

13,000 years of fire activity in a temperate
rainforest on the Central Coast of British Columbia, Canada

By:

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B.Sc. (Honours), University of Victoria, 2012

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In the School of Environmental Studies

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Abstract

While wildfire is globally most common in the savanna-grassland ecotone, the flammability of coastal temperate rainforests is considered low and little is known regarding historic fire activity. Reconstructing historical fire activity typically requires dendrochronological records from fire-scarred trees and post-fire cohorts, but this type of information is rare in perhumid temperate rainforests, which are dominated by dense fuels with high year-round moisture content. I reconstructed historic fire activity using fire scars, tree rings, soil charcoal, and remote sensing techniques in a 2000 km² island group located within the Hakai Lúxvbálís Conservancy on the coastal margin of central British Columbia. I broadly assessed 13,000 years of fire activity with charcoal deposited in soils, and reconstructed late Holocene fire events with a 700-year chronology built from living fire-scarred trees and stand establishment data. I used a weight of evidence approach to hypothesize the origins of fires and whether First Nations intentionally utilized fire for resource management. Low-severity fires occurred most frequently in forests surrounding former First Nations habitation sites, and lightning strikes do not occur often enough to explain the observed temporal or spatial patterns of fire activity in the study area. Low-severity fires occurred approximately every 39 years, and were 25 times more likely to occur than previously estimated. Fires influenced the composition and structure of vegetation by creating a mosaic of vegetation types in different stages of succession, and thus increased the abundance of culturally important food plants. Fire events have not occurred in the study area since 1893, which also coincides with the reduction of First Nations activities in their traditional territories. My data are consistent with the hypothesis that humans intentionally used fire to manage resources, though

further research and ethnographic data collected elsewhere in the region is required to corroborate these findings. Ecological legacies of historic fires remain visible on the present day landscape, and by reconstructing the historic range of fire cycle variability we gain a better understanding of human-driven fire activity and the abrupt changes that occurred in the 20th century.

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Dedication

To the forests that my son Ansel will someday roam

Chapter 1: Introduction

1.1 Background

Fire is both a natural and anthropogenic disturbance that is nearly ubiquitous in terrestrial ecosystems (Bowman et al. 2009; Whitlock et al. 2010). Humans have interacted and co-evolved with fire for millions of years, changing the temporal and spatial components of fire by igniting fires in landscapes that would otherwise be void of ignition sources (Bond and Keeley, 2005; Pausas and Keeley, 2009). The cultural importance of fire in western North America has been mostly examined in the savanna-grassland ecotone (Guyette et al. 2002; Bowman et al. 2011; Ryan et al. 2013) and in fire-adapted forests dominated by Ponderosa pine (*Pinus ponderosa*), Douglas-fir (*Pseudotsuga menziesii*), and Garry oak (*Quercus garryana*) (Veblen et al. 2000; Lepofsky et al. 2005; Gedalof et al. 2006; Odion et al. 2014; Moritz et al. 2014). There has been little documented evidence of historic fire activity in British Columbia coastal temperate rainforests, and previous disturbance studies have asserted that both human- and lightning-caused fires are infrequent and become progressively rarer in high-latitude regions ($> 50^{\circ}$ N) (Lertzman et al. 2002; Gavin et al. 2003a; Gavin et al. 2003b; Hallett et al. 2003; Daniels and Gray, 2006).

Low fire activity in high-latitude (hereafter ‘perhumid’) temperate rainforests is assumed due to the dominance of old growth forests on the landscape (Daniels and Gray, 2006), the lack of fire adaptations in dominant conifer species (Agee, 1993), the decreased flammability of large-diameter, moisture-laden surface fuels (Guyette et al. 2002; McWethy et al. 2013), and the absence of 20th century fire activity (Meyn et al. 2010). A

handful of studies have described temporally coarse reconstructions of Holocene fire activity in coastal temperate rainforests in British Columbia with charcoal stratigraphy in lake sediments (Brown and Hebda, 2002; Gavin et al. 2003a; Gavin et al. 2007) and radiocarbon dating of soil charcoal (Lertzman et al. 2002; Gavin et al. 2003b; Hallett et al. 2003). These studies depict patchy fires during the last few centuries embedded within landscapes experiencing long (sometimes >1000 year) fire intervals as well as widespread increases in fire activity in the mid- and late Holocene (Lertzman et al. 2002; Gavin et al. 2003b; Gavin et al. 2006) that may be related to increases in the use of fire by First Nations (Lepofsky et al. 2005; Marlon et al. 2012; Walsh et al. 2015).

The near absence of lightning and more than 13,000 years of human history on the Central Coast of British Columbia make it an ideal location to study anthropogenic burning and the ecological legacies associated with human- and lightning-ignitions. Coastal First Nations have a long history of utilizing fire to promote plant resources and have altered the structure and distribution of forests, woodlands, and meadows through intentional burning in wet and dense temperate rainforests on the Olympic Peninsula, Vancouver Island, and Haida Gwaii (Turner, 1999; Wray, 2009; Turner, 2014). However, most of the records of First Nations burning are after European contact and definitive evidence of pre-contact First Nations burning remains scarce (Lepofsky et al. 2005; Lepofsky and Lertzman, 2008; Turner et al. 2013). I am not aware of any studies that have examined how First Nations historically controlled the spatial and temporal components of fire to affect successional patterns, create patch mosaics of vegetation, and influence the type and abundance of specific species on the Central Coast of British

Columbia. Understanding the historic pattern of both human- and lightning-ignitions is critical to interpreting current species distributions and landscape patterns (Keeley, 2002; Marlon et al. 2008).

Forests on the Central Coast of British Columbia are dominated by western redcedar (*Thuja plicata*), western hemlock (*Tsuga heterophylla*), and Sitka spruce (*Picea sitchensis*). These forests are noted for their late successional characteristics, containing trees with a wide range of ages and sizes and an abundance of coarse woody debris (Meidinger and Pojar, 1991; Banner et al. 2005). The seeming absence of even-aged stands suggests that stand composition and structure has been influenced by processes operating at small spatial scales such as treefall and not disturbance events such as fires (Lertzman et al. 1996; Lertzman et al. 2002; Gavin et al. 2003b; Daniels and Gray, 2006). Although late successional characteristics imply long fire-free intervals at millennial scales, fire size and frequency have never been investigated in these very wet forests. Knowledge of historic fire activity provides a better understanding of fire effects on old growth forest structure, processes of species turnover, and long-term trajectories of forest composition (Gavin et al. 2006; Falk et al. 2011). In addition, the pattern and frequency of historic fire events can help guide current forest management strategies that attempt to simulate the rate, pattern, and recovery of forests from disturbances.

Characterizing historic fire activity in areas with proposed long fire-free intervals is challenging and requires combining longer-term paleoecological and shorter-term dendrochronological techniques to reconstruct fire histories (Marlon et al. 2012). Long-

term fire evidence can be derived from charcoal deposited by ancient fires in soil stratigraphy, which provides spatially explicit records of fire on the landscape (Gavin et al. 2001; Lertzman et al. 2002). More recent fire evidence (<1000 years) is often recorded in fire-scarred trees and post-fire cohorts, which have survived low- and mixed-severity fires (Daniels and Gray, 2006). Archaeological and ethnobotanical datasets can also be utilized to assess relationships between humans and patterns of historic fire activity (Lepofsky and Lertzman, 2008). Using a broadly interdisciplinary approach that combines several techniques across spatial and temporal scales is important to accurately describe and reconstruct historic fire activity in British Columbia coastal temperate rainforests (Lepofsky et al. 2005; Heyerdahl et al. 2012).

1.2 Overview

The goal of this dissertation is to describe 13,000 years of fire history and reconstruct the temporal and spatial patterns of fire in a perhumid temperate rainforest located on the Central Coast of British Columbia. I used a broadly multi-disciplinary approach, combining field surveys, ethnohistorical data, airborne light detection and ranging (LiDAR) techniques, and modelling approaches to reconstruct historic fire activity. Fieldwork was conducted on several small hypermaritime islands in a 2000 km² study area on the outer coast of central British Columbia. The majority of research took place in seven watersheds located on Hecate and Calvert Islands (North 51° 39' Latitude, West 128° 04' Longitude) within the Hakai Lúxvbálís Conservancy. Forest stand and age structure data including indirect fire evidence (basal core sampling of post-fire cohorts) and direct fire evidence (chainsaw wedges or ‘cookies’ of healed-over fire scars) and ages of more than 4000 trees were collected at 50 stands within a stratified random sampling

design representing the range of elevations, aspects, slopes, and dominant vegetation types.

The primary objective of Chapter 2, “13,000 years of fire history derived from soil charcoal in a British Columbia coastal temperate rainforest”, was to describe the long-term (>13,000 years) fire history of Hecate and Calvert Islands. Soil charcoal and fire-scarred tree samples were collected in the study area and used to infer time since the most recent fire event (TSF) and provide an estimate of the frequency of historic fire events. These data were also used to develop a model of inbuilt age, which corrects radiocarbon errors associated with dating soil charcoal derived from long-lived conifers with actual fire dates from living fire-scarred trees. Chapter 2 was published in the journal *Ecosphere* (Hoffman et al. 2016a) as a co-authored article with D. G. Gavin, K. P. Lertzman, D. J. Smith and B. M. Starzomski.

In Chapter 3, “700 years of human-driven and climate-influenced fire activity in a British Columbia coastal temperate rainforest”, the objective was to gain an understanding of the contribution of topography, climate, and humans to the spatial and temporal variation of historic fire activity at the local level and the contribution of large-scale climate oscillations at the regional level. To determine the drivers of spatial and temporal aspects of fire activity, the relationship between the locations of fire events and former First Nation habitation sites was evaluated. Fire events were modelled together with existing tree-ring reconstructions of large-scale climate oscillations, including El Niño-Southern Oscillation, the Pacific Decadal Oscillation and the Arctic Oscillation. Reconstructed fire

years were also compared to periods of antecedent annual and interannual drought using the Palmer Drought Severity Index (PDSI). This research builds on the baseline fire history data established in Chapter 2, but focuses specifically on tree-ring reconstructions of fire events and further examines ignition sources along with human and environmental variables to determine factors influencing historic fire activity. Chapter 3 was published in the journal *Royal Society Open Science* (Hoffman et al. 2016b) and was co-authored by D. G. Gavin and B. M. Starzomski.

In Chapter 4, “Ecological legacies of anthropogenic burning in a British Columbia coastal temperate rain forest”, I examine the ecological legacies of historic fire events in the study area by assessing the clustering patterns of specific trees and herbaceous plants traditionally associated with fire and utilized for food and medicine by coastal First Nations. I include a regional examination of fire occurrence in the 2000 km² study area and develop a weight of evidence approach with multiple hypothesis testing to infer whether fire ignitions were natural (lightning-caused) or human (either accidental or intentional). I use ethnobotanical references together with ecological modelling to assess how, why, and when historic fires occurred. I detected fire events in the study area approximately every 39 years, which is 25 times more likely than previous 1000-year estimates (Daniels and Gray, 2006). This chapter presents data that is consistent with First Nations intentional fire management systems at the regional scale and develops a better understanding of human-fire-environment interactions. Chapter 4 was published in the *Journal of Biogeography* (Hoffman et al. 2017) and was co-authored by K. P. Lertzman and B. M. Starzomski.

In Chapter 5, “Ecological legacies of fire detected using plot-level measurements and LiDAR in an old growth coastal temperate rainforest”, I use field-based ecological sampling together with airborne light detection and ranging data (LiDAR) to conduct a comparative analysis of plot and landscape level measurements. The structure and composition of 30 vegetation plots on Hecate Island with detailed fire histories were compared to 12 vegetation plots with no recent (<1000 years) aboveground fire activity on Calvert Island. This chapter assesses the utility of remote-sensing data to characterize historic fire disturbances in heterogeneous old growth forests with high natural variability. I compare existing reconstructions of stand age, density, and forest plot characteristics to determine the ecological legacies of historic fire events and how fire continues to shape present day vegetation. Chapter 5 is currently in review and is co-authored by A. J. Trant, W. Nijland and B. M. Starzomski.

There are several key findings of this dissertation. First, lightning is rare in the study area and does not explain the spatial or temporal characteristics of historic fire activity. My research strongly suggests that First Nations have a long and complex relationship with fire in the study area and my data are consistent with the hypothesis that First Nations have utilized fire to manage landscapes and specific plant resources in the study area for centuries. Although indigenous burning has not occurred since the end of the 19th century, legacies of historic fire events continue to shape the present day forest structure.

Second, humans (whether accidentally or intentionally) controlled the temporal and spatial aspects of burning and likely utilized favourable climate conditions, such as warm and dry periods associated with large climate oscillations, to promote fire activity. Unlike many other ecosystems in British Columbia, fire activity was not associated with topographical features such as south-facing and steep slopes occurring at high elevations.

Third, the presence of berry patches in repeatedly burned sites and culturally modified and charred western redcedar trees surrounding former First Nations habitation sites suggests long-term human-fire interactions. There has been no fire activity in the study area since the last detected fire on Hecate Island in 1893. This 123-year period coincides with the only time in over 13,000 years that First Nations have not consistently inhabited the study area.

Fourth, although remotely derived images are increasingly available, remote sensing techniques still require ecological field sampling and plot-level measurements to correctly detect low-severity ground fires. Field sampling is important to validate and compare vegetation metrics in heterogeneous landscapes such as British Columbia's old growth coastal temperate rainforests. Additional details on these conclusions and a discussion of areas in need of future research are presented in Chapter 6.

Chapter 2: 13,000 years of fire history derived from soil charcoal in a British Columbia coastal temperate rainforest

Adapted from: K. M. Hoffman^{1,2,†}, D. G. Gavin³, K. P. Lertzman^{2,4}, D. J. Smith⁵, and B. M. Starzomski^{1,2}. (2016) *Ecosphere* 7(7): e01415. DOI:10.1002/ecs2.1415

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2.1 Abstract

Little is known regarding the fire history of perhumid coastal temperate rainforests in the Pacific Northwest (PNW) of North America. While reconstructing historical fire activity typically requires dendrochronological records from fire-scarred trees or stratigraphically preserved lake sediment data, this type of information is virtually non-existent in this region. To describe the long-term fire history of a site on the Central Coast of British Columbia, Canada, I radiocarbon-dated 52 pieces of charcoal. Charcoal ages ranged from 12,670 to 70 yr BP. Fires occurred regularly since 12,670 yr BP, with the exception of a distinct fire-free period at 7500-5500 yr BP. Time Since Fire (TSF) estimates from soil charcoal and fire-scarred trees ranged from 12,670 to 100 yr BP (median = 327 years) and 70% of the sites examined had burned within the past 1000 years. An increase in fire frequency in the late Holocene is consistent with the widely held hypothesis that anthropogenic fires were common across the PNW. I evaluate TSF distributions and discuss the difficulties in assigning actual fire dates from charcoal fragments with large inbuilt ages in a coastal temperate rainforest setting. I determine that a comprehensive approach using soil charcoal and fire scar analyses is necessary to reconstruct general trends in fire activity throughout the Holocene in this region.

2.2 Introduction

Coastal temperate rainforests in the Pacific Northwest (PNW) of North America are believed to have a very low frequency of forest fires, reflecting their location in one of the wettest biomes on Earth (Veblen and Alaback, 1996). This low forest fire activity is typically attributed to the high amounts of precipitation and to the rarity of lightning in these settings (McWethy et al. 2013). Taken together, these attributes decrease the

probability of ignition in coastal forests characterized by large-diameter, moisture-laden fuels (Lertzman et al. 2002; Daniels and Gray, 2006). Although the role of lightning relative to human ignitions in coastal temperate rainforest settings remains unclear, sustained periods of warm and dry conditions are likely required to promote fuel conditions conducive to fire initiation and spread (Bowman et al. 2011). Moreover, despite mounting ethnographic evidence for the human use of fire by First Nations in the PNW over millennia (Walsh et al. 2015), little is known of the influence of these practices on the magnitude and frequency of fires in coastal temperate rainforests. Nor is there a long-term perspective on how fire activity may have been altered as a result of cultural changes related to European contact and 20th century policies of fire suppression (Lepofsky et al. 2005).

With the exception of the boreal forests, most contemporary forest types in Canada experience lightning- and human-caused fires in near equal proportions (Stocks, 1991). Human-caused fires are predominantly associated with the locations of industry, roads, and cities (Pew and Larsen, 2001). Lightning is rare in coastal temperate rainforests in British Columbia (average area burned as a result of lightning ignitions from 1959-1997 is <0.01%) (Stocks et al. 2002). Prior to the 20th century, as they were in other parts of North America (Ryan et al. 2013), First Nations could have been an important ignition source, by synchronizing ignitions with the most fire-conducive period of the year (Marlon et al. 2012; McWethy et al. 2013).

The purpose of this research was to reconstruct the Holocene fire history of a coastal temperate rainforest on the Central Coast of British Columbia, Canada. Reconstructing historical fire activity often involves high-resolution networks of geo-referenced fire-scarred trees (Falk et al. 2011). Although fire-scarred trees can provide annually-resolved records of fire activity, fire-scarred trees are rare in coastal temperate rainforests and do not fully reveal fire recurrence on time scales relevant to the slow dynamics of these forests (Daniels and Gray, 2006). Over longer time periods and at regional scales, fire occurrence may be explored by examination of charcoal deposited in wetland or lake sediments (Hallett et al. 2003; Whitlock et al. 2010). Unfortunately, this method is not spatially explicit and sometimes includes charcoal fragments from distant fires (Gavin et al. 2003a). Using wood charcoal deposited in soils, however, can provide a complementary approach to these other methods for reconstructing long-term fire histories in coastal temperate rainforests (e.g. Lertzman et al. 2002; Gavin et al. 2003b).

Wood charcoal is an inert form of carbon that can persist in soil for millennia (Carcaillet, 2001; Lertzman et al. 2002). However, various processes can disrupt the layering of temporal sequences of charcoal in soils. These include the decomposition or accumulation of organic material, bioturbation by roots or animals, tip up mounds, or more recent fires that consume humus and erase previous records of soil charcoal (Gavin et al. 2003b). Although microscopic charcoal may be transported significant distances by wind and water, macroscopic charcoal (>5 mm) found in depositional hollows within standing forests likely originated within a few tens of metres (Clark and Patterson, 1997). Thus, radiocarbon ages assigned to the latter can be used to date a fire that occurred at, or

very near, the site of deposition (Higuera et al. 2007; Payette et al. 2012). As the radiocarbon ages assigned to the charcoal reflect the date of carbon dioxide fixation rather than fire, radiocarbon dates may pre-date the actual fire event. Addressing the source of this ‘inbuilt age’ is critical in temperate rainforests where fires may burn old woody debris that may exceed ages of 1000 years (Gavin, 2001).

2.3 Methods

2.3.1 Study area

The research was completed within the Hakai Lúxvbálís Conservancy on Calvert and Hecate islands ($51^{\circ} 38' N$ latitude, $128^{\circ} 05' W$ longitude), a 160 km^2 island group located on the Central Coast of British Columbia, Canada (Fig. 2.1). The study area is situated in the very wet hypermaritime subvariant of the Coastal Western Hemlock zone of British Columbia’s biogeoclimatic classification system (Meidinger and Pojar, 1991). This region receives some of the highest annual rainfall in North America (~4000 mm) and is characterized by subdued and rugged topography, moderate year-round temperatures (~ 7° C), and low rates of evapotranspiration (Banner et al. 1993). There is no history of industrial logging in the study area.

Ancient human land use and settlement patterns have varied spatially and temporally throughout the study area and archaeological evidence suggests human presence in the region for at least 13,000 years (McLaren et al. 2014; McLaren et al. 2015). The study area contains three formerly occupied habitation sites with extensive shell middens (>3 m depth) and an abundance of near shore features including fish traps, clam gardens, and

root gardens that were continuously utilised throughout the Holocene (Stafford et al. 2009; McLaren et al. 2014). There are no records or evidence of fire in the study area in the 20th century, and few oral histories of pre-contact anthropogenic fire. However, controlled burning by First Nations, referring to the purposeful burning of vegetation for plant and animal management, was used in nearby and analogous ecosystems (Turner, 1999; Trusler and Johnson, 2008).

The bedrock geology of Calvert and Hecate islands is almost entirely of igneous origin, comprised of a mix of quartz diorite and granodiorite (Roddick, 1996). Compared to the adjacent mainland, there are very few glacial features or glaciogenic deposits. Upland mineral soils are generally nutrient-poor Ferro-Humic Podzols that are strongly influenced by the underlying bedrock geology (Banner et al. 1993). Shallow, imperfectly drained organic soils accumulate folic material (forest material, branches, roots) and are classified as Folisols. Poorly-drained soils with high rates of peat accumulation are Typic or Fibric Mesisols (Valentine et al. 1979). Because of the subdued and low-lying terrain, there is little evidence of down slope movement of soils.

Four main vegetation types defined by dominant species and closely associated with terrain attributes and soils are found in the study area: zonal forests, bog forests, bog woodlands, and blanket bogs (Banner et al. 2005). Productive zonal forests are associated with hill slopes and riparian areas, and are comprised of moderately-drained Folisols where thin organic horizons developed on gravelly colluvium over bedrock (Banner et al. 1993). These forests are dominated by large diameter western redcedar (*Thuja plicata*

Donn ex D. Don.) and western hemlock (*Tsuga heterophylla* [Raf.] Sarg.) with lesser amounts of yellow-cedar (*Cupressus nootkatensis* [D.Don] Farjon and Harder) and Sitka spruce (*Picea sitchensis* [Bong.] Carr.) (Meidinger and Pojar 1991).

Bog forests have similar soils to zonal forests and are found almost exclusively on hillslopes. They exhibit stunted growth forms and are dominated by western redcedar, yellow-cedar, western hemlock, and shore pine (*Pinus contorta* var. *contorta* Douglas ex Loudon) (Klinka et al. 1996). Bog woodlands are found in patchy mosaics of forested and unforested sites in subdued or rolling terrain, and are dominated by western redcedar, yellow-cedar, and shore pine with lesser amounts of mountain hemlock (*Tsuga mertensiana* Bong.) (Klinka et al. 1996). Blanket bogs with both minerotrophic and ombrotrophic characteristics occur in sparsely forested wetland areas (Banner et al. 2005). Minerotrophic bogs are comprised of vascular plants and sedges with shallow, nutrient-poor Ferro-Humic Podzols forming over bedrock while ombrotrophic bogs are dominated by sphagnum species and contain organically-enriched, nutrient poor soils (Mackenzie and Banner, 2001).

Forested sites in the study area are characterized by living and dead fuels in various stages of decay (Feller, 2003). The majority of surface fuels are comprised of moisture-rich, large-diameter coarse woody debris, which is often covered by a dense canopy and moss on the forest floor (Daniels and Gray, 2006). Surface fuels have low flammability compared to dead, standing fuels that can be significantly drier and may be an important fuel source in hypermaritime forests (Daniels, 2003; Dorner and Wong, 2003). Fire

events are thought to be most common in the late summer when high-pressure ridges form on the coast of British Columbia, promoting sustained warm and dry weather conditions (Lertzman et al. 2002; Macias Fauria and Johnson, 2006).

Thirty-nine sites stratified by the proportional representation of four vegetation types: zonal forest (15%), bog forest (26%), bog woodland (31%), and blanket bog (28%) were sampled in the 300-ha study area (Fig. 2.1). One soil pit was dug at each site and multiple pieces of soil charcoal were sampled from each soil profile. The most intact fire-scarred trees were sampled at each site where they were present. The study area was selected to overlap (75% of sites) a 287-ha fire dated with fire-scarred trees to 1893 AD. To compare the effects of time since the most recent fire event (TSF) on different fuel and vegetation characteristics apparent in the four vegetation types, I sampled the remaining 25% of sites with the same methods in forests with no recent aboveground fire evidence on Hecate and Calvert islands.

2.3.2 Charcoal sampling and analysis

Charcoal layered in discrete stratigraphy was apparent in soils throughout the study area, suggesting a long history of fire activity. Thirty-nine soil pits (one per sampling site) 20-150 cm in depth and 100-500 m apart were selectively sampled in concavities ($1-2 \text{ m}^2$) found on locally level terrain. To minimize erosional loss of charcoal, sites were chosen in locations not affected by downslope movement (Carcaillet, 2001; Lertzman et al. 2002). Macroscopic charcoal was sampled in the field and microscopic charcoal was collected in stratigraphy (bulk soil) and sampled under a dissecting microscope in the

laboratory. Detailed descriptions of soil charcoal stratigraphy and soil classifications are presented in Appendix A1: Table A1.1.

Samples were washed with deionized water in the laboratory, and charcoal fragments were sieved carefully through a 0.5 mm mesh. For charcoal pieces >5 mm, we used taphonomic keys developed by Mustaphi and Pisaric (2014) to obtain information about the species, morphology, and wood type. For these samples, radiocarbon dates were obtained from a single piece of charcoal. If charcoal fragments were <2 mm or did not contain enough mass for radiocarbon dating, two or three pieces of charred wood were combined from the same depth in the soil for radiocarbon dating (35% of samples). Multiple charcoal fragments deposited in the same stratigraphy likely formed from the same fire event (Lertzman et al. 2002). Gavin et al. (2003b) tested this hypothesis and found that 40 radiocarbon dates obtained from 2-3 microscopic charcoal fragments in the same stratigraphic layer yielded nearly identical radiocarbon dates. Although the total number of samples submitted for radiocarbon dating was constrained by the funds available, charcoal fragments were selected based on the quality, depth, and location of the sample in continuous stratigraphy and to best represent all vegetation and soil types.

Samples were radiocarbon dated (^{14}C) using accelerator mass spectrometry (AMS) at Keck Carbon (University of California-Irvine, United States). Radiocarbon dates were calibrated to calendar years with the INTCAL13 calibration curve (Rev 7.0.4 program) (Reimer et al. 2013). All dates are reported as calibrated years before present (yr BP [1950]) using the median of the probability distribution of each radiocarbon date. We

submitted 10 to 20 charcoal samples at a time for dating and the depth and age results obtained were used to adapt the strategy for further submissions.

I used Gavin's (2001) methodology and inbuilt age equation to create an inbuilt age model specific to the study area. When possible, I selectively sampled charred fragments of bark, needles, branches or outer wood to decrease inbuilt age and to provide a more accurate age for radiocarbon dating. Fourteen fire scars dating the most recent fire event were used to calculate the inbuilt age distribution and the adjusted error (Table 2.1). I assigned radiocarbon dates to fire scars when a series of two or more fire scars could be matched to a stratigraphic pattern of soil charcoal. The adjusted error was calculated as a weighted moving average of the probability distribution of the calibrated radiocarbon age using the inbuilt age distribution as a set of weights (Gavin, 2001). Inbuilt age adjustments were added to the 2-sigma radiometric error of each radiocarbon date.

2.3.3 Fire scar sampling and analysis

Sampling fire scars in the study area was necessary to create estimates of inbuilt age, record recent fire activity for TSF estimates, and to validate selections of soil charcoal for radiocarbon dating. I removed wedges with a chainsaw from the bases of living, fire-scared trees within the one-hectare area surrounding each study site (Arno and Sneck, 1977). In areas with a high density of intact fire scars, the most sound or the oldest of the trees (determined after coring and measuring) were collected (Arno and Sneck, 1977). To validate dating of fire scars, I obtained two 5 mm increment cores ~1.3 m from the ground from 90 trees (three species-specific chronologies) in forest stands outside of the 1893 fire perimeter with no aboveground fire evidence (Johnson and Gutsell, 1994).

In the laboratory, fire scar wedges and increment cores were processed using standard dendrochronological techniques (Stokes and Smiley, 1968). Samples were visually crossdated and then statistically verified using the computer program COFECHA (Grissino-Mayer, 2001). I combined fire scar dates from the study area into a composite record and identified fire years as those in which at least two trees had fire scars. I determined the exact calendar year of fire occurrence by crossdating fire scars to species-specific tree-ring chronologies (Johnson and Gutsell, 1994).

2.3.4 Interpretation of Time Since Fire (TSF)

TSF dates were acquired from fire scars or from the youngest calibrated radiocarbon date sampled at each site (Gavin et al. 2003b). Estimates for TSF were calculated and compared to lower and upper quantiles between vegetation types and between the two study areas (sites located within and outside of the 1893 fire perimeter) with randomization tests in the WRS2 package in R statistical software (R Development Core Team, 2016). The rationale for comparing TSF within the 1893 fire perimeter to outside of the 1893 fire perimeter was due to the close proximity of three former habitation sites within the 1893 fire perimeter which may include targeted resource sites that were repeatedly burned over millennia. Because I do not know the historic composition and distribution of vegetation in the study area throughout the Holocene I also assessed the relationship between terrain attributes derived from a digital elevation model (DEM) with TSF distributions.

2.4 Results

2.4.1 Radiocarbon dating and soil charcoal stratigraphy

In total, 97 soil charcoal samples were obtained. Thirty-five samples were deemed replicates with regards to soil type and stratigraphy. Ten samples that didn't fit my ecological criteria (see methods) were deprioritized due to budget allocation. Fifty-two samples from 39 sites were submitted for radiocarbon dating. The radiocarbon ages assigned to the charcoal samples ranged from 12,670 to 70 yr BP. Radiocarbon age, calibrated dates, and median calibrated probabilities with associated errors are presented in Appendix A1: Table A1.2. One radiocarbon date was rejected because it was based on a sample with low carbon (<1 mg). All radiocarbon dates were physically stratified by depth, with no age reversals, and charcoal fragments in organic soil types were consistently younger than charcoal in mineral soil types (Gavin, 2003) (Appendix A1: Table A1.1). This finding confirmed that the uppermost charcoal layer accurately represented TSF distributions (Fig. 2.4a) (Gavin et al. 2003b).

2.4.2 Fire scars and soil charcoal

Crossdating of 99 fire scars from 45 wedges revealed 16 distinct fire events during the period 1376-1893 AD. The majority of wedges were from western redcedar (73%) and yellow-cedar (20%), but I also sampled a small portion of shore pine (7%). Fourteen sampling sites contained multiple fire events recorded on fire-scarred trees and I was able to link the five most recent fires in the study area with soil charcoal buried in stratigraphy. The association between fire scars and soil charcoal was based on the expectation, supported by my investigations (Appendix A1: Table A1.1), that surficial

charcoal deposited near the soil surface (<5 cm) originated from the most recent fire event (Lertzman et al. 2002; Gavin et al. 2003b). My criteria for linking fire scars to soil charcoal to build the inbuilt age model included selecting depressions or hollows where charcoal was likely to accumulate, avoiding sampling when there was evidence of past soil disturbances (e.g. tip up mounds, mass wasting), and prioritizing charred bark, twigs, and macroscopic charcoal deposited in stratigraphy. This sampling procedure was critical to building accurate inbuilt age adjustments for the broader study area (Table 2.1).

2.4.3 Time Since Fire (TSF)

TSF was acquired from fire scars (54% of sites) or from the youngest calibrated radiocarbon date sampled at each site (46% of sites) (Appendix A1: Table A1.3). Several sites had shallow soils (median = 15 cm depth) with one charcoal date (67% of sites). Sites with >2 charcoal dates consistently demonstrated increased age with depth (Appendix A1: Table A1.1). Therefore, I am confident that using one radiocarbon date to estimate TSF had a high likelihood of correctly identifying the most recent fire (Gavin et al. 2003b).

Excluding dates from the 1893 AD fire, TSF estimates ranged from 12,670 to 100 yr BP (median = 327 years) and the median of the charcoal-based TSF was 2030 yr BP (Fig. 2.2; Appendix A1: Table A1.3). There was no significant difference ($P = 0.54$) in TSF between the four vegetation types in the study area (Fig. 2.2a), nor between TSF and terrain attributes derived from the DEM (Appendix A2: Fig. A2.1). Because there were no fire scars present outside of the 1893 fire perimeter, I compared TSF with the youngest calibrated radiocarbon dates acquired from each site (not including fire scars or

soil charcoal from the 1893 fire). The removal of charcoal and fire scar dates derived from the 1893 fire revealed that sites sampled within the 1893 fire perimeter experienced more fire events and had significantly shorter TSF than elsewhere in the study area ($P = 0.003$) (Fig. 2.2b). Examinations of landform controls including aspect, slope, elevation, and derived terrain attributes did not explain the spatial pattern of TSF estimates within or outside of the 1893 fire perimeter. The summed probabilities of fire occurrence and TSF for each vegetation type are displayed in Appendix A2: Fig. A2.2 and a photograph of the four vegetation types in the study area is presented in Appendix A2: Fig. A2.3.

