Forest encroachment drivers, dynamics, and wildfire resiliency of forest-grassland ecotones in interior British Columbia

By

Andrea Laura Robinson

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Thesis examining committee:

Jill Harvey (PhD), Research Chair and Thesis Supervisor, Faculty of Science

Wendy Gardner (PhD), Committee Member, Faculty of Science

Mike Flannigan (PhD), Research Chair and Committee Member, Faculty of Science

Colin Laroque (PhD), External Examiner, Department of Soil Science, University of Saskatchewan

ABSTRACT

Historically, fire was a dominant disturbance across interior British Columbia. Dry montane forests were maintained by a mixed severity fire regime dominated by low and moderate severity fires. This fire regime controlled the establishment of woody vegetation and increased diversity and species richness amongst herbaceous understories, particularly at the forest-grassland ecotone. Fire-adapted plant communities, including overstory trees like Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) thrived with frequent stand maintaining fires. For millennia prior to the 19th century, Indigenous communities stewarded landscapes in BC, using fire as a tool to promote restoration and resilience. Following European settlement, western North American forests have undergone significant changes in structure and function. This shift in management has resulted in widespread fuel accumulation and increased stand density across the landscape.

A restoration treatment utilizing mechanical thinning (2005) and prescribed fire (2006) was applied to a forest-grassland ecotone in interior British Columbia. The two objectives of this study are to (1) determine how fuel treatments influence plant communities and tree growth, and (2) describe the drivers and dynamics of conifer encroachment in the grasslands. I conducted a small-scale fire-history study, to confirm a 20–30-year fire return interval documented for the area. For Objective 1, I collected understory plant data from plots (n = 50) in the treated site, an adjacent encroached forest (untreated) site, and a representative grassland site. Tree cores (n = 160) from Douglas-fir trees in both the treated and untreated sites were analyzed for growth differences and sensitivity to climate. Understory species richness and tree growth both increased significantly post-treatment, but climate sensitivity of trees was not significantly different, To determine encroachment dynamics (Objective 2), I collected tree core and basal disk samples from two plots (n = 632 trees) including diameter breast height measurements and in-plot cartesian coordinates for stem mapping. Using Ripley's K analysis, results indicated trees were not clustered spatially across the plot but were clustered by age at small scales. A time series analysis also concluded three distinct pulses of establishment in the 1970s and late 1990s but did not have any significant association to regional climate variables including precipitation, drought, and temperature as well as broad scale climate drivers such as the Pacific Decadal Oscillation and El Niño Southern Oscillation These results indicate site specific variables

including microclimate, light availability, and competition along with the disrupted disturbance regime have more influence than climate on the establishment and perpetuation of Douglas-fir encroachment.

Improved understanding of how and why encroachment occurs can inform landscape management decisions and guide the application of restoration treatments. Although broad-scale implementation of these treatments is often not feasible, restoring a heterogeneous mosaic of forest and grassland conditions can enhance ecosystem resilience and reduce wildfire risk. Proactive stewardship of dry interior forests is therefore essential to prevent landscape-altering wildfires and the loss of these ecosystems. This study was done in partnership with Stswecem'c Xget'tem First Nation; including First Nations perspectives and Indigenous led stewardship is essential to the conservation of BC and Canada's lands.

Keywords: prescribed fire; thinning treatment; understory vegetation community; ecotone; wildfire, climate-growth relationships, tree rings, Douglas-fir

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It's hard to know where to begin, but you'll be grateful once you do.

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to reconciliate and understand the privilege I have, particularly in conducting western science research. Colonization has had a profound impact on Canadian and worldwide Indigenous Peoples, and I want to recognize the wisdom, reciprocity, stewardship, and relationships that give heart to these communities. I also acknowledge that I spent much of my time in T'kemlups te Secwepemc working on my research, and more recently, moved to Lhtako Dene territory where I have completed my thesis. I hope that progress continues to be made in returning rights and title back to these communities where it belongs. Kukwstsétselp.

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Dedication

To the legacy Douglas-fir trees standing tall amongst the wildflower meadows

When I am among the trees

When I am among the trees,

Especially the willows and the honey locust,

Equally the beech, the oaks and the pines,

They give off such hints of gladness,

I would almost say that they save me, and daily.

I am so distant from the hope of myself,

In which I have goodness, and discernment,

And never hurry through the world

But walk slowly, bow often.

Around me the trees stir in their leaves

And call out, "stay awhile".

The light flows from their branches.

And they call again, "Its simple", they say,

"And you too have come into the world to do this, to go easy, to be filled with light, and to shine"

-Mary Oliver

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CHAPTER 1: Introduction

Ecology of the Interior Douglas-fir biogeoclimatic zone

Interior Douglas-fir (*Pseutdotsuga menziesii* var. *glauca*) forests primarily occupy drier sites at mid-elevations within the south-central interior of BC. Interior Douglas-fir stands are typically open-canopied, with closed-canopy conditions occurring in more mesic environments (Hope et al., 1991). Understories are often dominated by herbs and grasses which support herbivory and understory fires (Hope et al., 1991). These stands are shaped by biotic and abiotic interactions including water availability, light conditions, disturbances, and neighbouring trees of different age classes and size (Oboite et al., 2025). In dry ecosystems, germination and establishment of Douglas-fir is often more sensitive to site specific variables like shading, competition, and available moisture (Fichtner et al., 2015; Haugo et al., 2013), whereas climate becomes a dominant factor controlling growth once the tree has matured (Littel et al., 2008). Information on masting for interior Douglas-fir is limited, but research suggests approximately once every 10 years a large cone crop is produced (Lastra, 2001). Prolific regeneration without controlling agents (e.g., a disturbance such as a low intensity fire) create a positive feedback cycle where competition for resources suppress tree growth leading to unhealthy, dense stands in the understory (Acquah et al., 2023). Seeds can disseminate up to 100 metres from the seed source, but other factors including wind and animals can contribute to dispersal (Hermann and Lavender, 1990). Rodents will opportunistically cache the seeds where they can remain viable for upwards of six years, perhaps supporting the biological factors influencing establishment (Hofmann, 1921). Due to the geographic extent and wide range of vegetation, the interior Douglas-fir (IDF) biogeoclimatic zone serves as winter range for many ungulates, including mule deer (*Odocoileus hemionus*) (Hope et al., 1991).

The IDF occurs primarily in the southern and central interior of BC, typically between 350 and 1450m above sea level. This zone is commonly located within rain shadow regions on the leeward side of mountain ranges, where limited moisture availability is a primary factor controlling ecosystem structure and productivity (Hope et al., 1991). The predominant climate has warm, dry summers and cool winters (Hope et al. 1991). The IDF zone can be further subdivided into 13 subzones: IDFxh, IDFdm, IDFdk, IDFww, IDFmw, IDFxw, IDFdc, IDFdh, IDFdw, IDFxc, IDFxk, IDFxx, and IDFxm. Common species such as pinegrass (*Calamagrostis*

rubescens), prickly rose (*Rosa acicularis*), showy aster (*Eurybia conspicua*), and kinnickinnick (*Arctostaphylos uva-ursi*) exist in the understory. The climax tree species in the IDF zone are Douglas-fir, where trembling aspen (*Populus tremuloides*) and lodgepole pine (*Pinus contorta*) exist as seral species (Steen and Coupé, 1997). Interior Douglas-fir are a drought tolerant and fire-resistant species that can live upwards of 400 years (Hermann and Lavender, 1990).

Often occurring at the upper edge of grasslands, the IDF zone is characterized by transitional areas between grassland and forest ecosystems known as forest-grassland ecotones. Forest-grassland ecotones primarily exist within the IDFxm (very dry-mild) subzone adjacent to the bunchgrass (BG) zone. Grasslands in BC are patchy, fragmented, and relatively limited in extent compared to prairie grassland ecosystems in the rest of Canada. Forest-grassland ecotones are also shaped by disturbance, formed through adapting to dynamic conditions including fire and herbivory (Strang and Parminter, 1980). Increased structural heterogeneity due to the overlap of two ecosystems often supports ecological niches often increasing diversity and species richness (Breshears, 2006; Liautaud et al., 2020). Although these ecotones do not occupy large areas of land, they are critically important habitats providing connectivity to adjacent ecosystems (Kark, 2012). But an altered disturbance regime in BC has led to the unregulated growth of Douglas-fir in the IDF where large swaths of grassland have been lost to encroachment (Bai et al., 2005; Strang and Parminter, 1980). Rapid conversion of grassland to forest is often highly driven by the large recruitment distances by seeds into open areas (Fogarty et al. 2022). These factors combine to perpetuate the spread of encroachment in forest-grassland ecotones in the IDF.

Climate and wildfire in interior British Columbia

Typically, wildfires are agents of change maintaining forest health and diversity, however, in recent years, the total area burned, and instances of extreme wildfire behavior are increasing quickly (Sankey et al., 2019). In addition to increasing wildfire activity, drought events are predisposing forests to burn (Wang et al., 2025). Hotter and drier weather due to climate change is contributing to the six biggest wildfire seasons on record in BC occurring in the last 10 years in 2017, 2018, 2021, 2023, 2024, and 2025. The heatwave of 2021 caused some of the hottest and driest conditions in western Canada on record and contributed to severe and devastating wildfires across the southern interior of BC (White et al., 2023). Twentieth century climate

change in the southern interior of BC is expected to cause warmer temperatures, lower relative humidity, and more drought events (Jain et al., 2022). These anticipated changes will precipitate even greater increases in fire activity and increase the susceptibility of forests and grasslands to burning, possibly at a higher severity than historically recorded (Parisien et al., 2023).

Fire regimes across North America are being increasingly researched and provide important insight into historical landscape structure and characteristics (Falk et al., 2010; Hagmann et al., 2021; Margolis et al., 2022). Fire regimes are defined by the frequency, intensity, size, and severity of fire occurring over a temporal and spatial scale (Agee, 2005; Krebs et al., 2010). In the interior dry Douglas-fir forests of BC, current literature points to a historical mixed severity fire regime with predominantly low and moderate severity fires occurring anywhere from every 10-30 years prior to the 20th century (Brookes et al., 2021; Copes-Gerbitz et al., 2023; Harvey et al., 2017; Heyerdal et al., 2012). The contemporary land base is now in a fire deficit with a century of departure in many cases from the historical fire regime; a shift largely driven by changes in land management practices and effects from climate change (Baron et al., 2025). These emergent changes have resulted from decades of fire suppression interacting with increasingly extreme fire weather forming conditions conducive to large, landscape-scale wildfires (Jain et al., 2022; Kreider et al., 2024).

Land use history in the Churn Creek Protected Area

The Churn Creek Protected Area (CCPA) is the largest area of protected grasslands within BC at 36,747 ha. With less than one percent of the province's land base being grasslands, this is a critically important habitat for many plant and animal species. This visually stunning area is also an important space for recreational users; the CCPA is used by the public for hunting, hiking, camping, and horseback riding. Surrounding Indigenous communities including Secwepemc and Tsilhqot'in Nations have utilized and occupied this area since time immemorial for cultural uses including berry-picking, gathering medicinal plants, fishing and hunting, and cultural burning (FREC, 2021; Lake and Christianson, 2019, Lewis et al., 2018; Parminter, 2023; Peacock et al., 2016). Land stewardship prior to European settlement often involved cultural burning for revitalization of berry patches, marking travel corridors, reducing fire risk around communities, and for ceremonial purposes (Blackstock and McAllister, 2004; Copes-Gerbitz et al., 2022;

Ignace and Ignace, 2017; Kimmerer and Lake, 2001; Turner, 1999). In turn, the surrounding dry interior forests were frequently cleared of understory debris by fire which revitalized the land and maintained the open character of Douglas-fir forests.

Churn Creek was established as a protected area in 1995 primarily for ecological conservation (Figure 1.1), but with the multi-use history of different groups, exceptions were made to accommodate continued ranching (BC Parks, 2000). The Empire Valley Ranch was incorporated as a part of the CCPA in 1998. After European settlement and prior to 1995, dominant land uses in the area included heavy ranching and agricultural practices, as well as logging (Lutz, 1980). Grazing went unregulated until 1919 when the *Grazing Act* was passed. These rangelands were some of BC's most pristine grasslands occupying the incised river valleys along the Fraser River corridor. Low, middle, and upper grassland represent a wide range of grassland systems, all habitat for many species including red and blue listed species (Gayton, 2004; Iverson, 2004).

Two of these historic ranches are still in operation today. Gang Ranch and the Empire Valley Ranch were established in the 1860s in the wake of the Cariboo Gold Rush. The increase in settlement in this area led to widespread changes on the landscape, and to the First Nations communities that resided here. The 1862/1863 smallpox epidemic in BC catastrophically affected local First Nations communities, where the disease proved fatal in up to thirty percent of victims (Boone, 2022). With these communities struggling to endure the smallpox epidemic, the effects of colonialism expanded across the interior of BC (BC Parks, 2000; McLean, 1982; Van Rijn, 2006) leading to reduced land stewardship and the loss of autonomy among these groups. Forced enrolment into residential schools where cultural practices and traditions were lost further contributed to the decline of traditional and cultural land practices. The further oppression of Indigenous People from the *Bush Fire Act* (1874) outlawed Indigenous fire stewardship and cultural burning. Without knowing future consequences, this act began the forestry revolution leading to dense, infilled forests across the province at risk from high intensity wildfire today. With over 4,000 head of cattle roaming what would become the CCPA and surrounding region during this period (Reid, 2010) a reduction in grass cover (fine fuels) that fuelled low severity fires further contributed to the loss of fire on the landscape. With the forest industry gaining traction, fire (lightning or human ignited) was viewed as financially destructive to the timber supply. Thus, the *Forest Act* (1912) was created and organized fire suppression began. In the

1950s logging companies bought rights to some of the timber supply within what is now the CCPA and ended the last burning practices used by local ranchers to control the establishment of trees (encroachment) in the grasslands (BC Parks, 2000). Selective logging practices during this time removed many large, old, fire-resistant trees thereby promoting second growth stands (Day, 1998; Hessburg et al., 2005).

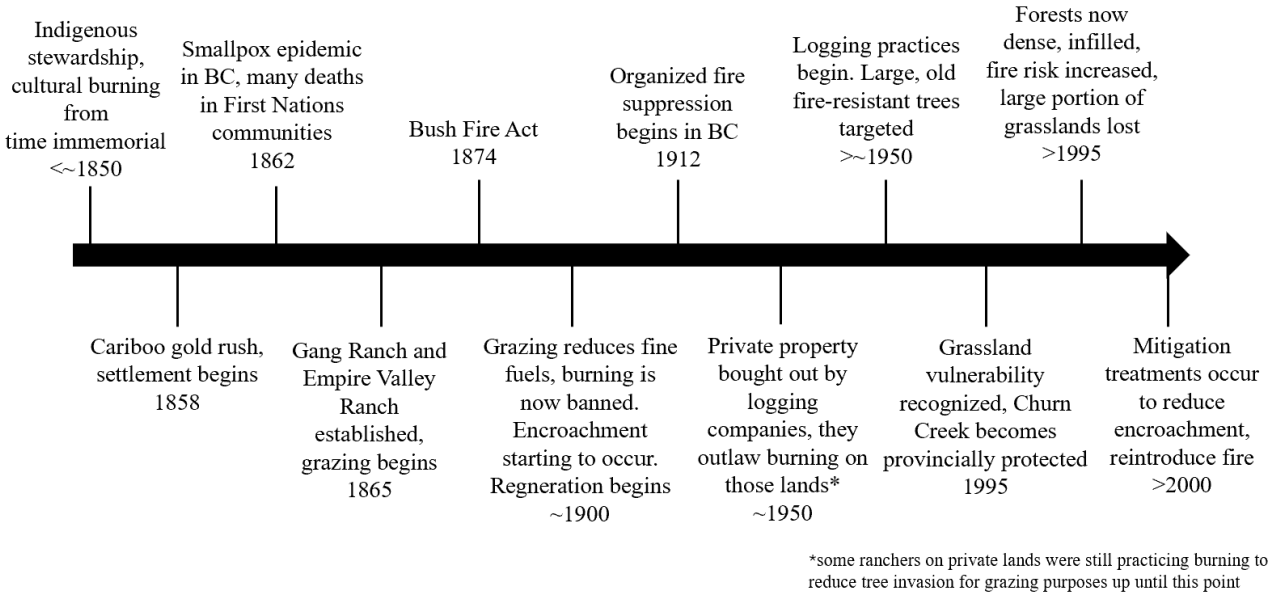

Figure 1.1 Timeline of the history of people and land in the Churn Creek Protected Area and surrounding areas in British Columbia.

The complex land use history of this area over the last 160 years has modified the once fire-adapted landscape and shaped it into a fire vulnerable landscape. Forest-grassland ecotones that formed the boundary between grasslands and forests once contained mature, fire-resistant Douglas-fir broadly spaced with an open understory. In contrast, these ecotones are now increasingly dominated by dense ingrowth and forest expansion into adjacent grasslands. In the absence of frequent fire, the dominant disturbance in this region (Harvey et al., 2017), positive feedback mechanisms have developed, allowing tree encroachment to proceed largely unchecked.

The Pitt Report (1998), Cariboo Chilcotin Land Use Plan (CCLUP; 1994), Cariboo Chilcotin Grasslands Strategy (2001), Churn Creek Management Plan (2000), and the Churn Creek Fire Management Plan (2001) all acknowledge the need to conserve and restore the fragmented and degraded grasslands of the Cariboo region. Managing encroachment is the primary focus of many of these documents, but other threats such as invasive species are of concern. With the early 2000's at the fore front of grassland conservation, how are grasslands faring today? Regulatory frameworks within the government often prolong the process of treating these areas, sometimes preventing it. This was not the intention, but policies including the Mule Deer Winter Range (MDWR), and Old Growth Management Areas (OGMA) inhibit effective fuel treatments due to requirements preventing excess removal of trees. These policies were formed to protect valuable ecosystems from past practices, but now create barriers to fuel reduction treatments, especially near communities or vulnerable areas, like forest-grassland ecotones (Preston, 2024).

In 2021 a 12,040ha wildfire burned in the CCPA. This fire burned at moderate and high severity in many locations where it would have historically burned with low severity impacts. Encroachment and forest ingrowth created continuous fuels in much of the former grassland adjacent forests, conditions that drove high intensity fire and high severity fire effects. The reintroduction of fire to this landscape is a management goal, however, the prolonged absence of fire over the past century has resulted in fuel accumulation and structural changes that increase the likelihood of severe fire effects when wildfires do occur, as evident in 2021.

Wildfire resiliency

The definition of resiliency by Gunderson et al. (2012) is the ability of an ecosystem to "recover to its former equilibrium state after disturbance" (p. xiv). In the context of wildfire on the landscape, resiliency refers to the ability of an ecosystem to regenerate following fire without transitioning to an alternative state (Gillson et al., 2019). This ability has often been shaped by fire over millennia, as over time species in fire prone areas became fire-adapted and have grown to coexist and persist alongside this disturbance (Keeley et al., 2011). But in recent decades, the capacity for ecosystems to be resilient to wildfire is changing and must be proactively supported by society. Supporting resilient ecosystems extends beyond basic recovery and includes adaptive and transformative resilience involving socioeconomic aspects (McWethy et al., 2019). Adaptive

resilience implies reducing wildfire risk (i.e. fuel management and community planning) whereas transformative resilience is truly a change in how society coexists with wildfire (McWethy et al., 2019).

Fuel management treatments aim to reduce the amount of fuel in an ecosystem and therefore reduce the risk of wildfire. Dry forests are often targets for fuel management in North America as they have been disproportionately affected from colonial forest practices and due to continuous and excessive fuel buildup, are increasingly volatile during wildfires (Agee and Skinner, 2005). Fire exclusionary policy across North America was legislated to protect forests in favour of the timber supply. With little regard for the ecological processes sustaining these ecosystems, a shift in the structure of dry forests over time has occurred (Hessburg et al., 2005). Identifying at-risk forests based on vegetation type and priority locations (i.e. proximity to communities, infrastructure, and ecological values) are the first steps in planning for resilient communities (Agee and Skinner, 2005). Once identified, these forests are often candidates for fuel management. Fuel reduction treatments include thinning, prescribed burning, mulching, or mastication. Thinning focuses on removing biomass from a system either by mechanical or manual processes to reduce stand density and decrease the risk of high severity wildfire. Reducing overstory tree density in combination with reducing the surface fuel loads can be a more effective method than either treatment alone (Ziegler at el., 2025). Prescribed fire is also widely used to reduce accumulated surface fuels while restoring historical disturbance regimes. In addition to lowering fuel loads, prescribed burning can enhance ecosystem resilience by promoting fire-adapted species, improving understory productivity, and increasing structural and compositional heterogeneity across the landscape (Crotteau and Keyes, 2020; Knapp et al., 2017). Using prescribed fire is also a form of transformative adaptation; “good fire” elicits community involvement, knowledge sharing, and proactively manages the potential outcomes of a future wildfire (Kaufmann and Shlisky, 2005).

There has been an increased interest in completing fuel treatments, particularly in wildland urban interfaces (WUI) or near vulnerable communities and research in BC has been primarily focused on quantifying fuels and fire-behaviour analyses. Another important but less well-known impact of fuel treatments are the effects on plant communities – particularly from a holistic ecosystem perspective. To fully embrace transformative adaptations to coexist with wildfire, all aspects of

ecosystem impacts must be clearly understood. My key research question addressing this gap aligned with wildfire resiliency, Douglas-fir ecology, climate and wildfire, and the effects of historical land use in the CCPA are:

Chapter 2

1. What are the effects of a mechanical thinning treatment and prescribed fire on understory plant community?
2. How do these treatments influence the growth and climate sensitivity of Douglas-fir trees remaining after the treatment?

Chapter 3

3. In a forest-grassland ecotone with encroachment, are Douglas-fir trees clustered spatially, by age, or by size?
4. Are moister, cooler periods associated with higher seedling establishment in a forest-grassland ecotone?

To address these questions, I conducted a field-based study in the CCPA. Using dendrochronological methods coupled with plant community and spatial analysis techniques at Thompson Rivers University I completed two studies. Chapter 2 answers question 1 and 2 and will be submitted for publication as an original research article to a currently undecided academic journal. Chapter 3 answers questions 3 and 4 and will also be submitted for publication as an original research article to a currently undecided academic journal. Chapter 4 summarizes the findings and management implications of my thesis.

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CHAPTER 2: Effects of prescribed fire and mechanical thinning on a forest-grassland ecotone in the Churn Creek Protected Area, British Columbia, Canada

ABSTRACT

In dry montane forests of interior British Columbia, Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) trees grow at the boundary of the upper grasslands forming transitional areas known as ecotones. Historically, forest-grassland ecotones were maintained by frequent, low- to moderate-severity fires that limited tree establishment and preserved open grassland structure. However, the exclusion of fire due to colonial forest policies has facilitated Douglas-fir infilling, resulting in widespread encroachment and loss of grassland habitat. Many understory plant species in this ecosystem are fire-adapted, and the shift away from frequent burning has altered plant community composition including declines in species richness and diversity. With this increase in vegetation, or fuels, these ecotones are also at risk from high severity wildfire reducing the resilience of these ecotones. Although studies of encroached forests often focus on fuel treatment effectiveness for wildfire behaviour, the ecological responses of plant communities remain less well understood. In the Churn Creek Protected Area, BC, a treatment utilizing mechanical thinning (2005) and prescribed fire (2006) were applied to a forest-grassland ecotone. The two objectives of this study are to (1) assess the effect of a thinning treatment and prescribed fire on understory plant community, and to (2) determine how these treatments influence the growth and climate sensitivity of Douglas-fir trees remaining after the treatment. To confirm the fire regime for the study site I conducted a small-scale fire-history which was consistent with a 20-30 year fire return interval which is within the historical range of variability for the area. Species identification and percent cover data was collected from 3.99m radius plots (n = 50) in the treated site, an encroached forest (untreated) site, and a representative grassland site for comparative analysis. Tree cores (n = 160) from Douglas-fir trees in both the treated and untreated site were analyzed for growth differences and sensitivity to climate. Understory species richness and tree growth both increased significantly post-treatment. Although the understory plant community did shift towards a more grassland-like composition compared to the untreated site, many bunchgrasses that exist in the surrounding grasslands have not established in the treated site likely due to competitive exclusion of pinegrass (*Calamagrostis rubescens*) which colonized the treated site as the dominant species. Although treatments did not alter climate–growth sensitivity, reduced tree density alleviated competition, resulting in a 72.3% increase in

growth. Climate-growth sensitivity was not significantly altered as trees in this region are already strongly correlated to climate due to moisture limitation. These findings suggest treatments including thinning and prescribed fire in a forest-grassland ecotone in BC's dry interior can enhance understory plant communities, increase tree growth, and reduce risk of catastrophic wildfire. Future treatments involving prescribed fire can be informed by the fire regime.

Keywords: prescribed fire; thinning; treatment; understory vegetation community; ecotone; wildfire, climate-growth relationships, tree rings, Douglas fir, British Columbia

INTRODUCTION

Grasslands are one of the most biologically diverse biomes on Earth covering approximately one-third of the terrestrial environment (Petermann and Buzhdygan, 2021). These ecosystems provide many important functions including carbon storage, water regulation, and cultural uses (Bai and Cotrufo, 2022; Li et al., 2016; Petermann and Buzhdygan, 2021; Sun et al., 2022). Globally, disturbances in grasslands include grazing and fire, and are key drivers of ecosystem processes that help maintain their openness, cycle nutrients, and favour species adapted to semi-arid and disturbance prone conditions (van Andel et al., 2012). The persistence of grasslands is threatened by increases in over grazing, land conversion, and the encroachment of woody plants, highlighting grassland conservation as an international priority (Gang et al., 2014).

Canada's prairies represent the greatest grassland areas within the country; however, British Columbia (BC) is home to a smaller, yet highly diverse area of grasslands. In BC, grasslands comprise less than 1% of the land base (Iverson, 2004) with the most extensive grasslands located in regions that experience dry, hot climates in the growing season. Driven by the rain-shadow effect of the Coast Mountain ranges combined with incised river valleys, regions of southern interior BC include topographical conditions suited to supporting grasslands. This xeric environment favours species adapted to semi-arid and arid conditions (Tisdale, 1947; Wikeem and Wikeem, 2004). Low-, middle-, and upper-elevation grasslands are found within BC's interior ranging in elevation from 300 to 900m above sea level (asl).

