetation control (including timing and time mixes needed). The escalating frequency of fires and droughts exacerbates the complexity, expense, and hazards associated with reforestation efforts.

Forest managers should be aware that the potential different growth outcomes of Douglas-fir seedlings in burned versus unburned sites can be attributed to various factors, including variations in microclimate, soil physical and chemical properties, soil moisture dynamics, and the intensity of competition with early-seral vegetation. These variations in post-fire environments, such as the reduction in SOM and AWHC in burned sites, as well as the presence of dead-standing trees, can adversely affect seedling survival and growth. In addition, VM treatments can create micro-site conditions that alleviate seedling mortality and help them overcome harsh site conditions. Therefore, it is crucial for forest managers to consider the complex interplay of these factors when designing post-fire reforestation strategies.

This study will be extended with additional measurements. Future plans include evaluating biomass allocation to foliage, stems, and roots of Douglas-fir seedlings growing under varying post-fire conditions. In addition, more detailed measurements of soil and air temperature at different heights in contrasting treatment plots are undergoing. All these assessments will improve our understanding of the impacts of wildfire on site conditions and Douglas-fir seedlings performance.

6 Appendix

Appendices Table 1:Results of statistical analysis with letters group for Height of 1-year-old seedlings during first (2021) and second (2022) year of study (C and SR treated plots planted at 2021 and C.D, SR.D, FSP.D, FSP.D+SR.D treated plots planted at 2022)

Site	VM Treatment	Estimate	Standard Error	Letter Group
B55y	FSP.D + SR.D	0.88	0.02087	A
B55y	FSP.D	0.848	0.02087	AB
B55y	SR.D	0.8477	0.02087	ABC
B55y	C.D	0.8257	0.02087	ABCD
B1y	SR.D	0.8085	0.02087	ABCDE
B1y	C.D	0.7837	0.02087	ABCDEF
UB	C.D	0.7792	0.02087	ABCDEF
B1y	FSP.D + SR.D	0.7694	0.02087	ABCDEF
UB	FSP.D	0.7571	0.02087	BCDEF
B12y	SR.D	0.7499	0.02087	BCDEF
B12y	FSP.D	0.7478	0.02087	BCDEFG
B12y	C.D	0.7444	0.02087	BCDEFGH
B1y	FSP.D	0.7441	0.02087	BCDEFGH
B12y	FSP.D + SR.D	0.7413	0.02087	BCDEFGH
UB	SR.D	0.7366	0.02087	BCDEFGH
UB	FSP.D + SR.D	0.7354	0.02087	CDEFGH
B55y	SR	0.7338	0.02087	DEFGH
B55y	C	0.7045	0.02087	EFGH
B1y	C	0.7007	0.02087	EFGH
B1y	SR	0.6916	0.02087	FGH
B12y	SR	0.6383	0.02087	GH
B12y	C	0.6346	0.02087	H
UB	SR	0.4957	0.02087	I
UB	C	0.3288	0.02087	J

Appendices Table 2: Results of statistical analysis with letters group for Height of 2-year-old seedlings during second year of study (2022) (C and SR treated plots planted at 2021)

Site	VM Treatment	Estimate	Standard Error	Letter Group
B1y	SR	0.8941	0.05743	A
B55y	SR	0.8812	0.05743	A
B1y	C	0.8059	0.05743	AB
B55y	C	0.7584	0.05743	AB
B12y	C	0.6913	0.05743	AB
B12y	SR	0.6851	0.05743	ABC
UB	SR	0.5794	0.05743	BC
UB	C	0.395	0.05743	C

Appendices Table 3: Results of statistical analysis with letters group for diameter of 2-year-old seedlings during second year of study (2022) (C and SR treated plots planted at 2021)

Site	VM Treatment	Estimate	Standard Error	Letter Group
B1y	SR	1.6209	0.08856	A
B55y	SR	1.4575	0.08856	AB
B1y	C	1.2815	0.08856	ABC
B12y	SR	1.1624	0.08856	BCD
B12y	C	1.1531	0.08856	BCD
B55y	C	1.1421	0.08856	BCD
C	SR	0.9563	0.08856	CD
C	C	0.8	0.08856	D