2.4.4 Inbuilt age

Inbuilt age adjustments ranged from 0-549 years and one sample required no adjustment (Table 2.1; Appendix A2: Fig. A2.4). Most radiocarbon dates had errors >160 years (25th percentile), >360 years (75th percentile), and <471 years (95th percentile). The greatest observed inbuilt age was 549 years. My inbuilt age model determined that in this vegetation type and with soil charcoal data, fires separated by <549 years cannot be considered separate events. I added the 549-year maximum inbuilt age to the 2-sigma radiometric error of each radiocarbon date to determine if fire events were distinct or overlapping. Using the maximum inbuilt age I recorded 16 distinct fire events with soil charcoal and seven additional fire events (<1000 yr BP) with fire-scared trees. Fire scar and soil charcoal derived TSF estimates are presented in Fig. 2.4a along with the adjusted inbuilt age distribution of every radiocarbon date sampled in the study area.

2.5 Discussion

My analyses of soil charcoal and fire scars from four vegetation types confirm that fire has been a feature of this landscape for millennia. The cumulative probability

distributions of the 51 radiocarbon dates demonstrate lengthy periods with a higher frequency of recurring fires from 12,700 to 7500 and from 5500 to 70 yr BP, with a distinct fire-free interval between ~7500 to 5500 yr BP (Fig. 2.3a). Because 23% of my samples recorded fires in the 12,700-7500 yr BP period, I can be confident that my interpretation of temporal trends in fire activity is not limited by a decrease in the number of samples through time (Gavin, 2003). There is an abrupt increase in fire evidence from ~3200 yr BP to 0 (representing the 1893 AD fire) across all sites (Fig. 2.3b). This increase could represent ignitions by First Nations and cultural shifts associated with increasing populations and a growing demand for plant resources surrounding village sites (Deur, 2002; Lepofsky et al. 2005).

Although forest fires are often linked to large scale climate variability (Macias Fauria and Johnson, 2006), an increase in the summed probability distribution of radiocarbon dates from ~3200 yr BP to 1893 AD is apparent in my data despite a transition to a cooler and wetter climate in the late Holocene (Brown and Hebda, 2002; McLaren et al. 2015).

When I compare these findings to those of a related study of soil charcoal in the Clayoquot Valley, ~300 km to the south on Vancouver Island, synchronous increases in fire activity appear in both records at 11,000-9000 and 3200 yr BP, as well as decreases between ~7500-5500 yr BP (Gavin et al. 2003b; Fig. 2.4b). Declines in fire activity in the mid-Holocene likely correspond to a complex mix of cultural and environmental shifts occurring in the region (Brown and Hebda, 2002). These include rising sea levels (McLaren et al. 2015), decreased summer insolation and an increase in the strength of the

Aleutian low (Bartlein et al. 1998), and the arrival of western hemlock and culturally significant western redcedar (Hebda and Mathewes, 1984).

The composition of the forests that burned over the past 13,000 years in the study area remains largely unknown. While the species of 54% of the radiocarbon samples was determined, I was unable to determine species assemblages in the early Holocene (Appendix A1: Table A1.2). Western redcedar, a rot-resistant and long-lived species that has been present in the study area for at least 6000 years dominated the remains of charcoal in the mid- and late Holocene (Pojar and MacKinnon, 1994; Brown and Hebda, 2002). Increased dating and identification of charcoal deposited in lower soil stratigraphy could reveal more accurate estimates of when western redcedar arrived in the region. I attempted to select short-lived species such as shore pine to decrease the inbuilt age of radiocarbon dates, but found that shore pine comprised only 13% of samples (Appendix A1: Table A1.2). This confirms that attempting to decrease inbuilt age estimates through selective sampling in this vegetation type may not be feasible.

Estimates of TSF varied from just over 100 years to several millennia across all vegetation types in the study area (Fig. 2.2). An abrupt increase in the summed probability of fire occurrence is clear in the late Holocene, with 70% of sites recording TSF of <1000 years (Appendix A1: Table A1.3). Broadly similar findings were reported by Lertzman et al. (2002) and Gavin et al. (2003a) in the Clayoquot Valley, where, despite 20% of the landscape not having burned for >6000 years, the TSF at 45% of sites examined was <1000 yr BP (Fig. 2.4b). Notably, in my study, 10% of sites examined had

not burned in over 7000 years and one blanket bog appears not to have burned since 12,672 yr BP. For both the Clayoquot Valley case and my own, though there has been fire present on the landscape over time, it is patchily distributed in space and time and the TSF distributions have an exceedingly long tail.

While TSF is often related to landscape features such as slope position, aspect, and elevation and also often varies with regard to vegetation type (Lertzman et al. 2002; Gavin et al. 2003b; Berg et al. 2006), there was no obvious such spatial pattern of fire occurrence in the study area. My results show no difference in TSF between vegetation types despite large differences in fuel availability and fuel moisture (Fig. 2.2a; Appendix A1: Table A1.3), nor were there any significant differences across terrain types (Appendix A2: Fig. A2.1). There were significant differences in TSF within and outside of the 1893 fire perimeter, suggesting the 1893 fire was located in a portion of the landscape more frequently experiencing fire (Fig. 2.2b). I suggest this pattern may reflect differences in available ignition sources due to the proximity of three continuously inhabited sites within the 1893 fire perimeter compared to one seasonally inhabited site outside of the 1893 fire perimeter (McLaren et al. 2015). If this is true, the fire cycle is more influenced by human activities than site-by-site inherent ecological or topographic characteristics.

To avoid misidentifying fire events, I only considered soil charcoal radiocarbon dates as a proxy for fire events when they were separated by at least the greatest observed inbuilt age (549 years) (Gavin, 2001) (Appendix A2: Fig. A2.4). This approach provided a

minimum estimate of the number of fire events in the study area throughout the Holocene. Although inbuilt age errors are likely significantly smaller in drier, fire-prone ecosystems, not adjusting for inbuilt age and solely identifying fire events using the assigned radiometric error would have, probably falsely, incurred a 93% increase in fire occurrence in the study area (29 compared to 15 fire events recorded with soil charcoal). I opted for a cautious approach, utilizing an inbuilt age model, to identify fire events and assess fire activity in forests with long-lived conifer species.

The marked increase in the summed relative probability of charcoal ages after ~3200 yr BP ends abruptly with the 287-ha fire in 1893 AD. Although First Nations continue to use the area for seasonal resource gathering, habitation sites have not been occupied since the late 19th century (McLaren et al. 2015). The 13,000-year pattern of coexistence of fire and people in the study area contrasts with the relative absence of both fire and people from the study area in the 20th century. This conjunction of a long history of human habitation with the presence of fire in a very wet, hypermaritime landscape is an important area for future study.

This is the first reconstruction of fire activity on the Central Coast of British Columbia and within hypermaritime forests in British Columbia and Alaska. I recorded 16 separate fire events in the study area with a network of living, fire-scarred trees from 1376-1893 AD. When soil charcoal was matched to fire scars in living trees, I found that the uppermost layer of charcoal dated the most recent fire event in both organic and mineral soils and that the most abundant charcoal was found at the interface of organic and

mineral horizons (Appendix A2: Fig. A2.5; Fig. A2.6). I did not find any examples where soil charcoal was present and had an age <500 yr BP with no aboveground evidence of fire activity. Because fire scars form on trees under specific physical and biological conditions, I can infer that portions of recent fire events were low and mixed in severity (Falk et al. 2011). Like in other systems (Heyerdahl et al. 2012), in this region, it is important to combine different types of evidence to reconstruct patterns of long-term fire activity. In my case, combining soil charcoal and fire scar analyses allowed me to describe millennia of fire activity, which I couldn't from either source alone.

Table 2.1 Fourteen radiocarbon dates from the uppermost soil layers were corrected with known fire dates from fire-scarred trees in the study area. Inbuilt age was calculated as the difference between the fire year and the median age calibrated AD intercept.

Site	Method	Year of fire (AD)	¹⁴ C Age BP	Median age cal AD (2-sigma error)	Inbuilt age
Hecate Island	Fire scar	1893	290 ± 20	1558 (1520- 1653)	335
Hecate Island	Fire scar	1893	120 ± 15	1842 (1684- 1953)	51
Hecate Island	Fire scar	1893	115 ± 20	1840 (1683- 1953)	53
Hecate Island	Fire scar	1719	375 ± 15	1488 (1451- 1618)	231
Hecate Island	Fire scar	1893	245 ± 15	1656 (1644- 1795)	357
Hecate Island	Fire scar	1893	595 ± 15	1344 (1306- 1404)	549
Hecate Island	Fire scar	1744	290 ± 15	1554 (1522- 1651)	190
Hecate Island	Fire scar	1893	205 ± 15	1784 (1653- 1950)	109
Hecate Island	Fire scar	1656	375 ± 40	1495 (1449- 1585)	161

Hecate Island	Fire scar	1778	200 ± 15	1779 (1654-1951)	0
Hecate Island	Fire scar	1893	510 ± 15	1422 (1409-1436)	471
Hecate Island	Fire scar	1778	570 ± 20	1346 (1313-1416)	432
Hecate Island	Fire scar	1893	250 ± 20	1654 (1643-1793)	239
Hecate Island	Fire scar	1893	280 ± 15	1591 (1521-1662)	302

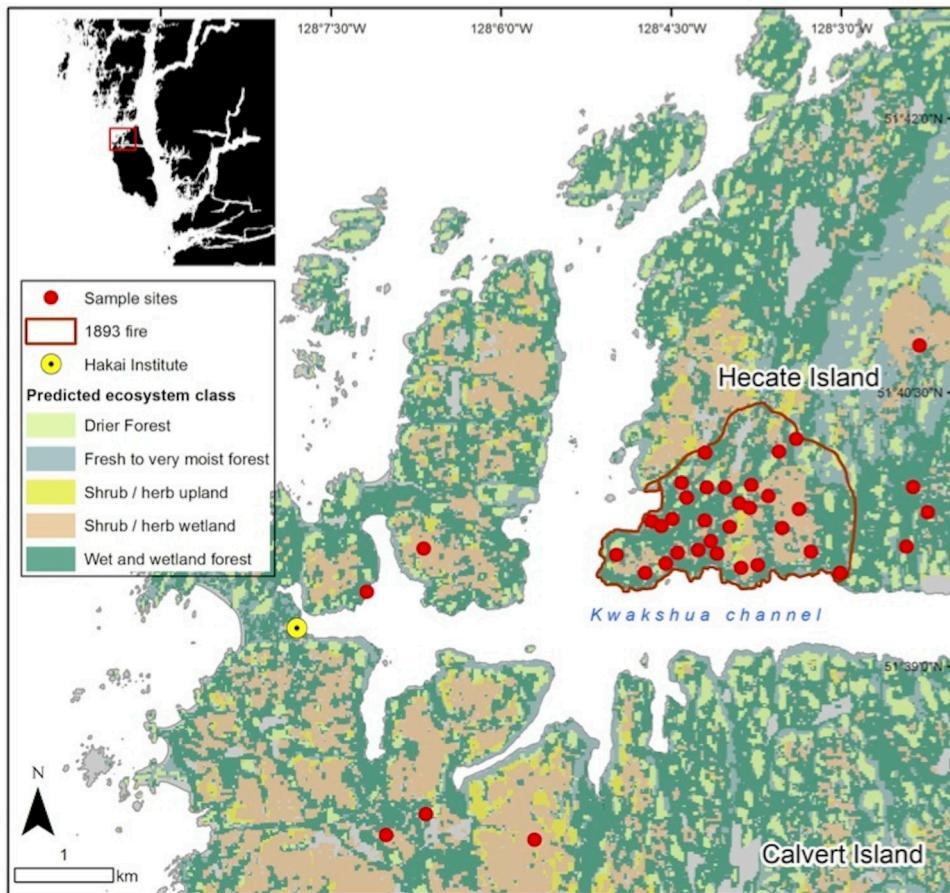
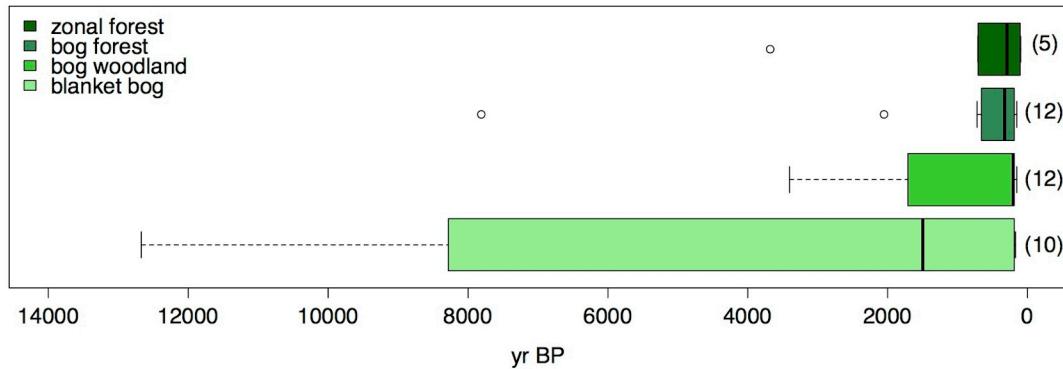


Figure 2.1 The location of sample sites (red circles) on Calvert and Hecate islands. The inset provides the location of the study site on the Central Coast of British Columbia, Canada.

a) Comparison of TSF across vegetation types



b) Comparison of TSF across site types

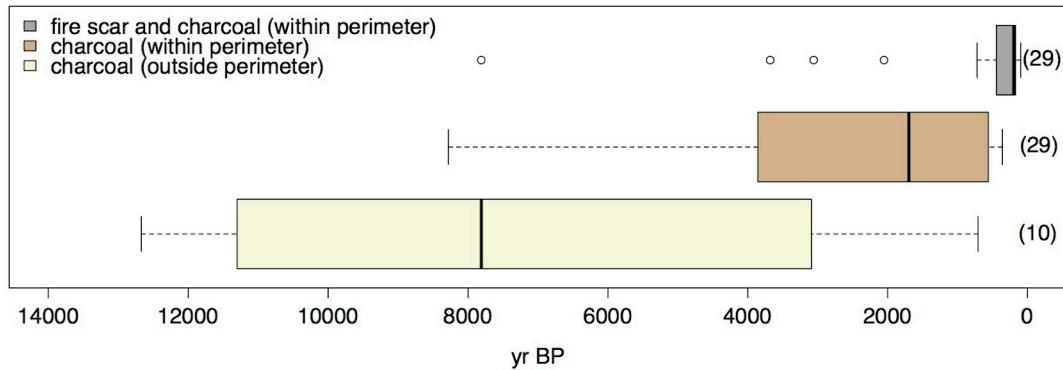


Figure 2.2 a) Box and whisker plots of Time Since Fire (TSF) for 39 sites in the study area. Sample sizes are in brackets to the right of box and whisker plots. Boxes represent the second and third quartile ranges and the centerline is the median. Circles represent outliers. TSF was not statistically different when comparing vegetation types ($P = 0.54$).

b) Box and whisker plots of TSF distributions within and outside of the 1893 fire perimeter (fire scars and soil charcoal dating the 1893 fire were removed from this analysis). Sites outside of the 1893 perimeter had significantly longer TSF when compared to sites within the 1893 fire perimeter ($P = 0.003$). Sites that included fire scars

in TSF distributions had significantly shorter TSF when compared to sites within the 1893 fire perimeter that had been dated with soil charcoal only ($P = 0.001$).

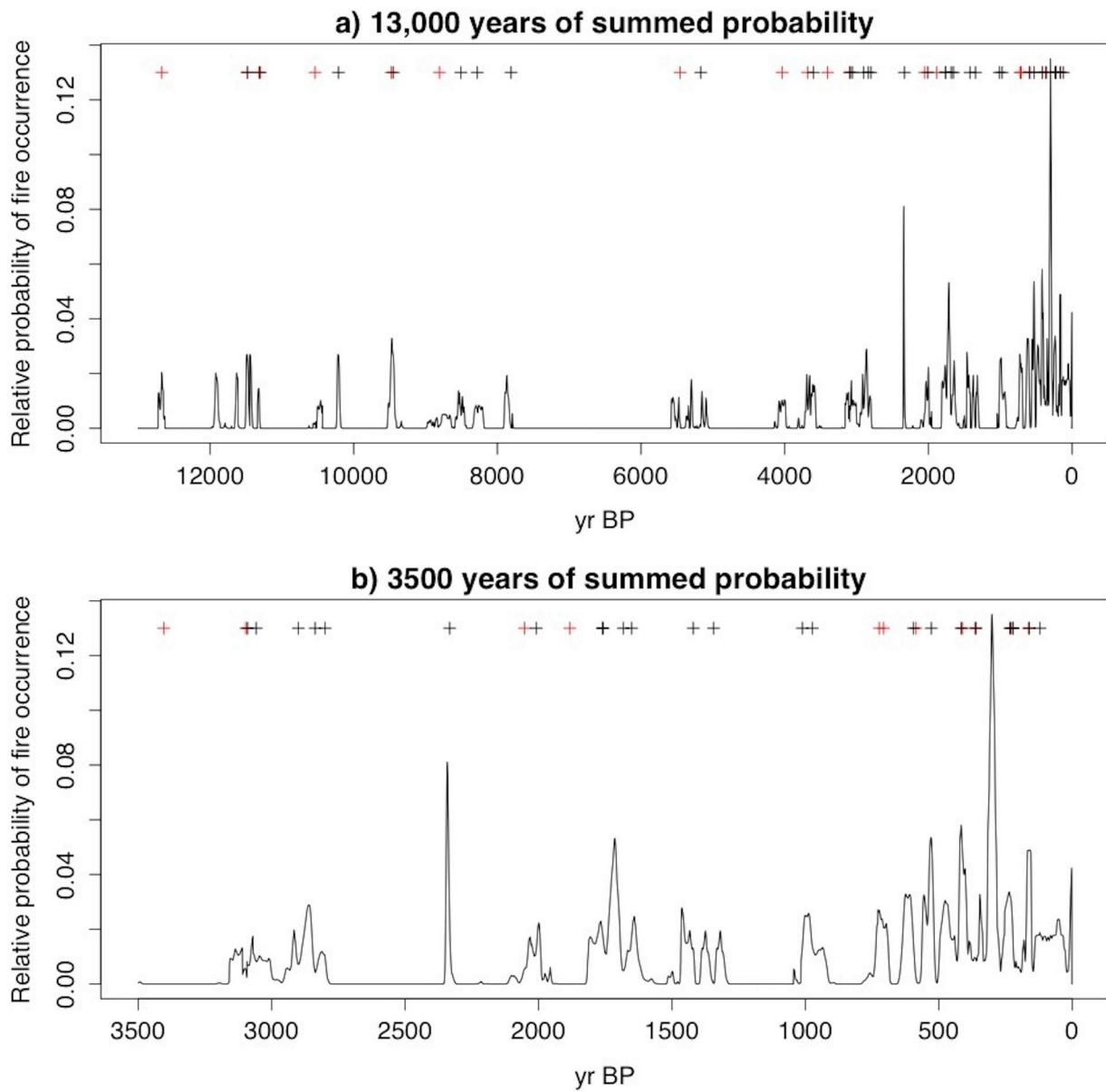


Figure 2.3 a) The entire record of soil charcoal based fire dates. Crosses (+) display the median age of each radiocarbon date. Red crosses indicate distinct fire events and black crosses indicate unconfirmed fire events due to overlapping probability distributions after

inbuilt age adjustments. The relative probability of fire occurrence is demonstrated by totalling the summed probability distributions of 51 radiocarbon dates across the study area from 13,000 yr BP to 0 (representing the 1893 AD fire). b) A subset of 34 dates, showing the most recent 3500 years at higher resolution. An increase in the probability of fire occurrence is recorded from ~3200 yr BP to 0 (representing the 1893 AD fire). There is no fire evidence in the study area after 1893 AD.

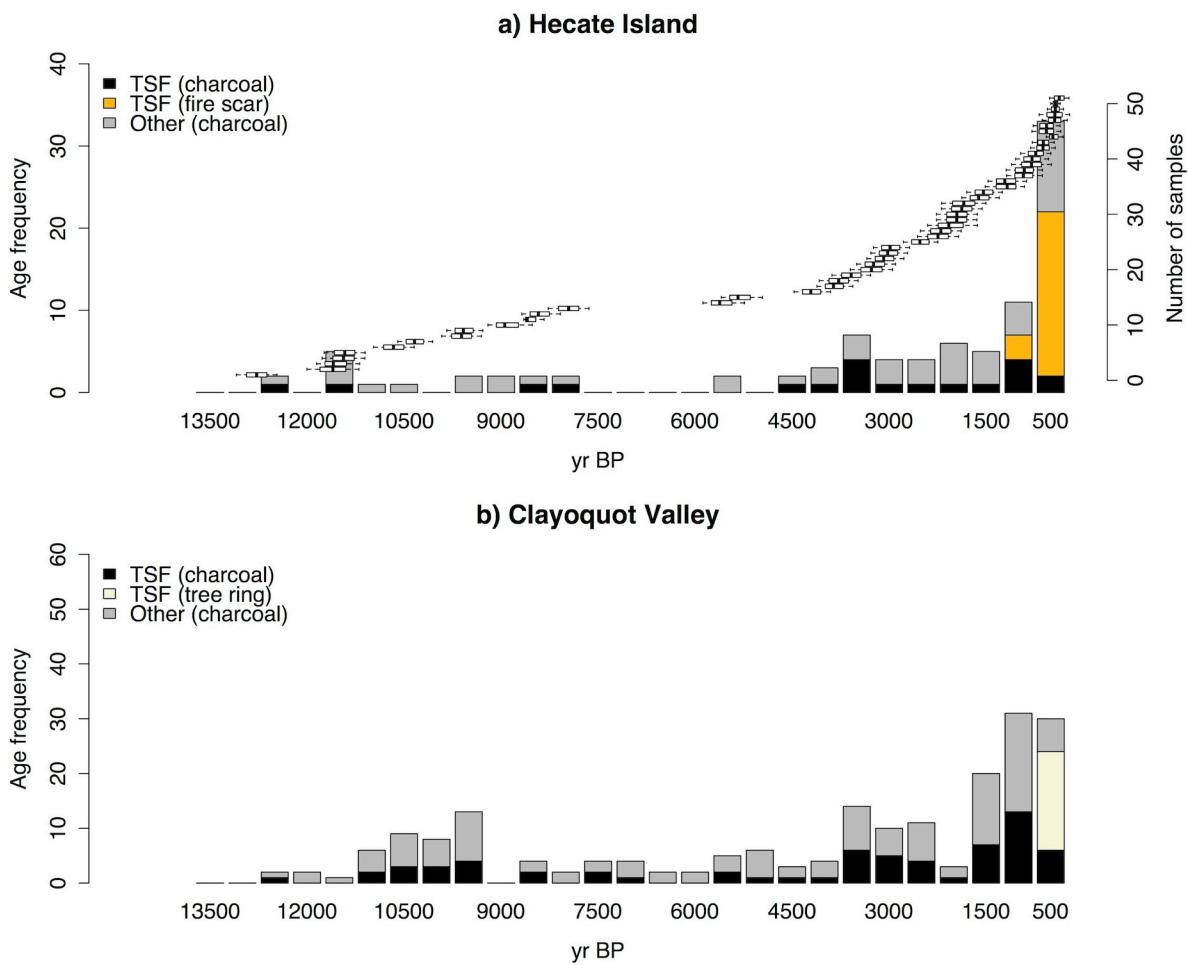


Figure 2.4 a) Age frequency of the calibrated median probability of 51 soil charcoal dates on Hecate Island from 13,000 years to present. Time Since Fire (TSF) estimates were derived from the youngest calibrated radiocarbon dates at 19 sites. The remaining

TSF dates were dated with fire scars at 20 sites. “Other (charcoal)” is the remaining soil charcoal dated but not included in TSF estimates. The small box and whisker plots are radiocarbon dates with adjustments of inbuilt age (additional to the 2-sigma radiometric and calibrated error) for each sample and the y-axis (right side) shows the number of samples distributed through time. b) For the purpose of regional comparison, TSF and age frequency of the calibrated median probability of 113 soil charcoal dates including radiocarbon samples used in inbuilt age adjustments from Clayoquot Valley over 13,000 years. TSF estimates were derived from the youngest calibrated radiocarbon date from 65 sites. The remaining TSF estimates were dated from forest stands (tree rings) at sites that established after fire events (Gavin et al. 2003b). Note the different y-axes.

Chapter 3: 700 years of human-driven and climate-influenced fire activity in a British Columbia coastal temperate rainforest

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3.1 Abstract

While wildland fire is globally most common at the savanna-grassland ecotone, there is little evidence of fire in coastal temperate rainforests. I reconstructed fire activity with a 700 year fire history derived from fire scars and stand establishment from 30 sites in a very wet (>4000 mm average annual precipitation) temperate rainforest in coastal British Columbia, Canada. Drought and warmer temperatures in the year prior were positively associated with fire events though there was little coherence of climate indices on the years of fires. At the decadal scale, fires were more likely to occur after positive El Niño Southern Oscillation (ENSO) and Pacific Decadal Oscillation (PDO) phases and exhibited 30-year periods of synchrony with the negative phase of the Arctic Oscillation (AO). Fire frequency was significantly inversely correlated with the distance from former First Nations habitation sites and fires ceased following cultural disorganization caused by disease and other European impacts in the late 19th century. First Nations were likely the primary ignition source in this and many coastal temperate rainforest settings. These data are directly relevant to contemporary forest management and discredit the myth of coastal temperate rainforests as pristine landscapes.

3.2 Introduction

Paleofire studies have found evidence that both humans and climate influence fire cycles, but it is often difficult to discern their relative importance and the drivers of fire and its ecosystem effects remain poorly understood (Bowman et al. 2011; McWethy et al. 2013). Humans affect fire activity by controlling ignition and suppression, and through altering vegetation and the availability of fuel types by practices such as land clearing and habitat modification (Foster et al. 2002; Guyette et al. 2002; Bowman et al. 2011; Whitman et al.

2015). Climate drives fire activity through interactions with local-scale, bottom-up controls such as available fuels and topography to affect fuel drying and fire spread and by influencing rates of natural ignitions (Lertzman and Fall, 1998; Heyerdahl et al. 2002; McWethy et al. 2013). In the Pacific Northwest (PNW) of North America it is often difficult to attribute historic drivers of fire activity and causes of fire ignitions due to both the abundance of natural ignitions from lightning and the impact of logging, grazing, and fire suppression that decreased fire occurrence soon after the loss of First Nations fire management (Guyette et al. 2002; Bowman et al. 2011; McWethy et al. 2013).

First Nations used fire for diverse purposes such as clearing land, creating fodder for game, and increasing the productivity of specific plants for food and medicine (Ryan et al. 2013; Turner, 2014). The cultural importance of fire has been mostly examined in the savanna-grassland ecotone (Guyette et al. 2002; Bowman et al. 2011; Ryan et al. 2013) and in fire-adapted forests dominated by Ponderosa pine (*Pinus ponderosa*) and Garry oak (*Quercus garryana*) (Veblen et al. 2000; Lepofsky et al. 2005; Gedalof et al. 2006). There has been very little documented evidence of historic fire activity in wet and dense coastal temperate rainforests located in the PNW and previous disturbance studies have asserted that both human- and lightning-caused fires are infrequent and become progressively rarer in high-latitude regions ($> 50^{\circ}$ N) (Agee, 1993; Wong et al. 2003; Daniels and Gray, 2006; Pearson, 2010). Low fire activity is assumed due to the presence of old growth forests (Daniels and Gray, 2006), the lack of fire adaptations in dominant conifer species (Agee, 1993), the decreased flammability of large-diameter, moisture-laden surface fuels (Guyette et al. 2002; McWethy et al. 2013), and the absence of 20th

century fire activity (Meyn et al. 2010). There have been no reconstructions of historic fire activity with fire-scarred trees in these wet and dense forests. Fires are hypothesized to be large, stand-replacing events occurring during unusually dry years when low fuel moisture promotes rapid fire spread (Agee, 1993; Bowman et al. 2011). A handful of studies in coastal temperate rainforests of British Columbia have described temporally coarse reconstructions of Holocene fire activity with charcoal stratigraphy in lake sediments (Brown and Hebda, 2002) and radiocarbon dating of soil charcoal (Lertzman et al. 2002; Gavin et al. 2003; Hoffman et al. 2016a). These studies depict patchy fires during the last few centuries embedded within landscapes experiencing long (sometimes >1000 year) fire intervals as well as widespread increases in fire activity in the mid- and late Holocene (Lertzman et al. 2002; Gavin et al. 2003; Hoffman et al. 2016a) that may be related to increases in the use of fire by First Nations (Hoffman et al. 2016a).

Wildfire occurrence is also influenced by variability in modes of sea surface temperatures (SSTs) such as the El Niño-Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO) that influence winter temperature and precipitation patterns and indirectly affect summer moisture availability (Hessl et al. 2004; Gedalof et al. 2005; Kitzberger et al. 2007). The ENSO is characterized by oscillating phases of warmer (El Niño) and cooler (La Niña) than average winter SSTs in the equatorial Pacific at interannual (2-7 year) frequencies (D'Arrigo et al. 2005). The ENSO affects climate and fire in distinct ways depending on ocean-atmosphere conditions and their associated circulation patterns (Kitzberger et al. 2007). The warm (El Niño) phase produces warmer and drier winter and spring conditions in the PNW, which enhances wildfire activity in

summer months (Gedalof et al. 2005; Kitzberger et al. 2007). The PDO is characterized by variations in SSTs in the North Pacific and positive (negative) PDO values produce relatively similar climate and circulation patterns to El Niño (La Niña) conditions though the periodicity is decadal (~20 years) (Mantua and Hare, 2002).

The Arctic Oscillation (AO) is the dominant mode of sea level pressure (SLP) variation north of 20° N and is similar to ENSO in its interannual periodicity (Thompson and Wallace, 2000). Variability in SLP has been shown to indirectly affect fire activity in high-latitude regions by affecting the position of the jet stream (Thompson and Wallace, 1998). It is characterized in its positive (negative) phase by negative (positive) pressure anomalies that form over the Arctic and a circumpolar belt of positive (negative) temperature anomalies at mid-latitudes, with the exception of low temperature anomalies that form over the PNW (Thompson and Wallace, 1998; Thompson and Wallace, 2000; Macias Fauria and Johnson, 2006). In western Canada and southeast Alaska, large fire incidents have been linked to the negative phase of the AO when high-pressure anomalies centered over the Arctic extend south to 30-50° N and result in warmer and drier conditions that are more conducive to fire initiation and spread (Macias Fauria and Johnson, 2006). Only a few studies have assessed fire-climate relationships between the ENSO, PDO, and AO in British Columbia. These studies have omitted coastal temperate rainforests from their analyses, suggesting that these forests are too wet to show summer moisture deficits and are therefore unable to support large wildfires (Skinner et al. 2002; Meyn et al. 2010).

In this study, I overcome the limitations of past studies by reconstructing the fire history of a hypermaritime coastal island with extremely low rates of lightning and a short summer dry season. First Nations were forcibly removed from the island more than 100 years ago and it never had Euro-American settlement, logging, grazing, or fire suppression (McLaren et al. 2014; McLaren et al. 2015). I hypothesize that if First Nations frequently used fire, the fire history record would be temporally and spatially associated with former habitation sites at decadal intervals and absent in the 20th century. If First Nations rarely used fire, fire would be consistently rare, occurring at centennial and millennial intervals and mostly explained by topographic and climatic controls. I evaluate the relationship between climate variability and humans on fire occurrence over ca. 700 years in a perhumid coastal temperate rainforest. I use a network of fire scars and stand establishment to address the following questions: 1) What is the relationship between the temporal occurrence of fire events with drought (The Palmer Drought Severity Index [PDSI]) and with single and interacting phases of climate drivers such as the ENSO, PDO, and AO; and, 2) Is fire occurrence best explained by human presence, regional climate variability, topography, or more likely, interactions among these drivers?