At elevations above 900m asl, grasslands transition to dry coniferous forests. This is where forest-grassland ecotones form and are a dynamic transition zone between open grasslands and forests and are shaped by complex ecological and climatic interactions. The transition between grassland and forest is primarily controlled by moisture and temperature (Meidinger and Pojar, 1991; Steen and Coupé, 1997). This gradual environmental shift of climatic conditions often allow species from both ecosystems to overlap. This overlap of ecosystems forms a mosaic of ecological communities often creating areas of higher biodiversity across the forest-grassland transition (Wikeem and Wikeem, 2004). Historically, fire maintained these forest-grassland ecotones (Harvey et al., 2017; Harvey and Smith 2018), where together with moisture limitations, low and mixed-severity fires shaped the ecotone boundary (Conver et al., 2018; Copes-Gerbitz et al., 2022; Harvey et al., 2017). Fire reduced understory vegetation, killed small

trees, and prevented a build-up of ladder fuels (Hessburg et al., 2005). In interior BC, frequent and low intensity surface fires maintained open canopy forests and reduced seedling establishment while rejuvenating the understory (Engber and Varner, 2012). These fires removed built up fuels allowing for the seedbank and fire adapted species to re-establish (Dickson-Hoyle et al., 2024).

Encroachment, or woody plant invasion, occurs when trees and shrubs expand into grasslands, often driven by changing land management approaches such as fire suppression and increased grazing pressure (Ratajczak et al., 2012; Sindelar, 1971). In BC, encroachment is one of the largest drivers of grassland conversion to forest and can result in densely forested areas leading to reduced biodiversity and increased fire risk (Bai et al., 2004). The encroachment of woody plants has been observed in over one third of grasslands in BC in the last century (Bai et al., 2005; Cariboo-Chilcotin Grasslands Strategy, 2001; Reimer, 2023). Greater moisture favours the establishment of woody vegetation (e.g., shrubs and trees), and cooler aspects (north and east) are more likely to have higher tree cover due to lower temperatures and moister soil conditions (Hamilton, 2020; Hamilton et al., 2022; Ratajczak et al., 2012). The proximity to a local seed source or increasing forest density adjacent to grasslands reduces the cover of grassland-associated plant species, thereby altering grassland community composition. Today, denser, infilled stands are observed at the upper elevation of contiguous grasslands, often with veteran trees scattered throughout as relics, or legacies, of historical forest structure. In addition to encroachment, invasive species are contributing to declines in both native plant diversity and overall grassland cover (Gibbons et al., 2017). Aggressive invaders such as cheatgrass (*Bromus tectorum*), spotted knapweed (*Centaurea stoebe*), and Canada thistle (*Cirsium arvense*) are particularly well-suited to establish in disturbed environments, often outcompeting native graminoids and forbs (Leung, 2002). Following European settlement, increased grazing pressure from ranching has further altered these ecosystems. The selective consumption of native grasses as forage can reduce their abundance, opening space for invasive plants and sometimes facilitating tree seedling establishment (Cariboo-Chilcotin Grasslands Strategy, 2001; Reid, 2010). Together, these pressures accelerate woody encroachment and are contributing to changes in the character of grasslands in BC.

Prior to European settlement and colonization (~1850), fire played a critical role in maintaining the structure and function of forest-grassland ecotones in southern interior BC. Landscape use and management was practiced since time immemorial by Indigenous people (Blackstock and McAllister, 2004). Secwepemcúl'ew is the 180,000km^2 area home to 17 Secwepemc Nations in interior BC. In this region, fire was sometimes caused by lightning, but many ignitions came from Indigenous Peoples (Lake and Christianson, 2019; Turner, 1999). Knowledge of the fire history for the southern interior of BC exists documented through oral and written records of local Indigenous groups (Copes-Gerbitz et al., 2022; Dickson-Hoyle et al., 2022; Nikolakis and Ross, 2022). Intentional fires were set to regenerate berries and medicinal plants, enhance travel, improve ungulate habitat, and reduce fire hazard (Blackstock and McAllister, 2004; Copes-Gerbitz et al., 2023; Dickson-Hoyle et al., 2022; Peacock et al., 2016; Wikeem and Wikeem, 2004). By working with the timing of the seasons and natural barriers, Indigenous People were able to coexist with wildfire for the benefit of themselves and the environment that often included increased species richness and diversity (Greenwood et al., 2024; Hoffman et al., 2021). Intentional burning by Indigenous Peoples was prolific and helped maintain fire regimes which in turn reduced the likelihood of severe landscape altering wildfires (Hessburg et al., 2022; Lake and Christianson, 2019). But, since the early 1900s colonial forest management has altered forest structure, and fire has largely been absent on these landscapes due to fire suppression practices and the *Bush Fire Act* of 1874 outlawing Indigenous and cultural burning (Copes-Gerbitz et al., 2022). This legislation in combination with introduced grazing practices for cattle, reduced fine fuels in the grasslands and contributed to the observed reduction of fire in this environment (Strang and Parminter, 1980; Van Auken, 2000), which has led to closed-canopy, dense, immature forests in many forest-grassland ecotones, particularly in the interior Douglas-fir (IDF) biogeoclimatic zone (Wong et al., 2003). Thus, colonial forestry policy and management have led to the accumulation of fuels in the dry interior forests of BC that were once resilient and generally adapted to fire.

This history of past fires, both human- and lightning-ignited, can also be recorded in tree-ring records (e.g., Margolis et al., 2022). Low and moderate severity fires often burn the lower bole of trees and can leave char or scorch marks and can form fire scars. Fire scars are damage to the trees when a fire "scars" or partially destroys the cambium on the lower trunk of a tree (primarily conifers) and the resulting wound heals forming a lobe within the rings (McBride, 1983). The

fire scars can be analyzed using dendrochronological methods and assigned a year, and sometimes season, when the fire occurred.

Restoration of open conditions in ecotonal and dry interior forests that were once ‘fire-adapted’ is an increasingly common practice. Treatments often include thinning, which is a silvicultural treatment used to reduce stems per hectare. Thinning is generally used for increasing tree growth and yield in forestry practices (Acquah et al., 2024), but as wildfire becomes increasingly prevalent on the landscape, thinning can be used as a wildfire risk reduction tool. Thinning reduces the live fuel load by removing and spacing trees out to a desired density of live tree stems (Harrod et al., 2009). Reducing stems per hectare to historical levels (<150) in dry interior forests decreases the risk of passive and active crown fire (Brookes, 2019; Brookes et al., 2021; Greene, 2021; Preston, 2024). The removal of fuel reduces canopy fuel continuity while providing more light to the understory. Prescribed fire is a tool that can be used in combination with mechanical thinning in dense unmanaged forests, and when applied strategically is an effective management technique (Weber and Taylor, 1992). Prescribed fire serves as a proactive strategy to reduce fuel loads, decrease the likelihood of intense wildfires, and rejuvenate fire-adapted vegetation (Harrod et al., 2009; Kral et al., 2018). Intentional management using fire as a tool can enhance biodiversity in grassland understory communities through reducing thatch which can prevent seed germination, and invigorating serotinous seeds and cones (He et al., 2019; Keeley, 2000; Kjaer et al., 2025). Studies on treatments using prescribed fire generally focus on quantifying the reduction in fuel and therefore the wildfire risk, but research is needed to determine how trees respond to the same treatments (growth, productivity) and if response to climate is altered. Grasslands and grassland-forest ecotones require periodic burning for maintaining ecosystem structure and function to control woody encroachment that would not typically establish under a historical fire regime of more frequent fire (McLauchlan et al., 2020; McLennan and Ronalds, 2000). Despite increasing application of thinning and prescribed fire, there is limited understanding of how these treatments influence understory community composition and the growth of trees remaining post-treatment.

The goal of this research is to determine how fuel treatments, including thinning and prescribed fire, influence plant communities, and tree growth in grassland-forest ecotones. At a site in the Churn Creek Protected Area (CCPA), BC, a mechanical thinning treatment (2005) and prescribed

burn (2006) were applied (Figure 2.1) to mitigate tree encroachment in a forest-grassland ecotone. This area was left generally undisturbed since the treatment with the exception of light grazing by livestock and wild ungulates. Specifically, the research objectives are to (1) assess the effect of a thinning treatment and prescribed fire on understory plant community, and (2) determine how these treatments influence the growth and climate sensitivity of Douglas-fir (*Pseudotsuga menziesii var. glauca*) trees remaining after the treatment. I also aim to expand on the fire history for the site; current records date back to the 1600s where Harvey et al. (2017) notes a mixed severity fire regime. I hypothesize (1) the fuel treatments applied at the CCPA site will increase grass and plant diversity in the understory plant community (Gordijn and O'Connor, 2021, Laughlin et al., 2008; Rice, 2005), and (2) following treatments, the remaining trees will have increased growth and increased climate sensitivity (Griesbauer and Green, 2010a; Acquah et al., 2024). Previous research has demonstrated that thinning and prescribed fire can support native species assemblages in grasslands (Ducherer et al., 2009; Wilkin et al., 2021). By looking at the understory community and overstory tree growth we can gain a better understanding of how the ecosystem as a whole is responding to the treatment. Indigenous communities and other land management agencies are increasingly leading landscape restoration efforts and results from this study can help inform decision makers on the ecological effects of using thinning treatments and prescribed fire as land stewardship tools.

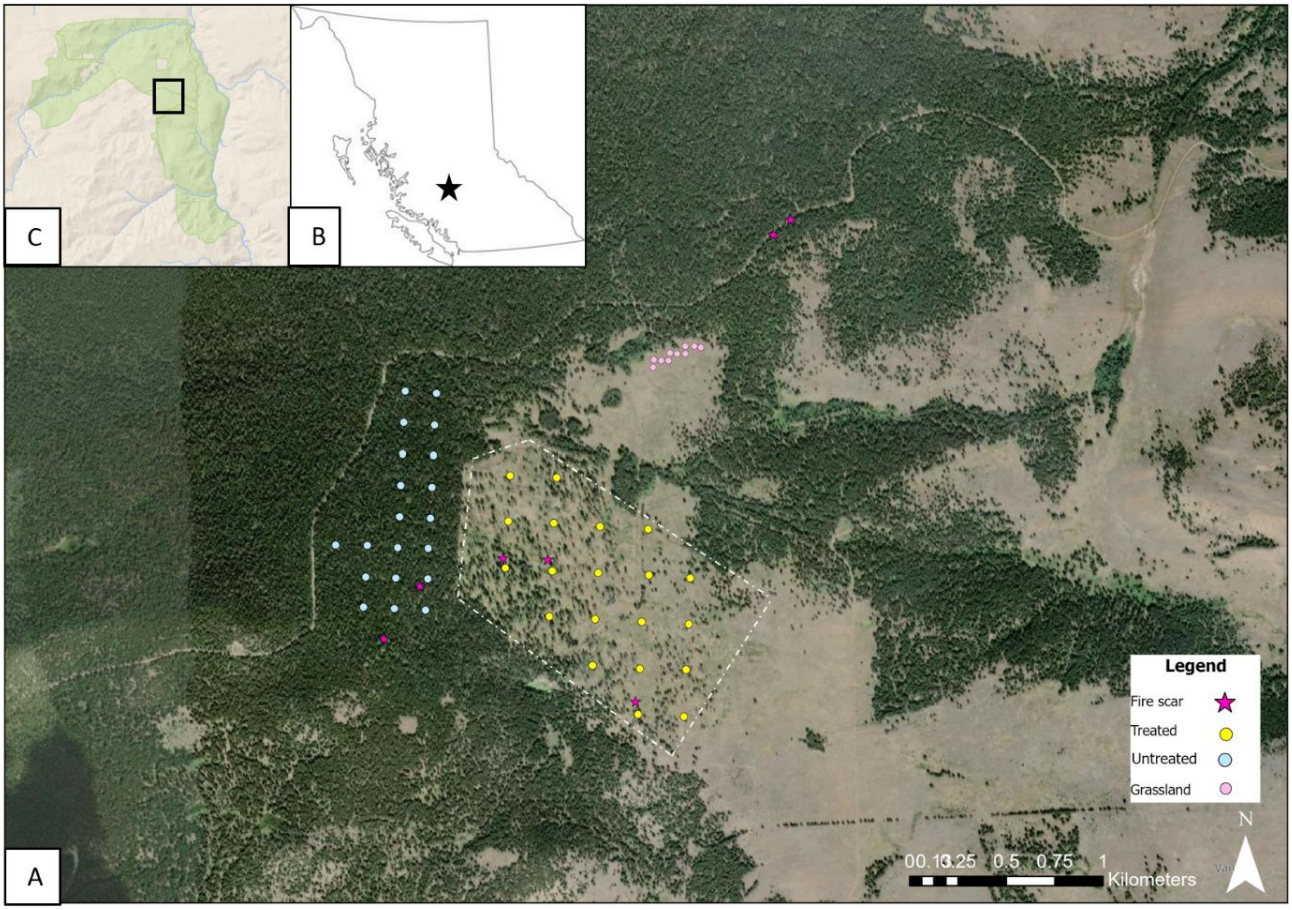

Figure 2.1. The study site (A) in British Columbia (B) within the Churn Creek Protected Area (shaded green; C). All surveyed plots in each site type are marked: treated (yellow circles), untreated (light blue circles), and grassland (pink circles). Sampled fire scar tree locations are marked with purple stars. The perimeter of the treated area is marked with a white broken line.

METHODS

Study Area

The study area is located in the CCPA in the central interior of BC (Figure 2.1) on the Fraser Plateau, part of the montane cordillera. This landscape is the traditional territory of the Stswecem'c Xget'tem (SXFN) First Nation within the greater Secwepemc Nation. The CCPA was established in 1995 for the conservation of grassland ecosystems. It covers 36,747 ha of grasslands and dry, interior Douglas-fir forests and is the largest protected area of native grasslands in BC. The CCPA is characterized by a mosaic of grassland and forest patches, wetlands, aspen copses, cliffs and clay banks along the Fraser River (Iverson, 2004). Low,

middle, and upper elevation grasslands are found within the protected area. Within the CCPA land, uses include managed cattle grazing (500 head; BC Parks 2000), backcountry horse-riding, hunting, and hiking. Indigenous communities gather traditional plants, fish, and hunt in the protected area. The ~60ha study area was selected with the BC Ministry of Forests where thinning and burning were used to mimic a historical, or legacy, forest-grassland ecotone (Figure 2.1).

The study site is in the Interior Douglas Fir very-dry mild (IDFxm) biogeoclimatic zone (elevation 800-1200m) where species such as bluebunch wheat grass (*Pseudorogeneria spicat*a), pinegrass (*Calamagrostis rubescens*), yarrow (*Achillea millefolium)*, prickly rose (*Rosa acicularisa),* Rocky-mountain juniper (*Juniperus scopulorum*), and Douglas-fir are commonly found. The target ecosystem of this study, forest-grassland ecotones, primarily occupy elevations between 850 m and 1100 m asl. This region is characterized by dry, warm summers and cool, moist winters where these dry, mature montane forests, and grasslands can be found (BC Parks, 2000; Steen and Coupé, 1997). Soils in the upper grasslands are generally dark brown chernozems (BC Parks, 2000). Average summer temperature is 16.2 ºC in July and -6.5º C in January and an average annual precipitation of 326.1mm for the closest weather station (Clinton, BC) (Government of Canada 30-year climate normal 1991-2020). The study site is on an east facing slope at the upper edge of the IDFxm at ~1100-1200m asl (Figure 2.1).

Sampling Methods

To assess the effect of the thinning treatment and prescribed fire on the understory grassland community and tree growth, 50 plots (*radius* = 3.99m; Figure 2.2) were established (Figure 2.1) in June 2023 including twenty plots in the treated site, twenty plots in an adjacent untreated (encroached) forest site and ten plots at a representative grassland site (Figure 2.3). Plot locations were determined by overlaying a grid randomly on a base map and plot centres were identified at the grid intersection points. Widespread grazing and/or Douglas-fir encroachment in the upper grasslands limited the area to situated plots in the grassland site. Ten plots in the grassland site were identified where minimal grazing was observed; these plots were located at similar elevations and aspect as the treated and untreated areas (Figure 2.1). At each plot, all vegetation species were identified, and percent canopy cover was ocularly estimated. Canopy cover was

defined as the area of ground underneath covered by the vegetation. A 2.5m radius plot nested in the 3.99m plot was used to count seedlings (<30cm in height) and saplings (>30cm <1.3m in height) (Figure 2.2). To assess the effect of the treatment on tree growth and climate sensitivity, two increment core samples were taken from the two closest Douglas-fir trees (>25cm DBH) to the centre of each plot in the treated and untreated sites (n = 80 trees). Cores were collected ~ 30cm above ground height. The grassland site did not have trees within or near plots so cores were not collected. In the untreated site, stems/ha was assessed using 5.64m radius plots for trees >5cm <12.5cm DBH, and 11.28m radius plots for trees >12.5cm DBH. In the treated site, 17.84m radius plots were used due to sparseness of trees in order to get an accurate representation. A total of three plots from each site were used to create an average stems/ha.

Following treatments in 2005 and 2006, invasive plants were removed on one occasion by pulling in 2007. In our study, if invasive species were detected, species presence and percent cover was recorded. Invasive species were determined using the Cariboo Chilcotin Coast Invasive Plant Committee and the Invasive Species Council of BC. Salsify (*Tragopogon* spp.) is considered a species of concern in the region, and Canada thistle is a provincially listed noxious weed species. Kentucky bluegrass (*Poa pratensis*) is not listed as a provincial invasive species but is an introduced agronomic species, which can outcompete native grasses in sensitive ecosystems such as the grasslands in interior BC (Holden et al., 2025) and was considered as invasive in this study. Common dandelion (*Taraxacum officinale*) is a non-native pioneer species but was considered as an exotic or invasive species for this study.

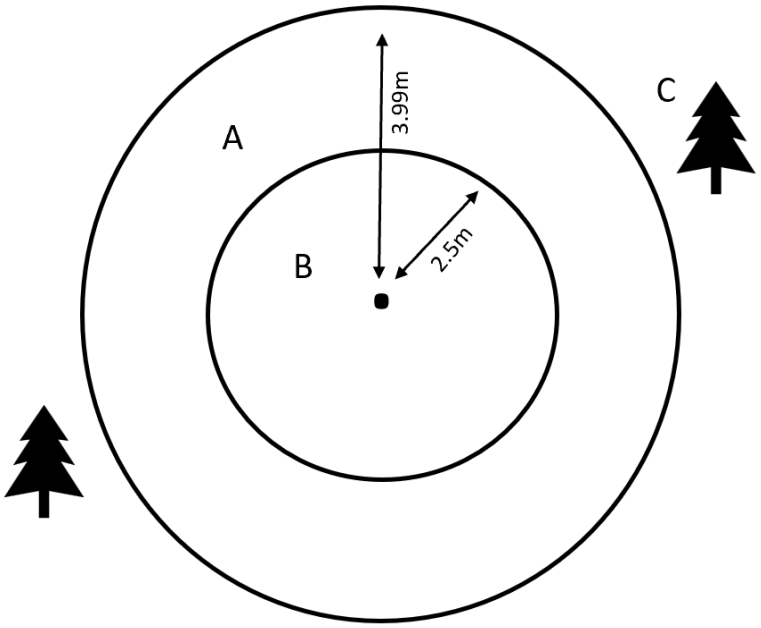

Figure 2.2. Plot layout design. (A) The 3.99m radius plot for plant cover surveys. (B) The 2.5m radius plots for seedling and sapling surveys. (C) The two closest trees over 25cm diameter at breast height to plot centre were cored.

To determine the fire history for this study site, I searched for fire-scarred trees opportunistically throughout the treated and untreated forest areas. The sampling was restricted to dead standing and fallen trees. Using a chainsaw, full or partial cross sections were obtained.

Figure 2.3. Left: Treated site where mechanical thinning and prescribed fire were completed in 2005 and 2006. Right: Untreated site showing dense infilling of the understory and located directly adjacent to the treated site. Photos taken in June 2023.

Sample Processing

Tree core samples were air dried and then mounted to slotted boards and both cores and fire scar samples were sanded progressively to 400 grit. All tree-ring samples were scanned using a high-resolution scanner and uploaded to Windendro (Regent Instruments Incorporated, 2001) for ring-width measurements. Ring-width measurement series were crossdated visually and statistically confirmed using COFECHA (Grissino-Mayer, 2001; Speer, 2009).

Statistical Analysis

Analyses were conducted in R version 4.1.2. Analyses were conducted on three datasets: (1) understory species community, (2) understory species by functional group, and (3) radial-tree growth. First, to assess the effect of a thinning treatment and prescribed fire on understory plant community, percent cover data was converted to an abundance ratio (proportion values between 0 and 1) and then transformed using an arcsine square root transformation. A beta dispersion test

was then conducted to test for homogeneity of variance between site types. For understory species by site type, a permanova was conducted using the *Vegan* package (Oksanen et al., 2022) to assess if site types are significantly different based on species composition and percent cover. Alpha (plot level) and beta (site level) diversity was also assessed by looking at differences in species richness within and across sites. A pairwise adonis post-hoc test was used to confirm these results from the *pairwiseAdonis* package (Martinez-Arbizu, 2020). A two-dimensional non-metric multidimensional scaling (NMDS) analysis was used to further compare and visualize species abundance for each plot by site type. A community species matrix was created and then the distance matrix using the Bray-Curtis method prior to NMDS. The Shannon-Weiner index (alpha diversity) was calculated along with the species richness. Species richness equates to the number of species detected in each site type. SIMPER (similarity percent) (Clarke, 1993) analysis for the community data was conducted to determine which species account for the largest differences between sites.

Second, differences between site types (i.e. grassland, treated, untreated) were examined by grouping understory species into functional groups. Functional groups were divided into tree (<1.3m height), shrub, graminoid, forb, and cryptogam. A Krukall-Wallis test was conducted using *R.statix* (Kassambara, 2023), and if significance was found a post-hoc pairwise Wilcoxon rank-sum test was completed to assess which groups were the most significantly different from one another. This analysis was performed on percent cover values and species richness. Cryptogams were not analyzed for species richness, as some species were grouped.

To assess radial-tree growth differences between treated and untreated sites, ring-width measurements were averaged between cores from the same tree, and if only a single core was collected, only one core was used. Ring width measurements were converted into basal area increment (BAI). Using *dplR* (Bunn, 2008) two chronologies were created using a bi-weight robust mean, one chronology for the treated area and one for the untreated area. Chronologies were detrended using the default cubic smoothing spline which is 50% frequency response at .67x the series length (Cook and Peters, 1981). Alternate detrending methods were tested (e.g., negative exponential curve) with little differences detected. Chronologies were then partitioned into two time periods: a 21-year period prior to treatment (1984-2005) and a 16-year post-treatment period (2006-2022) to generate four chronologies: pre-treatment treated site, pre-

treatment untreated site, post-treatment treated site, post-treatment untreated site. A t-test was used to assess for significant differences in growth between the two intervals of analysis for both the treated and untreated areas. BAI chronologies were then compared to climate data for the two intervals of analysis. Climate data included monthly maximum temperature, total monthly precipitation, and monthly climate moisture index (CMI) from March to September in the growing year. Aggregates of June/July, July/August, June/July/August, and May/June/July climate parameters were also computed for analysis. Climate data was obtained from Climate BC (Wang et al., 2016) that interpolates site-specific climate data based on the three closest weather stations to the study site since 1900. Pearson's correlation analysis was used to determine if significant relationships between tree growth and climate parameters were present using *Treeclim* (Zang and Biondi, 2024). This analysis examined the time periods 1984-2005 and 2006-2022 for both the treated and untreated chronologies.

Fire scars were dated and when possible, assigned a "latewood", "earlywood", or "ring boundary" position within the cellular structure. The fire history analysis and exploration system (FHAES) (Sutherland, 2017) was used to compile fire events and create a fire history for the study area. Fire events were cross-referenced with those identified in Harvey et al. (2017) and if samples from more than two trees (from this study, or one tree from this study and one tree from Harvey et al. (2017) that was within 1 km of the study area) recorded the same year, it was considered a fire event.

RESULTS

Understory Plant Community Analysis

A total of 98 understory plant species in the treated, untreated and grassland plots were recorded. Fifty-nine forb, 17 graminoid, eight shrub, three tree, and 10 cryptogam species were observed. Cryptogams were classified by species and genus where possible, but when unable to identify, "moss" or "lichen" was used as an overall grouping. The majority of species were uncommon, recording less than 0.5% cover (n=72). Twenty-six species were found to have more than 0.5% cover (Table 2.1). Exotic/non-native species recorded were: common dandelion, salsify, Canada thistle, and Kentucky bluegrass (Table 2.1).

Table 2.1. Species found with greater than 0.5% mean cover for all plots by site type. * Denotes non-native species. Underlining denotes site type with highest average.

Species with more than 0.5% - Mean Cover (%)			
Species	Grassland	Treated	Untreated
	Graminoids		
Calamagrostis rubescens	0	40.5	23.5
Festuca saximontana	0.5	0	0
Hesperostipa comata	4.84	0.01	0
Juncus balticus	1.95	0	0
*Poa pratensis**	23	7.4	0.03
Pseudorogeneria spicata	2.1	0.05	0
	Forbs		
Achillea millefolium	3.6	0.6	0.14
Antennaria spp.	5.6	0.12	0.35
Arctostaphylos uva-ursi	0	1.4	0.16
Arnica cordifolia	0	0	3.9
Cerastium arvense	3.6	0.03	0
Eurybia conspicua	0	0.51	0.6
Fragaria virginiana	0	0.53	0.3
Galium boreale	0.25	0.54	0.35
Lathyrus ochroleucus	0	0.29	0.51
Penstemon procerus	0.85	0.16	0.005
*Taraxacum officinale**	1.54	0.96	0.20
Tragapogon spp.*	8.2	0.5	0.03
Camandra umbellata	5.5	0	0
Linnea borealis	0	0	0.8
	Shrubs		
Juniperus spp.	0.1	0.08	2.51
Rosa acicularis	0.15	2.6	0.27
Symphoricarpos spp.	0.22	0.98	0.03
	Trees		
Pseudotsuga menziesii	0.01	1.43	0.56
	Cryptogams		
Moss	0	0	37
Lichen	2.7	0	0

Functional groups were compared between sites to assess for differences using a Kruskall-Wallis test for percent cover and richness. If significant results were found ($p<0.05$) then a Wilcoxon rank-sum test was run for each set of pairs (treated/untreated, treated/grassland, untreated/grassland). For species richness comparisons in the forb functional group there were significant differences between the plots in the treated and untreated sites and between the plots in the untreated and grassland sites (padj=0.018). All pairs of sites were not significantly different from one another (padj= <0.050) for graminoids. Shrubs differed significantly between the untreated and treated sites (p.adj=0.004), and between the grassland and treated sites (p.adj=0.021). Percent cover of seedlings and saplings did not have any significant differences, and cryptogams were not compared between sites due to grouping of species. Functional group richness is summarized in Figure 2.4.