Appendices Table 4: Results of statistical analysis with letters group for diameter of 1-year-old seedlings during first (2021) and second (2022) year of study (C and SR treated plots planted at 2021 and C.D, SR.D, FSP.D, FSP.D+SR.D treated plots planted at 2022)

Site	VM Treatment	Estimate	Standard Error	Letter Group
B1y	SR.D	1.3228	0.05179	A
B1y	SR	1.1923	0.05179	AB
B55y	FSP.D + SR.D	1.1688	0.05179	AB
B55y	SR	1.1187	0.05179	ABC
B1y	FSP.D + SR.D	1.1118	0.05179	ABC
B55y	C	1.0937	0.05179	ABC
B55y	SR.D	1.0796	0.05179	ABC
B55y	FSP.D	1.0638	0.05179	ABC
B1y	C	1.0587	0.05179	ABC
B1y	C.D	1.0373	0.05179	BC
B12y	FSP.D + SR.D	1.0146	0.05179	BC
B55y	C.D	1.0074	0.05179	BC
B1y	FSP.D	0.9998	0.05179	BC
B12y	SR.D	0.9938	0.05179	BC
B12y	SR	0.983	0.05179	BC
B12y	FSP.D	0.9812	0.05179	BC
B12y	C.D	0.9681	0.05179	BC
UB	SR.D	0.9448	0.05179	BC
UB	FSP.D + SR.D	0.9278	0.05179	BC
B12y	C	0.9224	0.05179	BC
UB	C.D	0.8825	0.05179	C
UB	SR	0.875	0.05179	C
UB	FSP.D	0.8726	0.05179	C
UB	C	0.8417	0.05179	C

Appendices Table 5: Results of statistical analysis with letters group for survival of 1-year-old seedlings during first (2021) and second (2022) year of study (C and SR treated plots planted at 2021 and C.D, SR.D, FSP.D, FSP.D+SR.D treated plots planted at 2022)

Site	VM Treatment	Estimate	Standard Error	Letter Group
B55y	FSP.D + SR.D	435.84	22.8886	A
B1y	FSP.D + SR.D	435.84	22.8886	A
B55y	SR.D	429.78	22.8886	A
B55y	C.D	429.78	22.8886	A
B55y	FSP.D	426.76	22.8886	A
UB	FSP.D + SR.D	426.76	22.8886	A
B12y	FSP.D + SR.D	426.76	22.8886	A
B12y	FSP.D	423.73	22.8886	A
B1y	SR	405.57	22.8886	A
B12y	SR.D	402.54	22.8886	A
B1y	SR.D	393.46	22.8886	A
UB	SR.D	390.44	22.8886	A
UB	FSP.D	387.41	22.8886	A
B1y	C	387.41	22.8886	A
B1y	C.D	381.36	22.8886	A
B12y	C	381.36	22.8886	A
B1y	FSP.D	366.22	22.8886	AB
UB	C.D	338.98	22.8886	ABC
B55y	SR	245.16	22.8886	BCD
B12y	SR	233.05	22.8886	CD
B55y	C	230.02	22.8886	CD
B12y	C	184.62	22.8886	D
UB	SR	48.4262	22.8886	E
UB	C	27.2397	22.8886	E

Appendices Table 6: Results of statistical analysis with letters group for survival of 2-year-old seedlings second (2022) year of study (C and SR treated plots planted at 2021)

Site	VM Treatment	Estimate	Standard Error	Letter Group
B1y	SR	369.25	32.5881	A
B1y	C	351.09	32.5881	A
B55y	SR	227	32.5881	AB
B55y	C	181.6	32.5881	BC
B12y	SR	169.49	32.5881	BC
B12y	C	115.01	32.5881	BC
UB	SR	24.2131	32.5881	C
UB	C	21.1864	32.5881	C

Appendices Table 7: P-values of the differences between vegetation management treatments and no-action control (C and C.D) treatments on abundance (cover) and species richness of total (native + introduced) forbs, graminoids, ferns, brambles, and shrub+trees across sites during first two years after planting. SR was compared against C; SR.D, FSP.D and FSP.D+SR.D were compared against C.D. Only cases with significant (P<0.05) or marginal (P<0.10) differences are shown.