3.3 Methods

3.3.1 Study area

The study area is located on Hecate Island (North 51° 39 Latitude, West -128° 04 Longitude) within the Hakai Lúxvbálís Conservancy on the Central Coast of British Columbia, Canada (Fig. 3.1). This region is characterized by a temperate climate with year-round cool temperatures (average annual ~7 °C, average summer ~12 °C) and annual rainfall sometimes exceeding 4000 mm (Banner et al. 2005). Hecate Island (67 km²) is

located in the very wet hypermaritime subzone of the Coastal Western Hemlock biogeoclimatic classification (Meidinger and Pojar, 1991). Excess soil water regulates this environment and subtle variations in slope or drainage result in significant differences in forest productivity (Banner et al. 2005). Four general vegetation types defined by dominant species and closely associated landforms are found in the study area (Banner et al. 2005). Productive (zonal) forests found in nearshore and riparian areas are characterized by large-diameter western redcedar (*Thuja plicata* Donn ex D. Don.) and western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), with lesser amounts of yellow-cedar (*Cupressus nootkatensis* [D.Don] Farjon and Harder), and Sitka spruce (*Picea sitchensis* [Bong.] Carr.) (Meidinger and Pojar, 1991). Bog forests exhibit stunted growth forms and are located on hill slopes dominated by western redcedar, yellow-cedar, western hemlock, and shore pine (*Pinus contorta* var. *contorta* Douglas ex Loudon) (Klinka et al. 1996). Bog woodlands are the most common vegetation type and are comprised of patchy mosaics of forested and unforested sites in subdued or rolling terrain (Banner et al. 2005). These forests contain roughly equal densities of western redcedar, yellow-cedar, and shore pine with lesser amounts of mountain hemlock (*Tsuga mertensiana* (Bong.) (Klinka et al. 1996). Blanket bogs are nutrient-poor, sparsely forested wetland areas that contain small amounts of shore pine and yellow-cedar (Banner et al. 2005). Zonal and bog forest vegetation types have closed canopies, larger diameter and more vertical fuel structures that are blanketed by thick moss compared to bog woodland and blanket bog vegetation types, which are more open, drier, and contain finer, more flammable fuel assemblages (Daniels and Gray, 2006; Hoffman et al. 2016a). Elevations in the study area range from

sea level to 150 m, and the geological substrate is homogenous quartz diorite and granodiorite (Roddick, 1996).

Although there are no oral histories of fire use by First Nations on Hecate Island, First Nations continuously inhabited the study area for at least 13,000 years until European contact in the late 18th and 19th centuries (McLaren et al. 2014). A team of archaeologists used radiocarbon dating of shell, faunal, and charcoal deposits to determine the locations of former habitation sites (McLaren et al. 2014; McLaren et al. 2015). This evidence together with the presence of culturally modified trees and nearshore features including shell middens, clam gardens, and fish traps confirm that Hecate Island supported three year-round habitation sites that were consistently utilized throughout the Holocene (McLaren et al. 2015; McLaren et al. 2015) and established millennia prior to the rise of contemporary vegetation in the region (Trant et al. 2016). Although First Nations continue to use the study area for seasonal resource gathering, habitation sites have not been persistently occupied since the late 19th century coinciding with two smallpox epidemics and the relocation of people to government-imposed reservations (Olson, 1955; Boyd, 1994; Trant et al. 2016).

3.3.2 Fire history sampling design

To reconstruct a 700 year fire history I sampled fire scars and stand establishment data from 30 plots (11.28 m radius [0.04-ha]) selected using a stratified random sampling design representing the range of elevations, aspects, and vegetation types within a 300-ha study area spanning three watersheds on Hecate Island (Fig. 3.1). To reconstruct low- and mixed-severity fires, I used a chainsaw to remove partial wedge sections of fire scars

from the bases of the oldest sound living trees (determined after coring and measuring) [40]. Fire-scarred trees were sampled in the one-hectare area surrounding every plot. I validated the year of each fire event recorded on fire scar wedges by sampling two 5 mm increment cores roughly 1.3 m from the ground from ~300 trees which were collected outside the fire area to ensure cross dating accuracy in the fire scar chronology (Johnson and Gutsell, 1994). In every plot, I recorded diameter at breast height (DBH) and removed two 5 mm increment cores from the root collar of every living tree >7.5 cm DBH to define the ages of post-fire cohorts within each plot and to estimate the year of the fire in which the cohort established (Johnson and Gutsell, 1994; Heyerdahl et al. 2001). Because fire scars provide more accurate records of fire occurrence, I assigned post-fire cohorts to the same year as nearby fire events if they occurred within the 10-year period of a nearby fire scar (Heyerdahl et al. 2001).

In the laboratory, cores and wedges were processed using standard dendrochronological techniques (Stokes and Smiley, 1968). Samples were first visually crossdated and then statistically verified using the computer program COFECHA (Grissino-Mayer, 2001). I crossdated fire scars to species-specific chronologies to obtain exact fire years (the calendar year in which the scar formed) (Heyerdahl et al. 2002). I was unable to determine the intra-ring position (seasonality) of the majority of samples due to rot and narrowing of tree rings at fire scar margins (Falk et al. 2011). I determined the fire history in each plot by compiling fire scars into a composite chronology, and identified fire events as those in which at least two trees had fire scars (Johnson and Gutsell, 1994).

Composite fire histories were graphed using Fire History Analysis and Exploration Software (Brewer et al. 2015).

3.3.3 Spatial controls of fire activity

I used a Generalized Linear Model (GLM) with a Poisson distribution and model selection via the Akaike Information Criterion (AIC_c) to predict two response variables 1) the abundance of fire scars; and, 2) the frequency of fire events using the explanatory variables (vegetation type, slope, aspect, elevation, distance to shoreline, and distance to former habitation site) in the ‘MuMin’ package in R statistical software (Barton, 2016; R Development Core Team, 2016) following the methods of Harrell (2015) and Burnham and Anderson (2002). Because there were no recorded fire events in the study area after 1893, I only calculated the MFI from the first detected fire in 1376 to the last detected fire in 1893 (517 years) and I assessed the composite MFI by combining all fire scar data from the study area over this period of analysis (Johnson and Gutsell, 1994). I was able to estimate fire effects on vegetation (severity) using the presence of fire-scarred trees, the MFI, and whether cohorts had established after fire events (indicative of mixed-severity fires) (Falk et al. 2011).

3.3.4 Climate datasets

The PDSI is a composite monthly index of regional climate calculated from current and archived instrumental data of precipitation and temperature (Palmer, 1965). Negative PDSI values indicate dry years and positive values indicate wet years (range -6 to +6). To characterize historic patterns of drought in the study area, I used an extended reconstruction of the PDSI from the gridded North American Drought Atlas (grid point

17 [127.5° W x 52.5° N]), which combines 28 tree-ring reconstructions of summer (June-August) temperature and precipitation (Cook and Krusic, 2004). In my analyses I used previous reconstructions of climate variability (PDSI, ENSO, PDO, and AO), which correlated to instrumental records and tree-ring chronologies sampled from the study area (Appendix B1: Table B1.1).

3.3.5 Annual fire climate analyses

At the annual scale, frequency analysis was evaluated with a G-test of goodness of fit in the ‘DescTools’ package in R statistical software (Signorell et al. 2015; R Development Core Team, 2016) to determine if fire events were more likely to occur during extreme climate years of single climatic indexes (negative and positive phases of the PDSI, ENSO, PDO, and AO). I used this test again to consider the influence of two-way and three-way interactions between the ENSO, PDO, and AO with fire occurrence (Schoennagel et al. 2007). I compared the number of fires during phase combinations assuming an equal likelihood of burning in all years. Sixty percent of fire events occurred between 1700-1893, therefore, I chose to analyze three-way interacting climate indexes during this period and compared all two-way interactions using their respective temporal windows (e.g. ENSO 1300-2004 with PDSI 1240-2006 [common period of analysis 1300-1893]) (Appendix B2: Fig. B2.1).

3.3.6 Interannual and multidecadal fire climate analyses

To characterize interannual and multidecadal relationships between fire occurrences and the PDSI, ENSO, PDO, and AO, I used Bivariate Event Analysis (BEA) with K1D software (Gavin, 2010). BEA is a temporal variant of spatial point pattern analysis, which

employs Ripley's K function to examine one-dimensional time series data (Gavin et al. 2006). This method is advantageous when testing time series data that are subject to serial autocorrelation and BEA assesses the response of fire years (synchrony, independence, or asynchrony) to climate events within a range of temporal windows. Monte Carlo simulations with 1000 replicates were used to randomize climate time series with 95% and 99% confidence envelopes (Carcaillet et al. 2009). BEA, as implemented here, assumes a one-directional process where fire events only respond to previous and current climate events (Bigler et al. 2007). I first tested single climatic indexes and fire years by defining extreme climate events as the 50 most positive or negative annual values of the PDSI, ENSO, PDO, and AO (Schoennagel et al. 2007). I then tested associations between two-way and three-way combinations among the phases of the ENSO, PDO, and AO and I defined extreme climate interactions by combining the 100 highest or lowest ranked annual values (Schoennagel et al. 2007).

3.4 Results

3.4.1 Fire history

I detected 15 fire events and three additional fires (recorded with one fire-scarred tree and verified with post-fire cohorts) from 99 fire scars on 45 partial sections of living, fire-scarred trees (Fig. 3.2). The majority of partial sections were from western redcedar (73%) and I found an average of two fire scars per sample (maximum 7) and 27% of trees had only one scar. I cored over 4000 trees (age range 52-953 years) and sampled 9 to 92 trees (mean = 52 trees) from each of 30 plots (Appendix B1: Table B1.2). My fire evidence was derived from fire scars and confirmed with stand establishment data, which

reliably extended to 1325 (689 years, sample depth >5 trees). The year of maximum fire synchrony and the last detected fire event in the study area was in 1893, when 90% of plots burned in a 287-ha mixed severity fire, in which 66% of trees survived (Fig. 3.2). The point fire interval (PFI [the MFI at the tree scale]) was 95 years and the composite MFI was 39 years in the period 1376-1893 (Appendix B1: Table B1.2).

3.4.2 Controls on the spatial variation of fires

The abundance of fire-scarred trees was found to consistently decrease with distance from former habitation sites (GLM Poisson regression: $b = -1.810\text{e-}03$, $z = -2.677$, $P < 0.01$; Fig. 3.3a). The most parsimonious model related the abundance of fire scars to distance from former habitation sites only (on the basis of AIC_c), and model averaging indicated that distance to former habitation site was significantly more influential than any other predictor or combination of predictor variables. Model validation confirmed that distance to former habitation site explained 65% of the variation (*pseudo R*²) in the abundance of fire-scarred trees (Appendix B1: Table B1.1; B1.3). The second response variable (frequency of fire events) modelled with a Poisson GLM confirmed that the number of fire events in plots decreased with distance from former habitation sites (GLM Poisson regression: $b = -0.0016$, $z = -4.750$, $P < 0.0001$; Fig. 3.3b). The most parsimonious model related the frequency of fire events to three main predictors (distance from former habitation site, aspect, and bog forest vegetation type; on the basis of AIC_c) (Table 3.1b). Model validation confirmed that distance to former habitation site was the most important predictor and together with aspect and bog forest vegetation type explained 52% of the variation in the frequency of fire events (Appendix B1: Table B1.1; Table B1.4).

3.4.3 Fire occurrence and single climate indexes

Frequency analysis confirmed that there was no significant relationship between drought (negative PDSI) and fire years (Appendix B1: Table B1.5). I found that 36% of fire events occurred during warm and dry years, 50% during average years, and 14% during cool and wet years (Appendix B2: Fig. B2.1). Fire years were not synchronous with years of significant departures from mean SSTs and SLP used as indexes for the ENSO, PDO, and AO (Appendix B1: Table B1.5; Appendix B2: Fig. B2.2). BEA analysis confirmed that antecedent drought (negative PDSI) values occurred more often than expected in the year preceding fire events (BEA, $P = 0.005$; Fig. 3.4a).

Positive phases of the ENSO (El Niño) occurred more often than expected 6-9 and 13-17 years prior to fire events (BEA, $P = 0.005$; Fig. 3.4b), and the positive phase of the PDO occurred more often than expected in the 10-year period prior to fire events (BEA, $P = 0.005$; Fig. 3.4c). Fires were consistently synchronous with the negative phase of the AO in the 33-year window prior to fire events (BEA, $P = 0.001$; Fig. 3.4d). Negative phases of the ENSO (La Niña) and the PDO were asynchronous with fire events and the positive phase of the AO was independent of fire events (Appendix B2: Fig. B2.3). To confirm that fire-climate relationships did not simply reflect signatures of natural climate variability, I performed BEA analysis on 16 randomly generated numbers (representing the 16 fire events) with nonsignificant results for all climate indexes (Appendix B2: Fig. B2.3).

3.4.4 Fire occurrence and interacting climate indexes

Fire frequency analyses of observed versus expected two-way and three-way interactions between phase combinations of the ENSO, PDO, and AO in the year of fire occurrence were nonsignificant (Appendix B1: Table B1.5; Appendix B2: Fig. B2.2). This demonstrated that fire years were not more likely to occur during specific phase combinations and their associated climate signatures. Although fire and climate interactions did not follow an observable pattern in the year of fire events, interannual and multidecadal interactions between two-way and three-way combinations of the ENSO, PDO, and AO were significant (Fig. 3.4). Fires were synchronous with the combined positive phases of the ENSO and PDO 3-14 and 20-30 years prior to fire events (BEA, $P = 0.005$; Fig. 3.4e). Similarly, the negative phase of ENSO (La Niña) and the positive phase of the PDO occurred more often than expected 1-12 years prior to fire events (BEA, $P = 0.001$; Fig. 3.4f). Fires were consistently synchronous with the positive phase of the PDO and the negative phase of the AO during the 6-33 year period prior to fire events (BEA, $P = 0.001$; Fig. 3.4g). BEA analysis of interacting three-way phase combinations of the ENSO, PDO, and AO were synchronous with fire events only when the ENSO and PDO were positive and the AO was negative 8-36 years prior to fire events (BEA, $P = 0.001$; Fig. 3.4h). All other phase combinations were nonsignificant (Appendix B2: Fig. B2.3).

3.5 Discussion

Drivers of historic fire activity remain largely unknown in coastal temperate rainforests in the PNW of North America (Whitlock et al. 2010; McWethy et al. 2013; Whitman et al. 2015). I present the first examination of the influences of and interrelationships between

humans and climate on fire activity in a very wet coastal temperate rainforest using a ca. 700 year record of fire scars and stand establishment. My data describe a low- and mixed-severity fire cycle in which fires occurred on average every 39 years over the 517 year period from the first detected fire in 1376 to the last fire in 1893 (Fig. 3.2). Although the frequency of fire events on Hecate Island may seem low relative to fire-prone landscapes, my data indicate that fire occurrence was 25 times more likely than previous estimates of ca. 1000-year intervals (Wong et al. 2003; Daniels and Gray, 2006; Pearson, 2010). Such a fire frequency would have had large impacts on forest composition and structure, specifically in the AD 1600-1893 period when fires were more frequent (Fig. 3.2). The combination of low and mixed severity fires recurring at multidecadal intervals, the absence of fire after First Nations relocation (Fig. 3.2), and the spatial pattern of fire relative to habitation sites (Fig. 3.3) suggests that fire may have been intentionally used as a tool for resource management (Turner, 1999; Turner, 2014). In addition, significant lagged effects of climate indexes prior to fire events indicate an added role of climate variability on fire occurrence (Fig. 3.4).

There is no evidence of fire activity after the last recorded fire in my study area in 1893 (Fig. 3.2). This 124-year fire-free interval (1893-present day) is distinct from the reconstructed fire history and is the longest fire-free period documented in the study area in seven centuries (Fig. 3.2). This period also coincides with the only time in roughly 13,000 years that the study area was not continuously inhabited by First Nations (McLaren et al. 2014; McLaren et al. 2015). Unlike many regions of British Columbia that experienced widespread fire suppression in the 20th century, there are no known

records of ignitions or suppressions in this remote coastal temperate rainforest during the 20th century, and very few in the entire coastal temperate rainforest as a whole (Stocks et al. 2002; Wong et al. 2003). However, several cycles of in-phase climate oscillations resulting in warmer and drier conditions conducive to fire spread occurred in the study area after the last recorded fire in 1893 (Appendix B1: Table B1.1).

Fire-climate studies in the PNW relate synchronous fire years to the year's climate and cite warm and dry spring conditions (Hessl et al. 2004), combined with low winter snowpack (Heyerdahl et al. 2002), as factors affecting fire years by drying fuels and resulting in longer fire seasons. Contrary to this, I found that antecedent drought in the year prior to fire events was significantly related to fire occurrence (Fig. 3.4a) and a closer examination of individual fire years revealed that fire events corresponded to 2-7 years of dry conditions (negative PDSI) (Appendix B2: Fig. B2.1). The largest fire in the study area (287-ha) occurred in 1893, during one of the wettest (10th percentile) reconstructed years on record and one of the strongest negative ENSO (5th percentile) and PDO (25th percentile) climate years. Although the combined effects of these climate indexes suggest unsuitable conditions for fuel ignition and spread, a closer examination of the 10-year period prior to the fire year reveals a period of sustained drought and three of the strongest reconstructed positive ENSO (5th percentile) and PDO (10th percentile) years. This pattern of antecedent drought in combination with positive in-phase ENSO and PDO indexes prior to large fire events in the study area repeats through the data set (Fig. 3.4) and reveals that antecedent drought may be a necessary precondition for fire initiation and spread at this study site. The effects of interannual and decadal drought on

forest growth and fuel dynamics are not well documented in this region. Such antecedent climate events may affect the available fuel profile (as occurs in grassy fuels types such as in the southwest United States (Swetnam and Betancourt, 1998; Kitzberger et al. 2007) or other unknown aspects of the fire cycle.

My results are comparable to other studies in the PNW and indicate that interannual drought and interactions with regional climate variability may have been an important prerequisite for fuel combustion and fire spread (Gedalof et al. 2005), but My results are unique in concluding that fires would not likely have occurred without human ignitions (Fig. 3.3). For example, in my analyses, I found that fire events were not spatially biased to expected locations such as south-facing aspects (Heyerdahl et al. 2001), steep slopes (Gavin et al. 2003), or rocky outcrops at middle and high elevations (Lertzman et al. 2002), but were strongly associated with the locations of former habitation sites (Fig. 3.3). I also found no difference in the frequency of fire events across the four vegetation types despite dissimilarities in fuel load, fuel availability, and potential flammability (Appendix B2: Fig. B2.4) (Hoffman et al. 2016a). Although the effects of topography and vegetation on fire frequency were not readily apparent, the variables aspect and bog forest vegetation type were included in my final model selection (based on weighted probabilities) along with distance to former habitation site (Table 3.1b). These variables may suggest other potential relationships between the spatial patterns of fire, vegetation, and humans such as the influence of prevailing summer winds, natural fuel breaks, and territorial boundaries. Despite being able to reconstruct historical fire events and gain insights into fire behaviour with the presence of fire-scarred trees (Falk et al. 2011), I

could not confirm whether anthropogenic burning was accidental or intentional, or how often burning was attempted but unsuccessful because fire weather was not conducive to fuel ignition and fire spread (Lepofsky et al. 2005).

Historically, low severity fires may have been intentionally set near habitation sites to clear land in otherwise dense forests, harvest timber, and promote important and accessible food species such as berry producing shrubs and bracken fern (*Pteridium aquilinum*), a fire follower species and an important starch in the diet of coastal First Nations (Turner, 1999; Turner, 2014). In these very wet forests, centuries of repeat burning surrounding habitation sites may have affected fuel loads, seral stages and the abundance and distribution of flammable vegetation (Kitzberger et al. 2007; Bowman et al. 2011; McWethy et al. 2013). Because biomass is not limited in coastal temperate rainforests, humans can exert control over natural conditions that promote or reduce limits on fire activity. Therefore, human drivers and their ecosystem effects are likely more pronounced relative to long-term climate in these very wet ecosystems (Bowman et al. 2011; McWethy et al. 2013; Whitman et al. 2015). As climate change continues to promote prolonged warm and dry conditions and human footprints expand (Ryan et al. 2013), fires in these ignition-limited systems may occur again, but have the potential to become large and uncontrollable (Bowman et al. 2011; McWethy et al. 2013; Ryan et al. 2013; Whitman et al. 2015). I propose that a greater understanding of the cultural traditions of burning and their ecological effects would provide more insight into human controls on historic fire activity and help assess vulnerabilities to future human-fire-climate interactions (Bowman et al. 2011).

3.6 Conclusions

Fires and their drivers have never been reconstructed with fire scars at centennial scales in perhumid coastal temperate rainforests. Although my study comprises only 300 ha in British Columbia, my results provide insight into the role of fire in very wet perhumid coastal temperate rainforests and the need to re-examine the fire regime concept to include human ignitions, fire as a tool for resource management, and longer analysis periods (Bowman et al. 2011). My results are consistent with other studies that suggest fire is rare in coastal temperate rainforest settings in the 20th century (Wong et al. 2003; Meyn et al. 2010; Pearson, 2010), but I also found that historical fires occurred on average every 39 years during the AD 1376-1893 period and were synchronous with large-scale climate oscillations. Both First Nations land use and climate variation appear to have interacted to create long-term trends in fire occurrence over the past 700 years and I find the fire history of my study site is best explained by relating cycles of anthropogenic burning, which varied on decadal timescales, to decadal variability in climate. The “ecological legacies” of historic anthropogenic burning, in the form of persistent effects on contemporary forest structure, likely abound in this region, and thus discredit the myth of coastal temperate rainforests as pristine landscapes. My clear evidence of precontact anthropogenic burning calls into question our current understanding of fire history in coastal temperate rainforest and how we define landscapes that were historically managed with fire.

Table 3.1 Results of the final model selection using Akaike Information Criterion (AIC_c)

for eleven Generalized Linear Models (GLM) that describe a) the abundance of fire-scarred trees and b) the frequency of fire events with nine predictor variables (vegetation [four types], elevation, slope, aspect, distance to habitation site in metres, and distance to shoreline in metres). Full results are provided in Appendix B1: Table B1.3; Table B1.4.

a) Abundance of fire-scarred trees						
Model	K	R²	ΔAIC_c	w_i	ER	Parameters
1	1	0.65	0.00	0.42	1.00	Distance to habitation
2	2		1.77	0.17	2.53	Distance to habitation, slope
3	2		2.02	0.15	2.80	Distance to habitation, elevation
4	2		2.39	0.13	3.23	Distance to habitation, aspect
b) Frequency of fire events						
1	3	0.59	0.00	0.30	1.00	Distance to habitation, aspect, bog forest type
2	2		0.75	0.20	1.50	Distance to habitation, aspect
3	1		2.31	0.09	3.33	Distance to habitation
4	5		2.53	0.08	3.75	Distance to habitation, aspect, bog forest type, zonal forest type, distance to shoreline
5	3		2.67	0.08	3.75	Distance to habitation, elevation, slope, distance to shoreline
6	4		2.76	0.07	4.28	Distance to habitation, bog forest type, zonal forest type, slope
7	3		3.58	0.05	6.00	Distance to habitation, elevation, bog forest type

Notes: Model averaging was conducted and only models within 95% confidence intervals are included. K = number of model parameters, R² = the pseudo R² value, ΔAIC_c = change in AIC score from the top model, w_i = AIC_c model weight, ER = top model weight divided by *i* model weight.

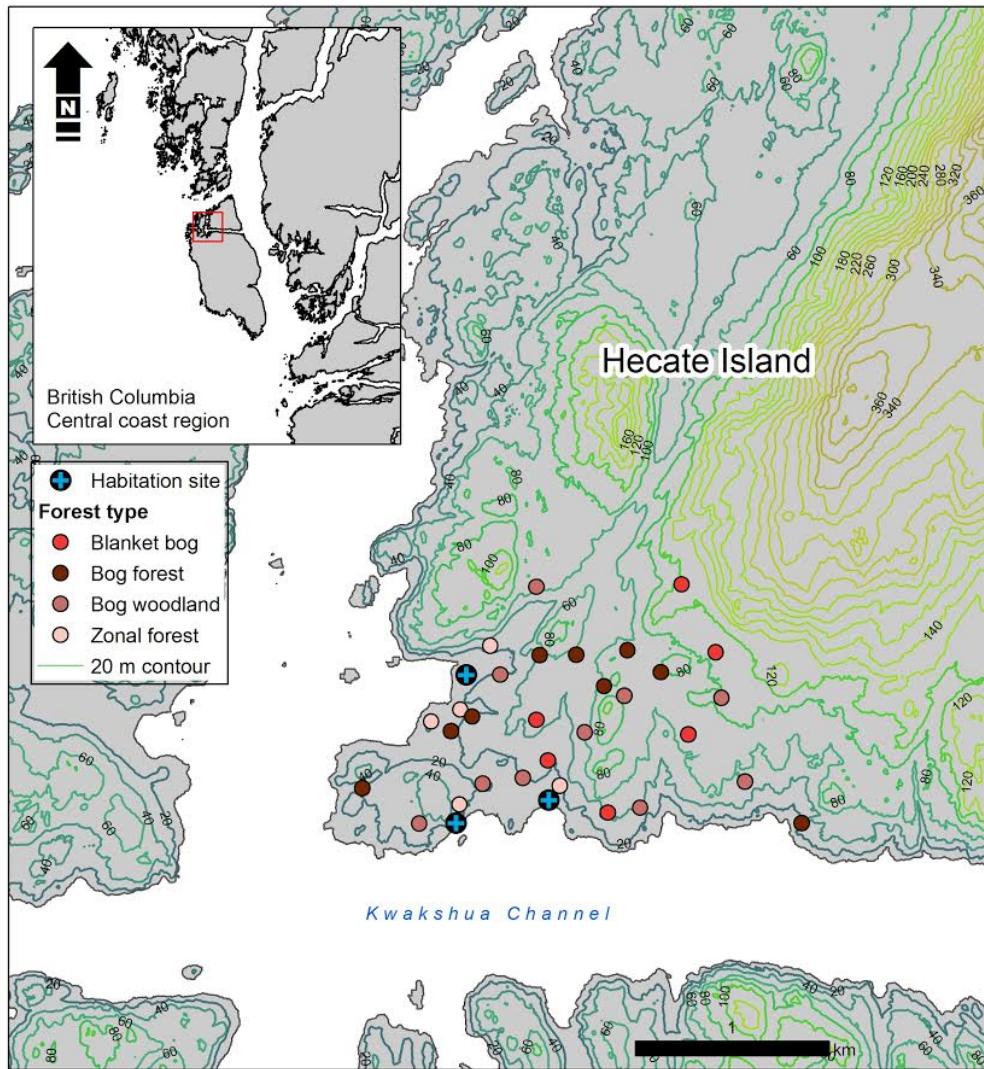


Figure 3.1 The locations of sample sites on Hecate Island (N 51°39'43 Latitude, W - 128°04'2 Longitude). The shaded circles (four vegetation types) represent the locations of 30 fire history plots and the blue crosses provide the general location of the three former habitation sites in the study area. The inset provides the location of the study site on the Central Coast of British Columbia, Canada.

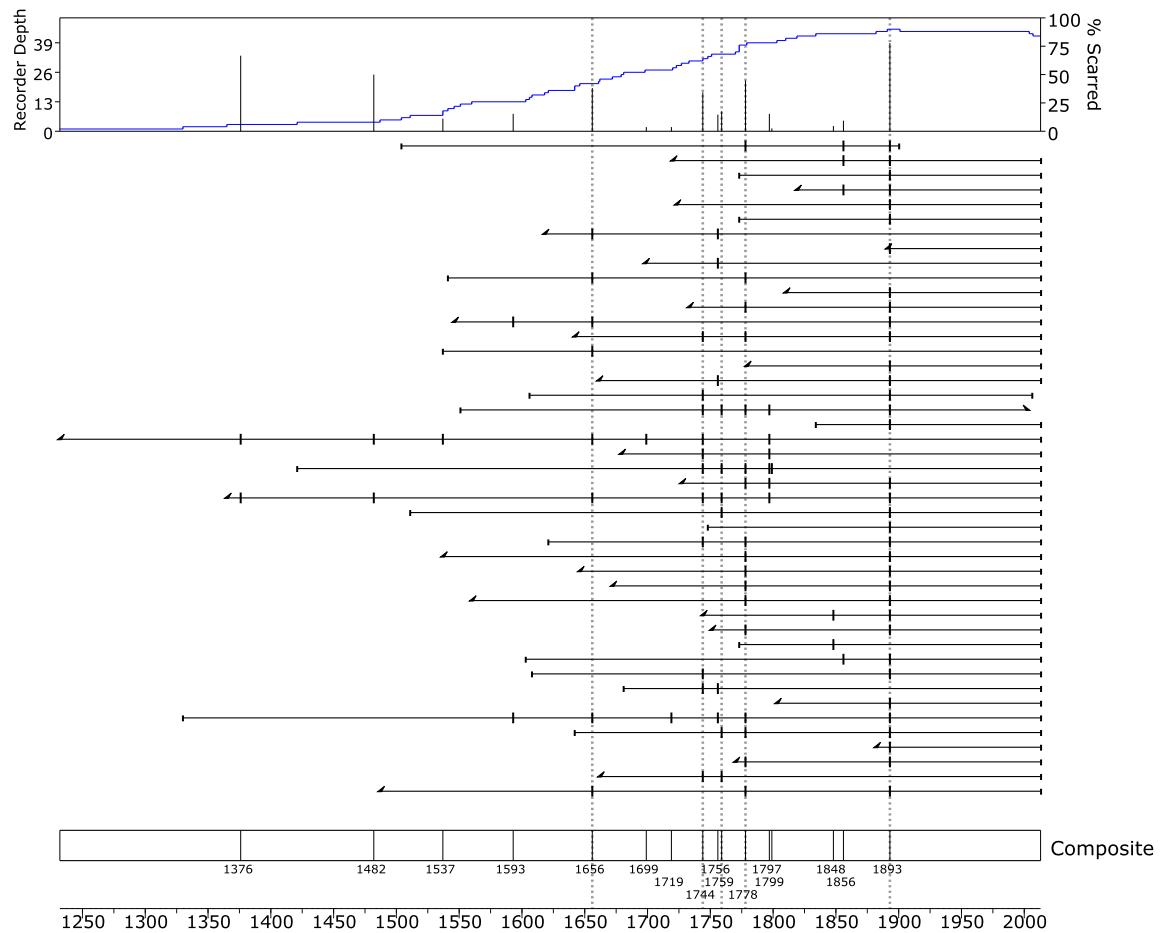


Figure 3.2 The composite fire history chronology of all fire scars in the study area. Each horizontal line shows the composite fire scar record at a single sampling plot through time. Vertical bars mark fire years and dashed vertical lines mark non-recorder years. Fine vertical lines at the beginning and end of each chronology mark pith or bark dates, while fine diagonal lines represent the earliest or latest ring dates for plots where pith or bark dates were not sampled. The sample recorder depth of the chronology is located on the top panel and all fire dates in the study area are recorded in the chronology in the bottom panel.

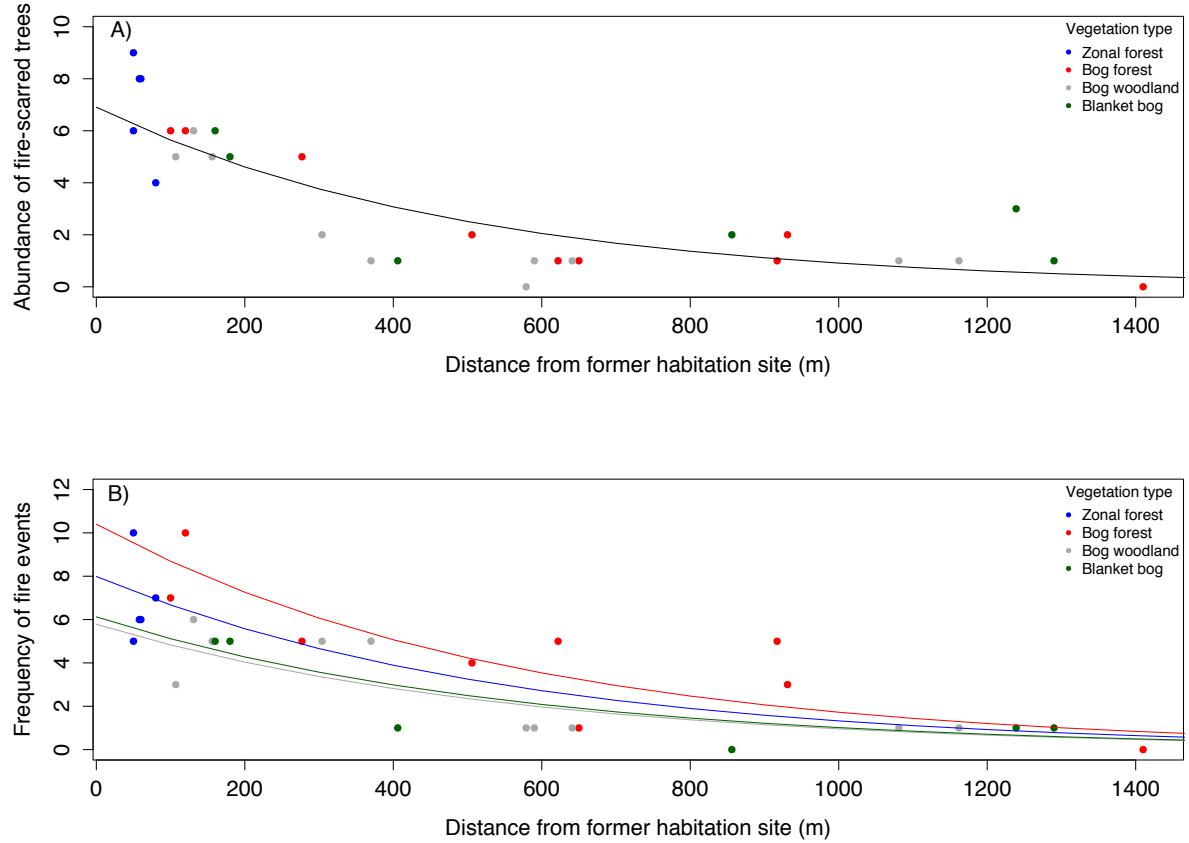


Figure 3.3 a) Poisson regression of the relationship between the abundance of fire-scarred trees and the distance from habitation sites in metres. The most parsimonious model related the presence of fire scars to distance from habitation sites only and model averaging indicated that the distance to habitation sites was significantly ($P < 0.01$) more influential than any other predictor or combination of predictor variables. b) Poisson regression of the relationship between the frequency (number) of fire events in each vegetation type and the distance from habitation sites in metres. The most parsimonious model includes three variables (distance to habitation site, aspect, and bog forest vegetation type) and distance to habitation site was the most important predictor variable ($P < 0.001$).