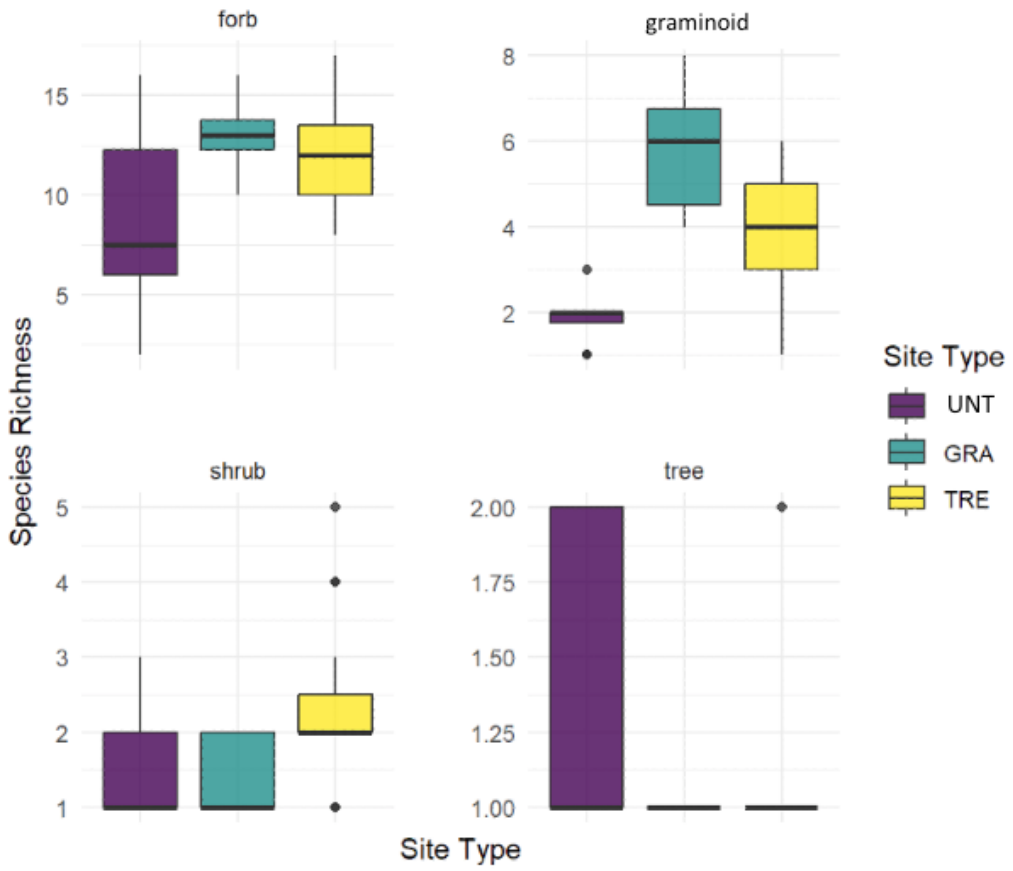

Figure 2.4. Species richness for each site type (untreated (UNT), treated (TRE), grassland (GRA)) for each functional group. Whiskers represent upper and lower quartile ranges, and dots are outliers. A pot-hoc pairwise Wilcoxon test was completed to determine significant differences between site types for each functional group.

Understory plant communities were significantly different between all three sites. Permanova results indicated site type (grassland, treated, untreated) has a significant result on the cover of plant species found ($p = 0.001$). Beta dispersion results indicate ($p = 0.06$) that the permanova assumptions were met and is a suitable test for this data. Post-hoc pairwise adonis results (Table 2.2) indicated that all site types are significantly distinct from each other ($p < 0.001$) (Table 2.2), with a higher level of variance between the grassland and untreated site types ($F=40.949$, sum of squares $=4.98$).

Table 2.2. Post-hoc pairwise adonis test results. All results indicate sites significantly differed from one another, but grassland plots compared against the untreated plots had the highest F value and sum of squares. This is likely due to a higher level of variance between sites.

Grassland vs Untreated

Df	Sum of Squares	R^2	F	Pr(>F)
1	4.9769	0.5939	40.949	0.001***

Grassland vs Treated

Df	Sum of Squares	R^2	F	Pr(>F)
1	3.1460	0.4032	18.916	0.001***

Untreated vs Treated

Df	Sum of Squares	R^2	F	Pr(>F)
1	2.7151	0.31857	17.765	0.001***

Average species richness (Figure 2.5) varied across the sites. Plots in the untreated, treated, and grassland sites recorded an average richness of 14.8, 19.4, and 21.2 respectively. Overall, the highest plot-level species richness was 28, recorded for two plots in the treated site. Species

richness increased in the treated site, but overall, the grassland site had the highest average richness. The Shannon index (alpha diversity), measuring richness and evenness, results indicate that the grassland site plots had highest diversity at 2.05, untreated site plots were 1.43, and treated site plots were 1.41.

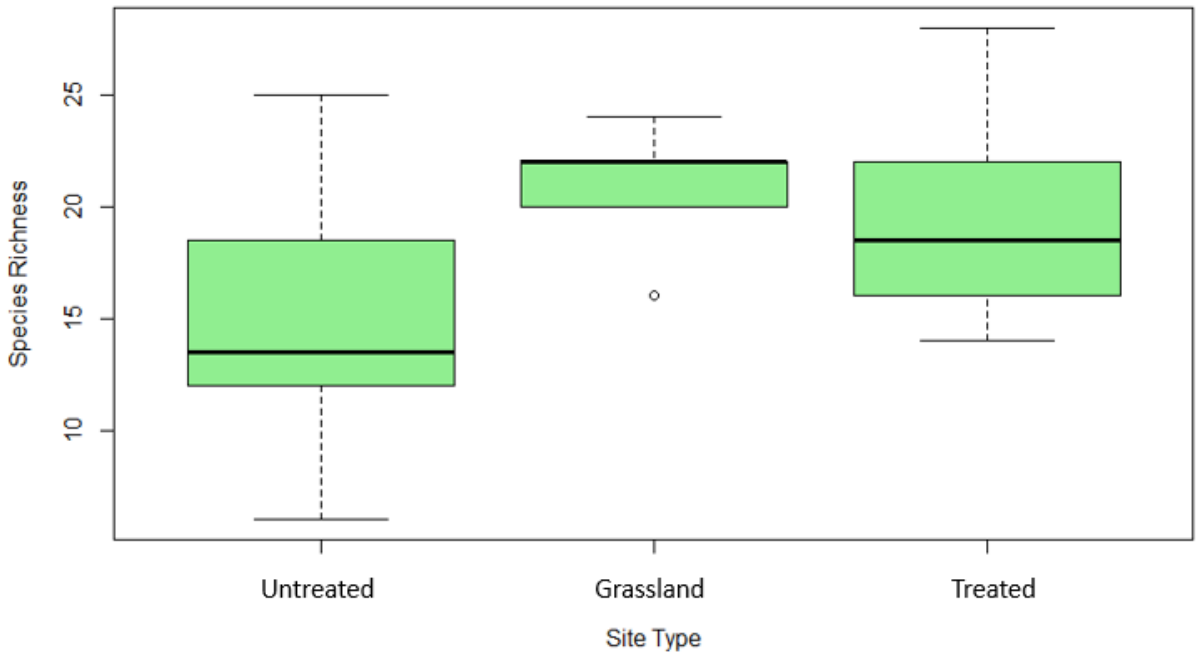

Figure 2.5. Total species richness for each site type (untreated, grassland, and treated). Whiskers represent upper and lower quartile ranges, and dots are outliers.

The analysis between cover of forbs in each site type indicate significantly different functional group composition between the plots in the untreated site and the plots in the grassland site and also between the plots in the grassland site and plots in the treated site (p.adj=0.0000219), but was not significantly different between plots in the untreated and treated sites. Grassland plots had the highest percent cover values of forbs. Graminoid percent cover between the untreated plots and treated site plots was significantly different (padj=0.002). The treated site plots had the highest graminoid percent cover. The functional group of shrubs was significantly different for plots between the untreated and treated sites (p.adj=0.011) and also between the plots in the grassland and treated sites (p.adj=0.011) but was not significantly different between plots in the

untreated site and plots in the grassland site. For trees (seedling and sapling), percent cover was significantly different between plots in the untreated site and plots in the grassland (p.adj=0.0000217) and between plots in the grassland site and plots in the treated site (padj=0.01). Lastly, cryptogams were significantly different (p.adj<0.05) between all sites. Functional group percent cover is summarized in Figure 2.6.

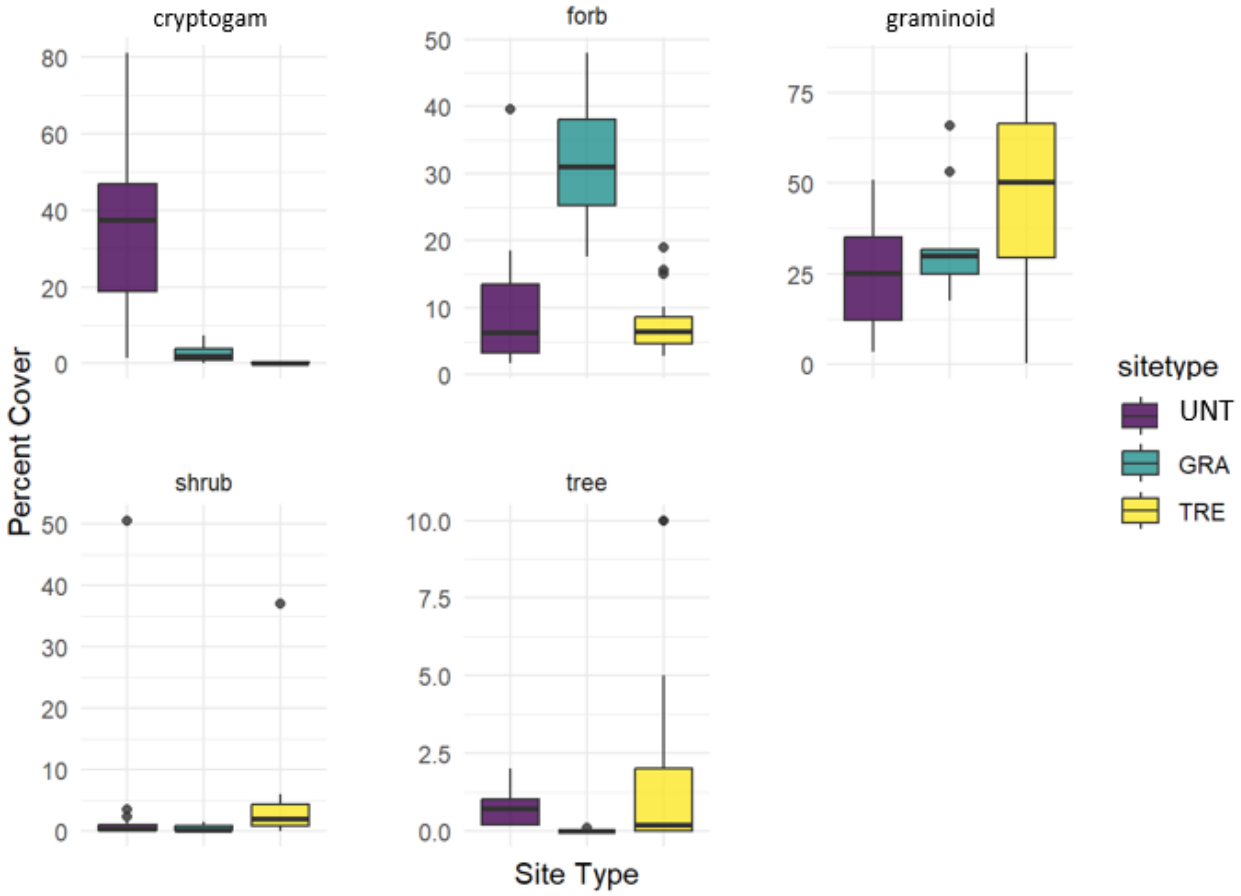

Figure 2.6. Average percent cover for each site type (untreated (UNT), treated (TRE), grassland (GRA)) by functional group. Whiskers represent upper and lower quartile ranges, and dots are outliers. A pot-hoc Pairwise Wilcoxon test was completed to determine significant differences between site types for each functional group.

A SIMPER (similarity percent) analysis was completed using the Bray-Curtis dissimilarity matrix to identify species that contributed most to the change in plant community between sites (Table 2.3). Pinegrass has a high contribution but was not significantly different between

grassland/untreated (p = 0.989) and untreated/treated (p = 0.183) (Appendix Table A1). Dominant species included pinegrass, moss, Kentucky bluegrass, and salsify (Table 2.3).

Table 2.3. SIMPER (similarity percent) analysis results for species with a cumulative contribution accounting for 75% of the differences in the Bray-Curtis dissimilarity matrix between site types. Sites are GRA (grassland), UNT (untreated), and TRE (treated).

Species	GRA vs. UNT	GRA vs. TRE	UNT vs. TRE
Calamagrostis rubescens	0.2514		0.3614
Poa pratensis	0.417	0.3395	0.6539
Tragopogon spp.	0.5805	0.5006	0.7344
Comandra umbellata	0.6399	0.5722	
Antennaria spp.	0.6802	0.6231	
Hesperostipa comata		0.719	
Cerastium arvense		0.749	

NMDS indicated the understory community of the treated site is transitioning away from a composition like that of the untreated community, with some of the plots more similar to grassland composition (Figure 2.7). NMDS stress values <0.2 indicates a good fit and Shepard's plot also indicates a good fit as points are clustered around the centre line. The untreated and grassland communities were the most dissimilar from one another. Distinct groupings are evident (Figure 2.7) between communities based on site-type indicating significant differences in composition, as evident in the permanova and pairwise adonis results.

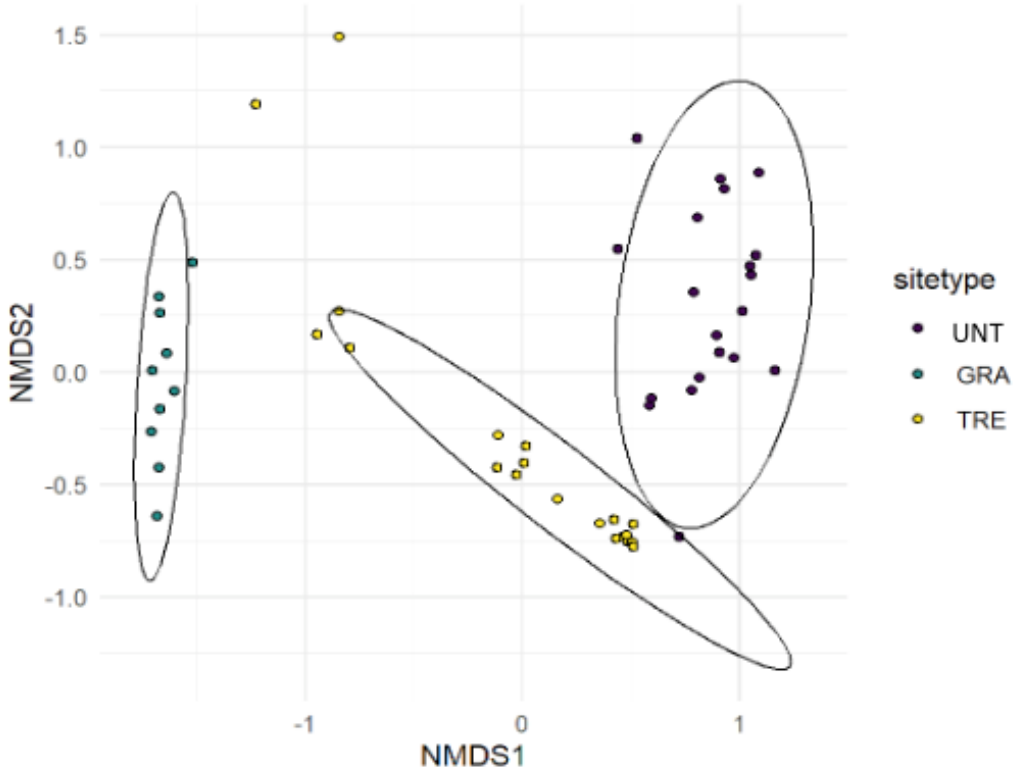

Figure 2.7. Non-metric multidimensional scaling (NMDS) bi-plot comparing plant community composition between the three site types (UNT (untreated), GRA (grassland), and TRE (treated)). Each point represents a plot (*n*=50). Points closer together have more similarity. The ellipses represent each site type, where distinct groupings are present.

Seedlings and saplings

Among the treated site plots, an average of 30 seedlings (range: 0-237) and two saplings (range: 0-8) were recorded for Douglas-fir. An average of zero trembling aspen (*Populus tremuloides*) seedlings were recorded, and only one plot recorded saplings (*n*=38) in the treated site; this plot fell directly in an aspen copse and these saplings were likely suckers from the surrounding mature trees. In the untreated site, there was an average of 77 seedlings (range: 6-276) per plot, and two saplings (range: 0-15) per plot for Douglas-fir (Appendix Table A2). In the untreated site, two plots recorded other sapling species: spruce (*Picea engelmannii* x glauca)(*n*=1) and trembling aspen (*n*=2). Thirteen plots were used for analysis in the treated site. No seedling and saplings were recorded in the grassland plots.

Tree-Ring Analysis

The average age of the mature trees sampled (two closest trees to plot centre with a DBH >25cm) in both the treated and untreated sites was 120 years as a minimum estimate (treated: 34-355 years; untreated: 84-173 years). The average stems/ha in the treated site was 130, and the average stems/ha in the untreated site was 1708 for all trees >5cm DBH. Forty trees were used in the treated site chronology and 39 trees were used in the untreated site chronology. All mature trees sampled were Douglas fir. When the growth of trees (represented as BAI) in the treated and untreated sites over the 2005-2022 interval (post-treatment) was compared the treated site had significantly greater growth than in the untreated site ($p = 0.0001266$; Table 2.4). Prior to the treatment (1987-2005) the growth of trees was not significantly different ($p = 0.8953$; Figure 2.8) between the untreated and treated sites.

Table 2.4. Results of t-tests that were conducted prior to treatment (before 2005) and after treatment (after 2005) on basal area increment (BAI) values over time for Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) trees within the study site. 'x' indicates the untreated stand and 'y' indicates the treated stand. * denotes significance.

	P-value	Avg. BAI	Degrees of Freedom	% Difference in growth
Pre-treatment (<2005)	0.8953	x = 1112.06 y= 1132.83	32	1.81
Post-treatment (>2005)	0.0001266*	x = 931.19 y = 1605.22	34	72.38

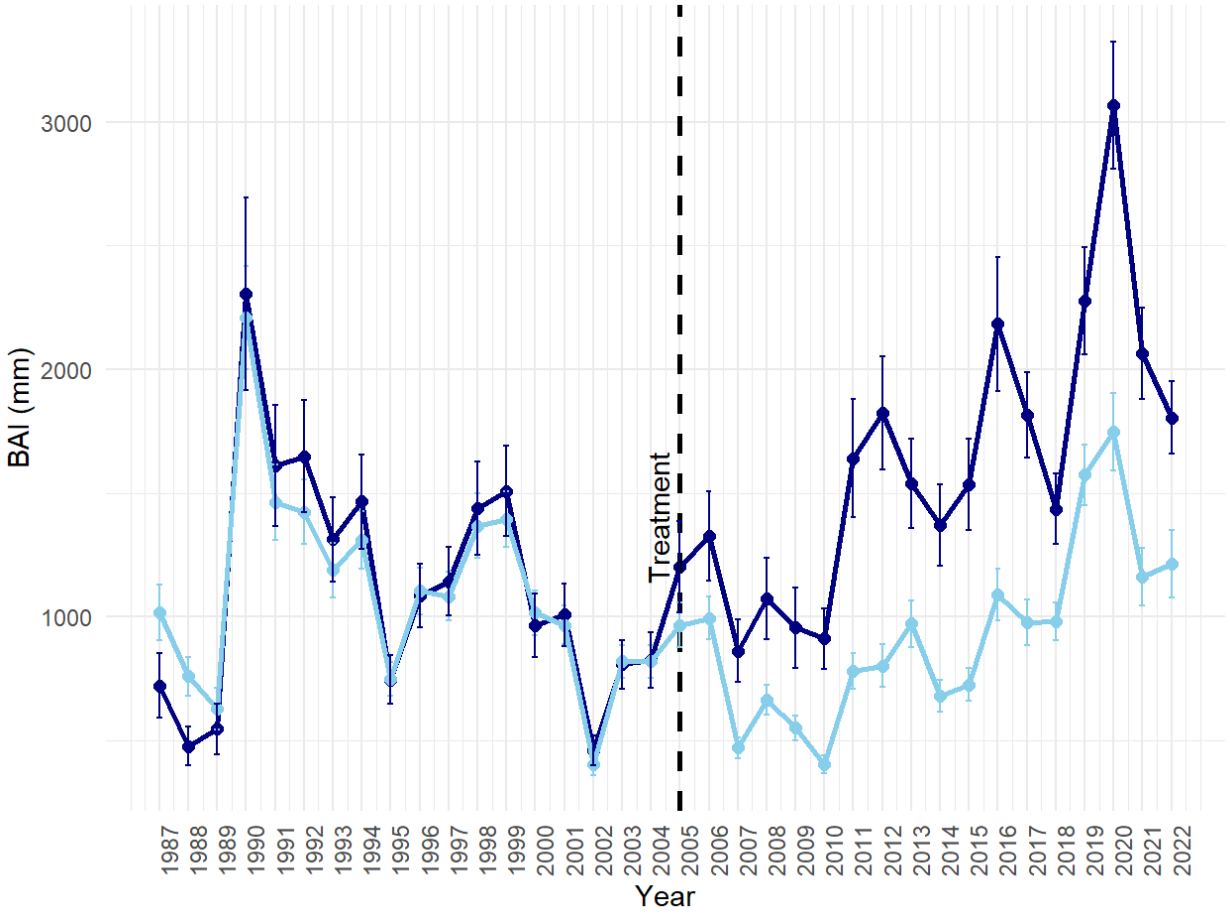

Figure 2.8. Annual basal area increment (BAI) measurements for averaged tree ring chronologies plotted for the period 1987 to 2022. The black dotted line indicates the year mechanical thinning occurred (2005) and prescribed fire was applied in 2006. Bars above and below each data point represent one standard deviation. Prior to the treatment, Douglas-fir (*Pseudotsuga menziesii var. glauca*) trees were growing at approximately the same rate (1.81% difference in growth). Growth of trees in the treated site (dark blue) showed an average increase in growth of 72.38% (2006-2022) when compared to the trees that remained in an untreated closed canopy stand.

To test for differences in climate-growth relationships between the untreated and treated study sites, correlation analysis was used. In Appendix Table A3, Pearson's correlation coefficient values are shown for both treated and untreated stands, in the before treatment and after treatment time periods in relation to three climate parameters: monthly precipitation, maximum temperature, and monthly CMI. The climate-growth relationships follow similar trends for the two chronologies (treated and untreated) where climate parameters in key growing season

months (June and July) were significantly correlated to tree growth. Tree growth was positively correlated to precipitation prior to treatment (1987-2005) in June (treated r = 0.552, untreated r = 0.524) and June/July (treated r = 0.539, untreated r = 0.562) with no significant negative correlations in the treated and untreated sites (Figure 2.9). For the post-treatment interval (2006-2022), the growth of trees in both the treated and untreated stand is positively correlated to July precipitation (treated r = 0.584, untreated r = 0.535), but only the treated stand had significant positive correlations with averaged May/June/July precipitation (r =0.424). Lastly, in the post-treatment interval (2006-2022) July/August precipitation is positively correlated (r = 0.520) to tree growth in the untreated stand. Similar climate-growth relationships were found for maximum temperature and CMI (Appendix Figures A1 and A2). I found only two significant correlations for tree growth in the treated stand during the post treatment interval (2006-2022) with CMI in July (r = 0.618) and July/August (r = 0.545). Correlations between tree growth and maximum temperature are generally negative for growing season months; greater maximum temperature was significantly related to decreased ring width in June (treated r = -0.444, untreated r = -0.462) and June/July (treated r = -0.336, untreated r = -0.318) prior to treatment. Post treatment, tree growth for the treated stand has a significant negative correlation (r = -0.498) with maximum temperature for July. In summary, the climate growth relationships for the untreated and treated chronologies are generally similar (Figure 2.9, Appendix Figures A1 and A2).

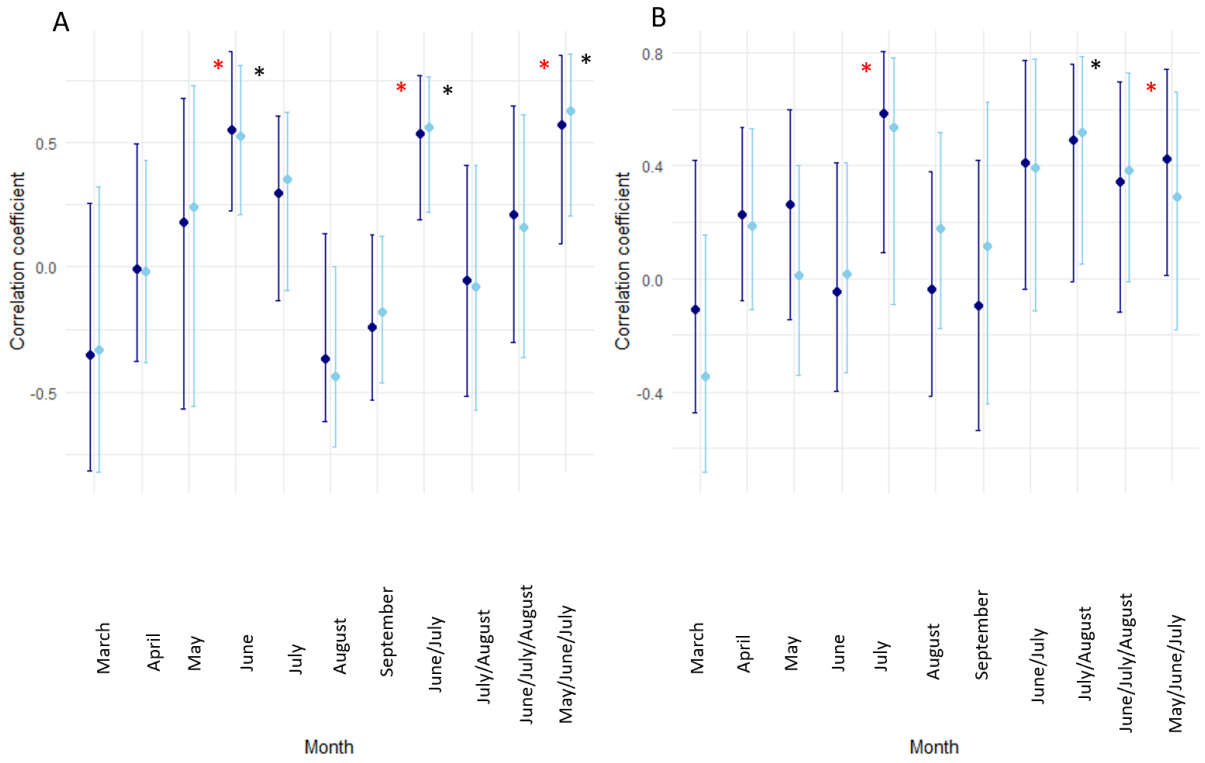

Figure 2.9. Pearson correlation coefficient values for tree ring growth and monthly precipitation for chronologies (A) prior (1987-2005) to and (B) after (2006-2022) treatment. Black * indicates significance (95% confidence intervals) for the untreated site, and red * indicates significance for treated site. Dark blue represents the trees in the treated site and light blue represents the trees in the untreated site. Monthly precipitation values for single months in the growing season starting in March until September were examined, as well as means of monthly values for June/July, July/August, June/July/August, and May/June/July.