			VM Treatments				
Habit	Trait	Site	SR 2021	SR 2022	SR.D. 2022	FSP.D. 2022	FSP.D.+SR.D. 2022
Forbs	Cover	UB					
		B1y					
		B12y			<0.001	0.077	<0.001
		B55y					
	Species Richness	UB					
		B1y					
		B12y					
		B55y					0.032
Graminoids	Cover	UB					
		B1y					0.013
		B12y					
		B55y					
	Species Richness	UB					
		B1y					
		B12y					
		B55y					

Ferns	Cover	UB			
		B1y			
		B12y			
		B55y			
	Species Richness	UB			
		B1y			
		B12y			
		B55y			
Brambles	Cover	UB			
		B1y			
		B12y			
		B55y			
	Species Richness	UB	0.069		0.012
		B1y			
		B12y			
		B55y			
Shrub+Tree	Cover	UB	0.008		0.037
		B1y			
		B12y			
		B55y			
	Species Richness	UB	<0.001		
		B1y			
		B12y			
		B55y			
Total	Cover	UB	<0.001		<0.001
		B1y	<0.001		<0.001
		B12y	<0.001	<0.001	<0.001
		B55y	0.052		0.023
	Species Richness	UB			
		B1y			
		B12y			
		B55y			0.018

UB: Unburned; B1y: Burned at 1-year-old; B1y: Burned at 12 years-old; B55y: Burned at 55 years-old; C: No herbicide, planted in winter after fire (2021); C.D: No herbicide, delayed planted in winter of following year after fire (2022).SR was compared against C; SR.D, FSP.D and FSP.D+SR.D were compared against C.D.

Appendices Table 8: P-values of the differences between vegetation management treatments and no-action control (C and C.D) treatments on abundance (cover), species richness of total (only native) forbs, graminoids, ferns, brambles, and shrub+trees across sites during first two years after planting. SR was compared against C; SR.D, FSP.D and FSP.D+SR.D were compared against C.D. Only cases with significant (P<0.05) or marginal (P<0.10) differences are shown.

			VM Treatments				
Habit	Trait	Site	SR 2021	SR 2022	SR.D 2022	FSP.D 2022	FSP.D.+SR.D 2022
Native Forbs	Cover	UB					
		B1y					
		B12y					0.020
		B55y					
	Species Richness	UB					
		B1y					
		B12y					
		B55y					
Native Graminoids	Cover	UB					
		B1y					
		B12y					
		B55y					
	Species Richness	UB					
		B1y					
		B12y					
		B55y					
Native Ferns	Cover	UB					
		B1y					
		B12y					
		B55y					
	Species Richness	UB					
		B1y					
		B12y					
		B55y					
Native Brambles	Cover	UB					
		B1y					
		B12y					
		B55y					
	Species Richness	UB					0.006
		B1y					
		B12y					
		B55y					

Native Shrub+Tree	Cover	UB	0.008		0.037
		B1y			
		B12y			
		B55y			
	Species Richness	UB	<0.001		
		B1y			
		B12y			
		B55y			
Native Total	Cover	UB	<0.001		<0.001
		B1y			
		B12y		0.046	
		B55y			
	Species Richness	UB			
		B1y			
		B12y			
		B55y			0.057

UB: Unburned; B1y: Burned at 1-year-old; B1y: Burned at 12 years-old; B55y: Burned at 55 years-old; C: No herbicide, planted in winter after fire (2021); C.D: No herbicide, delayed planted in winter of following year after fire (2022).SR was compared against C; SR.D, FSP.D and FSP.D+SR.D were compared against C.D.

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