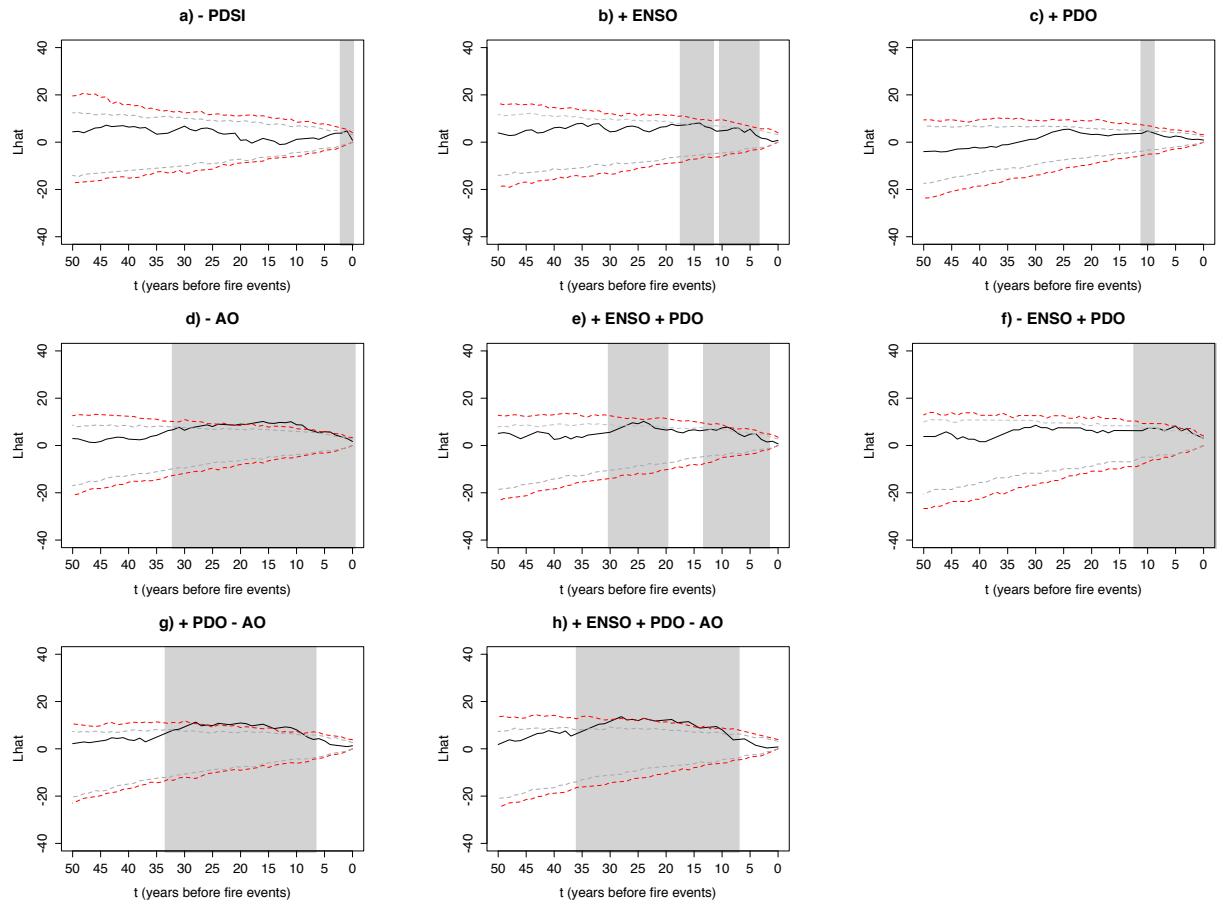


Figure 3.4 Bivariate Event Analysis (BEA) of the temporal association between fire years and extreme climate events. Abbreviations are the El Niño-Southern Oscillation (ENSO), the Pacific Decadal Oscillation (PDO), the Palmer Drought Severity Index (PDSI), and the Arctic Oscillation (AO). Black lines above the dotted red (99% confidence envelopes) and the dotted grey (95% confidence envelopes) lines indicate synchrony between the two events (events occurred more often than expected, t years prior to fire events). Confidence envelopes are based on 1000 Monte Carlo simulations and years of significant synchrony are shaded in grey. Further results are provided in Appendix B1: Table B1.5 and Appendix B2: Fig. B2.2; Fig. B2.3.

Chapter 4: Ecological legacies of anthropogenic burning in a British Columbia coastal temperate rainforest

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4.1 Abstract

Historic burning by First Nations is often considered to have little impact on coastal temperate rainforests, yet few long-term fire histories have been reconstructed in these forests. I use a multidisciplinary approach to assess the impact, scale, and ecological legacies of historic fires. I focus on perhumid temperate rainforests located on the Central Coast of British Columbia, Canada. I reconstructed 700 years of temporal and spatial aspects of fire activity with 30 plots on Hecate Island using fire scars and stand establishment. I then conducted a paired study of 20 former First Nations habitation and control sites on 15 islands to relate fire activity to patterns of human settlement. I mapped 15 years of lightning strike densities and use mixed-effects modelling to assess whether fire activity predicted the distribution and abundance of traditional plants. Sixteen low and mixed severity fires were recorded from 1376-1893. The abundance of traditional plants and the density of western redcedar trees were best predicted by the location of former habitation sites and shorter mean fire intervals (MFI). Lightning is too rare to explain the pattern of fire activity in the study area. No fire activity was detected after 1893, coinciding with the relocation of First Nations from the study area. First Nations occupations were associated with fire in space and time. People likely utilized fire as a tool for resource management to influence the densities of specific plants by creating mosaics of vegetation in different stages of succession. By assessing the ecological impacts of historic fire events we gain a better understanding of the abrupt changes that occurred in the 20th century. Our ability to understand temperate rainforest ecosystems may be compromised if we underestimate the role of humans in driving historic fire activity.

4.2 Introduction

Fire is one of the most important mechanisms driving landscape-scale ecological dynamics, plant community structure, and soil properties (Bowman et al. 2009; Moritz et al. 2014). Humans have altered fire activity for millennia in many different ecosystems by igniting and suppressing fires, and modifying fuel composition and structure (Pausas and Keeley, 2009). Because fire is both a stochastic, natural process and an intentional, cultural process there is much debate regarding the relative importance of lightning versus human ignitions (Marlon et al. 2012; Ryan et al. 2013; Krasnow et al. 2016; Rolstad et al. 2017). Climate is a primary driver of fire activity at regional scales, shaping the composition and distribution of vegetation and governing weather conditions that promote fire initiation and spread (Krawchuk and Moritz, 2011). At these broad scales, human influences are considered secondary to climate because their effects are geographically localized (Whitlock et al. 2015), and the relative roles of humans and climate are difficult to discriminate and generalize (Ryan et al. 2013). Although the historic and contemporary dynamics of fire activity in fire-prone landscapes are well studied, current fire-history methods which heavily rely on fire scar records are likely inadequate to decouple natural and anthropogenic contributions to historic fire activity in temperate rainforests (Bowman et al. 2011; McWethy et al. 2013). Globally, coastal temperate rainforests contain ample biomass for burning, but are considered both climate- and ignition-limited systems due to high year-round fuel moisture levels (Power et al. 2008; Bowman et al. 2009; Whitlock et al. 2015). Reconstructions of forest stand establishment following historic fire events and improved data on First Nations fire-

management systems may provide a greater understanding of fire activity in coastal temperate rainforests.

Perhumid or high-latitude ($>50^{\circ}$ N) coastal temperate rainforests extend from the northern tip of Vancouver Island to the panhandle of Alaska (DellaSala et al. 2011). These forests are the wettest variant of the temperate rainforest biome and are considered to have low natural resilience to fire (DellaSala et al. 2011). In these forests, fuel moisture levels rarely drop to the point of ignition and a thick layer of fog often blankets the coastline even in the summer months (Daniels and Gray, 2006). Despite the seemingly unsuitable conditions for fire, recent research suggests that short-term summer drought does occur (Hoffman et al. 2016b) and that fire has been an important component of forests in this region for millennia, with associated impacts on the abundance, diversity, and availability of plant resources (Trant et al. 2016; Hoffman et al. 2017).

Anthropogenic burning refers to both intentional and accidental ignitions by people (Guyette et al. 2002). In this paper, I consider intentional burning as the practice of periodic burning of landscapes or particular sites by First Nations over time to achieve specific management goals (Turner et al. 2013). Although oral histories of intentional burning are well documented in southern and interior areas of British Columbia (Turner, 2014), we know little regarding how various cultural groups utilized fire on the Central Coast of British Columbia and consequently how differing management practices affected ecosystem patterns and processes (Turner, 1999). Reconstructing the frequency and extent of intentional burning is challenging because many fire management practices

have subtle ecological legacies and many recollections of burning have faded from memory (Lepofsky et al. 2005; Turner 2014). Reconstructions are rendered more difficult by the arrival of Europeans on the coast of British Columbia in the late 18th and 19th centuries which in many regions resulted in cultural suppression and erased legacies of First Nations land-use practices through logging, agriculture, and livestock grazing (Turner et al. 2013).

The rarity of summer lightning (Lertzman et al. 2002), availability of biomass for burning (Daniels and Gray, 2006), and a >13,000 year history of human habitation (McLaren et al. 2015) make the Central Coast of British Columbia an ideal location to examine historic fire activity. The goal of this study is to use a weight-of-evidence approach combining quantitative and qualitative data to improve our understanding of the temporal and spatial attributes of historic fire activity, the probability of lightning and human ignitions, and how fire impacted landscape patterns and processes (Lepofsky et al. 2003). I utilized multiple lines of evidence to test three hypotheses: 1) Fires were natural (lightning-caused); 2) Fires were human (accidental ignitions); and, 3) Fires were human (intentional ignitions). I predicted the spatial and temporal attributes, severity, and cyclical patterns of historic fire activity along with changes to fire activity in the 20th century. My hypotheses, predictions for each test, and support for each model are described in Table 4.1. Further direct and indirect ethnobotanical evidence of anthropogenic burning is outlined in the Supporting Information (Appendix C1: Tables C1.1; C1.2).

4.3 Materials and methods

4.3.1 Study area

The study area is composed of a 2000 km² island group on the coastal margin of central British Columbia, Canada (Fig. 4.1). This region is characterized by a temperate climate with year-round cool temperatures (average annual ~7 °C, average summer ~12 °C) and annual rainfall that locally may exceed 4000 mm (Banner et al. 2005). The Pacific Ocean moderates temperatures throughout the year and the British Columbia Coast Mountains protect the islands on the outer coast from cold winter and hot summer continental air masses (DellaSala et al. 2011). Located within the very wet hypermaritime variant of the Coastal Western Hemlock biogeoclimatic zone, this region is characterized by low rates of evapotranspiration and soils that can remain saturated throughout the year (Meidinger and Pojar, 1991).

4.3.2 Vegetation

Productive forests are dominated by western redcedar (*Thuja plicata* Donn ex D. Don) and western hemlock (*Tsuga heterophylla* [Raf.] Sarg.), with lesser amounts of yellow-cedar (*Cupressus nootkatensis* [D.Don] Farjon and Harder), and Sitka spruce (*Picea sitchensis* [Bong.] Carr.). However, productive forests are restricted to steeper slopes or riparian and nearshore areas with nutrient-rich and moderately drained soils. This distribution is in contrast to the majority of vegetation types in the study area, which exhibit stunted growth forms as a result of prolonged saturation and acidic soil conditions (Banner et al. 2005). Less productive vegetation types include bog forests and bog woodlands, which grow in moderate and poorly-drained soils on rolling terrain and form a patchy mosaic of heterogeneous woodlands dominated by cedars, western hemlock, and

shore pine (*Pinus contorta* var. *contorta* Douglas ex Louden). Blanket bog vegetation types are located on subdued terrain and contain nutrient-poor soils with shrubby, short-statured vegetation (Banner et al. 2005).

4.3.3 Human history

Compared to other coastal regions of British Columbia, sea levels in the study area have been relatively stable (+/- 2 m) throughout the Holocene period and cultural sites remain well preserved on the landscape (McLaren et al. 2014). The combination of sea-level stability, a mild climate, and abundant terrestrial and marine resources supported continuous settlement in the region for at least the last 13,000 years (McLaren et al. 2015). The study area includes numerous former First Nations habitation sites with an accumulation of shell-midden deposits and nearshore features including clam gardens, fish traps, and culturally modified trees (CMTs) (McLaren et al. 2015; Trant et al. 2016). There are no ethnographic accounts of anthropogenic burning in the study area, but elders in the nearby community of Waglisla (Bella Bella) describe older generations repeatedly burning hillslopes and interior mountain sites to increase the abundance of blueberries (*Vaccinium spp.*) (Turner, 2014). There are also oral histories of burning off vegetation on small islands in nearby Haida Gwaii to establish ownership and to increase the productivity of food-producing plants (Turner, 1999). Although First Nations seasonal-resource gathering continues in the study area, habitation sites have not been occupied since at least the end of the 19th century (McLaren et al. 2015). The majority of the landscape has not been logged, potentially preserving ecological legacies of long-term First Nations land-use practices.

4.3.4 Fire history reconstruction: Hecate Island

I completed a fire-history case study in a historic 287 ha burn located on Hecate Island (N 51° 39' Latitude, W 128° 04' Longitude) (Fig. 4.1). This site was chosen for a detailed fire history because it contained abundant, rot resistant, and long-lived (>1000 years) fire-scarred trees. Fire-scarred trees are rare in perhumid coastal temperate rainforests due to long fire-free intervals (Lertzman et al. 2002), intolerance to fire of local tree species, and high rates of wood decomposition (Daniels and Gray, 2006). To reconstruct historic fire activity and identify contemporary vegetation patterns associated with anthropogenic burning, I sampled fire scars, forest-stand establishment, and described plant species richness and abundance from 30 plots (11.28 m radius [0.04 ha]) selected using a stratified random sampling design representing the range of elevations, aspects, and dominant vegetation types in the area (Fig. 4.1).

In the one-hectare area surrounding every plot, I used a chainsaw to remove partial wedge sections of fire scars from the bases of the oldest sound living trees (determined after coring and measuring) (Arno and Sneck, 1977). I sampled two 5 mm increment cores roughly 1.3 m from the ground from 100 trees in forests located outside of the 287-ha perimeter with no aboveground fire evidence to ensure cross-dating accuracy in the fire-history chronology (Johnson and Gutsell, 1994). In every plot, I removed two 5 mm increment cores from the root collar of every living tree >7.5 cm diameter at breast height (DBH) to determine the year of establishment (Tepley and Veblen, 2015). I used the presence of fire-scarred trees to assess fire severity and assigned post-fire cohorts to the same year as a fire event if they occurred within the 10-year period of a nearby fire scar.

(Heyerdahl et al. 2001). In the laboratory, cores and wedges were processed using standard dendrochronological techniques (Stokes and Smiley, 1968). Samples were first visually cross-dated and then statistically verified using the computer program COFECHA (Grissino-Mayer, 2001). For cores that did not reach pith, I used Duncan's (1989) method to calculate the distance to the chronological centre of each tree. I compiled fire scars into a composite chronology, and identified fire events as those in which at least two trees had fire scars (Heyerdahl et al. 2001). I calculated point fire intervals (PFI [years between successive fires at plot locations]) and composite mean fire intervals (MFI [years between successive fires anywhere in the study area]), and mapped the spatial and temporal extent of historical fires (Tepley and Veblen, 2015).

I completed vegetation surveys and estimated the percent cover and richness of all living and dead tree, shrub, herbaceous, and non-herbaceous species in every plot to assess the potential effects of historic fire activity on present-day plant communities. Specifically, I examined the relationship between fire events and the abundance and composition of traditional plants that were historically utilized for food and medicine (Turner, 2014). A list of all species sampled in the study area and ethnobotanical references of traditional plants associated with historic fire management systems in British Columbia are provided in the supporting information (Appendix C1: Tables C1.1; C1.2).

Because I expected clustering of the four vegetation types (zonal forest, bog forest, bog woodland, and blanket bog) in the data (vegetation types would be more similar within groups than between groups) I analyzed the data using a Generalized Linear Mixed-

effects Model (GLMM) with a Poisson distribution to account for spatial dependencies within sites. I used the ‘glmmML’ and ‘MuMIn’ packages in the computer program R (Brostrom, 2013; Barton et al. 2016; R Development Core Team, 2016) following the methods of Zuur et al. (2009). I analysed the distribution of each response with the predictor variables MFI, elevation, aspect, slope, distance to shoreline, and distance to former habitation site. All explanatory variables with high collinearity (Variance Inflation Factor [VIF] >3) were removed and models were compared with Akaike Information Criterion (AIC_c). Graphical methods were used to assess model adequacy.

4.3.5 Regional fire history surveys

I sampled belt transects (6 m wide x 30 m long) in a paired design to examine the presence of fire activity in forests surrounding former habitation and control sites ($n = 20$) located on 15 islands throughout the study area (Fig. 4.1). I selected former habitation sites containing established shell midden and archaeological records of occupation throughout the Holocene (McLaren et al. 2014; McLaren et al. 2015). These sites were compared to controls that were selected for their similar vegetation structure and site attributes (aspect, slope, and elevation), but lacked any above or belowground evidence of shell midden. Controls were either located on separate islands (75%) or several hundred metres away from former habitation sites to limit the potential of previous fires burning through treatment and control transects. The location of transects at each site was randomly assigned based on the size of the nearshore site area, and transects six m wide and 30 m long in replicates of three were completed at the present-day forest edge at each former habitation and control site. Thus, sites with wider shorelines would have the three transects spaced further apart, on average, than sites with narrower shorelines. The height

and decay class of all trees >15 cm DBH were sampled in transects along with the presence of CMTs, fire scars, and coarse woody debris (CWD).

4.3.6 Lightning strike densities

I assessed the exact locations of lightning strikes with a peak current > 5 KA (the minimum estimated current strength to promote fire ignition) in the 2000 km² study area for the period 1999-2014 (CLDN, 2016). I acquired the data from the Canadian Lightning Detection Network, which includes real-time strike locations with negative and positive polarity (CLDN, 2016). Lightning data were examined with gridded climate data to quantify simultaneous instances of low fuel moisture and summer lightning activity (Krawchuk and Moritz, 2011).

4.4 Results

4.4.1 Fire history reconstruction: Hecate Island

Sixteen fire events were recorded on 45 partial sections of living fire-scarred trees in the 287 ha fire perimeter located on Hecate Island (Fig. 4.2; Appendix C1: Table C1.3). The fire history chronology spanned the calendar years 1330-2014 (684 years). The first fire event occurred in 1376 and the last and the largest fire event occurred in 1893 when 90% of plots burned in a 287-ha low- and mixed-severity fire (Fig. 4.2). I dated 99 fire scars on 45 trees, the majority of which were recorded on living western redcedar trees. Trees containing fire scars were >400 years at the majority (65%) of plots. I was unable to determine the intra ring position (seasonality) on most samples due to rot and narrowing of tree rings at fire scar margins. Fire size estimates inferred from the location of trees bearing fire scars dating the same fire year and the locations of post-fire stand establishment ranged from 0.01-287 ha, with a median size of 3 ha (Fig. 4.2). The PFI

was 95 years and composite MFI was 39 years (based on a 517 year reconstruction from 1376-1893; Appendix C1: Table C1.3). I found no evidence of fire scars, post-fire stand establishment or char on trees after the last recorded fire in 1893.

4.4.2 Fire and plant communities

I found an average of 37 plant species in each plot; species richness was highest in the bog woodland vegetation type and lowest in the zonal forest vegetation type. I detected overdispersion in my model analysing the density of western redcedar trees; I therefore chose to reanalyse the model with a negative binomial distribution (Zuur et al. 2009). The density of western redcedar trees was significantly higher in vegetation plots with shorter MFI ($P < 0.01$) and model averaging delineated with 95% confidence intervals (based on AIC_c) indicated that the model containing only the variable MFI was the best predictor of western redcedar density (Table 4.2a). The abundance of traditional plants was significantly greater in plots with shorter MFI ($P < 0.01$), flatter terrain ($b = 0.021, z = 2.902, P < 0.01$), in close proximity to habitation sites ($b = -0.009, z = -1.993, P < 0.05$), and declined significantly at higher elevations ($b = -0.0139, z = -2.685, P < 0.01$; Table 4.2b; Fig. 4.4). Model averaging delineated with 95% confidence intervals (based on AIC_c) indicated that the predictors aspect, elevation, distance to habitation, slope, and MFI were of relatively equal importance and were included in the final model selection (Table 4.2b). Model selection criteria and validation are provided in Appendix C2 in the Supporting Information.

4.4.3 Transect data: aboveground fire evidence

Fire-scarred trees and burned CMTs were more prevalent in transects near former habitation sites when compared to control sites (*Welch two sample t-test*: $t = -4.1015, d.f. = 27.699, P < 0.001$; Fig. 4.3a fire-scarred trees], $t = -3.6088, d.f. = 26.64, P < 0.001$; Fig. 4.3b CMTs]). The majority of trees with fire evidence were found 10-20 m (35%) and 20-30 m (46%) from the present day forest edge. Dating of trees surrounding former habitation sites and control sites confirmed that most trees (80%) exceeded 200 years of age. Consistent with Trant et al. (2016), trees were also taller near former habitation sites ($t = -3.8868, d.f. = 37.587, P < 0.01$; Fig. 4.3c), and had larger DBH ($t = -2.7542, d.f. = 36.786, P < 0.05$; Fig. 4.3d). Most fire evidence was found on culturally modified western redcedar trees. There was no significant difference in the number of trees and the average number of pieces of CWD between transects in former habitation and control sites (Appendix C1: Fig. C1.1).

4.4.4 Temporal and spatial patterns of lightning

There have been two lightning strikes on Hecate Island since 1999. Both strikes occurred in December and were accompanied by heavy precipitation (>30 mm in the 24-hour period; Fig. 4.5). In total, 396 strikes occurred in the 2000 km² study area from 1999-2014, and of these strikes, 108 hit land and 17 occurred in summer months (May 1st – Sept. 30th; Fig. 4.5). The odds of lightning striking the sea were higher than striking the land in both winter and summer months ($P < 0.01$; Fig. 4.5). Cumulative precipitation in the 14-day period prior to 94% of lightning strikes in the summer months exceeded 20 mm, indicating that fuel moisture levels would likely not have lowered to a point (~30-40% moisture content) that fire could spread (Agee, 1993; Daniels and Gray, 2006; Whitlock et al. 2015).

4.5 Discussion

Multiple lines of evidence derived from fire scars, stand ages, plant community characteristics, and lightning strike densities are in opposition to the hypothesis that historic fires resulted from natural lightning ignitions. My data suggest that lightning is too rare to explain the temporal and spatial patterns of historic fire activity in the study area (Fig. 4.5). My data also support my second hypothesis that fires (whether accidental or intentional) were associated with human occupation of the land, and I find that fires of human origin occurred regularly in my fire history data, as reflected in their timing, location, severity, and effects to vegetation. My data show that my third hypothesis that humans intentionally used fire to manage resources is consistent with evidence collected in my study area, though further research on this is warranted. Although I strongly suspect that First Nations burning was intentional and widespread in the study area, evaluating this third hypothesis requires that additional fire history reconstructions and ethnographic data be collected elsewhere in perhumid rainforests (Table 4.1).

4.5.1 Temporal attributes of historic fire activity

Low- and mixed-severity fires occurred approximately every 39 years for almost six centuries prior to cultural changes in the region in the late 19th and early 20th centuries (Hoffman et al. 2016b). Because the process of fire scarring is highly variable and dependent on geographic location and fire tolerance of tree species (Tepley and Veblen, 2015), I provide a minimum estimate of the number of historic fire events (16) and acknowledge that the distribution of fire intervals recorded at the plot level is skewed to fire intervals that are longer than those of the composite MFI expressed at the stand or

landscape scale (Appendix C1: Table C1.3). This is expected since not every fire event recorded in the study area was intense enough to scar the majority of trees, and differences in fuel moisture and fuel availability resulted in unburned areas within fire boundaries (Johnson and Gutsell, 1994).

The ecological effects of fire activity in coastal temperate rainforests are assumed to be localized with year-round wet conditions overriding human ignitions (Whitlock et al. 2015). In perhumid rainforests, climate is assumed to be the primary driver of the distribution and composition of vegetation and lightning-ignited fires are estimated to occur every few hundred or even few thousand years (Lertzman et al. 2002; Daniels and Gray, 2006). Despite these findings, previous research on human and climate drivers of fire activity in my study area indicates that First Nations may have utilized favourable climate conditions such as periodic extreme droughts to manage resources with fire (Macias Fauria and Johnson, 2006; Hoffman et al. 2016b). Although I were unable to determine the exact timing of fire events, ethnographic accounts of intentional burning in nearby areas suggest that burning occurred in the late summer after berries had been harvested and the vegetation was drier (Turner, 1999; Turner, 2014).

Human-driven fire activity has also been documented in other island environments and temperate rainforests globally. For example, in New Zealand where lightning is rare, human ignitions explain much of the variation in historic fire activity and in regional patterns of vegetation (McWethy et al. 2013). Researchers have also demonstrated that First Nations were an important ignition source in coastal temperate rainforests in

Patagonia and Tasmania (Whitlock et al. 2015). Although lightning ignitions are also relatively rare in perhumid coastal temperate rainforests in British Columbia, to this point humans have not been considered drivers of historic fire activity.

4.5.2 Spatial aspects of historic fire activity

Several studies in North America have been unable to distinguish human influences on historic fire activity because the majority of research has taken place in fire-prone ecosystems where it is difficult to differentiate between lightning and human ignitions (Lepofsky et al. 2003; Bowman et al. 2011; Walsh et al. 2015). Though the impact and scale of historic anthropogenic burning in perhumid coastal temperate rainforests in British Columbia may seem geographically localized, a closer look at the study area reveals that although the majority of fires were relatively small (~ 3 ha) and low-severity, the location of fires near former habitation sites is a significant finding (Fig. 4.3). There are ~ 275 archaeological sites in the study area alone (McLaren et al. 2015) and thousands of sites in coastal temperate rainforests in British Columbia. The density of archaeological sites, together with compelling evidence of anthropogenic burning presented in this study, suggest that the scale of human-caused fires (whether intentional or not) in perhumid rainforests and their lasting ecological effects may be greater than previously believed.

4.5.3 Patterns of lightning strikes

Lightning-strike densities are strongly influenced by the proximity of cold-water bodies, elevation, and diurnal heating and cooling cycles over land masses (Burrows and Kochtubajda, 2010). Unlike interior and mainland areas of Canada with seasonal patterns

of lightning, lightning occurs year-round in offshore and Pacific coastal regions of British Columbia (Burrows and Kochtubajda, 2010). Inland areas of British Columbia have on average 15-33 days of lightning annually, compared to an average of 1-10 days of lightning in Pacific coastal regions (Burrows et al. 2002). Lightning strike densities in inland areas of British Columbia follow predictable temporal and spatial patterns, are positively correlated with elevation (most common at ~800 m asl) and occur in clusters where one fire ignition is estimated for every 50 lightning strikes (Burrows and Kochtubajda, 2010). This pattern is opposite to the spatial and temporal pattern of lightning observed in the study area, where lightning is most common in the winter months and strike densities are highest over the ocean (Fig. 4.5). Although I acknowledge that the lightning dataset used in this study is short (15 years) and may not be representative of previous climatic periods, I find that characteristics of contemporary lightning strike densities do not correspond to, or appear to explain, the observed patterns of fire activity in the study area (Table 4.1).

4.5.4 Ecological legacies of historic fire activity

Perhumid coastal temperate rainforests contain species that are poorly adapted to fire disturbances and lack traits to survive and colonize even after low- and moderate-severity fires (Banner et al. 2005). Western redcedar, western hemlock, and Sitka spruce have low resistance to fire because of their relatively thin bark and shallow root systems (Agee, 1993). Despite these characteristics, I found that western redcedar trees were able to repeatedly endure low-severity fires and may be more fire tolerant than previously assessed. Western redcedar is a culturally important and highly valued species with many uses including canoe and house plank construction, as well as for firewood, clothing, and

medicine (Hebda and Mathewes, 1984; Turner, 2014). Frequent landscape burning directly behind habitation sites may have encouraged western redcedar regrowth and increased the density of culturally modified and fire-scarred western redcedar (Table 4.1) (Turner, 1999).

First Nations may have also intentionally increased the abundance of specific plants and affected the diversity of plant communities by regulating the size, frequency, intensity, and timing of fires (Turner, 1999; Lepofsky and Lertzman, 2008; Turner, 2014).

Although it is difficult to infer intentional fire-management systems from a forest that last burned more than 120 years ago, I found that the abundance of culturally important plants was significantly positively correlated with fire activity (Fig. 4.4; Appendix C1: Table C1.4). These plants were clustered at low elevations near former habitation sites in open, flat, and easily accessible areas. An increase in the presence of fire-scarred trees and burned CMT's 20-30 m from the present-day forest edge (areas with no belowground evidence of shell midden) suggests that fires may not have started in the immediate vicinity of homes and were separate from habitation sites (Table 4.1). Oral histories have confirmed that habitation sites were often located very close (within a short walking distance ~ five minutes) to wetland burning sites on the Olympic Peninsula in Washington (Wray, 2009) and directly behind village sites in Haida Gwaii and on Vancouver Island (Turner, 1999). Potential fire-management techniques that may have been utilized close to habitation sites in the study area include using fire to fall large western redcedar trees to make canoes and partially girdling and burning trees to create dry and easily accessible firewood in wet winter months (Turner, 1999; Turner, 2014). A

counter-argument to the intentional use of fire in these sites may be the fact that fires were not more common on otherwise identical nearby control island sites, where the risk of intentional fire management to infrastructure was not present.

Although there are many documented plant-management techniques used by First Nations groups, burning was used in the Pacific Northwest of North America to maintain open habitats, which are typically richer in food resources compared to closed-canopy coniferous forests (Boyd, 2002; Storm and Shebitz, 2006; Wray, 2009). These open areas may have been easily accessible resource sites where berries were picked, trees were thinned, and plants were gathered after burning (Appendix C: Table C1.2). Burning in different years and seasons may have created a diversity of forests in various stages of succession, thereby increasing the abundance and productivity of food plants in differing life stages (Lepofsky and Lertzman, 2008). There have been no documented fire ignitions or suppressions in the study area since the last detected fire in 1893. The abrupt end to fire activity in the late 19th century coincides with the forced relocation of First Nations groups to other islands or mainland areas of British Columbia (Table 4.1).

First Nations burning left distinct temporal and spatial legacies in my study region. My evidence is consistent with the hypothesis of intentional human fire ignitions, but also partially consistent with accidental ignitions (Table 4.1). I conclude that the history of fire on this landscape is largely anthropogenic, likely representing a mix of fires that were part of a system of intentional management and fires that were ignited unintentionally near habitations. Reconstructing fire histories in old growth coastal temperate rainforests

is complex and combining multiple lines of evidence is required to test hypotheses about the origins of fires. I can pose the hypothesis, however, that centuries of First Nations burning (whether intentional or accidental) have left land-use legacies, which include a mosaic of vegetation types and successional stages, and an increase in the abundance of culturally important plants. A cessation of First Nations burning can explain why fire disappeared from the study area at the end of the 19th century, in contrast to its occurrence on average every 39 years over the previous six centuries.

Table 4.1 Three hypotheses were tested using a weight-of-evidence approach to infer the spatial and temporal attributes of historic fire activity and to assess whether fire ignitions were human- or lightning-caused. The hypotheses and predictions for each test are detailed in the table below. The bold text indicates whether the prediction is consistent with lightning or human ignitions. Multiple lines of evidence are consistent with intentional ignitions by humans.