Fire history

Nine samples from Douglas-fir trees with fire scars were collected (full or partial cross sections). One sample was collected but not used for analysis due to significant decay. All samples were taken from dead trees, and two were from the same tree and combined (Figure 2.1). The earliest fire event was in 1500 and the most recent occurred in 1948 (Figure 2.10). Sixty-two individual fire events were identified among the samples, with 24 fire events confirmed by two or more trees recording in the same year. Fire scars were clearly visible on cross sections (Figure 2.11). Most of the fires were assigned to the "dormant season" as they occurred at the ring boundary of the latewood and earlywood. Only two fire events were identified in the earlywood (1610, 1794).

The ages of sampled trees ranged from an estimated ~145 years to ~373 years of age indicating they were mature trees when they died (average age was 288 years). In some cases (*n*= 6) only minimum ages could be determined due to substantial decay. The average fire return interval for all samples is 27 years (range 11-46 years).

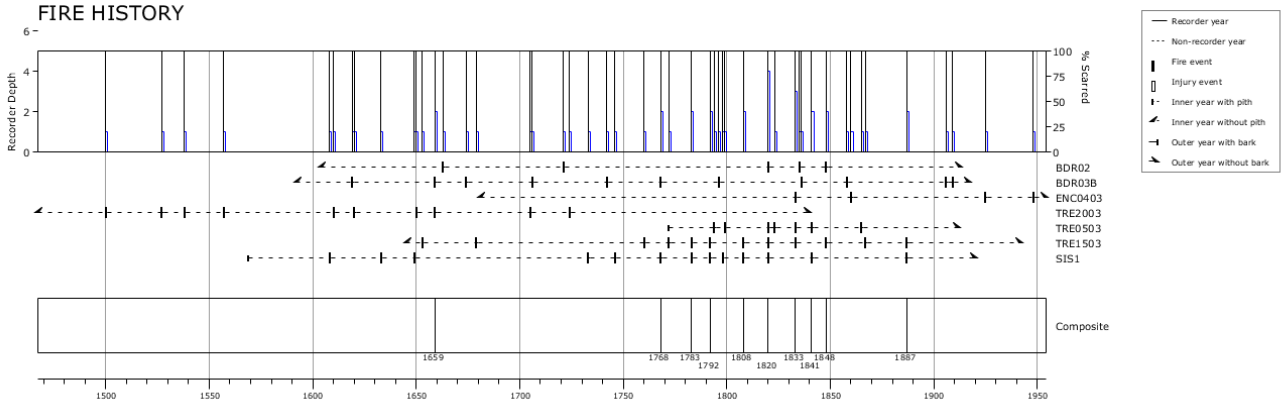

Figure 2.10. Reconstructed fire history for the study area in the Churn Creek Protected Area. Scars were collected from full or partial cross sections of dead and downed trees. Fire events date back to 1500 (only one recording sample), extending the known fire history in this area another ~120 years from Harvey et al. (2017).

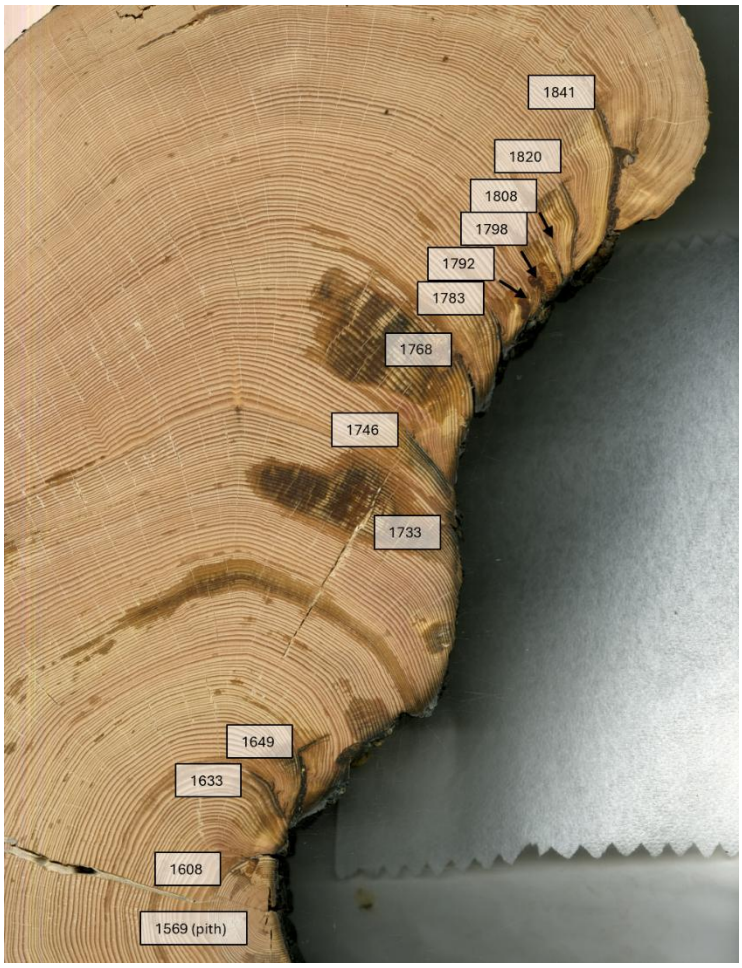

Figure 2.11. Sample from a fire-scarred Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) tree collected from the Churn Creek Protected Area (CCPA) with 12 past fires identified. The first fire recorded was in 1608 and the last fire was in 1841. This tree was found fallen over within the untreated stand in a closed-canopy forest. Pith is 1569.

DISCUSSION

Forest thinning and prescribed fire increase species richness but not diversity

Species richness, the number of species present in a given area, and species diversity, which accounts for both richness and the relative abundance of species, are key metrics for assessing overall biodiversity in an ecosystem (Gotelli and Colwell, 2001). In this study, species richness was higher in the treated site than in the untreated site; however, grasslands consistently exhibited the greatest average richness across all sites (Figure 2.5).

The treated site mimics a forest–grassland ecotone, as it exhibits characteristics of both ecosystems, including mixed understory species composition and structural heterogeneity. Ecotones are often thought to support elevated richness due to the overlap of species from both adjacent systems and the increased environmental variability (Kark, 2012). Studies specific to BC are limited, however, a study by Ross (2016) in Lac du Bois Provincial Park suggests that forest–grassland ecotones may, in some cases, exhibit lower richness than adjacent ecosystems. The results of this study indicated that ecotonal richness was not higher than the surrounding habitats as expected. In my study, overall richness in the treated site was lower than that of the grassland site aligning with these findings of Ross (2016)**,** however, two plots in the treated area recorded the highest richness across all sites (Figure 2.5). This indicates considerable spatial variation in post-disturbance community development across this 60ha ecotone site and is related to the size (scale) at which the study is conducted. Such fine-scale heterogeneity can enhance diversity and promote species persistence following disturbance (Goodwin et al., 2018). Considering the treatment occurred 17 years prior to data collection, the understory plant community is still undergoing successional change, and limited dispersal may restrict some species to localized patches. The greater richness in the treated site compared to the untreated site can be attributed to colonization of native forbs and grasses due to increased light, decreased competition, and decreased litter cover allowing for early establishment (Peterson and Reich, 2008; Williams et al., 2024). Although individual plots may vary in species richness and diversity in the treated site, the site as a whole becomes more ecologically diverse due to higher beta diversity, or variance across plots. Despite not always showing elevated richness, ecotones can still enhance biodiversity at larger spatial scales by supporting unique assemblages of species not found in either adjacent ecosystem alone (Sabo et al., 2005).

The Shannon–Wiener diversity index assesses alpha diversity, a function of both dominance and richness within each site. The grassland site exhibited the highest Shannon index (2.05). The untreated site had a slightly higher score (1.43) than the treated site (1.41) but are not significantly different. Despite similar values, this difference highlights how the community structure varies between sites. This variance is likely due to the untreated plots having fewer species overall (lower richness), but the percent cover values among those species were more uniform. In the treated site, community structure was characterized by strong dominance of a single species alongside many low-cover species, resulting in reduced evenness. Pinegrass

consistently exhibited the highest percent cover across treated plots (Table 2.1), driving the dominance effect observed. Many species in the treated site, particularly forbs, recorded less than 0.5% cover, which increased species richness, but because the Shannon-Wiener index accounts for evenness, it may underestimate the diversity present. Additionally, the site has experienced only a single treatment and is still changing through successional processes, which may contribute to its lower diversity compared to the neighboring grasslands or encroached stands. The Shannon index must be interpreted with care, as results are often biased towards abundance of species, rather than richness (Barrantes and Sandoval, 2009; Strong, 2016), which at this site, may not be the best representation of diversity.

Significant differences in functional group composition and percent cover between the three sites (grassland, treated, untreated)

Functional group cover for each of the three sites were significantly different. Cover of forbs was relatively similar between the treated and untreated areas, but richness was significantly increased in the treated area with species not found in the untreated site such as cutleaf anemone (*Anemone multifida*), small flowered penstemon (*Penstemon procerus*), and meadow death camas (*Toxicoscordion venenosum*) are contributing to this increased diversity. Some species such as heartleaf arnica (*Arnica cordifolia*), contributed to the slightly higher cover in the untreated site. Rhizomatous species such as heartleaf arnica can spread easily once established, covering a large area. In the plots in the treated site, there is a higher number of forb species present contributing to the higher richness (Figure 2.4), but percent cover is significantly lower than the grassland site (Figure 2.6). These low cover species (less than 0.5% average cover) can contribute more than common species to the increased species richness across fine-scale sites (Heegaard et al., 2013). As reported earlier, the grasslands had the highest average species richness, and this could be attributed to the increased number of grass and forb species (Figure 2.5). Microtopography and patchy resources (eg. cattle or ungulate manure, small depressions etc.) creates niches for different species that can coexist in grassland environments over a mosaic of small-scale patches (Petermann and Buzhgygan, 2021).

Pinegrass is the dominant cover of all species in the treated site and is a widespread species occupying the IDF zone (Meidinger and Pojar, 1991). In the SIMPER analysis (Table 2.3)

pinegrass had a high contribution to the differences between sites but was not statistically significant between grassland vs. untreated ($p = 0.989$) sites and untreated vs. treated ($p = 0.183$) sites, suggesting its influence is not consistent across site types. Pinegrass is a rhizomatous species and can quickly spread in a disturbed area, particularly in more xeric sites where competition for moisture is higher (Kremsater et al., 2009). An increase in light post-treatment is the driving factor for increased dominance and spread of this species, often establishing within five years and then persisting at a consistent level (Kremsater et al., 2009; Haeussler et al., 1990). This species also resprouts following fire but grazing cycles can inhibit spread and growth (Haeussler et al., 1990) helping manage competitive exclusion by pinegrass. However, repeated disturbances (frequent burning) can reduce competitive exclusion in some dominant species including pinegrass (Laughlin et al., 2008), although bunchgrasses may be susceptible to too much burning so a balance must be found. Many of the native bunchgrass species have not yet recovered following the treatments, in part due to the pervasive spread of pinegrass. Bunchgrasses including bluebunch wheatgrass, spreading needlegrass (*Achnatherum richardsonii*), and Columbia needlegrass (*Achnatherum nelsonii*) are often associated as later seral stage species (Meidinger and Pojar, 1991). These species had very low percent cover in the treated site, indicating the site has not yet achieved advanced successional stage. Further prescribed fire could reduce the cover of dominant species, creating opportunities for species already present but not yet fully established to increase in abundance.

Community assembly post-treatment a function of life-history strategies and seedbank

Fire historically shaped forest-grassland ecotones. Herbaceous species adapted to these dry and fire-prone conditions due to frequent low intensity surface fires. Different traits allow species to recover or re-establish post-fire from corms, bulbs, deep root systems/tap roots, rhizomes, seedbanks, or re-sprout from root crowns (Keeley et al., 2011; Rowe, 1983). Species such as Douglas-fir, snowberry (*Symphocarpus* spp.), prickly rose, bluebunch wheatgrass, pinegrass, yarrow, and sagebrush mariposa lily (*Calachortus macrocaprus*) each have life-history strategies that allow them to persist post-disturbance (Appendix Table A4). In the grasslands of BC, these species are also adapted to drought. Low amounts of precipitation require species to store water efficiently. This includes deep taproots, fine roots spread throughout the upper soil horizons,

bulbs, or ability to collect dew (Ocheltree et al., 2020). These traits are like those of fire adapted species and collectively enable these plants to thrive in an environment that is both arid and fire prone (Figure 2.10). The treatment applied to the study site in 2005/2006 emulated a historical disturbance, and as a result, more species with those adapted traits are re-establishing on the site. Specifically, more grassland species like small flowered pentstemon and lemonweed (*Lithospermum ruderale*) were found post-treatment which were not found within the untreated site. In the untreated site, species adapted to more moist sites including rattlesnake plantain (*Goodyera oblongifolia*), and sedges (*Carex* spp.) were more common. The release of nutrients post-fire may help support the new growth of species residing within the seedbed or soil (Reinhart et al., 2016). The increase in available light energy from thinning benefits the understory and allows for an increase in growth of initial colonizers or shift in species composition due to change in seral stage (Yearsley and Parminter, 1998). Community structure is not determined solely by disturbance such as fire, but also by species recruitment and physical heterogeneity across the site (Yearsley and Parminter, 1998).

The thinning treatment (2005) opened the forest canopy and increased available light for the understory plant community as well generated drier and warmer conditions at the ground surface. Pre-treatment, this area was very likely dominated by moss cover driven by lower light and higher soil moisture and similar in character to the current conditions in the untreated site (Figure 2.3, Table 2.1). Prescribed fire applied in 2006 decreased organic litter allowing for germination of grasses and forbs (Wayman and North, 2007). This disturbance created favourable conditions for colonizing species to establish and allowed previously dormant seeds within the seedbank to germinate. The re-emergence of many native species after the re-introduction of fire to the site can be facilitated by processes like scarification and serotinous-induced germination for certain seeds that have been dormant in the soil seedbank for decades (Kraft and Ackerly, 2014; Fleming et al., 2023). The last suspected fire (single fire-scar event) that occurred in the study site was in 1948, indicating the seedbank could have been dormant for 57 years. Legacies of past forest characteristics of herbaceous meadow understories persist to present day as evidenced by the species re-establishing at the site. Studies documenting seed bank persistence in grasslands are few (Clements et al., 2007; Kiss et al., 2018), and little is known about the seedbank in this area. Wind dispersal or animal deposition are also possible seed sources; the study site is within the

Mule Deer Winter range habitat (MDWR) and is home to many species of birds and small mammals.

Presence of invasive and non-native species higher in grassland, treated sites than the untreated

Salsify, an invasive species, had the highest average cover in the grassland plots and second highest in the treated site. Salsify was only recorded at one plot in the untreated site at a low percent cover (0.5%). This species requires high light availability and open growing conditions, and the untreated stand is not suitable for establishment (Government of BC, 2022). Canada thistle was only present in the treated site with a few individuals found across the site. This species is also shade-intolerant, making the untreated site a poor habitat (Government of BC, 2022). Each plant can produce upwards of 1,000 seeds and spreads aggressively through wind dispersal (SSISC, 2025). Common dandelion was present in all sites but again had the highest cover in the grassland plots, with the treated plots not far behind, and the least presence in the untreated site. These species are opportunistic invaders and were likely able to colonize after the prescribed fire where bare soil was available (Gioria et al., 2023). This also highlights the vulnerability of grasslands to invasive species, as most invasive species opportunistically establish in disturbed sites. If left unmanaged, some species could persist and spread aggressively (Canada thistle), but the overall cover of invasive species was quite low other than salsify. Salsify can be weedy in rangeland and grassland areas but can be managed through herbicide use (Mangold and Lansverk, 2011) and is also effectively reduced by 90% after exposure to fire (Government of BC, 2022; White and Boyd, 2017). For future management of this site, prescribed fire could be used to accommodate the reduction of this species (White and Boyd, 2017), however, additional considerations such as fuel curing and weather conditions must be carefully evaluated, as planning for prescribed burn objectives can be complex.

Nearly four times as many seedlings in the untreated site, but a higher number of saplings in the treated site

The treated site has fewer seedlings present, but a higher number of saplings compared to the untreated site. Establishment of Douglas-fir seedlings is based heavily on many factors which

include climate, solar availability, soil quality and moisture, proximity of seed source and competition from other vegetative species (Hermann and Lavender, 1990). High temperatures (>55 degrees Celsius) are lethal to conifer seedlings, and disturbances including fire can create warm conditions on the soil surface post-burn from higher insolation and lower albedo (Rank et al., 2022). In a more open stand, such as the treated site, conditions including increased light and high temperatures are more likely to occur as canopy cover is less. But once the tree is established (>30cm height), higher light availability may be beneficial as tree growth and size increases (Government of Canada, 2024). Douglas-fir seedlings also prefer more moist sites as tap roots have not yet developed, and fine roots near the surface require moisture (Hermann and Lavender, 19900; Heyerdahl et al., 2006). The untreated site provides more shade than the treated, creating a microclimate with cooler maximum temperatures than the untreated stand. This may contribute to the higher density of seedlings counted in the untreated plots (3.9 times more) (Appendix Table A2). Higher moisture in the spring contributes to higher germination rates in grassland areas (Bai et al., 2000); moisture may persist longer at the untreated site as evapotranspiration rates are less than in an open canopy stand. Needle litter is thought to inhibit seedling establishment in dense forests (Bai et al., 2000), but comparatively higher available moisture may outweigh this effect between the treated and untreated stands. A higher ratio of seedlings are surviving to reach sapling age in the treated site likely due to the increased light availability and reduced competition from surrounding trees. Higher moisture, shade, and seed source in the untreated site are likely beneficial for germination and early establishment of Douglas-fir seedlings leading to an overall greater number of trees, but reduced growth as the tree matures.

Treatment increased basal area increment (BAI) but did not change climate-growth relationships

Prior to treatment, growth patterns were similar between trees growing in the untreated and treated sites ($p = 0.8953$) (Figure 2.8), but tree growth in the treated site following the treatment exhibited a significant increase of 72.38% ($p = 0.0001266$) when compared to the untreated site trees. Increased light availability, along with reduced competition for resources such as water and nutrients, are key factors contributing to enhanced growth of trees in the treated site (Zald et al., 2022). While increased nutrient availability immediately after thinning and prescribed fire may

have contributed to the initial release, the persistence of elevated growth over the subsequent 17 years suggests a sustained improvement in growing conditions. The post-treatment growth release and increased vigor in conifers following thinning is well documented (Bose et al., 2018; Dagley et al., 2018; Fiedler et al., 2010); for example, Bose et al. (2018) found a 31% increase post thinning. However, the strong growth response observed in this current study (72.38%) represents the higher end of increases reported in the literature, contributing further evidence to the effectiveness of thinning in enhancing conifer growth. There was an average of 1708 stems/ha in the untreated site (>5cm DBH) and an average of 130 stems/ha in the treated site. Historical densities in the IDF have been found to be between 92-150 stems/ha (Brookes, 2019; Brookes et al., 2021; Greene, 2021). This treatment falls within the appropriate thinning range for this site's ecology and history. Increased growth and vigor following treatment also have important ecological implications, including greater crown development, enhanced carbohydrate storage, and improved wildlife habitat and forage availability (Crotteau & Keyes, 2020). Additionally, larger tree size is generally associated with increased resistance to wildfire and prescribed fire due to thicker bark and greater crown to base height (Crotteau & Keyes, 2020).

Despite the strong growth response to the treatment applied in 2005/2006, climate-growth analyses were broadly similar for the treated and untreated chronologies with minimal differences in Pearson correlation values (Figure 2.9, Appendix Figure A1 and A2, Appendix Table A3). Overall, the trees in the treated and untreated sites had similar climate-growth relationships with little change in strength or direction of climate associations between the two periods of analysis. In the treated and untreated sites, precipitation exerted the strongest influence on growth of trees, particularly during early summer, while higher growing-season temperatures were generally associated with reduced growth (Figures 9 and Appendix Figure A2). This finding is consistent with other studies documenting moisture limitation on the radial growth of interior Douglas-fir in BC (Beedlow et al., 2013; Chen et al., 2010; Fritts, 1974; Griesbauer and Green, 2010a; Watson and Luckman, 2002). Douglas-fir tend to be more influenced by site level factors (i.e., microclimate, shading) when establishing, but as the trees mature, they have a stronger relationship to regional climate variables (Hermann and Lavender, 1990). Since Douglas-fir are already sensitive to regional climate, climate-driven growth responses are oftentimes exacerbated in moisture limited environments and likely points to why the sensitivity did not change in the treated stand post-treatment. The influence of climate on

Douglas-fir trees at their edge range can be complex, particularly over short time spans (Griesbauer and Green, 2010b).

Fire return interval of 27 years

Nearby to the study area, Harvey et al. (2017) documents a fire return interval of 15-25 years and a historical mixed severity fire regime. Harvey et al. (2017) found that fires occurring in forest-grassland ecotones (within 400 m of large grassland areas) were more frequent than widespread fire events between 1600-1900 AD and supported variable forest structures maintaining a mosaic on the landscape. In this current study, evidence to further support these conclusions was found. Here, I report an average fire return interval of 27 years over 1500 to 1948 (Figure 2.10). The last recorded fire-scar in the study area (only one recording tree) occurred in 1948 and the frequency of fire events in all samples decreased after ~1850 coinciding with the introduction of the *Bush Fire Act* (1874) and introduction of smallpox to First Nations communities in BC. This inherently led to the reduction in land stewardship and the use of fire by these communities. With this known history of frequent fire prior to the ~1900's, impacts from ranching, fire-suppression, and forest practices post-settlement have led to widespread increases in forest density and encroachment across BC's interior. This increase of trees in the absence of fire has effectively replaced the open understory historically dominated by forbs and grasses, to one filled with ladder fuels and immature, unhealthy trees. The thinning and prescribed fire treatments that were done at the study site aimed to mimic a historical forest-grassland ecotone structure when frequent stand-maintaining fire occurred (K. Iverson, personal communication). Climatic warming and drying (Parisien et al., 2023), including increased and prolonged droughts, coincide with the increase in forest fuels creating environmental conditions susceptible to high intensity wildfire and severe wildfire effects (Ellis et al., 2022).

A majority of the fire events (96%) were identified at the ring boundary and marked as dormant season fires as the fire scar occurred between the latewood and earlywood of the following year (Figure 2.11). It is generally thought that these fires occurred in late summer or fall at the end of the growing season. A potential issue with this assumption is not knowing when the onset of the growing season in the spring of the following year occurs (Kuch et al., in revision; McBride, 1983). In this study, I cored trees in June and not all trees had commenced radial growth at that

time. With the known influence of Indigenous burning in the area, spring fires occurring by human ignition in April and May would also show in the same position at the ring boundary. Lightning fires typically do not occur until peak summer months (July/August) when weather and fuel conditions allow (Government of Canada, 2025), therefore spring fires in this region are often attributed to human ignitions (Copes-Gerbitz et al., 2023; Harvey et al. 2017; Hoffman et al., 2025). The onset of earlywood formation varies annually because the timing of sufficiently warm spring temperatures required for growth differs from year to year. Therefore, the interpretation of seasonality, and even year, may be incorrect at times. However, overall this would likely only cause dated events to be off by a year at times would not impact the general information this study provides on the fire return interval and historical fire regime. Future dendroecological studies involving timing of the growing season (Nehemy et al., 2023) and onset of earlywood/latewood formation (Torbenson et al., 2016) may need to be conducted for individual study sites to accurately assign seasonality to fire scars.

Management considerations

This study clearly points to increased tree growth, increased species richness, and a shift in understory community composition following mechanical thinning and prescribed fire. These findings indicate that in ecosystems with substantial woody encroachment, this method of restoration can successfully reduce fuels, promote resilient understory communities and generate enhanced tree growth that is stable in response to climate variability. Following thinning and prescribed fire treatments, increased health at an ecosystem level was evident and is a primary objective for restoration. By using a more holistic ecosystem approach in this study, I was able to collectively assess multiple ecosystem factors (tree growth, understory dynamics, site history) response to thinning and prescribed fire. In the context of terrestrial ecosystems, disturbances such as fire, insect outbreaks, or windthrow shape forests and the processes that occur within them (Shorohova et al., 2023) and understanding these complex relationships is important to ensure restoration methods are adequately meeting objectives. In this study, looking at only either the understory or overstory independently would not provide a holistic overview of the effect of treatment on the forest-grassland ecotone, particularly in the context of landscape stewardship. Careful planning and management for invasive species will need to be conducted

moving forward; burning or other treatments will need to be completed at the correct time of year to decrease the likelihood of survival for these species. Wildfire risk is also reduced through this treatment, by retaining 92-150 trees per ha or less in IDF zones (historical densities) (Brookes, 2019; Brookes et al., 2021; Greene, 2021) through thinning small diameter trees and leaving the large, fire-resistant trees. This reduces ladder fuels and canopy cover decreasing the likelihood of severe wildfire. In combination with prescribed or cultural fire post-treatment to reduce surface fuels, the risk for crown fire decreases significantly (Preston, 2024). Continual treatments informed by the fire return interval will need to be done to mitigate and maintain the wildfire resilience of this site.

Management activities should prioritize reducing stem densities, increasing heterogeneity at a stand level, and implementing fire on the landscape. Re-integrating Indigenous fire-stewardship (Lake and Christianson, 2019) and/or prescribed fire following thinning can increase understory species richness and control the establishment of woody species to prevent encroachment. Since time immemorial SXFN have inhabited Secwepemcul'ewc and stewarded this land, helping cultivate the fire regimes in this area which inherently structured the plant communities and shaped fire-adapted traits in these ecosystems (Copes-Gerbitz et al., 2022, Dickson-Hoyle et al., 2022, FREC 2021). These practices were disrupted in the mid 1800's through the loss of Indigenous people to disease and colonial practices (Copes Gerbitz et al., 2022; Hoffman et al., 2022). Opportunities for truth and reconciliation in inclusive land management present themselves in providing autonomy back to the Indigenous groups who historically stewarded this land.

At a landscape scale, this treatment has little impact, but by strategically linking and building a network among treatments across the IDF, forest-grassland ecotones can help restore a landscape-level mosaic. Findings from this study align with the Churn Creek Protected Area fire management plan (Blackwell et al., 2001) in using thinning and prescribed fire as a restoration tool in forest-grassland ecotones.

Limitations

Data including species percent cover and tree cores were collected in mid-June. I conducted surveys during a period when the majority of species were expected to be most easily identifiable. Nonetheless, I acknowledge that certain species may have been overlooked,

particularly early season ephemerals that had already senesced or late summer flowering species that had not yet emerged. When analyzing the tree cores, I noted that some trees had started growing for the 2023 year with small amounts of earlywood present, but that some trees had not yet started. This could influence the accurate dating of fire events in the fire scarred samples. Lastly, the climate growth analysis was based on short time series. This can lead to weak and spurious correlations (Speer, 2009). Examining longer time periods would increase the strength of the climate-growth analysis but due to the constraints of pre- and post-treatment intervals, the analysis period was shorter.

CONCLUSION

This study provides evidence that mechanical thinning and prescribed fire can increase species richness, increase tree growth, and re-invigorate an understory plant community from closed, moist forest conditions, to a more open canopy with a shift towards grassland species composition. Species richness increased with treatment, and IDFxm community composition is slowly re-establishing but long-term restoration and monitoring of the site will require periodic burning within the historic fire-return intervals to maintain and continue to improve the forest-grassland understory. Low severity wildfire and prescribed or cultural fire can be used to steward the land and restore fire resilient ecosystems in combination with careful management and planning.

Forest practices and colonial policy have led to denser forests less resilient to fire, particularly in the dry interior forests of BC. The primary disturbance in this ecosystem historically was low and moderate intensity fire (Harvey et al., 2017, Brookes et al., 2021) and contemporary forests have changed due to active wildfire suppression, logging, and cattle grazing. This can lead to positive feedback loops which encourage encroachment and establishment of woody vegetation without a control mechanism. Mature fire-resistant Douglas-fir trees can live for centuries and provide important wildlife habitat and forage to many species, but these stands must be maintained with natural disturbance or involve periodic treatments (prescribed fire or thinning) to prevent establishment of even-aged understory stands. These stands, as seen in our forests today, are susceptible to burning at a higher severity, and in turn, having negative ecological

outcomes. This was evidenced in a 12,040 ha wildfire that occurred in 2021 in the CCPA where much of the forests where Harvey et al. (2017) documented a frequent, low severity historical fire regime, burned at high intensity with severe ecological effects.

BC Parks have management plans (Blackwell et al., 2001; BC Parks, 2000) which are shaped by considering all systems within a park – social, ecological and cultural. When assessing the efficacy of a restoration treatment, it is important to review the findings in the same lens. Further research on the longer-term effects of treatment, and multiple treatments, should be completed to build further understanding of the relationship between the understory, overstory, and fire on the landscape. As ecotones shift in response to human and climate pressures, these transitional ecosystems serve as critical indicators of ecosystem health and targets for restoration in interior BC.

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CHAPTER 3: Drivers and dynamics of Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) encroachment at a forest-grassland ecotone in interior British Columbia, Canada

ABSTRACT

Encroachment in interior British Columbia (BC) is a growing concern for the conservation of grasslands. Grasslands in BC comprise less than one percent of the provincial land base and are targets for restoration to enhance native species, provide habitat, and reduce the risk of severe wildfire effects. Significant portions (>30%) of grasslands in the interior Douglas-fir (IDF) and bunchgrass (BG) biogeoclimatic zones have been affected by encroachment since the 1900s in the Cariboo-Chilcotin region. In the 21st century, effects from fire suppression and colonial forestry policy have led to dense stands of trees at the upper margins of grasslands altering the historical open structure. Historically, wildfire and human-ignited fire by Indigenous Peoples burned frequently in this area at a low-moderate severity every 20-30 years. This maintained an open character forest with mature fire-resistant Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) spaced throughout. While the role of altered disturbance regimes in driving encroachment is well established, important gaps remain in understanding the influence of climate and site-level dynamics. In particular, the spatial patterns of establishment and the climatic conditions associated with recruitment events are not fully described in forest-grassland ecotones. The objectives of this study are to determine (1) whether trees are clustered spatially, by age or size, and (2) if moister, cooler periods are associated with higher seedling establishment.

Results indicate Douglas-fir trees were not clustered spatially but did establish near other similarly aged trees. Trees were also found to not be clustered by size (diameter breast height - DBH), indicating possible competition effects between cohorts. Three pulses of establishment were found for periods in the 1970s and 1990s. These pulses were not associated with any climatic drivers including precipitation, temperature, standardized precipitation evapotranspiration index (SPEI), Pacific Decadal Oscillation (PDO), and El Niño Southern Oscillation (ENSO). This suggests site specific factors have a greater influence on establishment than regional climate variability. Although Douglas-fir growth at this site typically correlates well with cooler and wetter periods, establishment here is likely driven by microclimates, shading, and seed availability.

Key words: encroachment; Douglas-fir; establishment; recruitment; climate; British Columbia; grassland; ecotone

INTRODUCTION

Encroachment, or woody-plant invasion, is the process of trees and shrubs expanding into grasslands or open ecosystems. It is a widespread process that occurs globally across different biomes. For temperate grassland ecosystems, encroachment reduces overall grassland area contributing to changes in carbon storage, altered wildlife habitat, and biodiversity loss (Bai and Cortufo, 2022; Petermann and Buzhydgan, 2021). Considering grassland degradation and loss is occurring at an increasing rate (Bardgett et al., 2021), understanding the mechanisms of woody encroachment into grasslands is crucial to inform conservation and management actions.

In North America, encroachment primarily affects grasslands, savannahs, and forest-grassland ecotones. Historically, these ecosystems were open landscapes with either no trees, or mature trees sparsely located within a predominantly grassland ecosystem. Prior to European settlement, encroachment was limited by key processes including fire, native ungulate grazing, drought, and competition from other vegetation (Fuhlendorf et al., 2008; Van Auken, 2009). Encroachment has increased concomitantly with European colonization and settlement due to increased grazing pressure and policy regulating burning practices (Reid, 2008). Since the early 1900s, increased tree and shrub establishment has not only impacted grasslands but has led to dense understory and subcanopy tree layers in historically low-density savannahs and forest-grassland ecotones (Hessburg et al., 2005; Van Auken, 2009).

Ecological disturbances are an important process to maintain grasslands and open forests. In dry interior forest regions of the Inland Northwest, insect outbreaks, windthrow, blowdown, and wildfire are the primary disturbances shaping ecosystem dynamics. Together, these processes maintain a heterogeneous landscape mosaic by creating forests of varying age and structure, cycle nutrients, and generate canopy openings that facilitate ecosystem health and the establishment of understory vegetation (Geils et al., 1995). In forest-grassland ecotones, wildfire was a dominant disturbance for centuries (Harvey et al., 2017; Iverson et al., 2004). Wildfire limits tree regeneration when burned at a frequent interval to maintain open grasslands and understories (Hessburg et al., 2005). Research documenting historical fire regimes in dry forests of interior British Columbia (BC) indicate a mixed severity wildfire regime dominated by high frequency, low severity wildfires punctuated by infrequent moderate to high severity wildfires (Brookes et al., 2021; Copes-Gerbitz et al., 2022; Harvey et al., 2017; Heyerdahl et al., 2012).

Through dendrochronological records of fire scars, these studies indicate wildfire burned approximately every 20-35 years in the interior (Brookes et al., 2021; Harvey et al., 2017). The spatial and temporal variation in fire frequency and severity formed a heterogenous landscape, often dominated by open canopy stands with mature, fire-resistant trees and diverse, herbaceous understories (Peterson and Reich, 2007). But since the late 1800s colonial fire exclusionary policy has drastically reduced fire on the landscape. The *Bush Fire Act* (1874) prohibited Indigenous cultural burning, and the *Forest Act* (1912) started organized fire suppression across the province. The exclusion of Indigenous fire stewardship and effective and widespread fire suppression tactics has led to changes in structure and function of the forest-grassland ecotones of interior BC. Indigenous People used fire widely in land stewardship practices in North America, but these practices were disrupted by the arrival of European settlers and colonial practices (Christianson et al., 2022; Hankins et al., 2025; Hessburg et al., 2021). Frequent understory fires once controlled the establishment of saplings, which helped maintain the open character of these forest-grassland ecotones. The introduction of domestic cattle and livestock increased grazing pressure, reducing fine fuel loads that historically carried low-severity grassland fires (Hessburg et al., 2005). Present day forest-grassland ecotones are often densely infilled with immature trees with stunted growth competing for light and water (Cole and Newton, 1986; Lochhead and Comeau, 2012).

Light availability, moisture conditions, and interactions with neighboring vegetation strongly influence the establishment and early growth of interior Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) in dry forest ecosystems. Growth of Douglas-fir is positively correlated with spring and summer precipitation, particularly in moisture-limited environments (Brubaker, 1980), and early seedling survival is largely dependent on adequate soil moisture (Foster et al., 2020). A further mechanism at play regarding establishment is recruitment and germination of seeds. Cone crops generally occur at irregular intervals but a medium to heavy crop can be expected approximately every seven to ten years in a quarter of Douglas-fir trees in a stand (Isaac, 1943; Hermann and Lavender, 1990). The periodicity of cone production often coincides with anomalously wet climate years providing an ideal seedbed for germination conditions (Vacchiano et al., 2021). Atmospheric conditions further constrain establishment, as increased vapour pressure deficit (VPD) elevates transpiration demand and, in water-limited environments, substantially increases the risk of seedling mortality (Davis et al., 2019). Temperature extremes also affect survival;

when soil temperatures exceed 60°C for as little as one minute, approximately 50% seedling mortality is possible (Rank et al., 2022). Additionally, seedlings in the Cariboo region of BC are also vulnerable to mortality from late spring frost events (Day, 1996). Long grass and organic litter layers may enhance seedling survival by providing shaded, moisture-retaining microsites during early development (Gray, 1995); however, as seedlings mature and transition toward greater light requirements, successful growth depends on eventual emergence above surrounding vegetation (Hermann and Lavender, 1990). These facilitative effects are not limited to herbaceous vegetation. Neighbouring mature trees can help foster survival rates among seedlings during drought periods with water and nutrient transfer through ectomycorrhizal networks (Bingham and Simard, 2012). Climate interacts with local stand conditions which influence site level growth and mortality. These spatial relationships between surrounding trees add to the complicated dynamics of forests. Stand dynamics raise an important question of whether Douglas-fir growth and expansion outwards from the forest edge are driven primarily by site-specific controls (e.g., microsite moisture, shading) or broader regional drivers such as climate and disturbance regimes.

Tree growth and establishment are inherently tied to climate, particularly through interactions with temperature and moisture availability (Case and Peterson, 2005). With climate change projected to increase global land temperatures by a minimum of 1.5^{o} C and higher in the Arctic (IPCC, 2021), climate change driven ecological effects from increased VPD can include reduced soil moisture, and further drought stress (Allen et al., 2010; Hanson and Weltzin, 2000). Edge-range environments such as forest-grassland ecotones are more susceptible to changes in climate as they often experience more extremes (Mast et al. 1998; Rehm et al., 2015; Will et al., 2013). Douglas-fir is expected to contract at its lower elevation limit due to climate change and unfavourable conditions including prolonged drought and increased temperature (Davis et al. 2019, Will et al., 2013). Contrarily, expansion of the lower range of this species is currently being observed in the Cariboo region of BC and resulting in native grasslands being lost to encroachment (Bai et al., 2005; Reimer, 2023). Interannual variability in precipitation and temperature can create favourable windows for recruitment, possibly aligning with high cone crop years, where large numbers of seeds can germinate and persist into maturity (Foster et al., 2020). These episodic recruitment patterns support disturbance dynamics of Douglas-fir encroachment into grasslands.

The Pacific Decadal Oscillation (PDO) and El Niño Southern Oscillation (ENSO) are two broad scale patterns of climate variability based on sea-surface temperatures and can influence moisture availability within BC (Lo et al., 2017). Positive and negative phases of PDO change about every 30 years (Biondi et al., 2001), while ENSO is characterized by flips between positive and negative phases every 2-10 years (Philander, 1983; Yu and Zwiers, 2007). The effects on regional climate by these two patterns of variability is not consistent across BC, with the greatest impacts in coastal areas and in the southeast of BC (Meyn et al., 2010). These broad scale patterns of climate variability can influence tree growth and establishment (Lo et al., 2017; Watson and Luckman, 2002).

The goal of this study is to assess whether climate variability and biotic facilitation influence conifer encroachment into grasslands. Specific research objectives are (1) to assess whether trees are clustered spatially, by age or size, and (2) if moister, cooler periods are associated with higher seedling establishment. I hypothesize (1) that trees are clustered to favour biotic facilitation (Mast and Veblen, 1999; Rice et al., 2012), and (2) in the absence of fire over the past 100 years, episodic seedling establishment over the past 100 years will be associated with multiple years of persistent moisture (Vacchiano et al., 2021). Episodic seedling establishment can be related to interannual climatic variability leading to even aged cohorts of trees (Bai et al., 2004). Determining the drivers of tree encroachment into grasslands and understanding any associated spatial patterns will assist management agencies with understanding ecological dynamics in these sensitive ecosystems and in designing appropriate treatments to promote resilient ecosystems.

METHODS

Study area

The study area is in the Churn Creek Protected Area (CCPA) in interior BC (Figure 3.1). The CCPA is BC's largest protected area of grasslands at over 36,747 ha. Established in 1995, it is a critical habitat for many species including California bighorn sheep (*Ovis canadensis),* mule deer *(Odocoileus hemionus),* and the threatened Lewis' woodpecker (*Melanerpes lewis).* Prior to becoming a protected area, the region experienced extensive unregulated ranching following early European settlement (Reid, 2008; Thomas, 1976). The Empire Valley Ranch is an active

ranch (~500 head of cattle) included in the protected area and works in collaboration with BC Parks for conservation objectives (BC Parks, 2000; Reid, 2008). Other land uses in the CCPA include fishing, hunting, and other recreational activities such as horseback riding and hiking (BC Parks, 2000). The study area lies within the traditional territory of the Stswecem'c Xget'tem First Nation (SXFN), one of 17 nations that make up the larger Secwepemc Nation. Stswecem'c Xget'tem First Nation and surrounding Indigenous communities still use this area to gather traditional and medicinal plants, and hold ceremonies (FREC, 2021).

The study area is located at 1040m above sea level (asl) in a forest-grassland ecotone in the Interior Douglas-Fir very-dry mild (IDFxm) biogeoclimatic zone (Figure 3.1). Forest-grassland ecotones in this area occupy elevations between 850-1100m asl where understory plant species include blue bunch wheatgrass (*Pseudorogeneria spicata*), pinegrass (*Calamagrostis rubescens*), snowberry (*Symphocarpus* spp.), juniper (*Juniperus* spp.), and showy aster (*Eurybia conspicua*) (Steen and Coupé, 1997). Primary tree species include interior Douglas-fir, trembling aspen (*Populus tremuloides*), and spruce (*Picea engelmannii* x glauca). This area lies within the Fraser Basin and Chilcotin Plateau ecoregions (BC Parks, 2000) and is influenced climatically by the Coast Mountain ranges to the west. The rain shadow effect reduces precipitation and shapes the dry climate for this area. Average summer temperature is 16.2 ^{o} C in July and -6.5^{o} C in winter in January for the closest weather station and average annual precipitation of 326.1mm (Clinton, BC; Government of Canada 30-year climate normal 1991-2020).

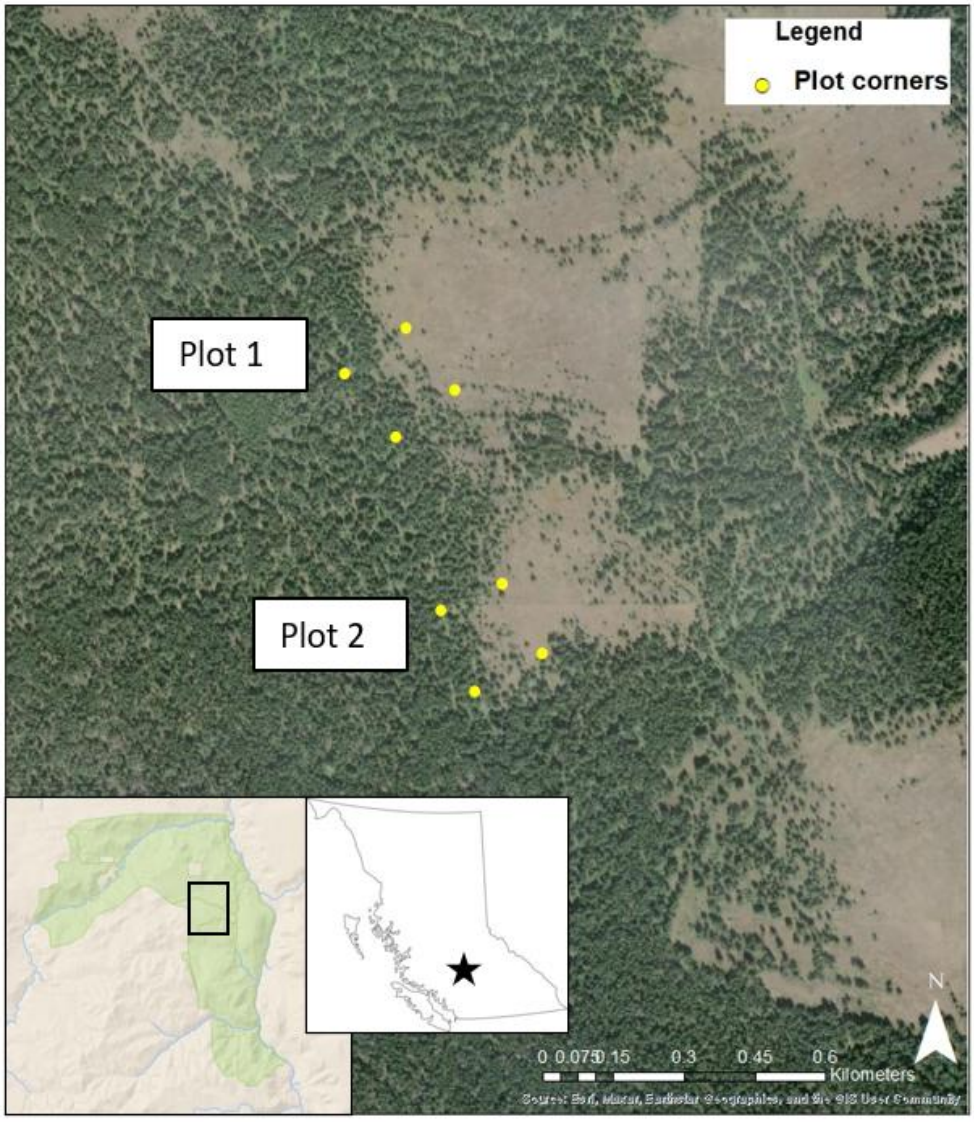

Figure 3.1. The study site in British Columbia (BC) in the Churn Creek Protected Area (CCPA). The inset maps show the location of the study site in the CCPA, and the location in BC. Corners of the two 1-hectare plots are marked in yellow.

In the interior of BC, the IDF biogeoclimatic zone has the largest area impacted by encroachment in the Cariboo-Chilcotin region (Reimer, 2023). Documentation through historical aerial imagery (Bai et al., 2004; Figure 3.2) and mapping efforts (Reimer, 2023) indicate that >30% of the upper grasslands of the Bunchgrass zone (BG) and IDF zone have documented woody plant

encroachment since the 1900s (Bai et al., 2005; Cariboo-Chilcotin Grasslands Strategy, 2001; Reimer, 2023).

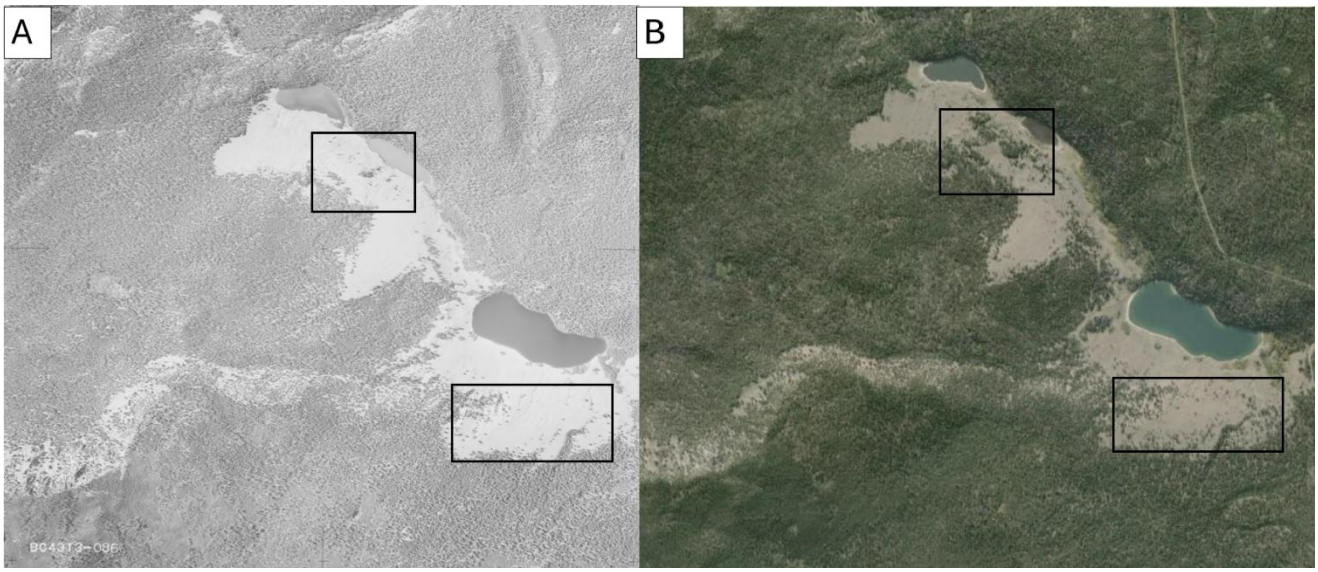

Figure 3.2. (A) Historical aerial image from 1965 of an area within the northern region of the Churn Creek Protected Area. (B) Satellite imagery from *iMap BC* taken after 2022 of the same area. Encroachment and forest infilling is evident in the grassland areas (black boxes).

Field methods

The location of the two 1 ha plots were selected to include a gradient from grassland to forest (ecotone). The southeast corner point of each plot was randomly selected in the field. Both plots are on a northeast aspect approximately 240m apart at 1040m asl (Figure 3.1). Site selection was limited due to accessibility, degradation, or treatment of sites by BC Parks to reduce encroachment. The two plots selected had no current or past encroachment mitigation work completed, but prior logging from the first half of the 20th century was observed. Plot 2 was also recently grazed with cattle noted near the study site (Figure 3.3).

Field data was collected in September 2023. Each 1 ha plot was divided into 100 10m x 10m subplots. To determine tree age, all trees with >10cm DBH (diameter at breast height) were cored using an increment borer 30cm above ground (two samples per tree taken at least 90˚ from each other), and all trees <10cm DBH but >1.3m tall were cut down and a basal disk was collected as

close to the root collar as possible. Trees <1.3m but > 30cm in height were considered saplings and counts were tallied by subplot. Trees <30cm height were considered seedlings and tallied by subplot. All trees >1.3m in height had the in-plot cartesian coordinates and DBH measured.

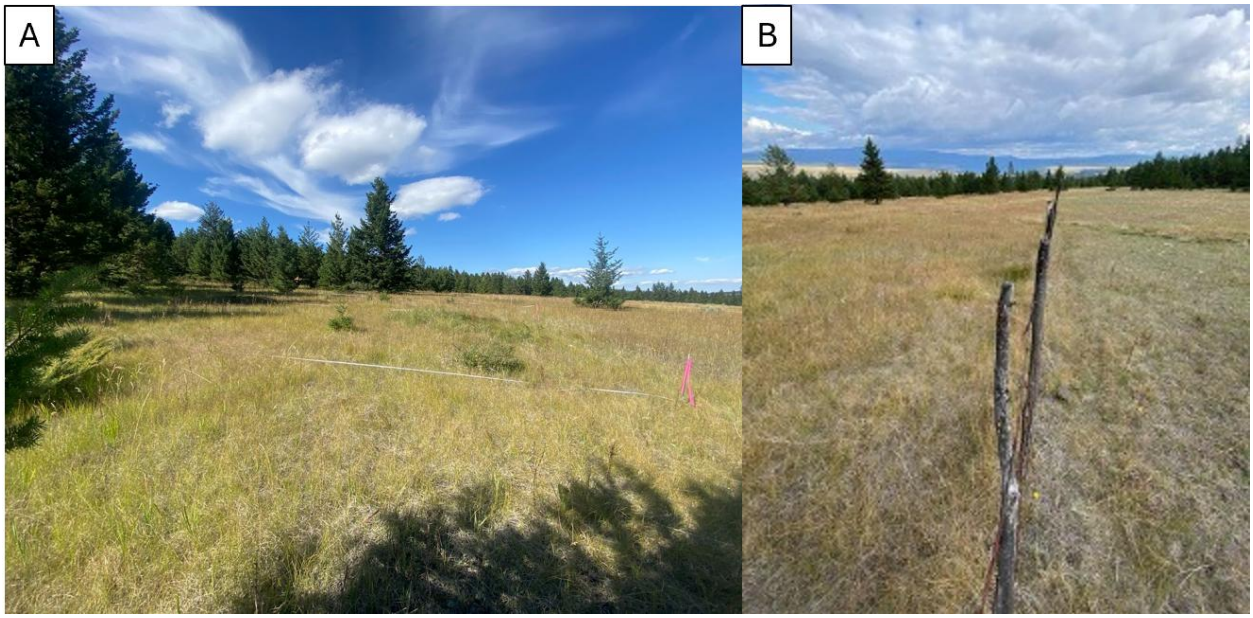

Figure 3.3. (A) Forest-grassland ecotone within the Churn Creek Protected Area. (B) Evidence of recent grazing on the right side of the fence line where Plot 2 was established, and Plot 1 was established on the left side.