Hypotheses	Ignition source	Prediction: frequency	Prediction: intensity	Prediction: size	Prediction: spatial attributes	Prediction: temporal attributes	Prediction: forest age	Prediction: plant community composition	Prediction: 20 th century changes	Findings
1. Non-human ignitions	Lightning	Rare, centennial/millennial scales (Lertzman et al. 2002) Not consistent	Mixed- and high-severity (Agee, 1996) Not consistent	Large, patchy fires (Agee, 1996) Not consistent	High and mid elevations, rocky outcrops (Burrows et al. 2002) Not consistent	Long fire return intervals (Lertzman et al. 2002) Not consistent	Old growth >250 years (Daniels and Gray, 2006) Not consistent	Natural pattern in distribution of food plants Not consistent	No changes to fire activity (Daniels and Gray, 2006) Not consistent	Evidence is not consistent with this hypothesis
2. Human ignitions (accidental)	Human	Less frequent decadal/centennial scales (Turner, 1999) Not consistent	Low- and mixed-severity (Turner, 1999; 2014) Consistent	Small fires (<3 ha) (Turner, 1999; Hoffman et al. 2016b) Consistent	Fires located within former habitation sites Consistent	No cyclical pattern Not consistent	Old growth, younger cohorts near former habitation sites Consistent	Natural pattern in distribution of food plants Not consistent	No fire ignitions in 20 th century (Hoffman et al. 2016b) Consistent	Some of the evidence is consistent with this hypothesis
3. Human ignitions (intentional)	Human	Frequent interannual/decadal scales (Turner, 1999; 2014; Lepofsky et al. 2003) Consistent	Low- and mixed-severity (Turner, 1999; 2014) Consistent	Small fires (<3 ha) (Turner, 1999; Hoffman et al. 2016b) Consistent	Fires located on the edges of former habitation sites (Turner, 1999; Wray, 2009; Hoffman et al. 2016a,b) Consistent	Cyclical pattern (~19 years) in plots surrounding former habitation sites (Turner, 1999) Consistent	All aged stands, open areas and evidence of even-aged cohorts from previous fires Consistent	Food plants associated with burned areas in higher densities (Turner, 1999; 2014) Consistent	No fire ignitions in 20 th century (Lepofsky et al. 2003) Consistent	Evidence is consistence with this hypothesis
Supporting Data	Real time lightning data, fire locations, fire timing	Fire history analyses, fire scars (N=99), post-fire cohorts (from 4000 tree cores)	Fire-scarred trees (N=99), fire maps	Point locations of fire scars (PFI), fire maps	Fire scars (N=99), post-fire cohorts (from 4000 tree cores), lightning strike data (15 years), transects	Fire scars (N=99), post-fire cohorts (from 4000 tree cores)	Fire history plots (N=30), transects (N=20)	Plant community and composition plots (N=30), transects (N=20), oral histories in nearby and similar ecosystems	Fire scars, fire maps, time since most recent fire, historical data	Ethno-historical accounts of the season and timing of fires and the sites

					(n=20), oral histories in similar systems					selected for burning are <i>required</i>
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Table 4.2 Model selection using Akaike Information Criterion for small sample sizes (AIC_c) for 12 Generalized Linear Mixed-effects Models (GLMM) that describe a) the density of western redcedar in vegetation plots; and, b) the abundance of traditional plants with six fixed predictor variables (elevation, slope, aspect, mean fire interval [MFI], distance to habitation site in metres, and distance to shoreline in metres), and one random model effect (vegetation type [four types]). K = number of model parameters, ΔAIC_c = change in AIC score from the top model, w_i = AIC_c model weight, ER = top model weight divided by i model weight. Full results and model validation are provided in Appendix C2.

a) Density of western redcedar in vegetation plots					
Model	K	ΔAIC_c	w_i	ER	Parameters
1	1	0.00	0.26	1.00	MFI
2	2	1.84	0.10	2.60	MFI, distance to habitation
3	2	2.14	0.10	2.60	MFI, slope
4	2	2.30	0.08	3.25	MFI, aspect
5	2	2.65	0.07	3.71	MFI, elevation
6	3	3.02	0.06	4.33	MFI, distance to habitation, slope
7	5	3.71	0.05	5.20	MFI, distance to habitation, slope, aspect, vegetation type

b) Abundance of traditional plants					
Model	K	ΔAIC_c	w_i	ER	Parameters
1	2	0.00	0.20	1.00	Distance to habitation, elevation
2	4	0.03	0.20	1.00	Distance to habitation, elevation, slope, MFI
3	4	0.49	0.16	1.25	Distance to habitation, elevation, slope, aspect
4	5	0.91	0.13	1.53	Distance to habitation, elevation, aspect, slope, MFI
5	3	1.80	0.08	2.50	Distance to habitation, elevation, MFI

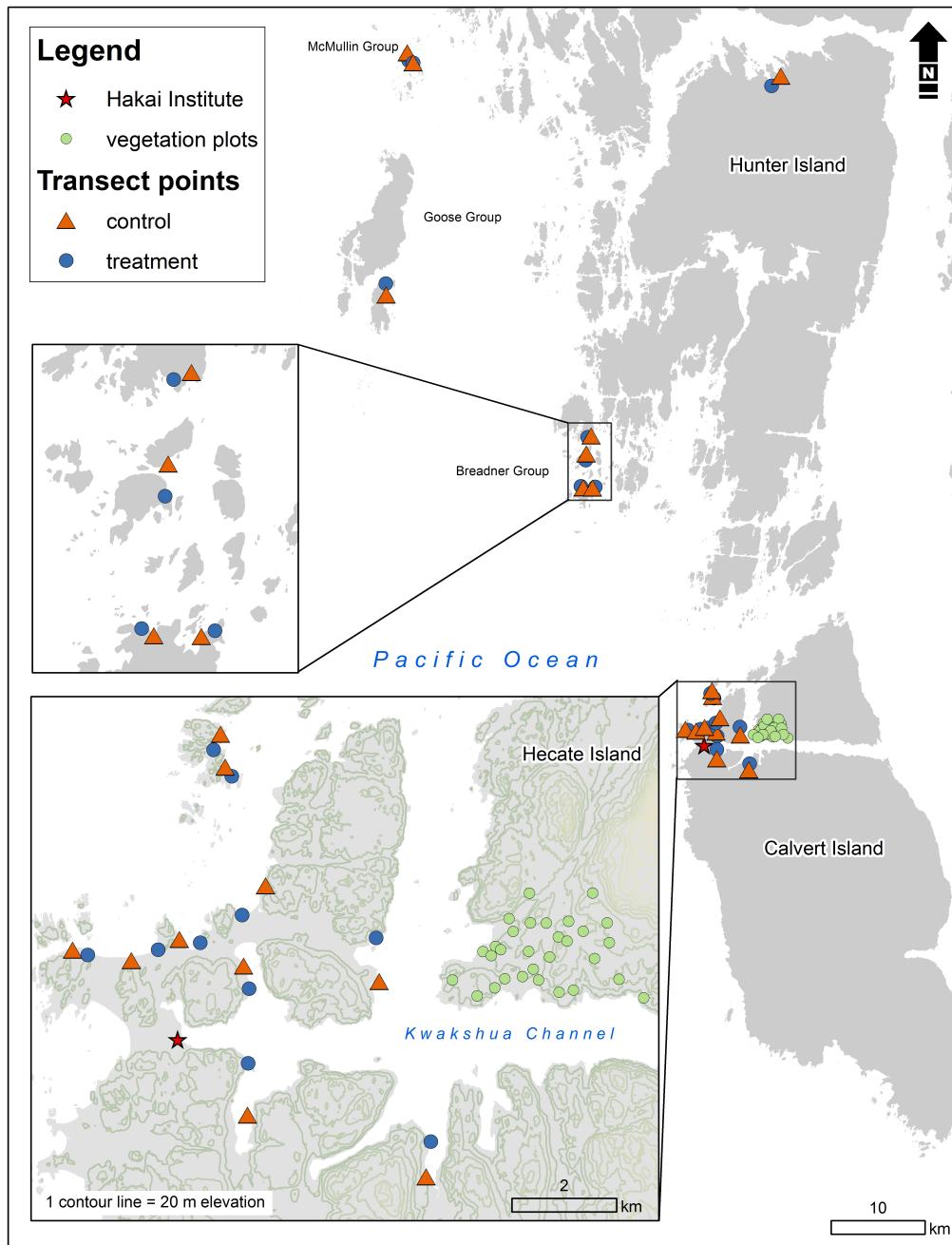
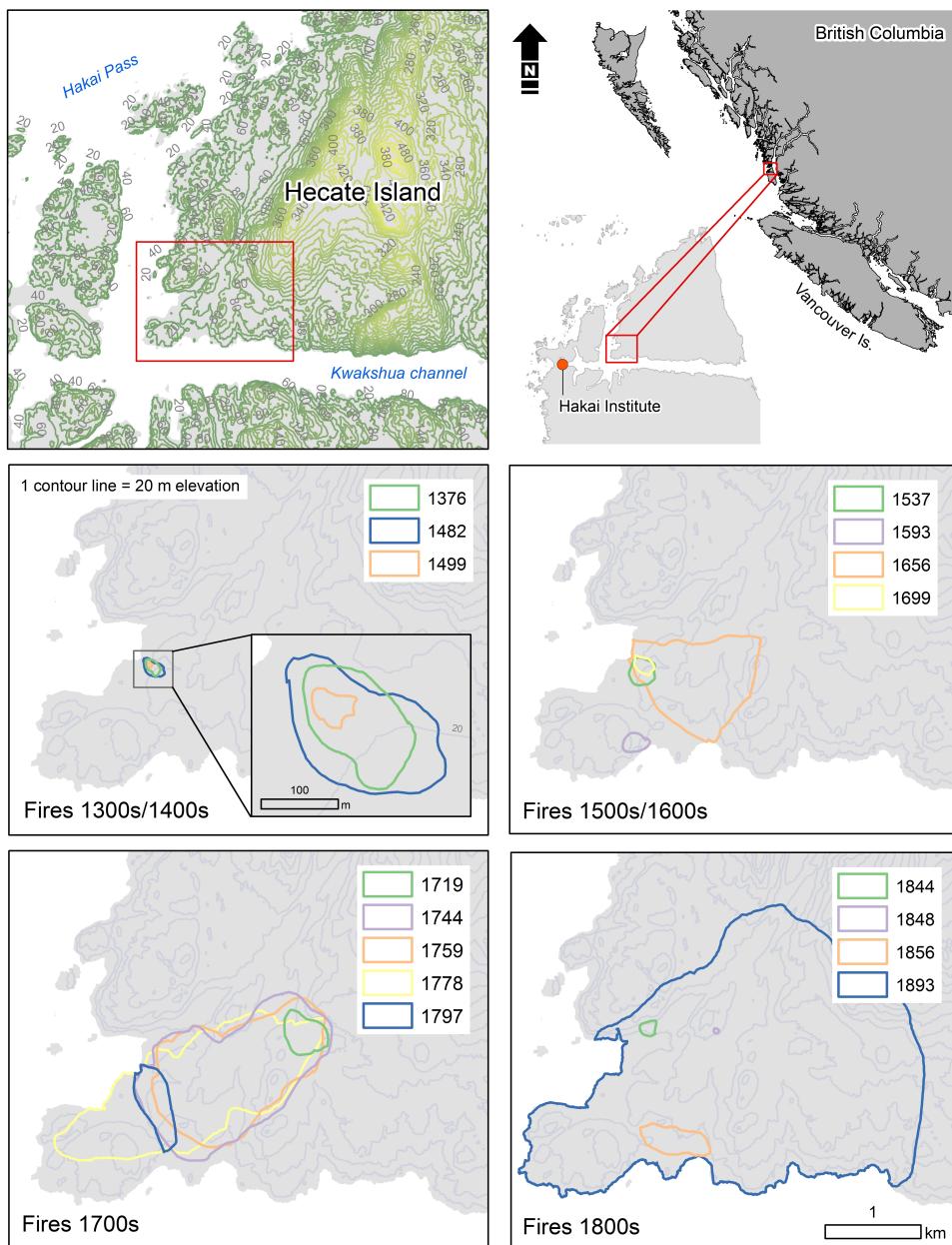


Figure 4.1 The location of the study area on the Central Coast of British Columbia, Canada. Triangles and blue circles represent the locations of the 20 transects (former habitation and control sites) used to assess the association between fire activity and humans. The star symbol is the location of the Hakai Institute. The map inset provides the location of the 30 fire history plots on Hecate Island.



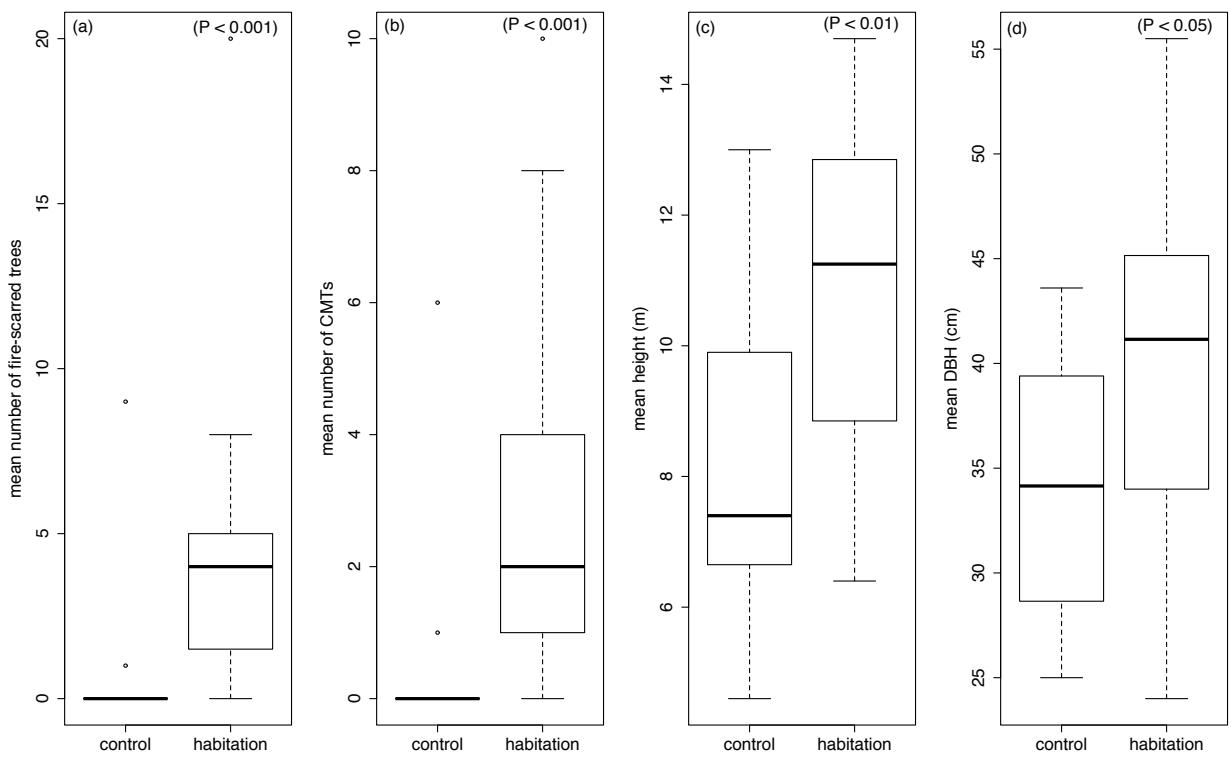


Figure 4.3 Box and whisker plots of the presence of fire evidence in forests surrounding former habitation and control sites ($n = 20$) located on 15 islands throughout the region (see Fig. 4.1). Boxes represent the second and third quartile ranges and the centreline is the median. Circles represent outliers. The abundance of fire-scarred trees ($P < 0.001$, Fig. 4.3a) and fire-scarred culturally modified trees (CMTs) ($P < 0.001$, Fig. 4.3b) were significantly greater in transects near former habitation sites. Trees were also significantly taller ($P < 0.001$, Fig. 4.3c) and had larger diameter at breast height (DBH) ($P < 0.05$, Fig. 4.3d) in former habitation sites compared to control sites.

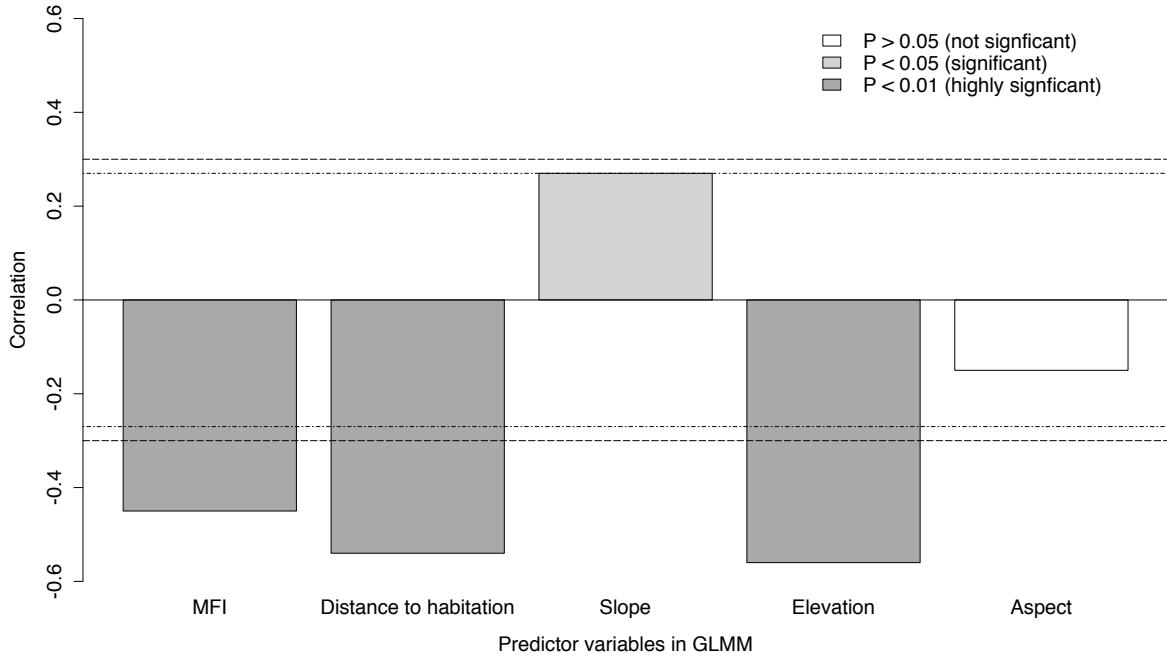


Figure 4.4 The correlation between predictor variables and the abundance (percent cover) of traditional plants found in the study area. Selected variables are the final predictors included in the Generalized Linear Mixed-effects Model (GLMM) (see Table 4.2b), and include the mean fire interval (MFI), distance to former habitation site, slope, elevation, and aspect. Lines and grey shading present 95% and 99% confidence intervals.

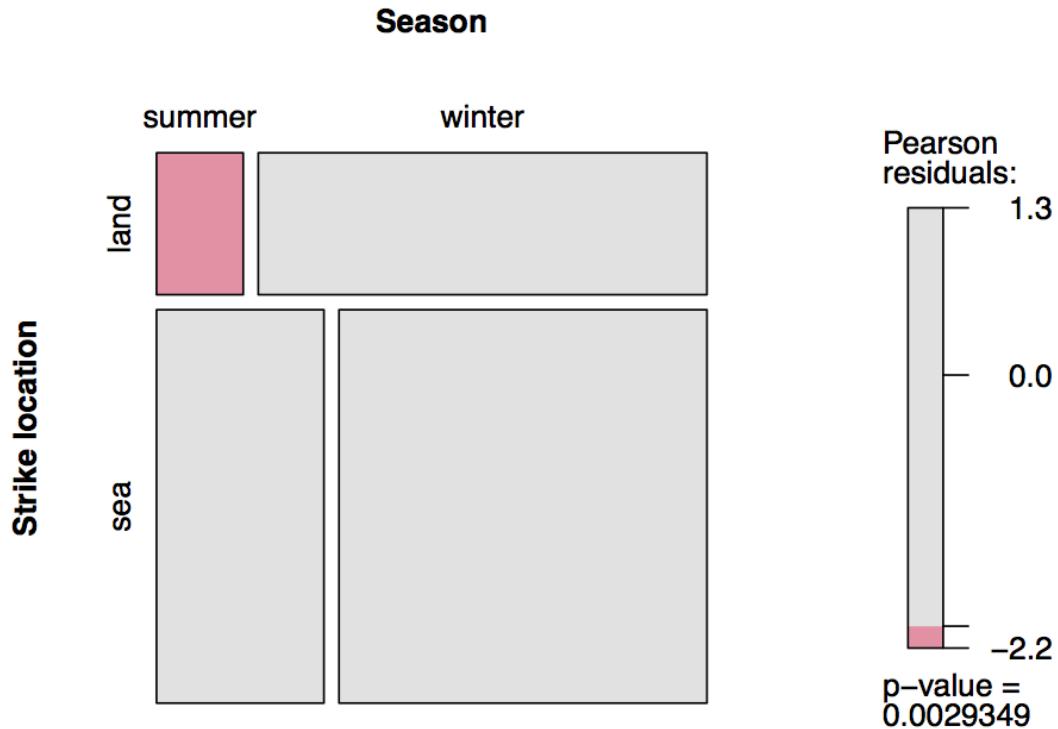


Figure 4.5 The probability of lightning striking land in summer (shaded in pink) is significantly lower ($P = 0.002$) than lightning striking the sea in winter. The size of each rectangle is proportional to the observed frequencies of lightning recorded during 1999–2014. Rectangles are shaded to reflect the magnitude and significance of the Pearson residuals. Low Pearson residuals indicate a good model fit.

Chapter 5: Ecological legacies of fire detected using plot-level measurements and LiDAR in an old growth coastal temperate rainforest

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5.1 Abstract

Vegetation succession following fire disturbances has long been of interest in ecology, but the evolution of landscape pattern and structure after low-severity ground fires is poorly understood. Historic fire disturbances in coastal temperate rainforest ecosystems are not well documented and time since the most recent fire is largely unknown. I use 6000 crossdated tree cores together with a 700 year fire history reconstruction derived from a British Columbia perhumid coastal temperate rainforest to assess differences in ecosystem heterogeneity in burned and unburned forest stands. I assess how vegetation has recovered after centuries of repeated, low-severity fires. My comparative analysis of airborne laser scans with field-based ecological methods focuses specifically on how remote sensing techniques can be used together with plot-level measurements to detect and characterize historic fire disturbances. I sampled 70 plots for stem density, stand structure, and stand composition. I expected lower biomass and stem density in plots with historic fire activity and overall higher conifer diversity. Trees in burned plots were significantly taller, more productive, and stem density was less than half that of unburned plots despite 124 years since the most recent fire. LiDAR analyses confirmed that historically burned forest plots had lower canopy cover and greater canopy complexity. While LiDAR is an important tool to bridge the spatial information offered by plot-level measurements to larger area characterizations, field-based measurements are still required to validate differences in community structure and composition in my temperate rainforest study area. My analysis provides an improved understanding of fire legacies and temperate rainforest structure, which increases our ability to detect fire disturbances in heterogeneous forests and is important for successful forest resource management.

5.2 Introduction

Ecological memory or the degree to which a landscape is shaped by its past patterns and processes is important to how ecosystems respond to disturbance and can be identified in the physical structure of vegetation, soil substrate, and resource availability (White and Pickett, 1985; Peterson, 2002; Johnstone et al. 2016). Fire can be both a natural and anthropogenic disturbance that is nearly ubiquitous in terrestrial ecosystems (Bowman et al. 2009; Whitlock et al. 2010). Fire affects successional vegetation patterns by opening canopies, consuming horizontal and vertical fuels, preparing seedbeds and changing soil substrate and water hydrology (McKenzie et al. 2011; Bolton et al. 2015). Although the feedback between fire and landscape pattern is culturally dependent and ecosystem specific (Bowman et al. 2011), recently burned forests often have characteristic patterns of spatial variability that can be detected with remote sensing techniques such as light detection and range systems (LiDAR; McKenzie et al. 2011). Nevertheless, there is little understanding of how these patterns change over larger spatial and longer temporal scales, specifically when fire disturbances are low-severity and much of the forest structure remains intact as standing live trees (Falkowski et al. 2010; Goetz et al. 2010; Krasnow et al. 2016). The rate of vegetation recovery and legacies of fire disturbance depends on several factors including time elapsed since the most recent fire event (TSF), and the fire frequency, fire severity, fire extent, and fire-sensitivity of vegetation (Foster et al. 1998; Johnstone et al. 2016; Stevens-Rumann and Morgan, 2016). Species life histories and local site factors such as topography also influence vegetation recovery (Foster et al. 1998; Bartels et al. 2016).

Ecologists are often challenged by the scale limitations of field assessments, which constrain their ability to compare plot-level measurements to landscape-level processes (Swetnam et al. 2015). This is especially true in heterogeneous landscapes where plot-level measurements provide important inventories of historic disturbances and vegetation succession, but are limited in their spatial application (Foster et al. 1998). Fortunately, the capacity to locate and characterize historic fire disturbances is increasing with the widespread availability of remote sensing techniques such as LiDAR, which provide site-specific measurements and regional characterization of vegetation (Bartels et al. 2016). LiDAR has the potential to greatly advance the spatial area of vegetation measured following disturbance, particularly canopy height metrics, and associated biomass (Houghton et al. 2009; Goetz et al. 2010; Bolton et al. 2015). However, the potential of this technology to reconstruct historic fire disturbances remains largely untested, especially in complex and heterogeneous landscapes where natural variability in forest structure is high (Swetnam et al. 2011). This is especially true of perhumid coastal temperate rainforests like those in British Columbia, Canada, where historic fire disturbances are not well documented (Daniels and Gray, 2006; Hoffman et al. 2016a, Hoffman et al. 2016b; Hoffman et al. 2017).

Analyses that bridge a range of spatial scales from small patches to broader, landscape-level legacies and are important when studying disturbances such as fire which are controlled by several processes operating at different scales (Falk et al. 2007). Quantifying relationships between fire history, forest structure, and forest productivity across large forested areas may provide an improved understanding of both spatial and

temporal variability in heterogeneous coastal temperate rainforests, which is important for ongoing forest resource management (Tepley et al. 2013). Utilizing comparative analyses of airborne LiDAR with plot-level measurements, I assess differences in burned and unburned forests in a perhumid temperate rainforest on the Central Coast of British Columbia. I ask the following questions: 1) What fire legacies are apparent in forest stand structure and composition over a 124 year post-fire period? 2) How does historic fire activity affect regeneration dynamics through changes to canopy structure and stand density? 3) Can LiDAR detect differences in forest stand structure that are apparent in plot-level measurements? I hypothesize that burned forests remain more open and contain more productive (taller and wider) trees with higher conifer diversity.

5.3 Materials and Methods

5.3.1 Study area

The study area encompasses a 20 km² area located on Hecate and Calvert Islands (North 51° 39' Latitude, West 128° 04' Longitude) within the Hakai Lúxvbális Conservancy on the Central Coast of British Columbia, Canada (Fig. 5.1). The coastal margin of British Columbia has many small islands characterized by exposed and rocky homogenous quartz diorite and granodiorite bedrock, subdued terrain, and elevations ranging from sea level to approximately 1000 m (Roddick, 1996). Cool temperatures (average annual ~7° C, average summer ~12° C) coupled with locally abundant (~4000 mm) and year-round rainfall distinguish this temperate climate region (Banner et al. 1993; Banner et al. 2005). The study area is located within the very wet hypermaritime subzone (CHWvh2) of the Coastal Western Hemlock biogeoclimatic classification (Meidinger and Pojar, 1991).

Excess soil water regulates this environment and subtle variations in slope or drainage result in significant differences in forest productivity (Banner et al. 2005). Although several vegetation types have been categorized in the study area (Thompson et al. 2015), four types dominate along a gradient of productivity and are defined by species and closely associated landforms (Banner et al. 1993; Banner et al. 2005). Productive (zonal) forests are found in nearshore and riparian areas with large-diameter western redcedar (*Thuja plicata* Donn ex D. Don.) and western hemlock (*Tsuga heterophylla* [Raf.] Sarg.), and lesser amounts of yellow-cedar (*Cupressus nootkatensis* [D.Don] Farjon and Harder) and Sitka spruce (*Picea sitchensis* [Bong.] Carr.) (Meidinger and Pojar, 1991). Bog forests exhibit stunted growth forms and are located on hill slopes dominated by western redcedar, yellow-cedar, western hemlock, and shore pine (*Pinus contorta* var. *contorta* Douglas ex Louden) (Klinka et al. 1996). Bog woodlands are the most common vegetation type in the study area and are comprised of patchy mosaics of forested and unforested sites in subdued or rolling terrain (Banner et al. 1993). These forests contain roughly equal densities of western redcedar, yellow-cedar, and shore pine with lesser amounts of mountain hemlock (*Tsuga mertensiana* [Bong.]) (Klinka et al. 1996). Blanket bogs are nutrient-poor, sparsely forested wetland areas that contain small amounts of shore pine and yellow-cedar (Banner et al. 1993).

Compared to most of coastal British Columbia, the Central Coast has experienced very small fluctuations in sea levels (+/- 2 m) during the Holocene (Shugar et al. 2014). This allowed First Nations to continuously inhabit the region for >13,000 years until European contact in the late 18th and 19th centuries (McLaren et al. 2014; McLaren et al. 2015).

Lightning-ignited fires are rare and First Nations likely played an important role in igniting fires and controlling the spatial and temporal aspects of historic fire activity (Hoffman et al. 2016a; Hoffman et al. 2016b). Ongoing research suggests that fire in the study area may have been intentionally used as a tool for resource management (Trant et al. 2016; Hoffman et al. 2017), but little specific ethnographic information is available regarding how First Nations used fire to control vegetation succession (Turner, 1999; Turner, 2014). Historic fire events were composed of low- and mixed-severity fires that did not result in significant stand mortality (Hoffman et al. 2016b). Although more than a century has passed since the most recent fire event, the ecological legacies of historic fire activity, such as fire-scarred trees and even-aged cohorts remain visible in the study area today. Colonists never settled the region and there is no history of industrial logging or mining (McLaren et al. 2015).

5.3.2 Ecological field sampling

Terrestrial ecosystem maps and satellite imagery taken in 2012 were used to select the locations of 38 plots (11.28 m radius [0.04-ha]) with a stratified random sampling design representing the range of elevations, aspects, slopes, and four dominant vegetation types on Hecate and Calvert Islands (Fig. 5.1). Twenty-seven plots were sampled within a low- and mixed-severity 287-ha fire on Hecate Island that most recently burned in 1893 (Fig. 5.1, hereafter ‘burned’ plots). Forest structure in burned plots was previously reconstructed with a network of 45 living fire-scarred trees (containing 99 fire scars) and 4000 tree cores (Hoffman et al. 2016b; Hoffman et al. 2017). Burned plots experienced repeated low- and mixed-severity fires of varying sizes (0.01-287-ha) from the first

detected fire in 1376 until the last detected fire in 1893 (Hoffman et al. 2016b). The 1893 fire affected all 27 burned plots.

The 27 burned plots were compared to 11 plots that were selected and sampled with the same methods, but contained no fire scars and had no other aboveground evidence of fire activity (hereafter ‘unburned’ plots) on Hecate and Calvert Islands (Fig. 5.1). Field surveys and the collection and crossdating of an additional 2000 tree cores confirmed the absence of fire scars and post-fire cohorts. This information together with previous radiocarbon dating of charcoal from historic fires deposited in soils confirmed that the 11 plots had not experienced fire activity for at least 1000 years (Hoffman et al. 2016a). The paired design allowed for comparison of plots with similar forest stand structure, vegetation, and topographical position but with differing fire histories. Time and financial constraints limited the ability to perform a balanced sampling design.

In all plots two, 5 mm increment cores were sampled from the bases (~15 cm height) of all trees >7.5 cm diameter at breast height (DBH). The DBH of all trees was measured at 1.3 m from the tree base and the height and species were also recorded. All saplings (<7.5 cm DBH, >15 cm height) and seedlings (< 15 cm height) were characterized (by species, height, and DBH), counted and then destructively sampled in five, 3 m² subplots. A diagram of how subplots were sampled is detailed in Appendix D: Fig. D1.1.

In the laboratory, fire scars, tree cores, and sapling discs were processed using standard dendrochronological techniques (Stokes and Smiley, 1968). Samples were measured and

counted using a Velmex® sliding stage micrometer (precision 0.001 mm) and then statistically verified using the computer program COFECHA (Grissino-Mayer, 2001). For cores that did not reach pith, I used Duncan's (1989) method to calculate the distance to the chronological centre of each tree. The exact year of fire events was determined by crossdating fire scar wedges and tree rings from post-fire cohorts (Johnson and Gutsell, 1994). For age structure analyses, I binned trees together by decade to reduce uncertainty in time to reach coring height (Tepley and Veblen, 2015). I calculated the density of all seedlings, saplings, and trees in four DBH classes in every plot (Table 5.1). All sites (burned and unburned plots) were also recorded using airborne laser scanning.

5.3.2 LiDAR sampling

In addition to the 27 burned and 11 unburned plots, I randomly selected 32 plots (11.28 m radius [0.04-ha]) that were stratified within the four vegetation types and assessed with LiDAR (Fig. 5.1). Each plot was ground-truthed to confirm the presence of fire, but no field sampling was conducted. Of the 32 plots assessed, four plots contained historically burned forests and 27 plots had no aboveground evidence of fire (Fig. 5.1). In total, 70 plots were utilized to characterize historic fire disturbances and train a predictive LiDAR fire disturbance model.

5.3.3 LiDAR data

Discrete return airborne scanning LiDAR was acquired across Hecate and Calvert Islands in August 2012 by Terra Remote Sensing Inc. (Sidney, BC. Canada). LiDAR was collected from 1150 m AGL at 100 Khz with a maximum scan angle of 26 degrees. The resulting point data have an average point density of $2 \text{ points} \cdot \text{m}^{-2}$ with an average vertical

accuracy of 15 cm. A digital terrain surface was generated from classified ground returns using triangular irregular network interpolation and rasterized at a spatial resolution of 1.0 m. The terrain model was used to derive elevation, slope, and aspect as well as to normalize non-ground LiDAR returns to height above ground surface. Vegetation greenness as expressed in the Normalized Difference Vegetation Index (NDVI) was calculated from SPOT6 images acquired on 10 August 2014 and corrected to surface reflectance. LiDAR data were processed using the FUSION software (version 3.5) (McGaughey, 2015). To compare differences in fire effects to vegetation with LiDAR I used vegetation metrics such as return height percentiles, percent returns above 1 m, average height, canopy relief ratio $[(h_{\text{mean}} - h_{\text{min}}) / (h_{\text{max}} - h_{\text{min}})]$, and topographical data. LiDAR vegetation metrics were calculated for every plot in the study area on 400 m² circular plots centred on the plot centre coordinates ($n = 70$) (Fig. 5.1).

5.3.5 Statistical analyses

I used a nested analysis of variance (ANOVA) within the four vegetation types to compare differences in stand composition and structure between burned and unburned plots using the linear and non-linear mixed-effects model package ('nlme') in R statistical software (R Development Core Team, 2016; Pinheiro et al. 2017). The site type (burned and unburned plots) represents a random factor at the top of the hierarchy and the sample type (four vegetation types) is the random nesting factor. Including sub-replication (the four vegetation types) in my hierarchical design reduces the unexplained variation and increases the power of the test for the main treatment effect (fire activity). Because the design was not balanced, it was best modelled using a linear mixed-effects model. The data were aggregated and the linear mixed effects model was fit. The variance

components of each random effect were assessed to verify the impacts of site and forest type. I calculated the 95% confidence intervals of the random effects (based on Markov chain Monte Carlo sampling) following the methods of Logan (2010). Detailed statistical analyses and model validation are provided in Appendix D: Table D2.1.

5.4 Results

5.4.1 Ecological field data

My analyses revealed significant differences in forest stand metrics between burned and unburned plots. Trees in burned plots were taller compared to unburned plots with the same vegetation types (*nested analysis of variance*: $F = 14.788$, $d.f. = 1$, $P < 0.0001$; Fig. 5.2a). Burned plots were also significantly younger than unburned plots in DBH classes >7.5 cm (*nested analysis of variance*: $F = 279.52$, $d.f. = 1$, $P < 0.001$; Fig. 5.2b). Only 16% of trees in burned plots were old growth (>250 years) compared to 58% of trees in unburned plots (Fig. 5.3; Appendix D: Table D2.2). Unburned vegetation plots had 132% more stems >7.5 cm DBH per hectare compared to burned plots (Table 5.1).

Regenerating seedlings and saplings <7.5 cm DBH in burned plots were significantly smaller when compared to unburned plots (*nested analysis of variance*: $F = 218.67$, $d.f. = 1$, $P < 0.001$; Fig. 5.2c). The average combined height of seedlings and saplings was 23 cm in burned plots compared to 53 cm in unburned plots. Seedlings and saplings in burned plots were also significantly younger (*nested analysis of variance*: $F = 211.72$, $d.f. = 1$, $P < 0.001$; Fig. 5.2d). The average age of seedlings and saplings in burned plots was 12 years compared to 43 years in unburned plots.

I found no significant difference between DBH in burned and unburned plots across the four forest types (*nested analysis of variance*: $F = 0.3926$, $d.f. = 1$, $P = 0.531$; Appendix D: Fig. D1.2). Six conifer species (western redcedar, yellow-cedar, western hemlock, shore pine, Sitka spruce, and mountain hemlock) were present in three of the four assessed DBH classes in both burned and unburned plots (Table 5.1). Sitka spruce and mountain hemlock were not present in the <7.5 cm DBH class in unburned plots (Table 5.1).

5.4.2 LiDAR data

LiDAR confirmed that trees in burned plots were significantly taller than trees in unburned plots based on 95th percentile height (*nested analysis of variance*: $F = 7.809$, $d.f. = 1$, $P = 0.006$; Fig. 5.4a). Canopy cover, measured by LiDAR as the proportion of returns above 1 m was also lower in burned plots compared to unburned plots (*nested analysis of variance*: $F = 7.86$, $d.f. = 1$, $P = 0.008$; Fig. 5.4b). Canopy relief ratio, an indicator of canopy complexity shows that burned plots had a greater spread in vertical biomass than unburned plots (*nested analysis of variance*: $F = 5.23$, $d.f. = 1$, $P = 0.028$; Fig. 5.4c). I found no difference in vegetation greenness measured with NDVI (*nested analysis of variance*: $F = 0.265$, $d.f. = 1$, $P = 0.609$; Appendix D: Fig. D1.3) between burned and unburned plots.

5.5 Discussion

In this study, I examine the legacy of historic fire activity to the height, density, diversity, and structure of forests in my temperate rainforest study region more than one hundred years after the last fire. Although burned plots had fewer trees (Fig. 5.3; Appendix D2:

Table D2.2), I found these plots contained higher conifer diversity than unburned plots (Table 5.1). Trees in burned plots were also taller and younger compared to trees in plots that have not burned in at least 1000 years (Fig. 5.2; Appendix D1: Table D1.2). The persistence of ecological legacies over long temporal and across broad spatial scales suggest that historic fire disturbances continue to affect present day forest structure and composition and will likely continue to shape responses to future disturbances (Peterson, 2002; Johnstone et al. 2016). My comparative approach supports my hypothesis that burned forest stands have taller trees and lower stem density (Fig. 5.2), but a few specific differences emerged between burned and unburned plots. First, trees in burned plots did not contain on average larger DBH as hypothesized (Appendix D1: Fig. D1.2), and second, the Normalized Difference Vegetation Index (NDVI) representing vegetation greenness was not significantly higher (Appendix D1: Fig. D1.3).

I establish through dendrochronological reconstructions of tree ages that surviving trees in burned plots persisted at densities of approximately 75 trees per hectare from 1650–1893 (Fig. 5.5). Sixty-five percent of burned plots contained fire-scarred trees >400 years and of these, 20 individuals exceeded 1000 years and had survived as many as 16 low- and mixed-severity ground fires (Fig. 5.5; Appendix D2: Table D2.3). Burned plots contained fire legacies, such as shade tolerant post-fire cohorts dominated by western hemlock, and pulses of shore pine recruitment (Fig. 5.5). These plots comprised old growth trees (>250 years) with patches of even-aged forest stands compared to unburned plots, which contained three times the amount of old growth trees (Fig. 5.3; Appendix D2: Table D2.3). Old growth trees with varying fire histories continue to undergo

compositional change related to stand histories and current population interactions (Tepley et al. 2013). Direct fire effects include changes to stand density, size structure and establishment rates, and the effects of local competition on growth and mortality (Foster et al. 1998).

The ability of researchers to detect fire legacies depends on the presence of fire-scarred trees and post-fire cohorts as well as the capacity of surviving mature individuals to act as seed sources between fire events (Agee, 1993). Seedlings can also germinate from the soil seed bank, or regenerate from germinating individuals that survived the fire (Bartels et al. 2016). The type of regeneration plays an important role in the timing and recovery of the disturbed forest and in structural development patterns (Oliver and Larson, 1996). Recovery is faster following low-severity fire disturbances that cause minimal impacts to vegetation, roots and soil substrates, and leave much of the existing forest structure intact (Agee, 1993; Foster et al. 1998).

In zonal and bog forest vegetation types, the majority of mature trees survived repeated low-severity ground fires and remained seed sources between fire events (Fig. 5.5). My data confirm that historic fire events occurred approximately every 39 years (Hoffman et al. 2016b). This fire return interval suggests that subsequent fires would have affected the survival of regenerating trees and their ability to sufficiently mature to produce seeds between fire events (Fig. 5.2c; Fig. 5.2d). Therefore, older surviving individuals likely remained an important seed source and may explain why there has been little change in species composition during the last six centuries (Table 5.1; Fig. 5.5). In bog woodland

and blanket bog vegetation types, regeneration occurred via surviving mature trees and through vegetative regeneration by root suckering, stem sprouting, and layering.

Vegetative regeneration is more common when fewer seed sources (mature trees) are present, and can be favourable as vegetation regeneration does not require mineral soil substrates for germination and less time is required for regeneration (Bartels et al. 2016).

Repeat low-severity ground fires likely reduced understory tree densities and decreased the amount of available fuel in burned plots (Fig. 5.4a; Fig. 5.5). Saplings (<7.5 cm DBH) and seedlings (< 15 cm) in burned plots were significantly younger than in unburned plots (12 compared to 43 years) indicating that a denser canopy and higher competition in unburned plots may have resulted in the suppression of understory trees (Fig. 5.2d; Fig. 5.4b). The majority of understory trees in unburned plots were comprised of western hemlock, which can remain suppressed in the understory canopy for decades until disturbances such as tree gap formation and replacement (Fig. 5.3; Lertzman et al. 1996; Bartemucci et al. 2002). Although western redcedar, yellow-cedar, and western hemlock were common regenerating species in both burned and unburned plots, less shade tolerant species such as Sitka spruce and mountain hemlock were not present when I examined saplings and seedlings in unburned plots (Table 5.1b; Appendix D1: Fig. D1.4). These species were found in higher densities in burned plots, which were less dense and had a more open canopy structure, which provided more available light for regeneration (Table 5.1b; Fig. 5.4).

Tree seedlings are less likely to successfully establish and survive when repeat fire disturbances are compounded by climate variability (Kemp et al. 2016). Therefore, differences in stand densities may be a product of fire history and climate combined (Stevens-Rumann and Morgan, 2016). Although I accounted for differences in underlying vegetation patterns in the four vegetation types, a warming climate in the 20th century may have played an important role in forest regeneration and stand dynamics following the 1893 fire (Tepley et al. 2016). Thus, examining differences through time by looking at multiple years since the most recent fire is important, as differences between burned and unburned plots may be explained, in part, by climate variability (Tepley et al. 2013). A comprehensive analysis of tree density along with tree seedling establishment through time with potential climate and topographic correlations would improve our understanding of the mechanisms of recovery following historic fires (Tepley et al. 2016). Unfortunately, this was not possible in my study because too much time has elapsed since the most recent fire.

LiDAR has the ability to represent complex vertical structures and ground surfaces with high precision (Bartels et al. 2016). In heterogeneous landscapes that have experienced high-severity, stand-replacing fires, canopy cover often recovers quickly, while canopy height takes several decades to return to pre-fire conditions (Bolton et al. 2017). Forest metrics derived from LiDAR detected differences in canopy height and complexity in burned plots 124 years after the most recent fire event (Fig. 5.4). I was able to detect fire legacies in our study area with LiDAR because centuries of repeat low-severity fires

continue to influence the present day stand structure and alter regeneration processes resulting in increased canopy complexity and lower canopy cover (Fig. 5.4).

NDVI is a commonly used index to detect and quantify post-fire vegetation recovery after recent fires, but ground vegetation can introduce errors when assessing the effect of fire on the tree canopy. This is especially true of low-severity fires, which require field surveys to determine which parts of the forest strata were affected (Cocke et al. 2005).

The recovery of canopy cover after fires usually returns NDVI to pre-fire values within two or three decades (Frazier et al. 2015). I found almost no difference between the NDVI of burned and unburned plots (Appendix D1: Fig. D1.3). I suspect that this lack of difference can be attributed to the long time elapsed since the last fire as well as the signal from ground vegetation in plots with reduced canopy cover.

The capacity of remotely sensed data to characterize post fire disturbance recovery is increasing as data sources are becoming more widely available and less costly (Swetnam et al. 2015; Gordon et al. 2017). How ecological communities change following fire disturbance is important for ongoing forest management and our understanding of old growth forests facing a rapidly changing climate (Bartels et al. 2016). While remote sensing techniques such as LiDAR are able to characterize broad geographical areas, plot-level measurements remain important calibration and validation tools (Bartels et al. 2016) and may identify processes that can be overlooked by remote sensing methods alone. For example, when I created a predictive model to examine potential locations of historic fire activity outside of the study area, I found that the

natural variability in temperate rainforest structure surpassed the effect of historic fire legacies, which impeded the predictive capabilities of my model. Although remote sensing techniques are often employed to detect and predict the locations of recent fire disturbances in boreal, arid, and temperate forest ecosystems (White et al. 1996; Díaz-Delgado, 2003; Bolton et al. 2017), I find these modeling techniques do not readily detect long-term fire legacies in high biomass, perhumid coastal temperate rainforests. Therefore, it is imperative to integrate plot-level measurements with remote sensing techniques to capture the full range of variability of a landscape or region and reliably inform site-specific forest management (McKenzie et al. 2011; Swetnam et al. 2015).

5.5.1 Management implications

Very little is known of fire disturbances in coastal temperate rainforests, and analyses of contemporary fire activity often do not consider the disruption of traditional fire management systems following the arrival of European colonists in North America (Ryan et al. 2013). In coastal British Columbia, fire suppression and cultural impacts to First Nations groups were initiated at the turn of the century and became widespread in the 1930s (Turner, 1999; Turner et al. 2013). Fire activity was also altered through widespread logging in the 1940s and 1950s, which was followed by the introduction of cattle and grazing in the 1970s (Turner et al. 2013).

The popular and widely accepted view of fire disturbance in perhumid coastal temperate rainforests emphasizes infrequent, high-severity stand-replacing fires occurring at centennial or millennial scales (Agee 1993; Daniels and Gray, 2006). Stand-replacing fire models have been assumed due to the availability of biomass and volatility of fuels,

which characterize western redcedar and western hemlock dominated forests (Agee, 1993; Daniels and Gray, 2006; Whitlock et al. 2010). Until recently, no fire histories had been completed with fire scars in perhumid coastal temperate rainforests (Hoffman et al. 2016a; Hoffman et al. 2016b). My 700 year fire history reconstruction suggests that low- and mixed-severity fires were once much more frequent, and occurred on average every 39 years prior to the forced relocation of First Nations groups in the late 19th century (Hoffman et al. 2016b). Applying stand-replacing models of fire activity with no direct empirical evidence in this region could limit our understanding of the effects of climate change on forest development and greatly underrepresent the diversity of old growth forest structures and development pathways in this ecosystem (Tepley and Veblen, 2015). Current reforestation guidelines in British Columbia hypermaritime forests (CWHvh2) promote diverse and dense plantations while encouraging patches of early seral vegetation (B.C. Forest Practices, 1995). While these practices are in line with stand replacing fire disturbances, they do not represent low- and mixed-severity fires, which historically characterized the region.

5.6 Conclusions

My analysis of LiDAR remote sensing and plot-level measurements of burned and unburned vegetation plots confirm that legacies of low-severity fires continue to shape structural and regeneration patterns in present day forests (Table 5.1). Although it has been more than 120 years since the most recent fire on Heceta Island, burned vegetation plots were distinct in their composition, structure, and productivity when compared to unburned plots. As remotely derived images become increasingly available, remote sensing techniques still require ecological field sampling and plot-level measurements to

accurately compare and validate vegetation metrics (Swetnam et al. 2015). I find that LiDAR is an important tool for mapping and characterizing coastal temperate rainforest structure and improves our understanding of the spatial components of fire. LiDAR can detect differences in forest productivity, stand density, and stand height between burned and unburned plots, but examining multiple spatial scales is necessary to verify species composition, and the severity and temporal properties of historic fire disturbances. My study reveals that fire history and the development and maintenance of old growth forests in heterogeneous landscapes such as British Columbia's old growth coastal temperate rainforests are more complex than previously estimated.

Table 5.1 The density of trees, saplings and seedlings per hectare were binned into four diameter at breast height (DBH) classes (1.5-7.5, 7.6-20, 20.1-40 and >40.1) for each of the six species, which characterize the four dominant vegetation types in the study area in a) 27 burned plots and, b) 11 unburned plots. The symbol (--) indicates that the species was not present.

a) Burned plots (n = 27)						
DBH class and vegetation type	Western redcedar	Yellow -cedar	Western hemlock	Sitka spruce	Shore pine	Mountain hemlock
Density of trees/ha >40.1 DBH						
Zonal forest	120	30	15	20	--	--
Bog forest	87	34	--	--	17	--
Bog woodland	35	10	3	--	--	--
Blanket bog	--	--	--	--	--	--
Density of trees/ha 20.1-40 DBH						
Zonal forest	150	50	100	5	35	30
Bog forest	180	232	76	--	118	11
Bog woodland	33	25	--	--	95	5
Blanket bog	7	4	--	--	--	--
Density of trees/ha 7.6-20 DBH						
Zonal forest	120	45	275	5	--	--
Bog forest	148	100	123	3	126	11

Bog woodland	137	190	3	3	478	15
Blanket bog	38	34	--	--	260	--
Density of trees/ha 1.5-7.5 DBH						
Zonal forest	39166	2600	25000	500	4500	--
Bog forest	29500	6000	21000	--	4000	1000
Bog woodland	35000	6500	5000	--	27500	1000
Blanket bog	11000	6000	2500	--	21500	500
b) Unburned plots (n = 11)						
DBH class and vegetation type	Western redcedar	Yellow -cedar	Western hemlock	Sitka spruce	Shore pine	Mountain hemlock
Density of trees/ha >40.1 cm DBH						
Zonal forest	225	50	--	50	--	--
Bog forest	75	25	--	--	--	--
Bog woodland	25	--	--	--	--	--
Blanket bog	--	--	--	--	--	--
Density of trees/ha 20.1-40 cm DBH						
Zonal forest	275	300	100	--	25	--
Bog forest	325	21	100	--	200	25
Bog woodland	125	100	25	--	50	25
Blanket bog	25	25	--	--	--	--
Density of trees/ha 7.6-20 cm DBH						
Zonal forest	275	100	550	--	25	--
Bog forest	400	125	125	--	350	25
Bog woodland	475	675	200	--	700	50
Blanket bog	25	--	--	--	213	--
Density of trees/ha 1.5-7.5 cm DBH						
Zonal forest	14500	3000	21000	--	--	--
Bog forest	24500	10000	6500	--	2500	--
Bog woodland	54000	29500	2000	--	4500	--
Blanket bog	9000	5000	1000	--	500	--

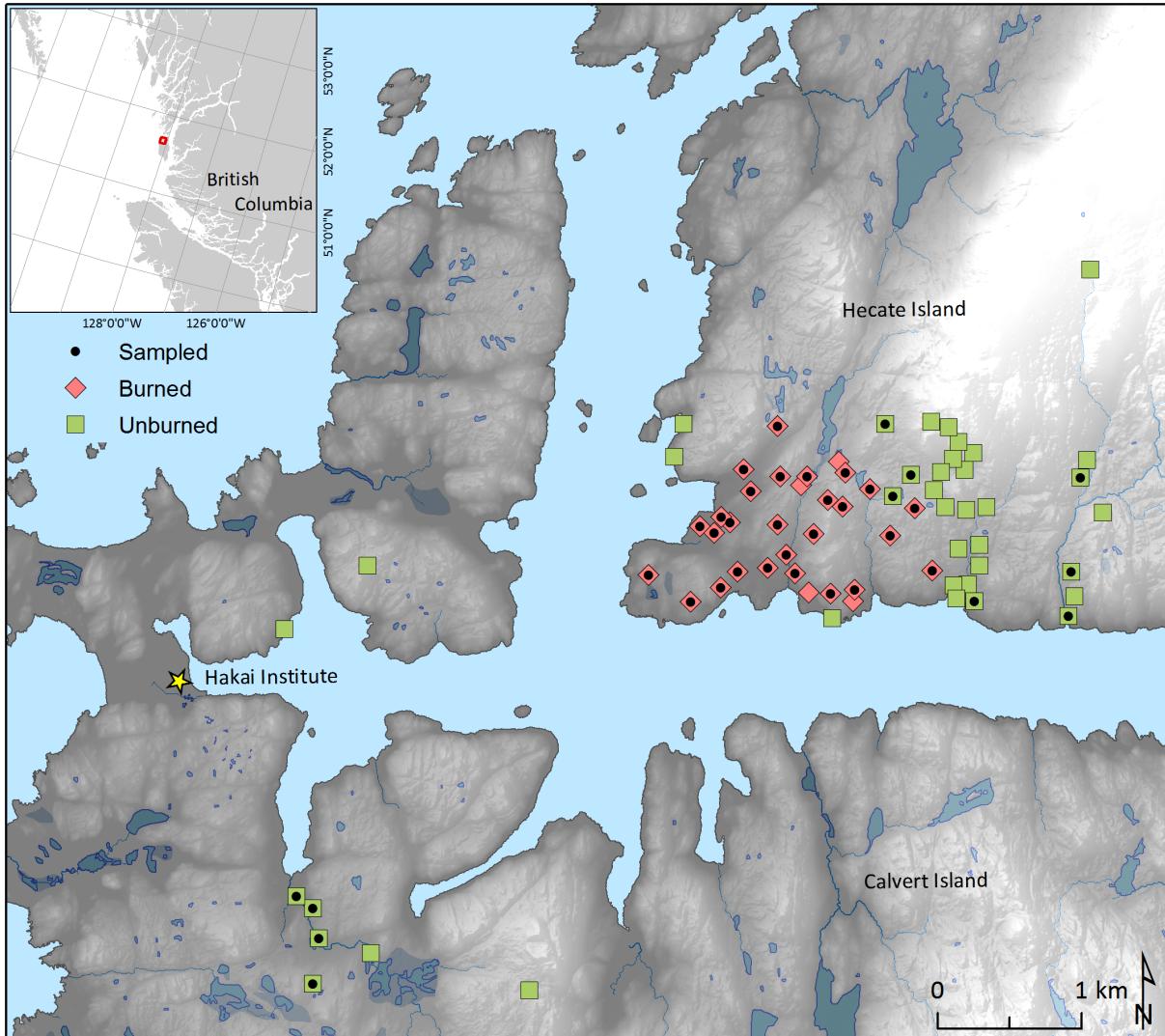


Figure 5.1 The study area encompasses a 20 km² area located on Hecate and Calvert Islands (North 51° 39' Latitude, West 128° 04' Longitude). Burned plots (red diamond symbol) on Hecate Island are within a low- and mixed-severity 287-hectare fire dated with fire-scarred trees to 1893 AD. Sampled burned plots (27) contain a black circle within the red diamond symbol. Unburned vegetation plots (green square) are located on Hecate and Calvert islands and have no evidence of aboveground fire activity (within the last 1000 years). Sampled unburned plots (11) contain a black circle within the green

square symbol. All plots were randomly selected, ground-truthed for fire evidence and included in the LiDAR analysis.

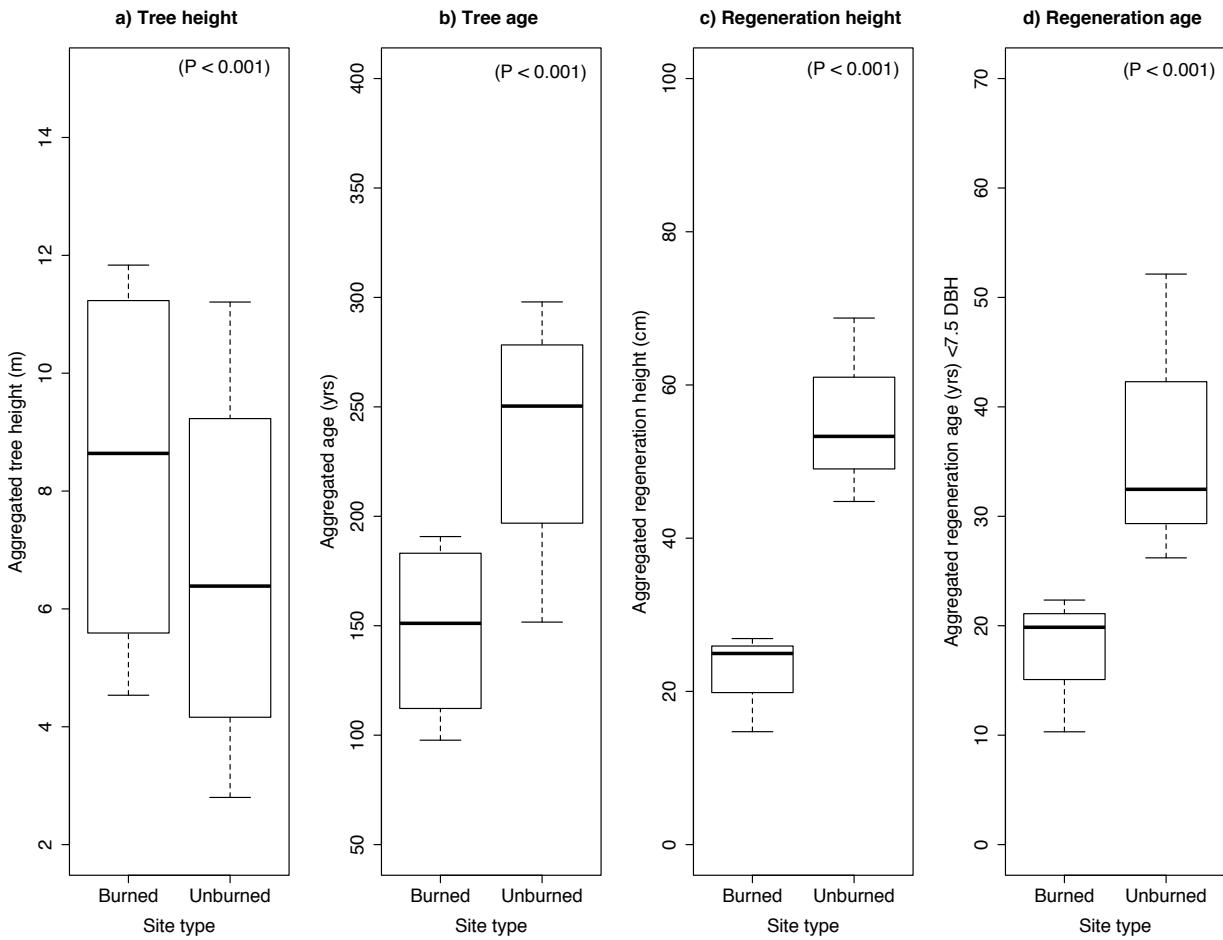


Figure 5.2 The results of the two factor model nested analysis of variance (ANOVA) with unequal sample sizes. Box and whisker plots describe differences in a) mature tree height in metres, b) tree age in years, c) regenerative tree height in metres, and d) regenerative tree age in years between burned and unburned forest plots aggregated across four vegetation types (zonal forest, bog forest, bog woodland, and blanket bog). Boxes represent the second and third quartile ranges, and the centreline is the median.

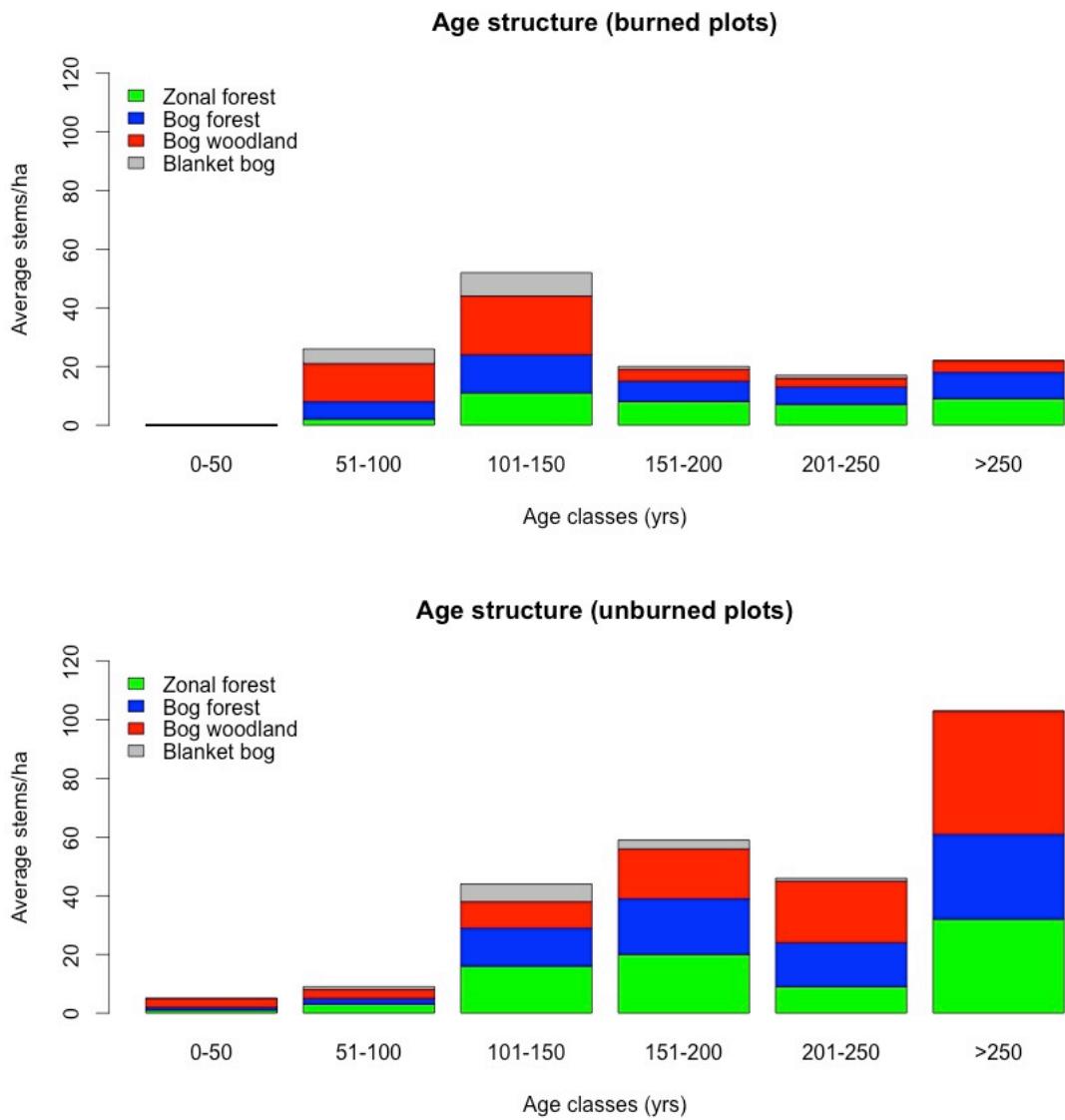


Figure 5.3 Six age class distributions are described for the four vegetation types in the study area: zonal forest (green), bog forest (blue), bog woodland (red), and blanket bog (grey) in 27 burned plots (upper panel) and 11 unburned plots (lower panel) with no aboveground evidence of fire activity. The average number of stems per hectare is explained on the y-axis.