Processing methods

Tree core samples were air dried and mounted to slotted boards, and basal disks were air dried prior to sanding. Samples were progressively sanded to a fine polish from 80 to 400 grit. Samples were scanned using a high-resolution scanner and uploaded into WinDendro ring measurement software (Regent Instruments Incorporated, 2001) for ring counts to estimate age. A stereoscope was used when ring boundaries were difficult to distinguish on scanned images. Where possible, visual crossdating of marker years was used to assist age estimations. If the two core samples for each tree had different ring counts, the sample with the higher number of rings and/or pith present was used. Crossdating young trees (<60 years) is often challenging because site-specific factors such as microclimate and light availability can exert a stronger influence on

juvenile growth than regional climate signals (Chhin and Wang, 2007). The increased competition among the high density of saplings can also cause suppression of growth resulting in ring widths in these trees to be indeterminate (Appendix Figure B2) (Tonelli et al., 2020). For samples where no pith was captured (n = 198) a pith correction factor was applied using the Duncan method (1989). A coring height correction factor (2.36 years) was used to account for the coring height (30cm) above the root collar (Copes-Gerbitz et al., 2022). This correction factor uses a Douglas-fir age-on-height linear regression model to approximate age to growing height:

(correction = 0.54 [coring height in centimetres]/6.87)

(0.54 × 30) / 6.87 = **2.36 years**

Analytical methods

Plot 1 and 2 were analyzed separately for stand structure to determine if there were differences in general stand characteristics. Age class and DBH of trees were delineated into 10 year and 10 cm bins for this analysis. To compare the influence of grazing, plots were also analyzed separately for seedling and sapling analyses.

To assess whether trees were clustered spatially, by age, or by diameter breast height (DBH), R software (version 4.4.3) was used. The plot data including stem locations was uploaded and converted to the BC Albers coordinate system. Stem maps were created by converting the plot coordinates into spatial objects and then uploaded to QGIS (QGIS Development Team, 2009) on a 100 x 100m grid. Trees were binned by age (10-year age groups), however trees greater than 100 years of age were placed in a single bin. Tree DBH was also binned into 10cm diameter groups.

Ripley's K (*Kr*) was used to detect spatial patterns of trees in each plot following the methods of Rice et al. (2012) and Halpern et al. (2010). This analysis can quantify whether trees are clustered, random, or regularly spaced through a point pattern analysis. Packages *spatstat* (Baddeley et al., 2005) and *sf* (Pebesma and Bivand, 2023) were used. Because the plots were located within a grassland-forest ecotone, an inherent gradient, or clustering of trees already exists. To reduce bias, an inhomogeneity test must be conducted to determine which method of Ripley's K will be used. To confirm inhomogeneity, a chi squared test to assess complete spatial

randomness (CSR) was conducted, if $p<0.05$ then data is inhomogeneous. Then, the K_{inhom} (*Ki*) function was used to correct for inhomogeneity as it corrects for variance in point density (Rice et al., 2012; Baddeley, 2000). Again, I defined a plot window of 100 x 100m and created a planar point pattern object using the cartesian coordinates of each tree. Intensity was estimated using a kernel smoothing density function; this was used to correct for non-uniform point density across the plot in the *Ki* function. *Ki* was computed with an isotropic edge correction (this compensates for neighbours that exist outside the plot boundary) and transformed into *Li (Ki - r)* for ease of interpretation (Acquah et al., 2023). *Li* is a variance-stabilized square root transformation of *Ki* where if observed values are above 0 it indicates clustering, and below 0 indicates dispersal. Simulation envelopes were used to test for statistical significance using 500 Monte Carlo tests under the null hypothesis of CSR. If observed results deviate outside of the envelope, then there is significant clustering (positive) or dispersal (negative). Next, *PCFi* (pair correlation function) with simulation envelopes was computed. Since *Ki* is cumulative, *PCFi* can determine at which exact distances clustering is occurring. Lastly, the marked correlation function was conducted for tree DBH and age for each plot (Moradi and Eckhardt, 2025). This approach permitted the investigation of whether tree clustering is based on tree age or size. This function is cumulative, and patterns at these smaller scales can influence large scale outputs (>10m). Outputs were restricted to smaller spatial scales (10m) to assess for biotic interactions where fine scale variability generally occurs (Rice et al., 2012).

Trees were binned into five-year age groupings for establishment cohorts due to potential error with ring counts/tree ages. Because the tree core and basal disk samples were not crossdated and had correction factors applied, there is potential for the age estimations to be off. If greater than 10% of trees established in a five-year bin it was considered a regeneration pulse (Hankin et al 2019). To assess if moister, cooler periods are associated with higher seedling establishment climate data was compared to establishment pulses. The standardized precipitation evapotranspiration index (SPEI) was calculated for monthly and annual values from 1900-2023 following the methods of Vicente-Serrano et al. (2010), Z-scores (number of standard deviations from the mean) were calculated for precipitation (annual), and growing season mean temperature (March-September) to assess for anomalous conditions in relation to establishment years since 1900-2023. Total monthly precipitation for peak growing season months (May, June, July) is often positively correlated with higher growth rates in Douglas-fir (Watson and Luckman, 2002)

and is known to facilitate increased establishment of Douglas fir (Case and Peterson, 2005; Rother et al., 2017) and was compared in relation to establishment years. Precipitation values for the five years prior to and post episodic establishment years were visualized compared to the mean monthly precipitation from 1900-2023. A Wilcoxon rank-sum test was conducted to look for statistical significance between years with high growing season precipitation, mean temperature, SPEI, and high establishment years.

To assess relationships between pulses of tree establishment and climate variability, intervention analysis was conducted for PDO data to define persistent cool and warm phases. Methods followed Trouet and Taylor (2010) using the NOAA PDO index (2025). A 30-year moving window was used with a two-sample t-test to obtain shift years. Nino3.4 (NOAA, 2025) anomalies was used for ENSO data. These were compiled into a barcode plot for dominant warm or cool phases of each climate variable. Douglas-fir establishment years were then visually compared with the barcode plots to explore whether high regeneration pulses are associated with warm or cool phases.

RESULTS

Stand structure

Overall stand structure characteristics between Plot 1 and Plot 2 were similar, but Plot 1 overall had more trees and a higher mean age and DBH (Table 3.1, Figure 3.4). Trees in Plot 1 had a mean age of 54 years (8-139), and trees in Plot 2 had a mean age of 40 years (6-115). The trees in Plot 1 had a mean DBH of 16.8cm (0.4-48.6cm), and Plot 2 trees had a mean DBH of 13cm (0.16-61.5cm). Basal area was 9.85m^2 for Plot 1 and 6.51m^2 for Plot 2. The total stems per plot was 334 (stems/ha) for Plot 1, and 298 for Plot 2 (Table 3.1). There were approximately the same number of saplings in both plots (Plot 1: $n = 51$, Plot 2: $n = 50$), but seedlings for Plot 1 ($n = 5308$) are double that of Plot 2 ($n = 2384$).

Table 3.1. Stand structure characteristics of live Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) trees >1.3m height for Plot 1 and Plot 2 in the Churn Creek Protected Area. Each plot is 1 ha in size on a northeast facing slope at 1040m asl elevation.

	Plot 1	Plot 2
Tree Age		
Mean	54	40
Min	8	6
Max	139	115
Tree DBH (cm)		
Mean	16.8	13
Min	0.4	0.16
Max	48.6	61.5
Trees/ha	334	298
Basal area (m^2)	9.85	6.51
Seedlings	5308	2384
Saplings	51	50

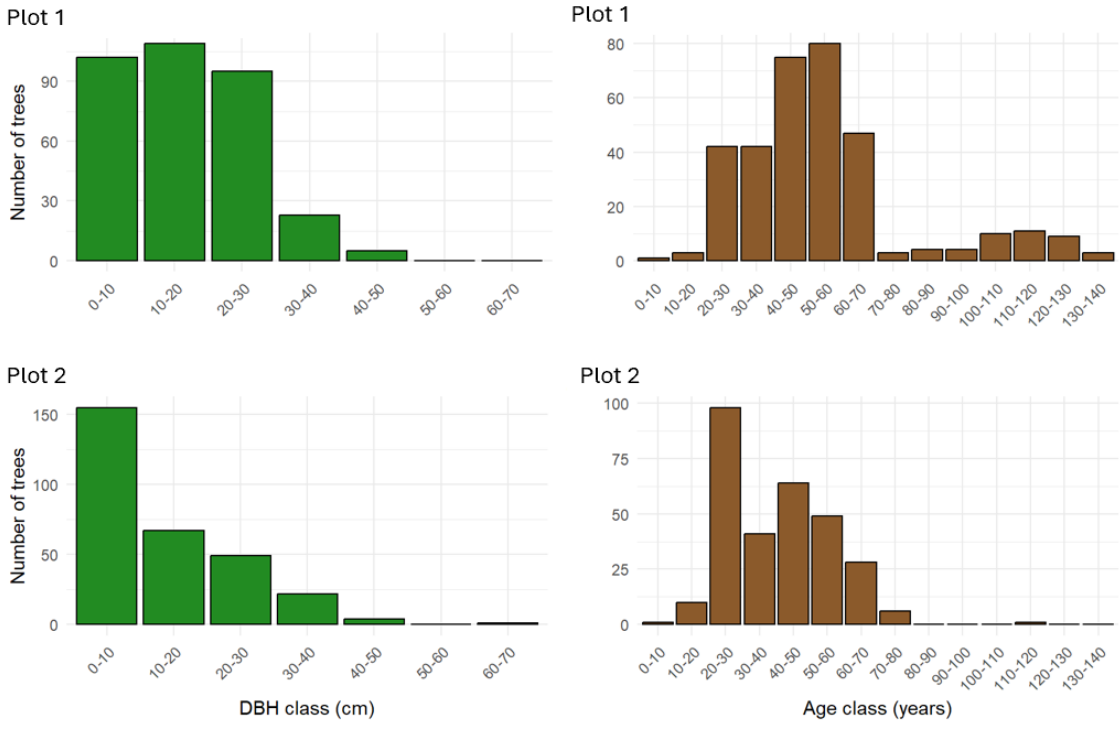

Figure 3.4. Stand demography for Plot 1 and Plot 2 including tree diameter breast height (DBH) and age. Tree DBH was delineated by 10cm increments and grouped together; tree age was grouped into 10-year age bins.

Sapling and seedlings

Higher concentrations of seedlings were found in the more open, grassland side in Plot 1 (Appendix Figure B1). Total number of seedlings in plot 1 (n = 5308) doubled that of Plot 2 (n = 2384). There was active cattle grazing in Plot 2 during data collection. The subplot with the highest number of seedlings was in Plot 1 (n = 516) in subplot 3C with the centre of the plot at 25m x 25m (Appendix Table B1). The mean number of seedlings in Plot 1 was n = 56 (+/- 99), and in Plot 2 was n = 24 (+/- 52). The mean number of saplings in Plot 1 was n = 0.5 (+/- 1), and in Plot 2 was n = .5 (+/- 1).

Spatial analysis

Stem maps (Figure 3.5) demonstrate the temporal invasion of Douglas-fir from the back of the plot (forested) to the front (grassland) from 1891 until 2023. Using the 2023 tree location information, a significant pattern of spatial clustering was not identified in Plots 1 or 2 (Figure 3.6). There is evidence of small amounts of clustering as the observed values are above zero and reach the envelope boundary, particularly at ~2m in Plot 1. But since observed results were within the simulation band, this indicated no significance based on the Monte Carlo tests suggesting the pattern was consistent with the null hypothesis. This approach focused on purely the spatial arrangement of trees within the plots accounting for the gradient within the forest-grassland ecotone. The pair correlation function (PCFi) results indicated weak small-scale clustering at distances <2m in both plots, although not significant (Figure 3.7).

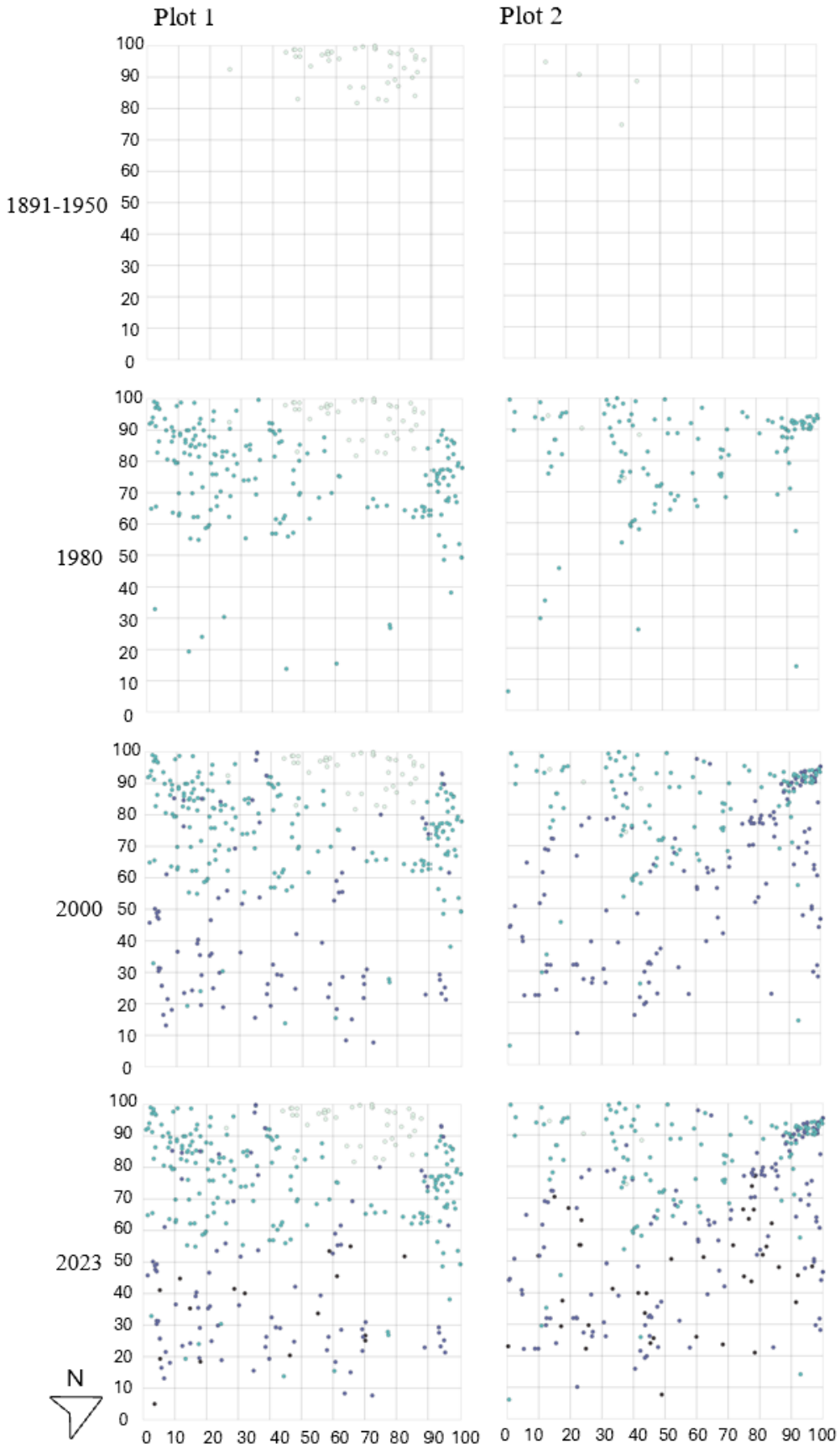
Plot 1
Plot 2
1891-1950
1980
2000
2023
N
0 10 20 30 40 50 60 70 80 90 100

Figure 3.5. Stem maps showing the spatial distribution of Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) trees in Plot 1 (left) and Plot 2 (right) for all trees >1.3m height. Initial establishment (1891-1950) captures all sample trees that established prior to 1950. The years 1980 and 2000 were identified to show encroachment associated with the pulses of establishment (~1970-1980) and (~1995-2000) identified. In the 2023 stem map, the location of all trees in each plot at the time of sampling are displayed. All distances are measured in metres (x, y).

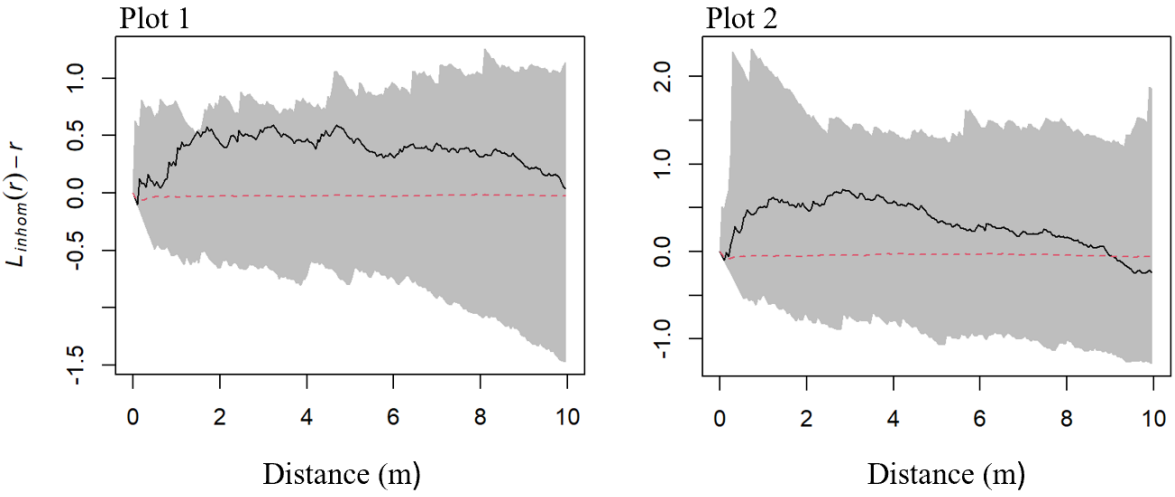

Figure 3.6. Strength of clustering, L_{inhom}, a transformed function of inhomogeneous Ripley's K (K_{inhom}) for all Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) trees >1.3m height in two 1 ha plots within a forest-grassland ecotone of the Churn Creek Protected Area. When L_{inhom} is equal to zero there is complete spatial randomness (CSR), greater than zero there is clustering, and less than zero there is dispersal of trees. This is based on a cumulative function over distance. The grey band represents 500 Monte Carlo simulation envelopes under the null hypothesis of CSR (red dashed line) and results are significant if the observed data (Plot 1 n = 334, Plot 2 n = 298) (black line) exist outside of the envelope.

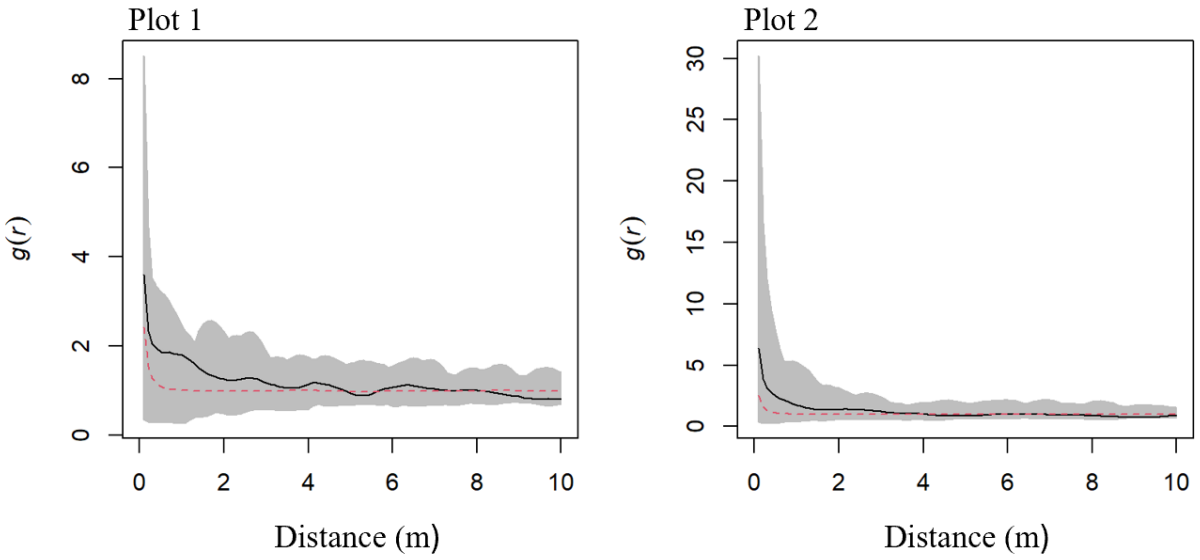

Figure 3.7. Strength of clustering at specific distances using PCFi (pair correlation function) of Ripley's K corrected for inhomogeneity for all Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) trees >1.3m height in two 1 ha plots within a forest-grassland ecotone of the Churn Creek Protected Area (CCPA). The grey band represents 500 Monte Carlo simulation envelopes under the null hypothesis of complete spatial randomness (CSR; red dashed line) and results are significant if the observed data (Plot 1 n=334, Plot 2 n=298) (black line) exist outside of the envelope.

When the dataset was separated into tree age and DBH (as an indicator of tree size), the mark correlation function results indicate clustering by age, but not by DBH (Figure 3.8). Significant clustering (K >1.4) at distances <2m is detected, but then decreases at 3m before rising above the significance envelope again for age groupings in Plot 1 (Figure 3.8). Tree age in Plot 2 exhibits significant clustering at distances >2m. Clustering relationships of similarly aged trees exist in both plots but at different scales. When looking at clustering by tree DBH, Plot 1 does not show any significant relationship. If any, there is weak dispersal at small distances. However, tree DBH in Plot 2 elicits a significant response to dispersal at distances <5m indicating larger trees are near smaller trees. This relationship can also be observed visually in the stem maps (Appendix Figure B1).

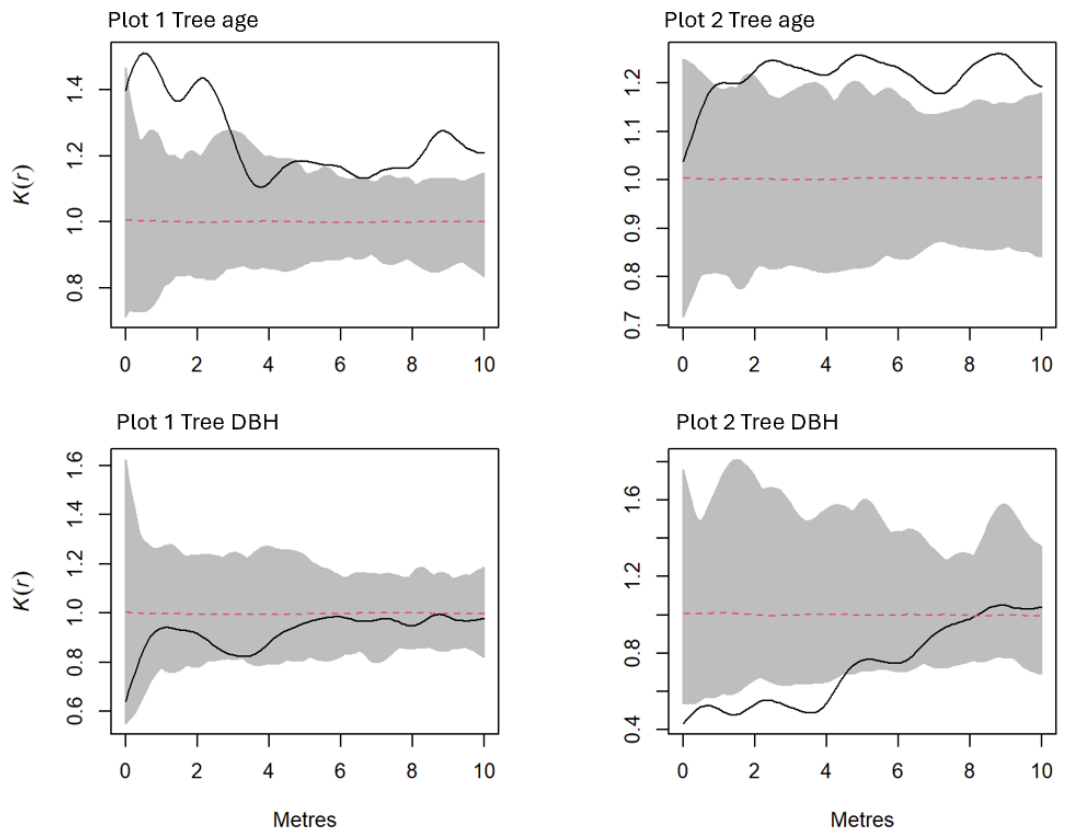

Figure 3.8. Spatial correlation of quantitative attributes age and DBH (diameter at breast height) for all Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) >1.3m height using the inhomogeneous Ripley's K mark correlation function. The grey band represents 500 Monte Carlo simulation envelopes under the null hypothesis of CSR (red dashed line) and results are significant if the observed data (Plot 1: $n = 334$, Plot 2: $n = 298$) (black line) exist outside of the envelope.

Pulses of establishment

Three five-year bins met the 10% threshold and were considered as "pulses": 1970-1975, 1975-1980, 1995-2000. Initial establishment began in the early 20th century by the larger and older trees the western sides of both plots; establishment increased greatly beginning in the 1950s. The pulses began in approximately the early 1970s and mid 1990s, persisting for a few years. After the pulses, few trees established in either plot (Figure 3.9F). The low establishment after 2000 could be in part due to immaturity of the stand as ~50 saplings were present in each plot (total $n = 101$) but were not yet considered trees and therefore excluded from the pulse analysis. Once these saplings mature, they would likely be considered another pulse.