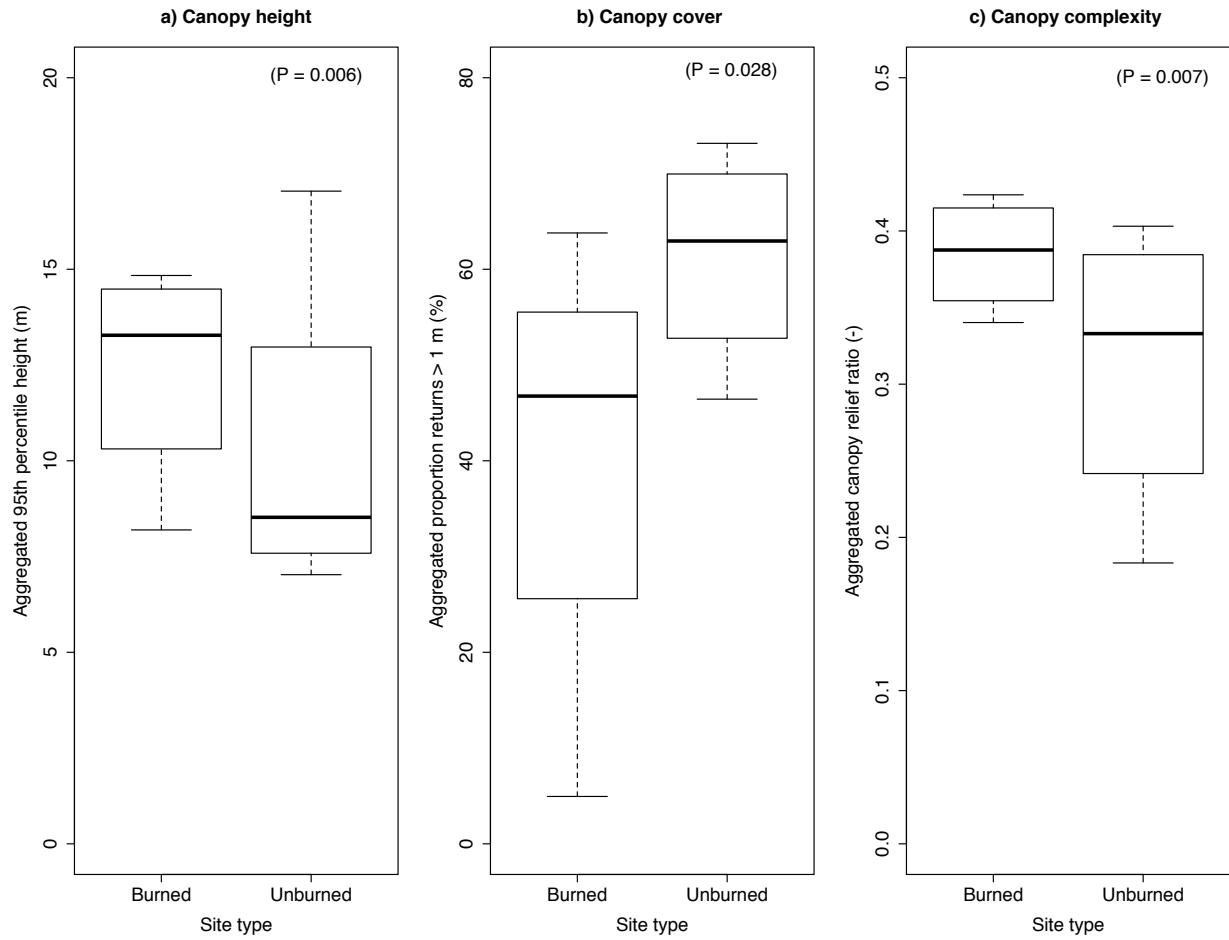


Figure 5.4 The results of the two factor model nested analysis of variance (ANOVA) with unequal sample sizes. Box and whisker plots describe differences in airborne light detection and ranging (LiDAR) measurements for a) canopy height, b) canopy cover, and c) canopy complexity between burned and unburned plots aggregated across four vegetation types (zonal forest, bog forest, bog woodland, and blanket bog). Boxes represent the second and third quartile ranges, and the centreline is the median.

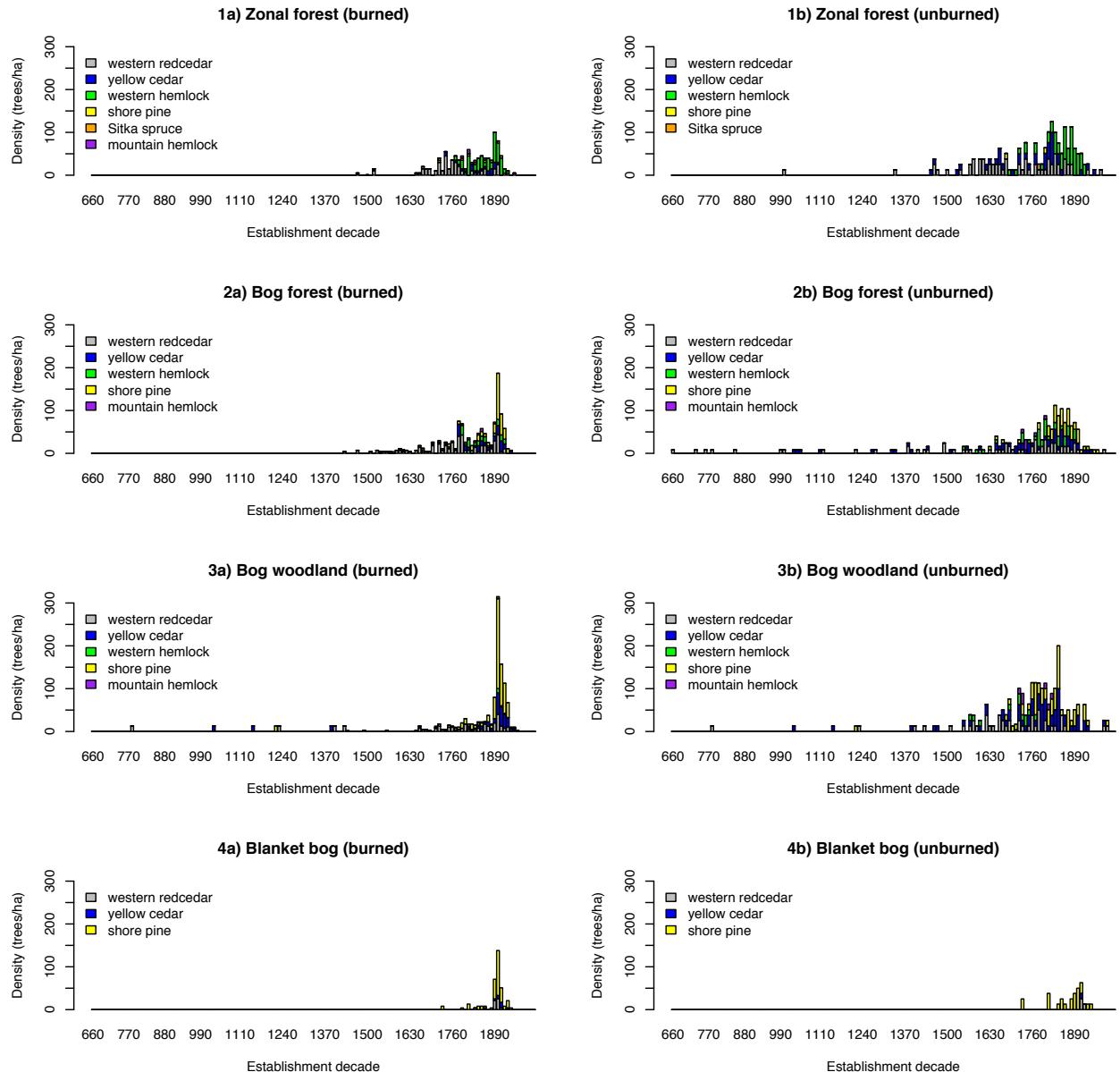


Figure 5.5 The 1353 year record (spanning 660-2013) binned by decade of establishment in burned versus unburned plots across four vegetation types (zonal forest, bog forest, bog woodland, and blanket bog) in the study area. Species are western redcedar (grey), yellow-cedar (blue), western hemlock (green), shore pine (yellow), Sitka spruce (orange), and mountain hemlock (purple). The density of trees per hectare is explained on the y-axis.

Chapter 6: Concluding remarks and future research

6.1 Concluding Remarks

Through time, humans have had a significant impact on their surrounding environments, including using fire to alter the structure and distribution of vegetation communities (Pausas and Keeley, 2009; Bowman et al. 2011; Ryan et al. 2013). Fire occurs in all of Earth's vegetated biomes, but the degree to which human ignitions have shaped vegetation patterns by amplifying fire activity is dependent on geography, densities of humans, and background rates of lightning (Bond and Keeley, 2005; Bowman et al. 2009; Bowman et al. 2011; Ryan et al. 2013; Whitman et al. 2015; Whitlock et al. 2015). Anthropogenic predictors of fire activity vary greatly among ecoregions, and few locations exist where lightning-ignitions are absent and humans are the sole determinant of fire activity (Pew and Larsen, 2001; Bond and Keeley, 2005; Bowman et al. 2009; Whitlock et al. 2015; Fusco et al. 2016). Lightning-ignited fires are especially rare in perhumid coastal temperate rainforests because of the distinct hypermaritime climate, which results in low lightning strike densities (Veblen et al. 1999; Burrows et al. 2002; Lertzman et al. 2002; Dorner and Wong, 2003; Daniels and Gray, 2006; Whitlock et al. 2010; McWethy et al. 2013).

Perhumid temperate rainforests are distributed throughout the coastal margin of British Columbia, Patagonia, New Zealand, and Tasmania (DellaSala et al. 2011). In these wet and dense forests, high-severity stand-replacing fire disturbances are estimated to occur at centennial or even millennial intervals (Lertzman et al. 2002; Gavin et al. 2003a; Daniels and Gray, 2006; Bowman et al. 2009; McWethy et al. 2013; Bowman et al. 2011; Holz et

al. 2012; Whitlock et al. 2015). In New Zealand, temperate rainforest species have few adaptations to withstand fires, and the arrival of indigenous peoples to the islands 800 years ago resulted in widespread loss of forest cover (McWethy et al. 2010; Bowman et al. 2014; Tepley et al. 2016). In Patagonia, millennia of burning by indigenous people significantly amplified fire activity so that new patterns of vegetation and fuel characteristics developed (Veblen et al. 1999; Holz and Veblen, 2011; McWethy et al. 2013). In Tasmania, high-severity fires lit by European settlers in the 1960s resulted in the population collapse of a subalpine endemic conifer that previously persisted under a low-severity fire cycle (Holz et al. 2014; Bowman et al. 2014). In these perhumid temperate rainforest settings, indigenous groups directly affected fire cycles by providing an ignition source, managing fires, and modifying vegetation and corresponding fuel patterns (Agee 1993; Veblen et al. 1999; Whitlock et al. 2010; McWethy et al. 2013; Holz et al. 2014; Whitlock et al. 2015).

Widespread fire suppression in the 20th century eliminated many indigenous burning practices globally, and decades of fire exclusion have resulted in dense, closed-canopy forests (Lepofsky et al. 2003; Lepofsky and Lertzman, 2008; Bowman et al. 2011; Turner et al. 2013; Ryan et al. 2013; Holz et al. 2014; Whitlock et al. 2015). In many coastal temperate rainforest settings, humans have altered and controlled fire activity for millennia, and the long-term effects of humans and climate on fire activity remains poorly understood (Marlon et al. 2008; Holz et al. 2012; Gavin et al. 2015; Whitlock et al. 2015). The reduction of indigenous fire activity in coastal temperate rainforests

throughout much of the 20th century also means that the forest structure we see today is likely different than the forest structure that existed pre-1900 (Bowman et al. 2011).

The coastal margins of British Columbia and southeast Alaska contain the largest and most intact tract of coastal temperate rainforest in the world (Veblen and Alaback, 1996; DellaSala et al. 2011). The absence of 20th century fire activity and the year-round wet and cool climate in this region suggest that these forests are not prone to frequent fire disturbances (Daniels and Gray, 2006). These factors, together with low rates of summer lightning strikes, make this region an ideal location to examine human-driven fire activity. This region also contains abundant old growth forests, the majority of which have not been logged, preserving ecological legacies of long-term First Nations land-use practices (Lertzman et al. 2002; McLaren et al. 2015; Trant et al. 2016). My research provides the first reconstruction of fire activity within perhumid temperate rainforests in British Columbia and Alaska, and on the Central Coast of British Columbia specifically. The work presented in this dissertation suggests that First Nations on the Central Coast have a long and complex relationship with fire. Although fire events have not occurred in the region since the end of the 19th century, legacies of historic fire activity continue to shape patterns in the present-day forest structure (Trant et al. 2016).

My research focused on reconstructing the temporal and spatial characteristics of historic fire activity on 15 islands located across a 2000 km² study area on the Central Coast of British Columbia. I utilized multiple lines of evidence across several disciplines to test three hypotheses about the origins of fires and the patterns of specific vegetation

commonly associated with First Nations burning. My data are consistent with the hypothesis that First Nations utilized fire to manage landscapes and specific plant resources in the study area for at least 700 years, but does not rule out the possibility that fires may have occurred accidentally on the landscape. Although I was not able to confirm whether ignitions were accidental or intentional, the 13,000-year pattern of coexistence of fire and people in the study area contrasts with the relative absence of fire and the significantly-impacted and diminished presence of First Nations (related to disease and colonialism) from the study area in the 20th century. These data are directly relevant to contemporary forest management and discredit the myth of old growth temperate rainforests as landscapes unaltered by humans. The four most important findings of this dissertation are:

- 1) Dating of soil charcoal, in conjunction with archaeological data from First Nations habitation sites collected by colleagues, reveals that fire and people have coexisted in this region for at least 13,000 years, prior to the establishment of present day coastal temperate rainforests.
- 2) Seven hundred years of fire scar and stand establishment data suggest that low-severity fires occurred approximately every 39 years until the end of the 19th century. Drought and warmer temperatures in the year prior were associated with fire events and fires were 25 times more likely than previous estimates. The absence of fire activity in the 20th century coincides with the forced relocation of First Nations from the study area.

- 3) Lightning-strike densities cannot explain the pattern of historic fire events, which were associated with the locations of nearshore First Nations habitation sites, not with south-facing, steep slopes at high elevations (as in most other studies in the interior of British Columbia). The forests surrounding former First Nations habitation sites had shorter fire return intervals. These forests also contained higher densities of western redcedar trees that were culturally modified with fire, as well as specific food plants commonly associated with burning by British Columbia First Nations groups.
- 4) LiDAR and field sampling confirmed that historically burned vegetation plots contained younger trees in lower densities than unburned plots, but also had higher overall conifer diversity. Plot-level measurements are important calibration and validation tools for remote sensing techniques and may identify historic fire events that have been previously overlooked by air photos and remote sensing methods alone.

6.2 General Implications

There has been a shift in recent years to attempt to reinstate fire to its historic ‘natural’ range of variability prior to 20th century land-use changes (Pausas and Keeley, 2009; Bowman et al. 2011). This is problematic, as the forests encountered by early explorers and non-native settlers on the coast of British Columbia had been shaped for millennia by both biological and cultural forces (Lepofsky and Lertzman, 2008; Turner et al. 2013; Turner, 2014), and human presence on the Central Coast of British Columbia predates contemporary vegetation and climate (McLaren et al. 2014). The mandate to restore ‘natural fire activity’ often does not include humans and their role in fire ignition,

suppression, and the modification of vegetation and corresponding patterns of fuel distribution (Bowman et al. 2011; Tepley et al. 2016). Further, neglecting human factors in historic fire activity undermines our ability to understand human-environment interactions, which directly relate to the management of present day forests (Guyette et al. 2002; Ryan et al. 2013). Current forest management practices are still primarily based on models of fire exclusion, and long-term investments in prescribed burning, improved fire safety in communities, and applied fire science are critical to avoiding large and uncontrollable wildfires (Collins et al. 2017). My research in perhumid coastal temperate rainforests underlines the importance of understanding how humans historically influenced, and in some cases determined, fire activity to create desirable vegetation patterns. How, and under what circumstances, First Nations carefully and sustainably manage British Columbia's diverse forests is an important area of future study.

6.3 Future Research

Most people perceive British Columbia's old growth coastal temperate rainforests as pristine or untouched ecosystems, but emerging research suggests that First Nations have managed coastal environments for millennia (McLaren et al. 2014; McLaren et al. 2015; Lepofsky et al. 2015; Jackley et al. 2016; Trant et al. 2016; Lepofsky et al. 2017). Unfortunately, the process of colonization resulted in cultural disorganization and the reduction or cessation of many traditional land-use practices by the end of the 19th century (Turner et al. 2013). Although First Nations continue to utilize traditional practices to harvest resources, in most cases, controlled burning has been greatly diminished or ended (Boyd, 1994; Turner, 1999; Turner, 2014). Reconstructions of historic fire events and an improved understanding of traditional First Nations fire-

management systems can help inform current forest practices. Integrating traditional fire knowledge and current fire science is also an important step to safely and effectively carry out prescribed burning in British Columbia. Regular prescribed burning can reduce fuel loads and decrease the risk of large uncontrollable wildfires like those experienced in British Columbia in 2003, 2016 and 2017 (Collins et al. 2017). Further, restoring fire to forests may help shift the general public's perception of fire from a frightening, once-in-a-lifetime occurrence, to a regular and healthy component of forest ecosystems (Bowman et al. 2011).

By suppressing natural and cultural fires, not learning from elders or fire specialists, and not engaging in prescribed burning, British Columbia has created a reality where forest fires are large, intense, unpredictable, and often uncontrollable. There is also the added component of climate change, which is expected to lengthen the fire season and increase fire frequency and intensity (Littell et al. 2016; Schoennagel et al. 2017). Though interior and coastal forests are quite different, the principle is the same: First Nations understood fire as a necessary part of the natural cycle of forests, required fire to harvest food, and also used it as a tool for forest management. How we think about fire in coastal temperate rainforests is important, because climate change, increased human access, and just plain carelessness will create more fire ignitions in these forests in the future.

Engaging in regular low- and mixed-severity prescribed burning may reduce the effects of extreme fire behaviour such as property loss, carbon release, and the destruction of timber resources (Marsden-Smedley and Kirkpatrick, 2000; Schoennagel et al. 2017).

Whether ignitions were intentional or accidental in the study area, regular burning by First Nations reduced ground fuels, thinned trees, and created more fire adapted landscapes surrounding habitation sites. Memory of traditional fire management practices still exists in First Nations communities. Indigenous knowledge and other ways of knowing can support and inform contemporary wildfire science. This presents an opportunity to engage in interdisciplinary research, and to better understand how fire exclusion has affected forest structure and resilience by employing prescribed burning in forests that were once regularly managed with fire (Collins et al. 2017; Schoennagel et al. 2017).

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Appendices

Appendix A1: Descriptions of soils and macrocharcoal sampled

Table A1.1 Detailed descriptions of the 39 soil pits sampled in the study area.

The site code, lab number, depth in centimetres, and description of charcoal stratigraphy sampled from each pit is provided along with the Canadian soil classification. Organic soils are as follows: TY.M (Typic Mesisol), FI.M (Fibric Mesisol), HU.M (Humic Mesisol), ME.H (Mesic Humisol), HU.FO (Humic Folisol), HI.FO (Histic Folisol). Mineral soils are as follows: O.FHP (Orthic-Ferro-Humic Podzol), Orthic-Humo-Ferric Podzol (O.HFP), and Orthic-Humic Podzol (O.HP).

Site code	Lab number (UCIAMS)	Pit depth (cm)	Soil class	Stratigraphic descriptions
KHEPBW14	151414 151415	34	O.FHP	Macroscopic charcoal was deposited in two distinct stratigraphic layers in an imperfectly drained mineral Podzol. A 2 cm band of charcoal was apparent at 12 cm and an intermittent band at 25 cm depth. Both samples were radiocarbon dated.
KHEPBF08	151416	28	HI.FO	No soil charcoal was apparent in the upper layer of this enriched organic Fibrisol. Abundant macroscopic charcoal was apparent in stratigraphy at 28 cm depth at the interface of an organic Fibrisol and a mineral Podzol layer. One sample was radiocarbon dated.
KHEPBW45	151417	34	TY.M	Abundant macroscopic charcoal deposited in ~3 cm thick stratigraphy was sampled at 2 cm depth in an organic Mesisol. Soil pipes were present at the bottom of this poorly drained pit. No other charcoal layers were present and one sample was radiocarbon dated.
KHEPZF52	151418 151419	45	HI.FO	Two stratigraphic layers containing abundant charcoal were found deposited in organic tiers over a mineral Podzol. The first layer contained macroscopic charcoal and was sampled at 10 cm depth in an organic Mesisol. A second layer containing microscopic charcoal was sampled in

				an organic Humisol at 30 cm depth. Both samples were radiocarbon dated.
KHEPZF39	151420 151421	50	TY.M	Abundant macroscopic charcoal was sampled in this deep organic Mesisol. The first charcoal stratigraphy was sampled at 8 cm depth and a second layer of charcoal was sampled in the same Mesisol layer at 15 cm depth. A third stratigraphic layer was sampled in a Humisol at 47 cm depth deposited near the bottom of the pit surrounding a large piece of wood. The upper and lower samples were prioritized for radiocarbon dating.
KHEPBF16	151422	28	HU.FO	Microscopic charcoal was apparent in intermittent stratigraphy in the first 10 cm of an organic Humic Mesisol. A sample was collected from this layer, but not submitted for radiocarbon dating. Abundant macroscopic charcoal was sampled from stratigraphy at 14 cm depth in the same organic Humic Mesisol. The lower sample was radiocarbon dated.
KHEPBF06	151423	52	HI.FO	I found little evidence of charcoal in the upper organic Fibrisol layer (0-10 cm depth). I sampled macroscopic charcoal in stratigraphy at 15 cm depth between two mineral Podzol horizons. No other charcoal layers were apparent and one sample was radiocarbon dated.
KHEPBW35	151424	40	HU.FO	Charcoal was littered throughout this profile but only found in two stratigraphic layers lower in the soil profile. The first sample was taken from a ~3 cm thick layer of macroscopic charcoal at 16 cm depth within an organic Mesisol. A second layer was sampled at 37 cm depth near the bottom of the pit within a Humic Podzol mineral horizon. I sampled both soil charcoal layers but prioritized radiocarbon dating of charcoal from the 16 cm layer to estimate TSF.
KHEPBW03	151425	26	F.I.M	Soil charcoal was deposited in two separate ~2 cm thick stratigraphic layers. The first layer contained abundant macroscopic charcoal and was sampled in an organic Fibrisol at 16 cm depth. A lower stratigraphic layer containing microscopic charcoal was sampled at 25 cm in an organic Mesisol. This was a nutrient-rich, poorly drained site with abundant seepage in the soil pit. I sampled both layers, but prioritized radiocarbon dating of charcoal from the 16 cm

				layer to obtain TSF estimates.
KHEPBW33	151426	22	HU.FO	I found a distinct charcoal layer at 5 cm depth within an organic Fibrisol. Charcoal fragment sizes were small <2 mm and I collected several pieces from the same layer for radiocarbon dating. There was evidence of intermittent charcoal layers buried in an organic Humisol layer but the presence of a large root at the base of the soil pit may have been the source of this charcoal, therefore, I prioritized radiocarbon dating of the charcoal sample from the upper layer to obtain TSF estimates.
KHEPBF03	151426 151427	48	F.I.M	Two stratigraphic layers ~3 cm thick of soil charcoal were sampled at 17 cm depth in an organic Fibrisol and at 34 cm depth in an organic Mesisol. Macroscopic charcoal was sampled in the upper layer and microscopic charcoal in the bottom layer. This was a poorly drained soil with evidence of soil piping in the bottom tiers. I submitted both samples for radiocarbon dating.
KHEPBW10	151429 151430	34	HI.FO	Soil charcoal was sampled from three distinct layers in both organic and mineral soils. The first layer was sampled at 5 cm depth beneath the litter layer in an organic Fibrisol. The second layer contained a ~2 cm thick layer of macroscopic charcoal at 16 cm depth and this sample was radiocarbon dated. A third sample was taken from an organically enriched mineral Podzol with microscopic charcoal found in a ~1cm layer at 28 cm depth and this sample was also submitted for radiocarbon dating. Because there were several fire-scarred trees at the site, which provided TSF in, this plot I prioritized the lower two samples for radiocarbon dating.
KHEPZF04	151431	40	TY.M	One stratigraphic layer was sampled in a Humic Mesisol layer at 30 cm depth and was submitted for radiocarbon dating. Well-decomposed, black organics in this soil profile made it difficult to identify charcoal. Although there was evidence of microscopic charcoal throughout the soil profile, it was found in intermittent stratigraphy and I collected but did not submit these samples for radiocarbon dating. There were several fire-scarred trees at this site, which confirmed TSF.
KHEPBF20	151432	26	HU.FO	Microscopic charcoal was sampled at the interface of a thick organic Humisol and a thin

				organically enriched mineral Podzol at 20 cm depth. I did not find any other evidence of charcoal throughout the soil profile and only submitted one sample for radiocarbon dating.
KHEPBB43	151433	64	TY.M	Macroscopic charcoal was apparent in two layers of thick ~5 cm stratigraphy in an organic Mesisol over bedrock. I sampled a charcoal layer at 16 cm depth and at 40 cm depth. Because of the presence of fire-scarred trees, I prioritized the lower sample for radiocarbon dating.
KHEPBB23	151434 151435	22	O.HFP	Two charcoal layers in stratigraphy were apparent in two distinct mineral Podzols. The first charcoal layer was thin with microscopic charcoal pieces at the interface of an organic Humic layer and a leached Podzol mineral layer. A more defined >2 cm thick macroscopic charcoal layer was apparent at 14 cm depth in a second Podzol layer. There were no fire-scarred trees at this site and I submitted charcoal from both layers of stratigraphy for radiocarbon dating.
KHEPBB40	151436	23	O.HFP	This moderately-drained mineral Podzol had abundant macroscopic charcoal deposited in stratigraphy at 17 cm depth and contained no other evidence of charcoal in the soil or aboveground fire-scarred trees. I submitted one sample for radiocarbon dating.
KHEPBF49	151437 151438	37	O.HP	Upper layers of the mineral Podzol horizon were organically enriched and contained two stratigraphic charcoal layers. The first layer contained macroscopic charcoal deposited at 14 cm depth and the second layer contained microscopic charcoal deposited at 22 cm depth. The two stratigraphic layers contained the only charcoal found in the soil pit. Both samples were submitted for radiocarbon dating and there was no evidence of fire-scarred trees at this site.
KHEPBF01	151439	44	HI.FO	Three charcoal layers were apparent in the upper soil profile dominated by an organic Mesisol. All layers were thin and contained microscopic charcoal. The first layer was deposited at 4 cm depth and subsequent layers at 12 cm and 20 cm depth. Because of limited funds and several fire-scarred trees at the site I chose to sample the lower layer to avoid replication in radiocarbon dates.

KHEPBF05	151440	28	HU.FO	Charcoal was found in two distinct stratigraphic layers within an organic Mesisol. The first layer contained microscopic charcoal buried at 5 cm depth within a well-defined Humic layer. The second layer contained macroscopic charcoal in a thick ~5 cm layer at 12 cm depth. Because there were fire-scarred trees at the site with known fire dates I prioritized the lower sample for radiocarbon dating.
KHEPBB09	151441	27	O.HFP	Charcoal was apparent in two distinct stratigraphic layers within separate Podzol mineral horizons. There were no fire-scarred trees at this site and macroscopic charcoal from stratigraphic layers ~5 cm thick were sampled at 10 cm depth and 22 cm depth. I prioritized the upper charcoal layer for radiocarbon dating to obtain TSF estimates.
KHEPBB19	151442	20	O.FHP	Charcoal was stratified in two distinct layers within separate mineral Podzol horizons. Macroscopic charcoal was sampled at 8 cm depth and microscopic charcoal was sampled at 16 cm depth. No fire-scarred trees were present at the site and I prioritized the upper charcoal layer for radiocarbon dating.
KHEPBF44	151443 151444	50	HU.FO	The presence of a well-decomposed Humic layer at this site made it difficult to detect charcoal in the top 10 cm of the soil profile. I was able to sample microscopic charcoal from two stratigraphic layers at 28 cm depth and 48 cm depth. The first sample was collected in an organic Mesisol and the second sample in a mineral Podzol horizon. Both samples were submitted for radiocarbon dating.
KHEPBB55	151445	38	HI.FO	Two distinct layers of charcoal stratigraphy were apparent in an upper organic Mesisol. Microscopic charcoal was sampled from a thin layer at 5 cm depth and from a second layer ~3 cm thick containing macroscopic charcoal at 8 cm depth. A third stratigraphic layer was sampled in a Podzol mineral horizon at 22 cm depth. Due to the presence of fire-scarred trees at the site I prioritized the middle layer for radiocarbon dating.
KHEPBW42	148877 148878 148879	32	HU.FO	Macroscopic charcoal was abundant in this Humic Folisol. I sampled two stratigraphic layers contained macroscopic charcoal in the

				organic layer at 12 cm and 16 cm depth and a third stratigraphic layer within a Podzol mineral horizon at 28 cm depth. The third layer contained abundant pieces of partially burned wood. There were no fire-scarred trees at this site and I submitted all three macroscopic charcoal pieces for radiocarbon dating.
KHEPZF38	148880 148881	22	HU.FO	Charcoal was apparent in three thin but distinct stratigraphic layers in shallow organic and mineral horizons. The first two stratigraphic layers contained microscopic charcoal and the first layer was sampled at between the litter and humus layer at 3 cm depth and the second layer was sampled within the humus layer at 8 cm depth. The third layer contained macroscopic charcoal and was sampled in a Podzol mineral horizon at 15 cm depth. There were no fire-scarred trees at this site and the upper two samples were prioritized for radiocarbon dating.
KHEPZF01	NA	60	ME.H	Charcoal was difficult to distinguish in the upper layers of this rich organic Humisol. I sampled one intermittent charcoal layer containing microscopic charcoal pieces at 50 cm depth. I did not submit this sample for radiocarbon dating because I was not confident that it was in situ and there were fire-scarred trees present at this site.
KHEPBW22	148883	28	HI.FO	There was only one distinct layer of stratigraphy in this soil pit containing both organic and mineral horizons. I sampled macroscopic charcoal from a ~5 cm thick stratigraphic layer at 26 cm depth within a Podzol mineral horizon. This sample was radiocarbon dated and there were no fire-scarred trees at this site.
KHEPBW07	148884 148885	35	HI.FO	Microscopic charcoal was sampled in stratigraphy at the interface of the humus and organic Mesisol layer at 10 cm depth. A second thicker layer containing macroscopic charcoal was sampled at 18 cm depth within a poorly-drained Podzol mineral horizon. Both samples were submitted for radiocarbon dating. Fire-scarred trees were present at this site.
KHEPBW02	148886 148887	50	HI.FO	Two layers of stratigraphy were apparent in this poorly drained soil containing organic and mineral horizons. The first stratigraphic layer contained microscopic charcoal and was

				sampled at 10 cm depth in an organic Fbrisol. The second stratigraphic layer was thicker ~3 cm and containing macroscopic charcoal situated between two Podzol mineral horizons at 18 cm depth. Both samples were radiocarbon dated and there were no fire-scarred trees at this site.
13ABP08	134770	45	O.FHP	One stratigraphic layer containing macroscopic charcoal pieces was sampled at 32 cm depth between two Podzol mineral horizons. No fire scars were present at the site and no other charcoal was identified in this soil pit. One sample was submitted for radiocarbon dating.
13KHSP1	134771	48	O.HFP	One stratigraphic layer containing abundant macroscopic charcoal was sampled at 32 cm depth from a Podzol mineral horizon. No fire scars were present at the site and no other charcoal was identified in this soil pit. One sample was submitted for radiocarbon dating.
13ECP16	134772	35	HI.FO	Two stratigraphic layers containing macroscopic charcoal were sampled in this soil pit containing organic and mineral horizons. The first stratigraphic layer was sampled at 25 cm depth in a Podzol mineral horizon and the second stratigraphic layer was sampled near the bottom of the soil pit at 35 cm depth in a Podzol mineral horizon overlaying bedrock. There were no fire scars present at this site and charcoal from the upper stratigraphic layer was prioritized for radiocarbon dating.
1311-9763	134773	100	TY.M	Two distinct stratigraphic layers of macroscopic charcoal were sampled from this deep organic profile. Both layers were ~ 5 cm thick and the first stratigraphic layer was sampled at 80 cm depth in an organic Mesisol. The second stratigraphic layer was sampled at 100 cm depth in a Humic-enriched tier at the base of the soil pit. Although both samples were collected, no fire-scarred trees were present and only charcoal from the upper stratigraphic layer was radiocarbon dated.
13ABP03	134774	19	O.FHP	One stratigraphic layer was apparent in this shallow mineral Podzol overlaying bedrock. I sampled macroscopic charcoal at 8 cm depth and found no other evidence of soil charcoal or fire-scarred trees at the site. One sample was radiocarbon dated.

13ABP04	134775	60	HU.M	This deep, organically enriched soil contained one stratigraphic macroscopic soil charcoal layer at 50 cm depth in a Humic Mesisol. This was the only charcoal identified and there were no fire-scared trees at the site. The sample was radiocarbon dated.
13ABP07	134776 134777	95	TY.M	This organic Mesisol contained two macroscopic soil charcoal layers at 34 cm depth and 80 cm depth. The first layer was ~ 5 cm thick and contained abundant macroscopic charcoal. The second layer was thinner and less distinct with large roots present. Both samples were radiocarbon dated and there were no fire scars at the site.
13WBP09	134778	21	O.FHP	One 3 cm thick stratigraphic layer was sampled at 15 cm depth within a Podzol mineral horizon overlaying bedrock. This sample was submitted for radiocarbon dating and no fire-scared trees were present at this site.
13ATMC1	134779	100	HU.M	This deep, organically enriched soil contained one macroscopic soil charcoal layer at 80 cm depth in an organic Humic Mesisol. This was the only charcoal identified and there were no fire-scared trees at the site. One sample was submitted for radiocarbon dating.