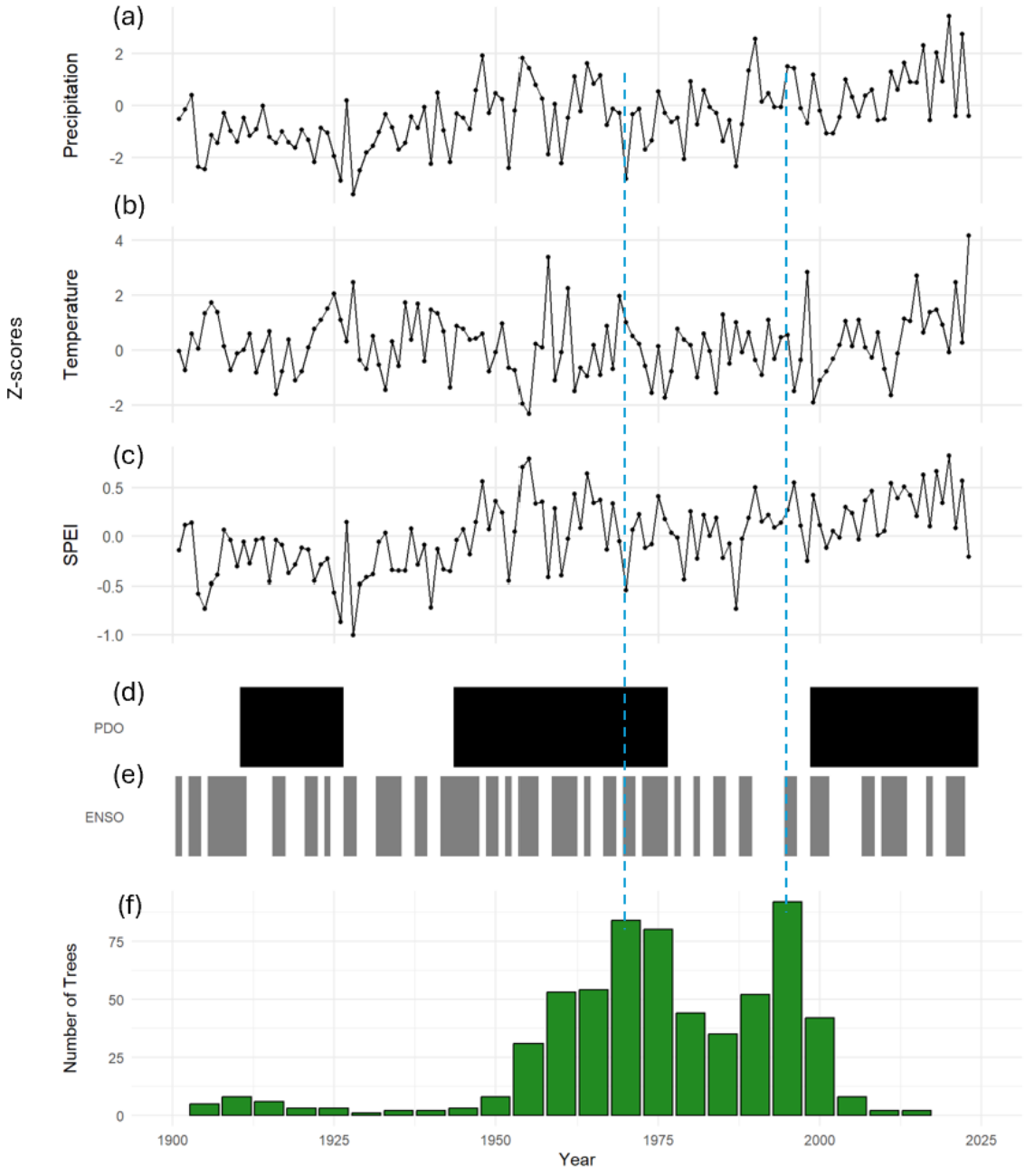

Figure 3.9. Variation in climate from 1901 to 2023 for (A) annual total precipitation, (B) annual mean temperature, (C) and annual standardized precipitation evapotranspiration index (SPEI) expressed as Z-scores (standard deviations from the mean). (D) Persistent phases of the Pacific Decadal Oscillation (PDO) and (E) El Niño Southern Oscillation (ENSO) identified through

intervention analysis, are defined as warm (black) and cool (white) periods. (F) Establishment bins (5-year periods) for Douglas-fir (*Pseudotsuga mensiezii* var. *glauca*) are represented (green) for 1ha plots combined (n = 632). Blue dashed lines highlight peak establishment intervals.

Climate analysis

PDO had no discernable effect on the pulses of establishment at the study site. The pulses (>10% of trees) were visually identified in periods of both warm and cool phases of PDO (Figure 3.9). ENSO also had no distinguishable pattern on the timing of establishment as it is variable during and before the periods of high establishment peaks (1970, 1975, 1995 bins) (Figure 3.9). Figure 3.9 allowed for visual interpretation of the relationships between PDO, ENSO, and establishment pulses and were not verified statistically. Due to the relative short time span in relation to broad scale climate and only three pulses occurring, statistical analysis would not be suitable. Results of the Wilcoxon-rank sum test indicated no significant relationships exists between high establishment years and years with high precipitation during peak growing season months May, June, and July ($p = 0.2254$) (Figure 3.10). The Wilcoxon-rank sum test was also conducted for mean temperature and SPEI; there was no significant relationship determined between May, June, and July mean temperature ($p = 0.1329$), drought indices (SPEI) ($p = 0.22$), and associated high establishment years. Overall, there are no discernible relationships between regional and broad scale climate drivers and bins or years of high establishment.

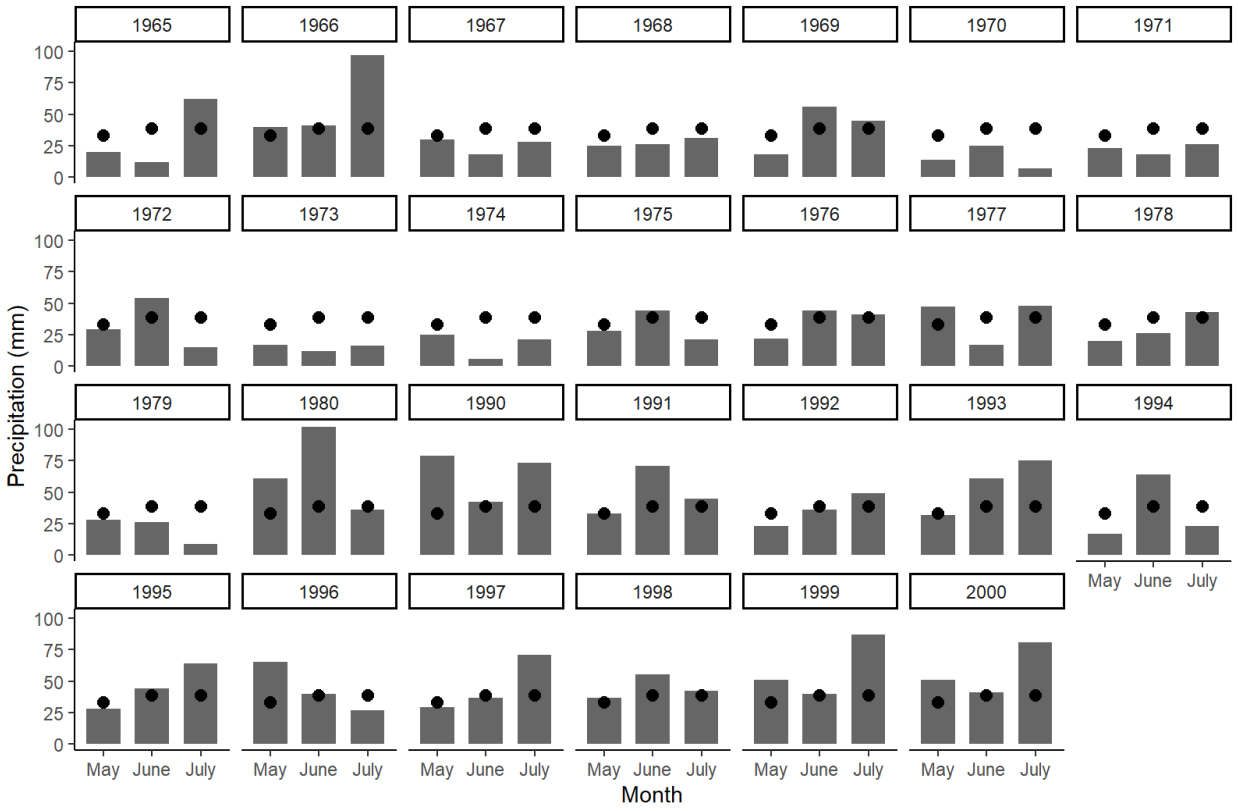

Figure 3.10. Total precipitation (mm) for May, June, and July of the five years preceding and following establishment bins. Black dots indicate mean precipitation for each month from 1900-2023. A Wilcoxon-rank sum test did not find any significant association between years with increased precipitation and high establishment rates.

DISCUSSION

Age-based clustering of trees evident in a forest-grassland ecotone

When tree position was evaluated independently no particular patterning or clustering was detected (Figures 3.6 and 3.7). However, when trees were analyzed by attributes such as age and DBH, patterns were detected (Figure 3.8). This analysis focused on small scales (<10m) at which biotic interactions are more likely to occur. In Plot 1, trees were significantly clustered by age at <2m and >4m, indicating a pattern of age-based clustering. Similarly, in Plot 2, trees were found to be clustered by age but at scales >2m. Clustering of trees by age is often indicative of disturbance-driven recruitment (Duncan and Stewart, 1991). However, given the land-use history and prolonged absence of fire at this site, the observed age clustering is more likely the result of localized recruitment. Small-scale clustering of similarly aged trees suggests establishment under

favourable microsite conditions rather than a single disturbance event. This is consistent with the episodic pulses of Douglas-fir trees establishing within the plots. Stem maps (Appendix Figure B1) provide insight into the age based and size-based patterning found with Ripley's K results and into the extent of former grassland areas prior to encroachment (Figure 3.5).

In contrast, trees were significantly dispersed according to size (DBH) at <4m scale in Plot 2, but tree size was random, or very weakly dispersed in Plot 1. For Plot 2, this suggests that larger diameter trees are often located near smaller trees, which could be indicative of facilitative interactions such as shading and seed rain (Navarro-Cano et al., 2021). Alternatively, patches of similarly aged trees with different size DBH could indicate competition dynamics, where a subset of individual trees were able to dominate and outcompete surrounding trees for resources (Mast and Veblen, 1999). During the process of counting rings to age trees, it was often noted that trees less than 50 years old exhibited suppressed growth in the last two decades (Appendix Figure B2). The high density of trees in the understory at small spatial scales contributes to the reduced growth as light availability, moisture levels, and nutrients are less abundant (Day,1996; Oboite et al., 2025), and likely leads to supressed radial growth. With a few select trees outcompeting their neighbours, we see the dynamics between age relationships where mature seed trees, and even-aged offspring are present in the same locale, but observe the dispersal, or difference in DBH. There is often an inverse relationship between tree size and stand density (Oboite et al., 2025) and this was evident at the study site. Trees in the grassland (recent encroachment) often had a larger DBH than trees infilling existing areas (Appendix Figure B1). This is most likely due to higher light availability and reduced competition further reinforcing this concept. As discussed by Halpern et al. (2010) care must be taken when interpreting biotic interactions based on spatial analysis and factors other than location alone including life history, environment, and timing of establishment are important context.

The initial trees that establish in the open grasslands act as recruiters for further establishment. As the trees mature their cone crops continue to increase in quantity until maturation (Hermann and Lavender, 1990). These seed trees, those bearing viable cones, can disseminate seeds upwards of 100m (Hermann and Lavender, 1990). The initial wave prior to 1980, had the farthest-reaching recruiters (~70m away from seed source, Figure 3.5). Further cohorts likely spread out and infilled meadow gaps. Douglas-fir trees 10 to 15 years old can begin producing

cones when grown in open conditions perpetuating the spread into grasslands at an accelerated rate (Isaac, 1943).

Site specific controls and biotic interactions exert larger pressure than broad-scale climate patterns on establishment of Douglas-fir over a 120-year period

The temporal pattern of tree encroachment at this site indicates three distinct pulses of establishment by Douglas-fir trees, occurring between 1970-1975, 1975-1980 and 1995-2000 during which greater than ten percent of overall trees established in each pulse between 1891 and 2023 (Figure 3.9F). Episodic establishment of Douglas-fir commonly correlates with regional climate signals including cooler temperatures and increased precipitation (Watson and Luckman, 2002). However, other studies suggest local site-specific conditions including biotic facilitation have been found to be more important than climate in driving establishment (Haugo et al., 2013; Karst, 2011; Tingstad et al., 2015). At this site, a complex history of land use and site-specific factors are the primary drivers of establishment.

Prior to 1950 recruitment was generally low at this site (Figure 3.9F). This coincides with records of logging and possible burning by local ranchers occurring until ~1950 (BC Parks, 2000), although exact locations are not known. Between 1891 and 1950 few trees are present in the plots (Figure 3.5), but by 1950 some trees would be mature enough to produce cones. Because there are low numbers of trees establishing during this period (Figure 3.9F), it is thought some form of disturbance occurred such as livestock grazing, forest harvesting, or burning. Cut tree stumps are present directly west of Plot 1. The removal of mature trees through logging may have reduced local seed availability, which could explain the delayed recruitment increase approximately 20 years later (1970) when post-logging cohorts reached reproductive maturity. Douglas-fir seeds typically start to decline in winter after seed fall in autumn (Hermann and Lavender, 1990), but rodents will opportunistically cache the seeds where they can remain viable for upwards of six years, perhaps supporting biological factors including prolonged lifespan of the seedbank inherently influencing establishment (Hofmann, 1920). Interior Douglas-fir exhibits masting, or periodic high cone production, approximately every 7–10 years (Hermann and Lavender, 1990; Lastra, 2001); however, research in British Columbia is limited. It is likely

when these events coincide with disturbance or unusually wet conditions, seedling establishment could be enhanced (Peters et al., 2005).

Establishment dates of trees were assessed in relation to broad scale climate variability including PDO and ENSO (Figure 3.9). Establishment peaks during the 1970s, and again during the 1990s occurred during both positive (warm) and negative (cool) phases of PDO and ENSO indicating no consistent relationship to broadscale climate variability. Although PDO and ENSO are both highly variable, the intervention analysis was able to identify dominant cool and warm phases. It is possible that small fluctuations in climate not captured with the barcode plot (Figure 3.9D, 3.9E) influenced trees at this site, but with no pattern found for climate variables tested, thus, I conclude climate is not a dominant influence on establishment. Due to uncertainty associated with age estimation methods, establishment dates were grouped into bins, limiting the ability to identify specific years of high recruitment. However, studies from climatically similar regions provide useful context including a study in Colorado by League and Veblen (2006) where the authors identified 1973 and 1979 as high establishment years coinciding with positive ENSO phases (La Niña). I found pulses between 1970-1975 and 1975-1980 so it is possible the years these authors found to be high establishment years coincide with our data. La Niña is typically associated with increased moisture and cooler temperatures in the interior region of western North America.

Regional interannual climate variables including annual and growing season precipitation, mean temperature, and SPEI were assessed for associations with tree establishment and no significant relationships were found. Climate patterns indicate variable but overall increasing annual precipitation in the last century (Wang et al., 2023). Precipitation was of particular interest because it is generally the strongest climatic control on Douglas-fir growth in the interior (Griesbauer et al., 2010). There are six years where two out of three months in May, June and July precipitation far exceed the monthly mean (1977, 1980, 1990, 1993, 1999, 2000; Figure 3.10) but establishment rates were not significantly higher during or near these wetter periods. High tree establishment years were generally non-synchronous with regional climate. But due to the associated error with age estimations of the trees (ring counts only, pith estimations, and coring height correction factor), results must be interpreted carefully as establishment years might not be accurate, and further research is warranted to be conclusive.

The absence of clear associations between climate variables and pulses of establishment indicates site specific factors are likely more influential to the overall recruitment and establishment of trees at the study site. Interactions between competition and shading on survival of seedlings in Douglas-fir is demonstrated by Haugo et al. (2013) where shading proved to be the most important factor in establishment of seedlings. Microclimates likely play an important role by providing shade, reducing VPD, and moderating air temperature (Wright et al., 2014). The variation in direction and strength of biotic and abiotic factors alter the constraints on successful establishment and growth highlighting the importance of interactive effects (Haugo et al., 2013). Competition from surrounding herbaceous vegetation can also reduce survival due to decreased soil nitrogen and water availability (van der Waal, 2009). Although we did not discern any relationship between increased moisture and higher rates of encroachment, Douglas-fir still require adequate levels of moisture for recruitment processes and prolonged periods of drought would increase seedling mortality (Heiner and Lavender, 1972).

Negative influence on Douglas-fir seedling survival associated with cattle grazing

Seedling establishment appeared to be strongly influenced by grazing pressure as seedling density was approximately two times higher in the plot with no recent or active cattle grazing (Figure 3.3b). The role of grazing (by cattle or native ungulates) is complex, as conflicting evidence on whether the presence of cattle is associated with increased seedling mortality or an increase in survival. Bai et al. (2000) found that manure enhanced seedling emergence when compared to mineral soil alone, and that Douglas-fir needle accumulations were not beneficial. But areas with bare mineral soil are prone to higher soil temperatures, which as Rank et al. (2022) and Davis et al. (2019) found to be detrimental to seedling survival. Further, a reduction in seedlings due to trampling and browsing is evident in areas with cattle grazing (Dunwiddie, 1977). Other research indicates that seedlings may prefer a duff layer and surrounding vegetation supporting a moisture rich environment (Hermann and Lavender, 1990), but there is some contradictory evidence in which exposed mineral soil provides a preferable seedbed with reduced competition (Rummel, 1951). In this study, I found a twofold higher density of seedlings in Plot 1 (n = 5308) when compared to Plot 2 (n = 2384, Table 3.1, Appendix Figure B1). At the time of sampling, active cattle grazing was evident in Plot 2, whereas Plot 1 had no evidence of recent

grazing or cattle activity (no manure, reduced soil erosion) (Figure 3.3). However, saplings had almost identical counts in Plot 1 and 2 indicating their ability to persist cattle disturbance (Table 3.1). Our results indicate seedling survival is potentially inhibited by cattle presence. The disturbance of trampling and grazing likely outweighs the factors of bare soil (open seedbeds) and reduced herbaceous competition.

Further analysis using seedling density maps (Appendix Figure B1) points to why seedlings may prefer conditions in Plot 1. For both plots, the highest density of seedlings was found to be in more open areas. Grasslands can provide an adequate seedbed for over wintering and stratification of Douglas-fir seeds (Bai et al., 2004). These open areas within the grasslands or canopy gaps likely had more bare soil, lowered competition, and occurred on the north side of trees (Marshall et al., 2023). With conditions in each plot comparable other than the presence of cattle, we can infer this is likely influencing establishment.

Lack of fire contributing to encroachment

Wildfire is a dominant disturbance in forest-grassland ecotones promoting species diversity, environmental heterogeneity, and can act as a control mechanism for encroachment (Peterson and Reich, 2007). In interior BC, wildfire occurrence has been drastically reduced since European settlement in the late 1800's contributing to the change in forest structure observed on the landscape today (Figure 3.1). Typically, lightning ignited fires burned periodically in this area constrained by natural barriers and heterogeneity among vegetation (Hessburg et al., 2022). Secwepemc First Nations communities within the area also contributed greatly to the documented fire regime through cultural burning and land stewardship techniques involving fire (Copes-Gerbitz et al., 2022; FREC, 2021; Hoffman et al., 2021; Lake and Christianson, 2019). In addition to the removal of wildfire from the landscape, modern forestry practices have reduced natural fuel breaks, introduced large-scale monoculture plantations, and harvested mature fire-resistant trees (Hessburg et al. 2019). However, in the CCPA documentation exists describing the last known burning to have occurred in the 1950's (BC Parks, 2000) as logging companies purchased the land from ranchers (Empire Valley Ranch) and halted the practice of burning to enhance grassland forage through decreasing the establishment of encroachment. A mixed severity fire regime dominated by low and moderate severity fire (Harvey et al., 2017) controlled

the establishment of Douglas-fir trees in the upper grasslands in part due to historical burning by First Nations in the area, as described above (Gruell et al., 1986; Iverson et al., 2002). Frequent understory fires killed seedlings and saplings, while large fire-resistant trees were left unharmed. With the lack of frequent understory fire in this area, Douglas-fir have encroached and continue to spread into the grasslands (Figure 3.2, Figure 3.5). Not only does this reduce the area of grasslands but it contributes to lowered species richness, lowered forage availability, and increased wildfire risk (Archer et al., 2017; Ratajczak et al., 2012, Strang and Parminter, 1980). Considering the contemporary changes to the character of interior forests, driven by fire suppression and forest management activities (patchy, uneven aged forests have been replaced with homogenous, dense, and immature stands) fuel loads on the landscape support moderate to high intensity wildfires, which lead to substantial forest mortality and ecosystem alteration in areas that used to be resilient to wildfire. Fire exclusion over the past century has resulted in the accumulation of forest fuels that would historically have been consumed by frequent burning, contributing to the emergence of large, severe wildfire events. A large (12,040ha) wildfire burned in the CCPA in 2021 and in some areas burned very severely, often in areas with dense encroachment (BCWS, 2026). This fire is the result of a fire regime deficit within the CCPA, and the across the province.

Management Implications

Encroachment in the CCPA, similar to grasslands elsewhere in the Inland Northwest, is a long-term issue that has resulted in a wide range of negative ecological effects including loss of biodiversity, increased fire risk, and a shift in carbon stores (Van Auken, 2009). Elevations between 700-1000m had the greatest change in vegetation cover types from grassland to forest over a 30-year period (1960-1990) in the Cariboo region (Bai et al., 2004). Recent spatial analyses by Reimer (2023) indicates 30,715ha of encroachment has occurred in the IDFxm subzone in the Cariboo Natural Resource Region since 1997. This highlights the rapid rate at which grassland loss is occurring. This study can help guide management practices as evidence points that a reduced disturbance regime and favourable site level parameters (i.e. shading, available seedbed, microclimate) can occur concomitantly driving pulses of Douglas-fir invasion. Incorporating grazing or burning in areas after a high seed year can pro-actively prevent the

further spread of trees in the grasslands and reduce current levels of encroachment. Ensuring local involvement from First Nations communities, including co-management will also help support reconciliation initiatives. As stands mature and become more difficult to manage, costs increase as well as treatment length, as mechanical or hand thinning treatments are then required prior to prescribed burning. But, continued thinning and removal treatments in areas previously encroached is still recommended for ecological and wildfire risk reduction objectives. Often stands designated as MDWR are overgrown and deer cannot effectively navigate through them; mature Douglas-fir open canopies are also important for intercepting snowfall where ground forage can be found (Day, 1998). Conservation of grassland areas in the Cariboo region is a top conservation priority and has been highlighted in multiple reports (CCPA Management Plan (2000) and Fire Management Plan (2001), Cariboo Chilcotin Land Use Plan (1994), Cariboo Chilcotin Grasslands Strategy (2001)). With ecological change accelerating, encroachment needs to be addressed across the Cariboo for community safety, habitat preservation, and the natural integrity of BC's grasslands.

Limitations

Tree cores and basal disks (all samples) were not crossdated. Basal disks were generally from young trees (<50 years), and accurately aging using statistical verification is difficult due to growth irregularities and in several cases at this site, stunting in the most recent ~20 years of growth which makes discerning the ring boundaries difficult (Chhin and Wang, 2007). To account for rings missed due to sampling height, a coring height correction factor was used. When pith was missed, a pith estimation method (Duncan, 1989) was used as well, contributing to the possible error. Both of these methods use estimation and add to the error in accurate aging. Due to these combined factors, using analytical methods that are year and seasonally specific to make statistical conclusions could not be used and therefore only associated patterns and multi-year groups were analyzed.

CONCLUSION

Encroachment has been documented in the CCPA for over a century and represents a natural ecological process; however, the current scale and rate of expansion threaten the ecological integrity of these grassland ecosystems (Wieczorkowski and Lehmann, 2022). Understanding the mechanisms driving tree encroachment into grasslands is therefore critical for informing management practices. This study contributes to addressing this knowledge gap by examining the temporal, spatial, and ecological factors influencing establishment. Here, I examined forest encroachment dynamics in the CCPA by incorporating analyses of the land use history, temporal patterns, and spatial structure. Both abiotic and biotic interactions are influential to the dynamics occurring at this site. Overall, climate is less influential in facilitating pulses of encroachment than was hypothesized and site-specific factors have a greater effect. Thus, the structure of trees at this site is based on the complicated relationships between microsites, life-history characteristics, disturbance, and moisture availability. As evident throughout the study site years in which site-specific factors including sufficient seed production, reduced grazing, lack of fire, and appropriate microclimate conditions contribute to the episodic establishment of Douglas-fir.

Continued and future management of forest-grassland ecotones is needed to help conserve what there is left of native grasslands in BC. Techniques involving prescribed fire should be used as a management technique (Blackwell et al., 2000) incorporating local Indigenous communities in planning and execution. Longer term studies or studies that can be replicated within forest-grassland ecotones in the CCPA could provide further insight into the specific mechanisms responsible for facilitating these pulses of tree establishment and encroachment.

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Chapter 4: CONCLUSION

Important research on the efficacy of fuel treatments continues to be done in western Canada, but gaps remain on how fuel treatments impact surrounding plant communities. As policymakers, land managers, and communities advocate for increased fuel treatments, understanding treatment effects on both understory vegetation and residual trees is critical for promoting ecological resilience to wildfire (Daniels et al., 2025; Stevens et al., 2014). Informing these treatments with the historical fire regime for each specific region can aid in and provide context for land management, potentially providing insight into areas that burned frequently, or areas that acted as wildfire refugia (Parks et al., 2025). In addition to understanding the effects of treatments, it is important to understand the processes that lead forests to reach conditions requiring thinning. Identifying the drivers and patterns of forest encroachment and infilling can help guide management strategies that address these changes at an earlier successional stage. This research study examined forest encroachment, evaluated treatment using thinning and prescribed fire as a potential management method, and assessed the ecological effectiveness of these treatments to inform future management strategies. By integrating multiple components, this study provides a more holistic understanding of how treatments influence ecosystem dynamics.

The complex relationships between fire, climate, and forest-grassland ecotones is not yet fully understood, but we were able to advance the knowledge base needed to inform management of forest encroachment dynamics and restoration in dry interior ecosystems in British Columbia (BC). The findings from Chapter 2 indicate mechanical thinning and prescribed fire can create conditions suitable for recovery where species richness is increased, functional group composition is shifted to more forb and grass dominant, and tree size is significantly increased. The understory plant community and overstory trees influence one another's success, supporting holistic ecosystem management. But treatments such as these must be continually managed over time as one treatment is often not enough to create lasting change (Nagelson et al., 2024). Here, I saw the spreading nature of pinegrass (*Calamagrostis rubescens*) and competitive exclusion resulting in the slow recovery of native bunchgrasses such as Bluebunch wheatgrass (*Pseudoroegneria spicata*) and spreading needlegrass (*Achnatherum richardsonii)* which often indicate later seral stages in the interior Douglas-fir biogeoclimatic zone (IDF) (Meidinger and Pojar, 1991). Further treatments can help mitigate these effects but require planning and maintenance. Although a thinning only site was not available for comparison, these findings

support other research suggesting that the combination of thinning and prescribed fire is more beneficial to the sites ecology than thinning alone supporting patchy heterogeneity increasing species richness across the site (Dodson et al., 2008; Harrod et al., 2009; Metlen and Fiedler, 2005).

Chapter 3 highlights the complex interactions that shape the process of encroachment in the Churn Creek Protected Area (CCPA). The role of climate, disturbance, and site-specific factors like shade, aspect, grazing, and microclimates on tree establishment and encroachment have not yet been fully understood, but this study provides insight into these relationships. Although trees were not found to be clustered spatially, trees were found to be clustered by similar age yet dispersed or randomly distributed by tree size (DBH). In areas heavily infilled with young Douglas-fir (*Pseudotsuga menziesii* var. *glauca*) an even aged cohort was present but select trees were able to outcompete neighbours and grow larger in size. Surrounding mature seed trees add to this dispersal or randomness in tree size. The even aged cohorts are likely establishing based on site-specific factors aligning for recruitment, rather than driven primarily by climate. With three main pulses of establishment found during the 1970s and 1990s, the visual association with climate records indicated no clear relationship to cool, wet years. However, our analyses are limited on this topic, and I recommend further research in this area. A history of disturbance at this site prior to the 1950s points to historical fire or grazing as the primary limiting factor on recruitment, but with the altered disturbance regime post 1950s an increase in recruitment is evident.

Findings from this study align with BC Parks Fire Management Plan for the Churn Creek Protected Area (2001) in using thinning and prescribed fire as a restoration tool in forest-grassland ecotones. The Churn Creek Protected Area management plan (2000) and the Fire Management Plan Churn Creek Protected Area (2001) specifically reference managing for encroachment and ingrowth using prescribed fire. Managing encroachment and grassland degradation is critical to support the long-term health of grassland-forest ecotones in the CCPA, and the rest of the Cariboo-Chilcotin region. By re-introducing fire into forest-grassland ecotones which were once shaped through fire, we can support wildfire resiliency goals for transformative adaptation. In addition, fire can revitalize understory vegetation and increase seedling and sapling mortality, helping to limit further forest encroachment. Increasing research on wildfire

resiliency calls for paradigm shifts in forest management through community and Indigenous collaboration to coexist with wildfire (Daniels et al., 2025). Management should prioritize reducing stem densities, increasing heterogeneity at a stand level, and implementing fire on the landscape by incorporating Indigenous led fire-stewardship or prescribed fire post thinning in suitable ecosystems. Integrating a mosaic across the greater region can enhance resiliency through restoring “natural” barriers to high intensity wildfire and increasing heterogeneity amongst ecosystems (Hessburg et al., 2019; Zaiats et al., 2024). Adapting to a future fire regime informed by legacy forests, evidence-based research, and the current context of wildfire on the land is critical to meet the transformative adaptation goals society needs (Daniels et al., 2020; McWethy et al., 2019). Proactive, multi-stakeholder stewardship is essential, as the combined effects of fire exclusion and climate change are driving an increase in large, high-severity wildfire events, whether society is prepared or not. Treatments can enhance ecosystem resilience by reducing the severity and landscape-level impacts of wildfire when done in conjunction with other mitigation strategies including Firesmart practices and principles around communities, increasing public awareness, and management of the landscape as one entity, not separate parcels. Treatments are one tool of many to support coexistence with wildfire.

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APPENDIX A

Appendix Table A1. SIMPER analysis pairwise results for each of the site types showing the contribution of each species for up to 75% of the differences. Table includes average dissimilarity of each species (average), standard deviation (SD), ratio of average contribution to the overall dissimilarity/SD (ratio), average abundance in group A (Ava), average abundance in group B (Avb), cumulative contribution to dissimilarity between groups (cumulative), P-value, and if the P-value is significant (significance).

Grassland vs untreated SIMPER results

Species	Average	SD	Ratio	Ava	Avb	Cumulative	P-value	Significance
Moss	0.24796	0.12924	1.92	0.00	36.90	0.251	0.001	***
Calamgrostis rubescens	0.16335	0.09236	1.77	0.00	23.45	0.417	0.989	ns
Poa pratensis	0.16123	0.06903	2.34	23.00	0.03	0.581	0.001	***
Tragopogon spp.	0.05856	0.04424	1.32	8.20	0.03	0.640	0.001	***
Camanadra umebellata	0.03981	0.03895	1.02	5.47	0.00	0.680	0.001	***
Antennaria spp.	0.03788	0.02693	1.41	5.55	0.34	0.719	0.001	***

Grassland vs. treated SIMPER results

Species	Average	SD	Ratio	Ava	Avb	Cumulative	P-value	Significance
Calamgrostis rubescens	0.29229	0.17991	1.6246	0.00	40.5	0.340	0.001	***
Poa pratensis	0.13873	0.08187	1.6945	23.0	7.36	0.501	0.001	***
Tragopogon spp.	0.06167	0.04704	1.3112	8.20	0.49	0.572	0.001	***
Camanadra umebellata	0.04384	0.04278	1.0250	5.47	0.00	0.623	0.001	***
Antennaria spp	0.04302	0.02920	1.4734	5.55	0.12	0.673	0.001	***
Hesperostipa comata	0.03956	0.05752	0.6877	4.84	0.01	0.719	0.001	***
Cerastium arvense	0.02555	0.03737	0.6838	3.57	0.03	0.749	0.001	***

Treated vs untreated SIMPER results

Species	Average	SD	Ratio	Ava	Avb	Cumulative	P-value	Significance
Moss	0.26280	0.14051	1.8703	36.9	0	0.361	0.001	***
Calamgrostis rubescens	0.21273	0.1415	1.5028	23.4	40.5	0.65	0.183	ns
Poa pratensis	0.05855	0.08191	0.7148	0.03	7.36	0.734	0.997	ns

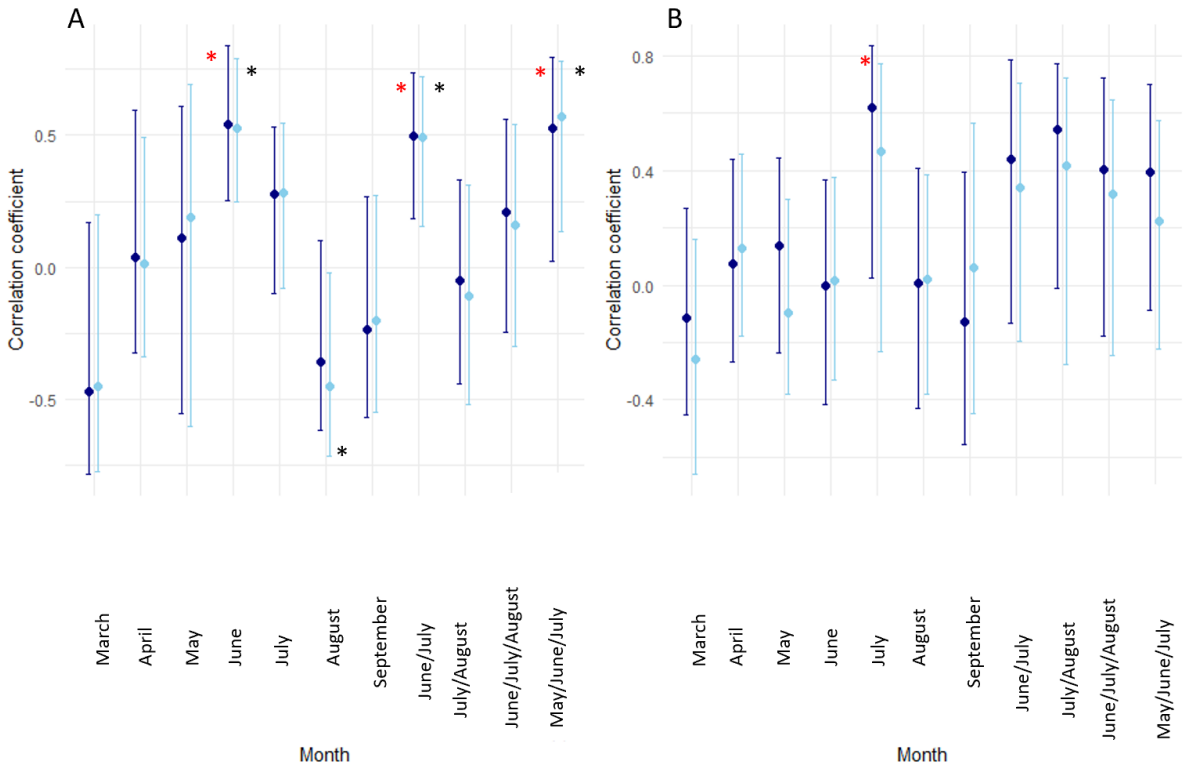
Appendix Table A2. Seedlings and saplings counted in the treated and untreated sites.

	Treated		Untreated	
	Seedlings	Saplings	Seedlings	Saplings
Pseudotsuga menziesii	389	25	1534	29
Picea engelmannii x glauca	0	0	0	1
Populus tremuloides	0	38	0	2
Total	389	63	1534	32
Average	30	5	77	3

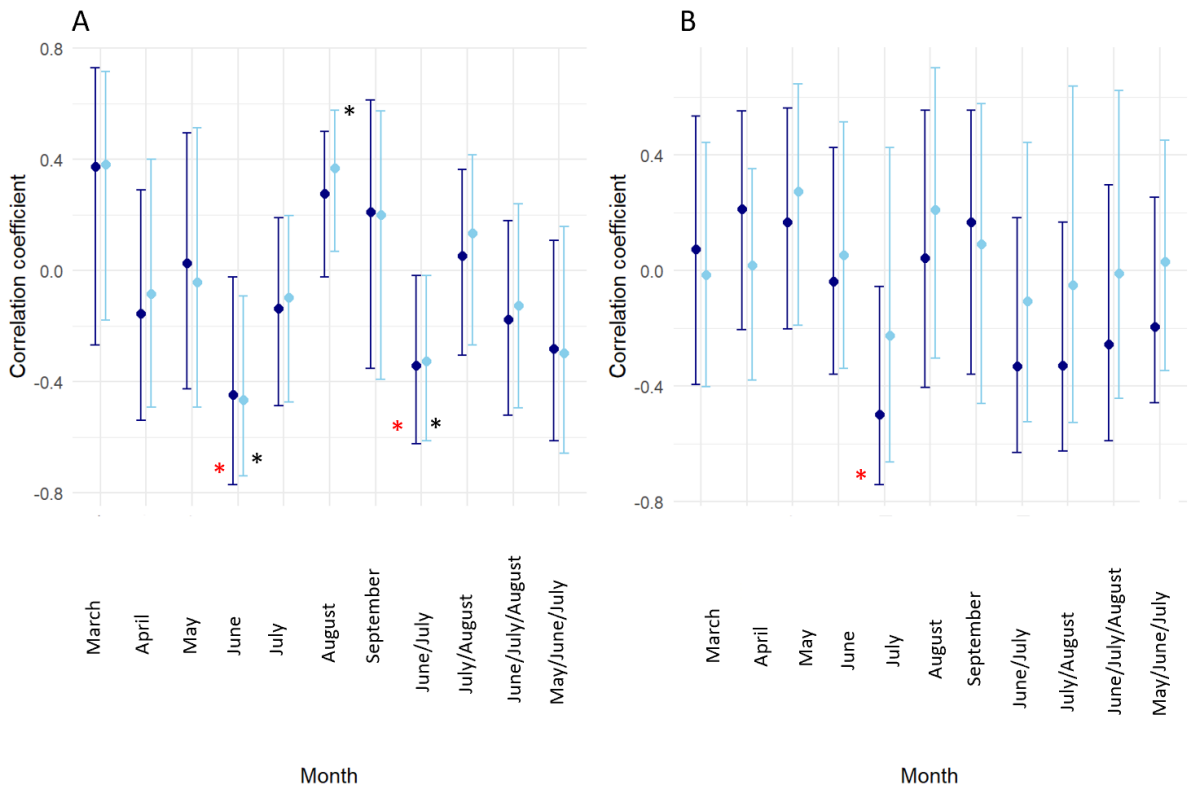
Appendix Table A3. Pearson correlation coefficient values for precipitation, CMI (climate moisture index), and maximum temperature for the climate growth relationship analysis between trees in the treated site and trees in the untreated site. Monthly values for single months in the growing season started in March until September were examined, as well as means of monthly values for June/July, July/August, June/July/August, and May/June/July

	Treated trees		Untreated trees	
Precipitation	Post-treatment	Pre-treatment	Post-treatment	Pre-treatment
March	-0.111	-0.356	-0.347	-0.332
April	0.228	-0.009	0.186	-0.019
May	0.262	0.182	0.01	0.241
June	-0.048	*0.552	0.014	*0.524
July	* 0.584	0.298	*0.535	0.351
August	-0.04	-0.367	0.175	-0.44
September	-0.098	-0.244	0.112	-0.184
June/July	0.408	*0.539	0.391	*0.562
July/August	0.492	-0.052	*0.520	-0.078
June/July/August	0.345	0.21	0.385	0.161
May/June/July	*0.424	*0.571	0.291	*0.627
Maximum Temp				
March	0.074	0.361	0.001	0.38
April	0.215	-0.149	0.022	-0.078
May	0.169	0.031	0.267	-0.035
June	-0.037	*-0.444	0.032	*-0.462
July	*-0.498	-0.13	-0.245	-0.093
August	0.043	0.278	0.204	*0.374
September	0.167	0.195	0.077	0.202
June/July	-0.331	*-0.336	-0.138	*-0.318
July/August	-0.329	0.064	-0.064	0.137
June/July/August	-0.255	-0.163	-0.023	-0.119
May/June/July	-0.195	-0.278	0.01	-0.302
CMI				
March	-0.114	-0.469	-0.26	-0.451
April	0.074	0.039	0.129	0.013
May	0.137	0.111	-0.095	0.191
June	0	*0.538	0.017	*0.526
July	*0.618	0.275	0.467	0.284
August	0.006	-0.355	0.019	-0.449
September	-0.13	-0.234	0.06	-0.202
June/July	0.439	*0.497	0.342	*0.492
July/August	0.545	-0.051	0.416	-0.109
June/July/August	0.403	0.209	0.319	0.158

May/June/July	0.394	*0.525	0.222	*0.568



Appendix Figure A1. Pearson correlation coefficient values for tree ring growth and CMI (climate moisture index) for chronologies (A) prior (1987-2005) to and (B) after (2006-2022) treatment. Black * indicates significance (95% confidence intervals) for the untreated site, and red * indicates significance for treated site. Dark blue represents the trees in the treated site and light blue represents the trees in the untreated site. Monthly CMI values for single months in the growing season starting in March until September were examined, as well as means of monthly values for June/July, July/August, June/July/August, and May/June/July



Appendix Figure A2. Pearson correlation coefficient values for tree ring growth and maximum temperature for chronologies (A) prior (1987-2005) to and (B) after (2006-2022) treatment. Black * indicates significance (95% confidence intervals) for the untreated site, and red * indicates significance for treated site. Dark blue represents the trees in the treated site and light blue represents the trees in the untreated site. Monthly precipitation values for single months in the growing season starting in March until September were examined, as well as means of monthly values for June/July, July/August, June/July/August, and May/June/July

Appendix Table A4. Most common fire adapted species found within the study site and their associated traits. ***Achillea millefolium* was reclassified from native to exotic in 2019 as research found the taxonomy of many plants found in the wild was originally of Swedish descent (Ramsey et al. 2008, BC Conservation Data Centre, 2025). It is thought that a mix of native and exotic plants exist in BC but are not able to be separately identified without genetic testing.

Fire adapted species found in the study site		
Species	Species Status	Fire adapted traits
Calachortus apiculatus (sagebrush mariposa lily)	Native, Yellow-list	Bulbs likely unaffected by fire underground and can resprout. Can increase seed production post fire. First Nations used fire to increase this species (Blackwell et al. 2000)
Calamagrostis rubescens (pinegrass)	Native, Yellow-list	Rhizomic species: top can be killed by fire but resprouts and can increase post-fire.
Pseudoroegneria spicata (bluebunch wheatgrass)	Native, Yellow-list	Basal buds on root-crown resprout post fire. Plant tops is highly flammable and burns quickly does little damage to root crown (Willms et al 1981)
Rosa acicularis (prickly rose)	Native, Yellow-list	Seeds are fire resistant. Rhizomic species which can resprout at base of stems
*Symphoricarpos albus (*common snowberry)	Native, Yellow-list	Can handle infrequent fire and resprouts from rhizomes as top is killed and recolonizes quickly (Miller 1977).
Pseudtosuga menziesii var. *glauca (*interior Douglas-fir)	Native, Yellow-list	Mature trees resistant to moderate surface fires - thick insulating bark, lower limbs dropped. Immature trees susceptible to burning. Seeds dispersed by wind into open burned areas providing good seed bed, shaded under remaining mature trees
Poa pratensis (Kentucky bluegrass)	Exotic, Agronomic	Above ground stems consumed by fire, and if burnt in the spring post new-growth, plant will be most damaged. During dormant season plants are not affected much and can initiate new growth.
Tragopogon spp. (salsify)	Exotic, Noxious weed	Top killed by fire, will regrow from root crowns or re-seeded by neighboring plants in exposed burnt areas. Burn after bolting before seed sets to reduce

Achillea millefolium (yarrow)	**Exotic/Native,	Regeneration post fire by rhizome and wind dispersed seeds. Plant stimulated by low/moderate severity fire but can reduced if burnt in spring

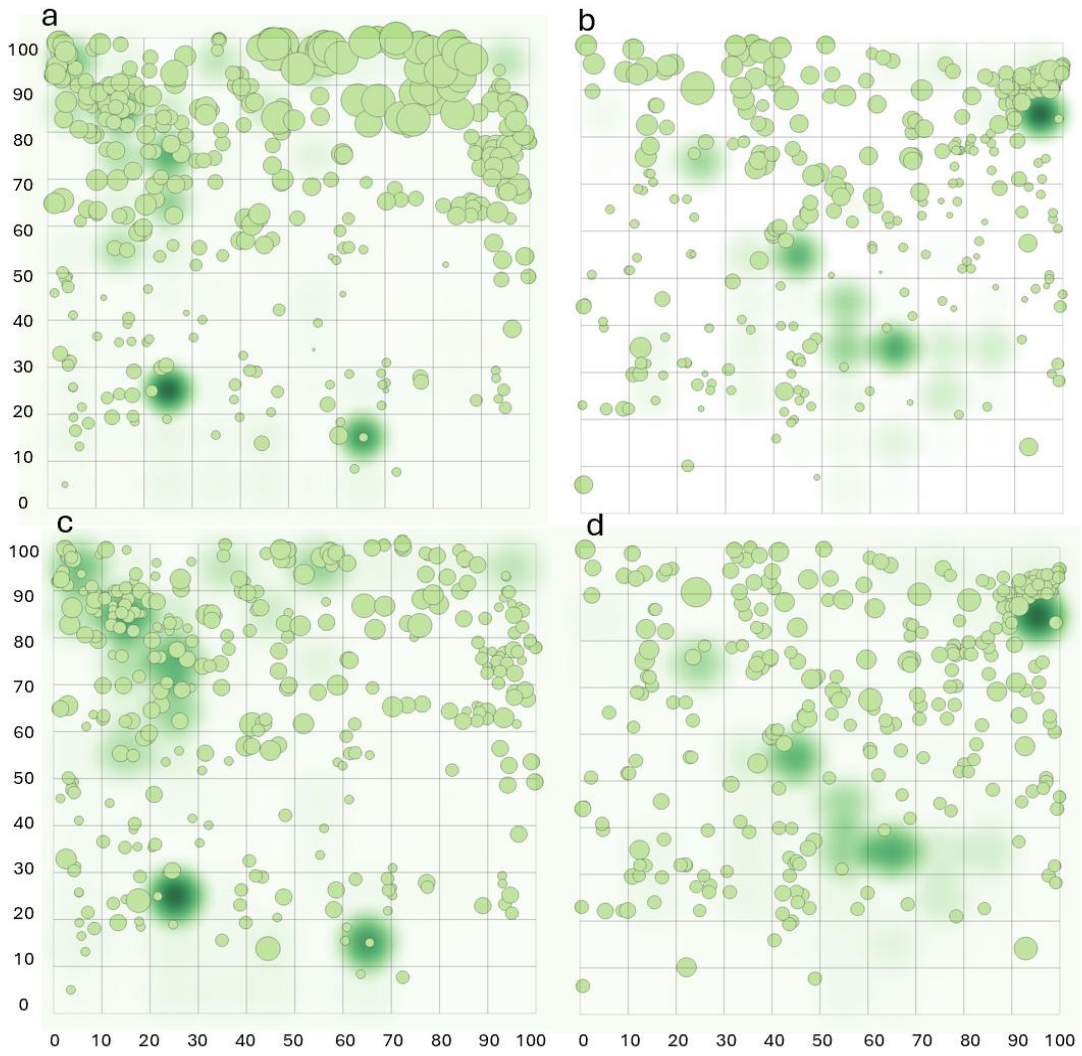
APPENDIX B

Appendix Table B1. Counts of seedlings and saplings for Plot 1 and 2 by subplot

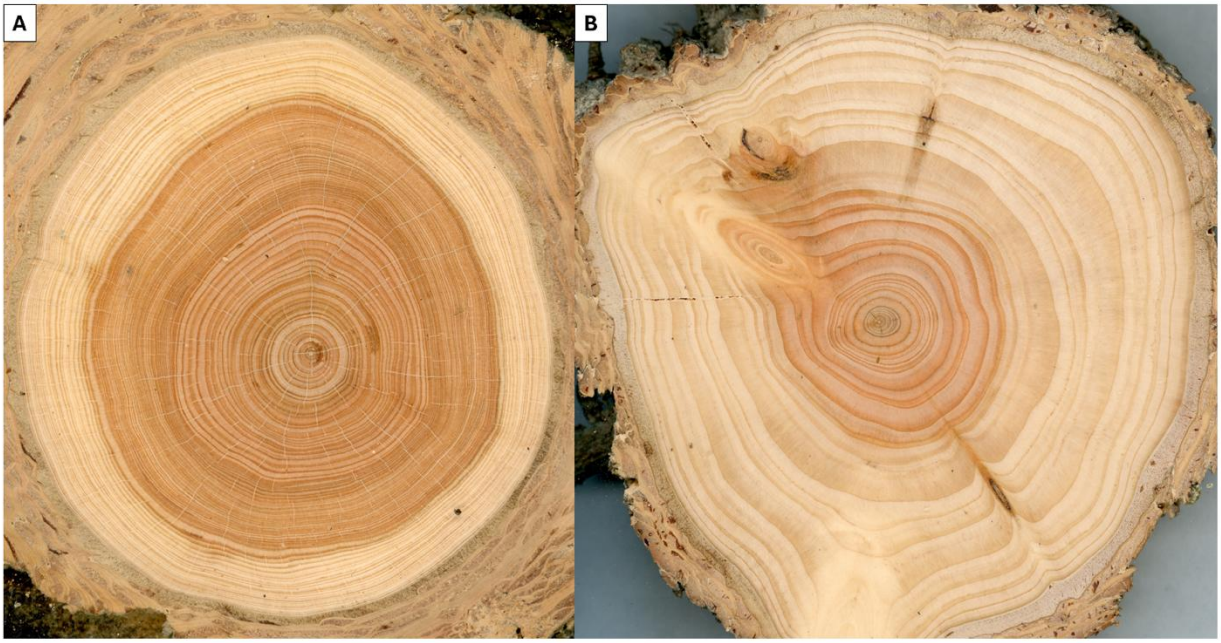
Plot1				Plot 2		
subplot	saplings	seedlings		subplot	saplings	seedlings
1a	0	19		1a	0	0
1b	0	57		1b	0	0
1c	0	35		1c	3	0
1d	0	15		1d	1	0
1e	2	20		1e	0	1
1f	1	46		1f	0	1
1g	0	0		1g	0	0
1h	0	5		1h	0	8
1i	0	176		1i	0	27
1j	0	309		1j	0	2
2a	1	9		2a	0	0
2b	0	22		2b	0	0
2c	0	8		2c	0	34
2d	0	25		2d	3	33
2e	0	3		2e	0	1
2f	1	208		2f	0	0
2g	0	82		2g	0	4
2h	0	197		2h	3	25
2i	0	392		2i	0	0
2j	0	62		2j	1	7
3a	2	63		3a	0	0
3b	1	68		3b	0	4
3c	0	516		3c	0	4
3d	0	0		3d	0	1
3e	1	33		3e	0	0
3f	1	80		3f	0	0
3g	0	252		3g	0	2
3h	0	354		3h	2	148
3i	0	117		3i	1	0
4a	2	51		3j	0	1
4b	1	36		4a	0	0
4c	1	34		4b	0	0
4d	3	2		4c	0	35
4e	1	22		4d	1	41
4f	0	0		4e	1	31
4g	0	10		4f	0	65
4h	1	29		4g	0	0
4j	0	165		4h	0	4
5a	0	57		4i	0	4

5b	0	72		4j	1	0
5c	0	1		5a	0	1
5d	0	0		5b	0	11
5e	0	0		5c	0	17
5f	0	13		5d	0	12
5g	0	0		5e	0	26
5h	0	5		5f	0	220
5i	0	133		5g	0	5
5j	0	39		5h	0	9
6a	1	16		5i	0	11
6b	1	0		5j	0	4
6c	0	25		6a	0	28
6d	2	29		6b	0	28
6e	1	51		6c	1	39
6f	0	1		6d	1	156
6h	0	87		6e	2	143
6j	0	246		6f	0	0
7a	0	61		6g	0	24
7b	1	417		6h	0	0
7c	1	40		6i	0	3
7d	2	0		6j	0	9
7e	0	2		7a	0	12
7f	0	10		7b	0	53
7g	0	2		7c	0	13
7h	0	1		7d	0	230
7i	0	0		7e	0	14
7j	1	63		7f	0	9
8a	0	1		7g	7	1
8b	0	3		7h	1	0
8c	0	13		7i	0	4
8d	0	1		7j	0	25
8e	0	3		8a	0	0
8f	1	2		8b	0	12
8g	0	0		8c	1	79
8h	0	2		8d	0	73
8i	3	2		8e	0	0
8j	0	54		8f	0	11
9a	0	0		8g	0	0
9b	0	1		8h	5	2
9c	0	13		8i	2	4
9d	0	13		8j	0	46
9e	1	6		9a	0	2
9f	0	0		9b	0	5
9g	1	1		9c	0	16
9h	0	3		9d	0	77

9j	3	61		9e	0	14
10a	0	1		9f	0	1
10b	0	0		9g	0	0
10c	0	0		9h	5	0
10d	0	2		9i	0	31
10e	0	0		9j	0	20
10f	0	0		10a	0	0
10g	0	5		10b	0	0
10h	0	15		10c	0	3
10i	4	18		10d	0	0
10j	5	174		10e	0	0
				10f	1	0
				10g	2	0
				10h	1	0
				10i	1	330
				10j	0	50



Appendix Figure B1. (A) Plot 1 tree age, (B) Plot 2 tree age, (C) is Plot 1 tree size (diameter breast height -DBH), (D) is Plot 2 DBH. The size of the symbol (green circle) corresponds to increasing DBH or age. Each size increment is in 10cm or 10 year age groups. Trees >100 years in age are grouped together. Heat map clusters (darker green) represent seedling density per 10 x 10m subplot.



Appendix Figure B2. Basal disk samples collected from Plot 1 where (A) has 132 rings with a diameter breast height (DBH) of 6.4cm whereas (B) has 25 rings with a DBH of 9.8cm. (A) This tree was located at the western most subplot in Plot 1 where the ecotone transitioned to dense forest. (B) This tree was located in the second most eastern subplot in the more open grassland side of Plot 1. This demonstrates the differences in growth by location.