Table A1.2 52 radiocarbon dates displayed in chronological order from 39 sites in the study area. The geographic location of each soil pit, type of dated material, species of charred samples (when known), size of charcoal fragments (mm), and the depth of stratigraphic layers (cm) of each soil charcoal sample are provided. All radiocarbon ages are reported in age Before Present (BP, before common era [1950]) with 2-sigma radiocarbon errors, calibrated years BP with 2-sigma errors, and median calibrated age BP.

Lab number (UCIAMS)	Coordinates	Dated material	Species	Size (mm)	Depth (cm)	Age BP	cal yr BP (2-sigma)	Med. age cal yr BP
151417	N51 39 31.8 W128 04 47.5	small branch	<i>Thuja sp.</i>	>5	2	70 ± 20	0-259	103
151418	N51 40 01.5 W128 04 27.9	bark	<i>Picea sp.</i>	>5	10	120 ± 15	59-266	109
151420	N51 39 35.9 W128 04 10.7	bark	<i>Thuja sp.</i>	>5	8	115 ± 20	55-267	111
151440	N51 40 00.4 W128 03 51.0	outer wood, bark	<i>Tsuga sp.</i>	>5	12	200 ± 15	148-293	221
151427	N51 39 59.8 W128 04 14.6	bark	<i>Pinus sp.</i>	>5	17	205 ± 15	149-296	223
151424	N51 39 38.1 W128 03 19.9	wood, bark	<i>Pinus sp.</i>	<5	16	245 ± 15	156-306	231
148881	N51 39 34.9 W128 04 36.6	wood	<i>Thuja sp.</i>	<5	3	250 ± 20	153-314	234
148884	N51 39 52.8 W128 03 52.0	small branch	<i>unknown</i>	<5	10	280 ± 15	293-426	360
151426	N51 39 38.4 W128 04 30.4	needle	<i>Pinus sp.</i>	<5	5	290 ± 15	299-428	364
151416	N51 39 54.4 W128 03 57.4	outer wood, bark	<i>Thuja sp.</i>	>5	28	290 ± 20	297-430	364
151428	N51 39 59.8 W128 04 14.6	wood	<i>Thuja sp.</i>	<5	34	375 ± 40	316-506	411
151423	N51 39 56.6 W128 03 42.1	bark	<i>Tsuga sp.</i>	>5	15	375 ± 15	332-499	416
148877	N51 39 34.0 W128 03 48.1	outer wood	<i>Thuja sp.</i>	>5	12	510 ± 15	515-541	528
148880	N51 39 34.0 W128 03 48.1	small branch	<i>unknown</i>	<5	8	570 ± 20	534-637	586
151425	N51 40 11.1 W128 04 15.2	outer wood	<i>unknown</i>	>5	16	595 ± 15	546-644	595
134779	N51 39 26.8 W128 07 15.0	wood pieces	<i>Thuja sp.</i>	<5	80	790 ± 15	684-729	707
151414	N51 39 52.2 W128 03 25.9	outer wood, bark	<i>Thuja sp.</i>	>5	12	815 ± 20	686-763	723

151439	N51 39 59.7 W128 04 04.8	outer wood, bark	<i>unknown</i>	<5	20	1110 ± 40	932-1089	1011
151434	N51 39 46.1 W128 03 34.9	wood pieces	<i>unknown</i>	<5	2	1460 ± 15	1310-1378	1344
151422	N51 39 49.5 W128 04 32.9	bark	<i>Pinus sp.</i>	>5	14	1525 ± 15	1357-1483	1420
151429	N51 39 39.2 W128 04 19.5	branch, wood	<i>Thuja sp.</i>	>5	16	1105 ± 20	964-1057	1493
151433	N51 40 11.2 W128 03 36.1	bark	<i>Pinus sp.</i>	>5	40	1755 ± 30	1565-1736	1651
148886	N51 39 56.5 W128 04 25.3	wood pieces	<i>unknown</i>	<5	12	1785 ± 15	1623-1740	1682
151415	N51 39 52.2 W128 03 25.9	small branch	unknown	>5	25	1805 ± 15	1699-1816	1758
151432	N51 39 47.2 W128 04 38.5	wood	<i>unknown</i>	<5	20	1810 ± 15	1707-1814	1761
151445	N51 39 49.1 W128 04 14.6	twig	<i>unknown</i>	<5	8	2055 ± 20	1950-2065	2008
134772	N51 38 06.8 W128 07 06.2	wood	<i>unknown</i>	>5	25	2070 ± 15	1992-2111	2052
151443	N51 39 37.9 W128 05 02.8	needle	<i>Thuja sp.</i>	<5	28	2305 ± 20	2314-2352	2333
148878	N51 39 34.0 W128 03 48.1	wood	<i>unknown</i>	>5	16	2690 ± 25	2755-2845	2800
148879	N51 39 34.0 W128 03 48.1	outer wood	<i>Thuja sp.</i>	>5	28	2760 ± 20	2786-2888	2837
151430	N51 39 39.2 W128 04 19.5	wood pieces	<i>unknown</i>	<5	28	2790 ± 20	2844-2955	2900
151437	N51 39 31.1 W128 03 04.8	bark	<i>Tsuga sp.</i>	>5	14	2915 ± 20	2972-3144	3058
134774	N51 39 59.0 W128 02 25.3	wood	<i>unknown</i>	>5	8	2935 ± 20	3020-3159	3090
134771	N51 38 13.5 W128 06 45.1	wood	<i>unknown</i>	>5	32	3170 ± 20	3363-3445	3404
148885	N51 39 52.8 W128 03 52.0	bark	<i>Pinus sp.</i>	>5	18	3355 ± 25	3556-3644	3600
151431	N51 39 48.8 W128 04 43.9	bark	<i>Picea sp.</i>	>5	20	3435 ± 15	3638-3721	3680
151441	N51 39 59.8 W128 03 27.0	small branch	<i>Pinus sp.</i>	>5	8	3700 ± 15	3983-4087	4035
151442	N51 39 42.1 W128 04 12.8	wood pieces	<i>unknown</i>	<5	8	4500 ± 20	5049-5289	5169
148883	N51 39 46.7 W128 04 02.8	wood pieces	<i>unknown</i>	<5	26	4735 ± 20	5330-5581	5456
134773	N51 38 49.7 W128 04 09.3	wood	<i>unknown</i>	>5	80	6985 ± 20	7753-7867	7810
151436	N51 39 33.2 W128 03 56.9	bark	<i>Tsuga sp.</i>	>5	17	7465 ± 30	8199-8361	8280
148887	N51 39 56.5 W128 04 25.3	outer wood	<i>Tsuga sp.</i>	>5	26	7740 ± 15	8453-8563	8508
151421	N51 39 35.9 W128 04 10.7	outer wood, bark	<i>Tsuga sp.</i>	<5	47	7930 ± 25	8636-8976	8752
151444	N51 39 37.9 W128 05 02.8	bark	<i>Tsuga sp.</i>	<5	43	8410 ± 25	9405-9494	9450
151419	N51 40 01.5	wood	<i>Tsuga sp.</i>	<5	30	8445 ± 35	9429-9527	9478

	W128 04 27.9							
151438	N51 39 31.1 W128 03 04.8	wood pieces	<i>unknown</i>	<5	22	9030 ± 20	10190-10233	10212
134775	N51 39 59.8 W128 06 42.4	wood	<i>unknown</i>	>5	50	9325 ± 25	10487-10589	10538
134776	N51 40 45.5 W128 02 21.0	wood	<i>unknown</i>	>5	32	9910 ± 25	11241-11359	11300
134777	N51 40 45.5 W128 02 21.0	wood	<i>unknown</i>	>5	80	9920 ± 30	11244-11392	11318
151435	N51 39 46.1 W128 03 34.9	wood	<i>unknown</i>	>5	14	10015 ± 30	11325-11630	11478
134770	N51 38 04.6 W128 05 47.9	wood	<i>unknown</i>	>5	32	10020 ± 25	11339-11628	11484
134778	N51 39 40.8 W128 06 44.6	wood	<i>unknown</i>	>5	15	10710 ± 30	12615-12728	12672

Table A1.3 Time Since Fire (TSF) was calculated from fire-scarred trees or the youngest calibrated radiocarbon date at each site excluding samples dating the 1893 AD fire. TSF was compared across the four vegetation types as well as between sites found within the 1893 fire perimeter and outside of the 1893 fire perimeter.

Vegetation type	Zonal forest	Bog forest	Bog woodland	Blanket bog
Number of sites	5	12	12	10
Number of fire scar TSF dates	4	8	6	4
Number of radiocarbon TSF dates	1	4	6	6
Median TSF (years)	294	206	327	2217
Maximum TSF (years)	3680	3404	7810	12,672
Minimum TSF (years)	97	153	153	172

Appendix A2: Age calibration and additional site figures

Figure A2.1 The Topographic Wetness Index (TWI) was derived from a hydrologically-corrected Digital Elevation Model (DEM) of the study area. This model includes water flow direction, accumulation, and slope for all plots in the study area (blue circles).

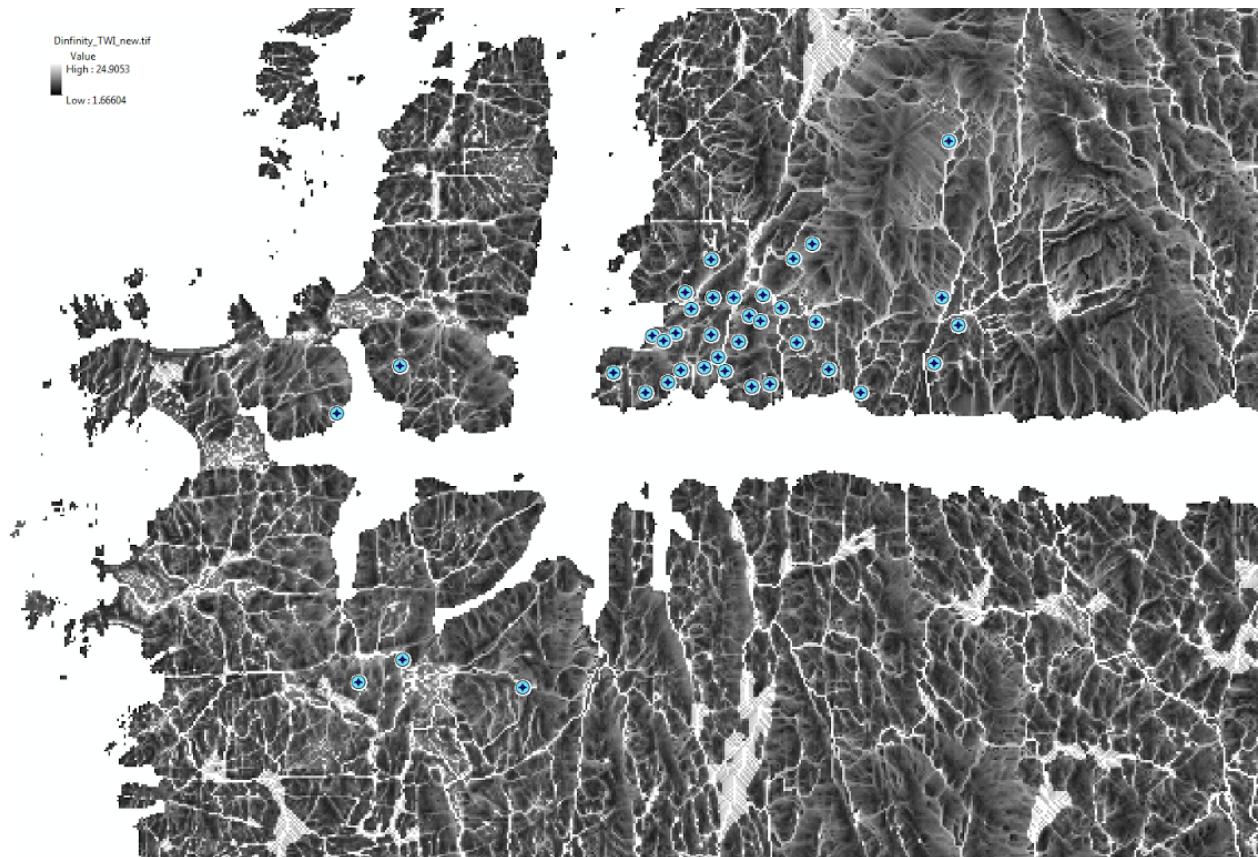


Figure A2.2 Age frequency of the calibrated median probability of 51 soil charcoal dates arrayed by vegetation type on Hecate and Calvert islands from 13,500 years to present. Time Since Fire (TSF) estimates were derived from the youngest calibrated radiocarbon date or from fire-scarred trees at each sampling site. The small box and whisker plots are radiocarbon dates with adjustments of inbuilt age (additional to the 2-sigma radiometric and calibrated error) for each sample and the y-axis (right side) shows the number of samples distributed through time.

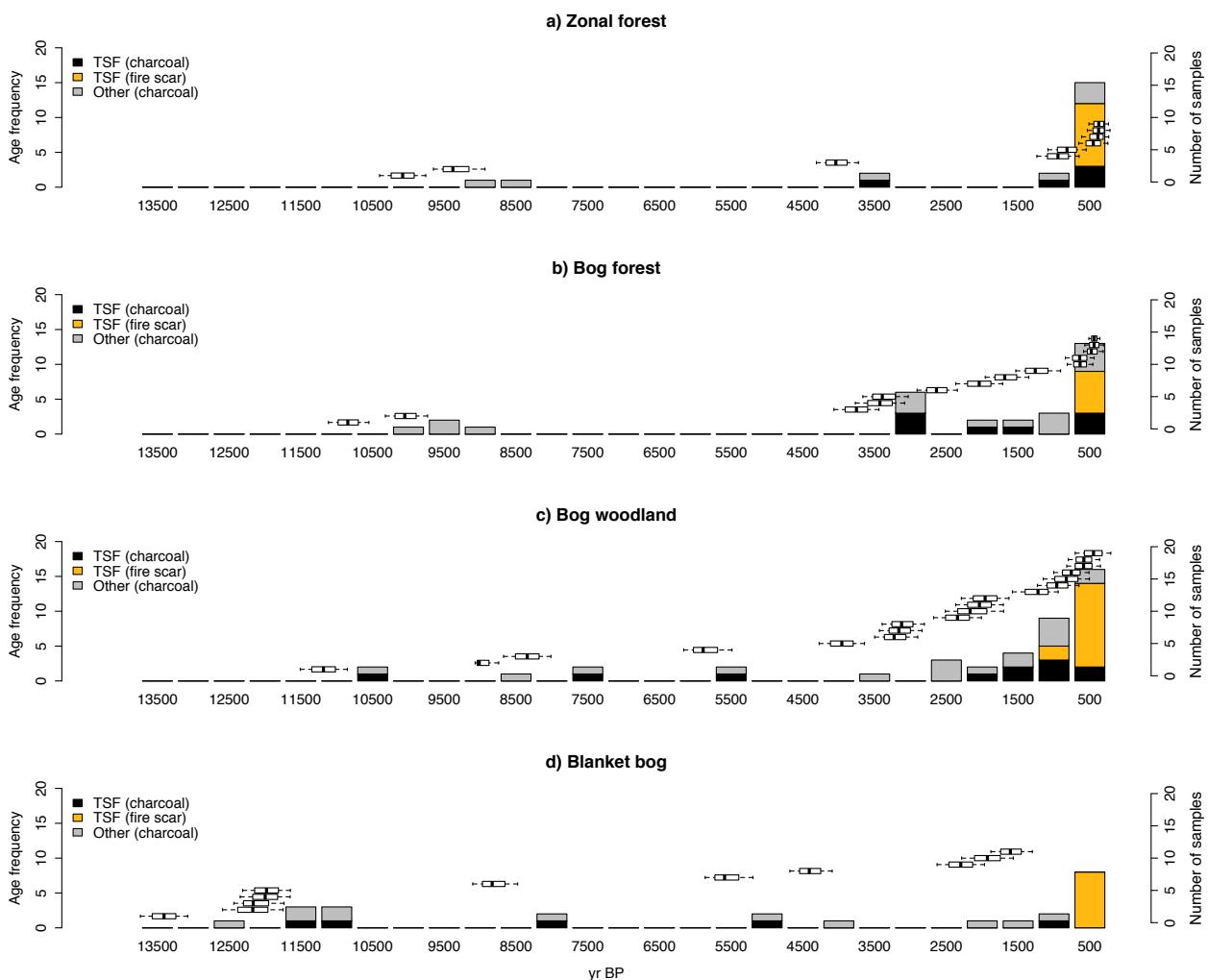


Figure A2.3 An aerial view of the study area looking west. The four vegetation types are represented in this photograph. Photograph by Kira Hoffman.



Figure A2.4 The distribution of inbuilt age was constructed by comparing 14 calibrated radiocarbon dates of surficial soil charcoal with the known year of fire derived from fire-scarred trees. The known year of fire was subtracted from each calibrated date and the probability distributions from each date were averaged (dashed grey line). The mean probability distribution was smoothed with a LOWESS filter (solid black line). Negative inbuilt age values correspond to calibrated radiocarbon ages that spanned the actual age of the fire. For more details on inbuilt age calculations and equations see Gavin (2001) and Gavin et al. (2003a).

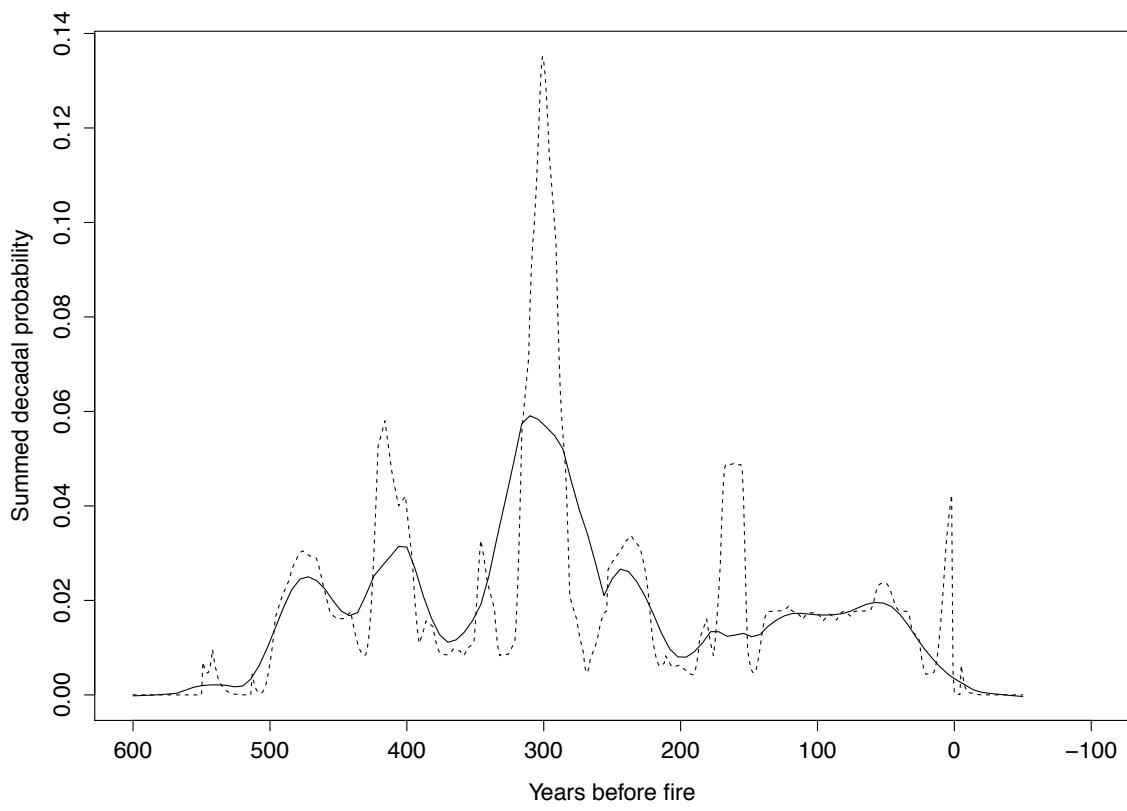


Figure A2.5 Macroscopic soil charcoal (2 cm depth) deposited in stratigraphy and sampled from an organic soil profile in a bog woodland (KHEPBW45, laboratory number: UCIAMS 151417). Photograph by Ian Giesbrecht.



Figure A2.6 Soil charcoal stratigraphy sampled in a shallow (~20 cm in depth) minerotrophic bog on Hecate Island (KHEPBB19). Macroscopic charcoal in stratigraphy was sampled at 8 cm depth above the white Ae layer and microscopic charcoal was sampled in an organically enriched Podzol mineral layer at 16 cm depth. Photograph by Kira Hoffman.



Appendix B1: Climate chronologies, fire histories, and statistical procedures

Table B1.1 Tree ring reconstructed climate chronologies used in analyses of fire activity on Hecate Island, British Columbia, Canada. Abbreviations are the El Niño-Southern Oscillation (ENSO), the Pacific Decadal Oscillation (PDO), the Palmer Drought Severity Index (PDSI), and the Arctic Oscillation (AO). Note correlations with instrumental record of temperature are in summer months unless indicated by (DJF [December, January, and February]). Statistically significant relationships are indicated with (*).

Climate index and time period (AD)	Season reconstructed	% variance explained in the instrumental record	Correlation with instrumental record of temperature in the study area (AD 1900-1975)	Correlation with residual tree-ring chronology in the study area (AD 1330-2014)
ENSO ¹ (AD 1300-2006)	Dec.-Feb.	51% (AD 1870-2002)	0.40 <i>P < 0.001*</i>	0.28 <i>P < 0.05*</i>
PDO ² (AD 1700-1997)	Dec.-Feb.	53% (AD 1900-1979)	0.34 <i>P < 0.001*</i>	0.28 <i>P < 0.01*</i>
PDSI ³ (AD 1249-2004)	June-Aug.	40% (AD 1928-1978)	-0.28 (DJF) <i>P < 0.001*</i>	-0.21 <i>P < 0.05*</i>
AO ⁴ (AD 1650-1978)	April-Sep.	48% (AD 1900-1975)	-0.33 (DJF) <i>P < 0.005*</i>	-0.19 <i>P = 0.10</i>

Note: All reconstructions were downloaded from National Climatic Data Center, National Oceanic and Atmospheric Administration and are freely available (<http://hurricane.ncdc.noaa.gov/pls/paleox/>)

References:

- ¹ Li, J. and others. 2013. El Niño modulations over the past seven centuries. *Nature Climate Change* 3(9):822–826.
- ² D'Arrigo, R. D., R. Villalba, and G. Wiles. 2001. Tree-ring estimates of Pacific decadal climate variability. *Climate Dynamics* 18:219–224.

³ Cook, E. R., and P. J. Krusic. 2004. North American summer PDSI reconstructions.

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⁴ D'Arrigo, R. D., E. R. Cook, M. E. Mann, and G. C. Jacoby. 2003. Tree-ring reconstructions of temperature and sea-level pressure variability associated with the warm-season Arctic Oscillation since AD 1650. *Geophysical Research Letters* 30.

Table B1.2 Summary of fire scar and stand establishment data

used to reconstruct the fire history of 30 plots on Hecate Island, British Columbia, Canada. Data from each plot include: the vegetation type (ZF = zonal forest, BF = bog forest, BW = bog woodland, and BB = blanket bog), the location (latitude and longitude), the number of fire scars, the number of years recording more than two fire scars, the year of the fire, and the length in years of the fire scar record. The age range and post-fire recruitment decade of trees and the point fire interval (PFI [the mean fire interval at the tree scale]) for each plot are provided.

Type	Coordinates Lat/Long	# of fire scars	# of yrs with > 2 scars	First fire scar yr	Last fire scar yr	Fire scar event yrs	Stand age range (yrs) and AD distribution	Post-fire cohort (decade)	Point fire interval (yrs)
ZF	51°39'50.7"N 128°4'36.1"W	14	4	1376	1893	1376, 1482, 1537, 1699, 1744, 1759, 1778, 1797, 1893	54-380 (1634- 2014)	1760, 1770, 1830	47
ZF	51°39'48.8"N 128° 4'43.5"W	2	0	1778	1893	1778, 1893	95-345 (1669- 2014)	1660, 1740, 1770, 1880	103
ZF	51°39'34.9"N 128° 4'36.6"W	1	0	1593	1893	1593, 1778, 1893	83-493 (1521- 2014)	1750, 1800, 1890	103

ZF	51°39'35.8"N 128° 4'10.7"W	21	4	1656	1893	1656, 1744, 1759, 1856, 1893	79-496 (1518- 2014)	1740, 1760, 1890	73
ZF	51°40'1.4"N 128° 4'27.9"W	3	2	1744	1893	1744, 1759, 1844, 1893	115-620 (1394- 2014)	1670, 1740, 1790, 1810	86
BF	51°39'59.7"N 128° 4'4.8"W	7	3	1744	1893	1744, 1759, 1778, 1848, 1893	84-464 (1550- 2014)	1700, 1850, 1880	73
BF	51°39'59.7"N 128° 4'14.5"W	3	2	1744	1893	1744, 1759, 1778, 1893	85-563 (1451- 2014)	1850	103
BF	51°40'0.3"N 128° 3'50.9"W	6	3	1719	1893	1719, 1744, 1759, 1778, 1893	71-414 (1600- 2014)	1670, 1890, 1900	73
BF	51°39'56.6"N 128° 3'42.1"W	6	4	1719	1893	1719, 1744, 1759, 1778, 1893	73-445 (1569- 2014)	1890, 1900	103
BF	51°39'54.3"N 128° 3'57.4"W	5	5	1656	1893	1656, 1744, 1759, 1778, 1893	63-533 (1481- 2014)	1720, 1900	73
BF	51°39'49.5"N 128° 4'32.9"W	25	7	1376	1893	1376, 1482, 1537, 1699, 1744, 1759, 1778, 1797, 1893	112-443 (1571- 2014)	1800, 1890	51
BF	51°39'47.2"N 128° 4'38.4"W	21	5	1537	1893	1537, 1656, 1744, 1778, 1797, 1893	92-350 (1664- 2014)	1790, 1890	86
BF	51°39'37.8"N 128° 5'2.8"W	5	2	1778	1893	1778, 1893	52-579 (1435- 2014)	1630, 1720, 1900	129
BF	51°39'31.0"N 128° 3'4.8"W	0	0	NA	NA	No fire scar evidence	120-953 (1061- 2014)	1790, 1840	No fire evidenc e
BW	51°39'56.5"N 128° 4'25.3"W	6	5	1656	1893	1656, 1744, 1759, 1778, 1893	72-463 (1551- 2014)	1800, 1890	86
BW	51°40'11.1"N 128° 4'15.2"W	0	0	1893	NA	1893	118-361 (1653- 2014)	1800, 1860, 1890	172
BW	51°39'52.7"N 128° 3'52.0"W	6	3	1744	1893	1744, 1759, 1778, 1893	52-570 (1441- 2014)	1890	129
BW	51°39'39.1"N 128° 4'19.5"W	18	6	1656	1893	1656, 1744, 1759, 1778, 1856 1893	59-531 (1483- 2014)	1890	73
BW	51°39'52.1"N 128° 3'25.9"W	0	0	1893	NA	1893	72-310 (1704-	1890, 1900	1 fire event

							2014)		
BW	51°39'46.7"N 128° 4'2.7"W	5	3	1656	1893	1656, 1744, 1759, 1778, 1893	82-119 (1895- 2014)	1890, 1900	103
BW	51°39'38.3"N 128° 4'30.3"W	15	3	1656	1893	1656, 1744, 1778, 1797, 1856, 1893	59-311 (1703- 2014)	1900	86
BW	51°39'37.7"N 128° 3'19.1"W	0	0	1893	NA	1893	70-360 (1654- 2014)	1900	1 fire event
BW	51°39'34.0"N 128° 3'48.0"W	3	0	1893	NA	1893	76-293 (1721- 2014)	NA	1 fire event
BW	51°39'31.8"N 128° 4'47.5"W	2	1	1893	NA	1893	49-533 (1481- 2014)	1890, 1900	1 fire event
BB	51°39'59.7"N 128° 3'27.0"W	0	0	1893	NA	1893	71-171 (1843- 2014)	1890	1 fire event
BB	51°39'42.0"N 128° 4'12.7"W	26	10	1656	1893	1656, 1744, 1759, 1778, 1893	52-120 (1894- 2014)	NA	103
BB	51°39'46.1"N 128° 3'34.9"W	1	0	1893	NA	1893	57-120 (1894- 2014)	1900	1 fire event
BB	51°39'33.1"N 128° 3'56.8"W	6	0	1893	NA	1893	74-110 (1904- 2014)	NA	1 fire event
BB	51°40'11.2"N 128° 3'36.1"W	1	0	1893	NA	1778, 1893	120-163 (1851- 2014)	1780	258
BB	51°39'49.1"N 128° 4'14.6"W	4	2	1656	1893	1656, 1744, 1759, 1778, 1893	91-219 (1705- 2014)	1890	103

Table B1.3-B1.4 Candidate Poisson generalized linear models were selected with

Akaike Information Criterion (AIC_c) for the two response variables: 1) abundance of fire-scared trees, and; 2) frequency of fire events with nine predictor variables (vegetation type [four types], elevation, slope, aspect, distance to habitation site in metres, and distance to shoreline in metres). Model averaging was conducted and the most

parsimonious model related the abundance of fire scars to distance from former habitation sites only and this was confirmed with model validation of residuals. In the second analysis, the most parsimonious model related the frequency of fire events to three variables (distance from former habitation site, aspect, and bog forest vegetation type). The explained deviance (*pseudo R*²) is the quality of fit achieved by maximum likelihood for the model.

B1.3 Abundance of fire-scarred trees					
Step 1: Assess collinearity in predictor variables	All variables have Variance Inflation Factor (VIF < 3), no evidence of correlation among variables				
Step 2: Specify parameters (variables) to include in the model	Elevation, slope, aspect, distance to habitation, distance to shoreline, vegetation type (zonal forest, bog forest, bog woodland and blanket bog)				
Step 3: Choose an appropriate distribution	Poisson distribution for count data				
Step 4: Graphically check the variances in the data, outliers, etc.	No apparent issues and no assumptions violated				
Step 5: Fit the beyond optimal (global) GLM and undertake model selection with AIC _c	Elevation, slope, aspect, distance to habitation, distance to shoreline and vegetation type. AIC _c 126.0. Model uncertainty exists; there are several models within the 95% confidence set of models. Model averaging is warranted. Note that vegetation type was not a parameter in the 95% confidence interval set of models.				
Step 6: Apply model averaging and 95% confidence intervals	Parameter	Estimate	Unconditional Standard Error	95% Confidence intervals	Relative importance
	Intercept	1.963e+00	1.707e-01	(1.591, 2.372)	
	Elevation	-4.302e-03	6.790e-03	(-0.0178, 0.009)	0.15
	Distance to shoreline	2.458e-04	8.440e-04	(-0.001, -0.001)	0.13
	Distance to habitation	-2.002e-03	4.383e-04	(-0.002, -0.001)	1.00
	Slope	-1.114e-02	1.350e-02	(-0.038,	0.17

				0.0166)	
	Aspect	3.255e-05	9.327e-04	(-0.002, 2.508)	0.12
Step 7: Select the most parsimonious model from the selection with AIC _c	Parameters	Log-Likelihood	AIC_c	Delta	Weight
	Distance to habitation	-51.81	108.06	0.00	0.42
	Distance to habitation, slope	-51.45	109.83	1.77	0.17
	Distance to habitation, elevation	-51.58	110.08	2.02	0.15
	Distance to habitation, aspect	-51.76	110.44	2.39	0.13
Step 8: Select the most parsimonious model	The most parsimonious model contained the parameter distance to habitation only and was the most significant predictor.				
Step 9: Assess model adequacy	I used several graphical methods to assess model adequacy and found no evidence of overdispersion or departures from model assumptions. Distance from habitation explained 65% of the variation in the abundance of fire-scared trees (<i>pseudo R</i> ²).				
B1.4 Frequency of fire events					
Step 1: Assess collinearity in predictor variables	All variables have Variance Inflation Factor (VIF < 3), no evidence of correlation among variables				
Step 2: Specify parameters (variables) to include in the model	Elevation, slope, aspect, distance to habitation, distance to shoreline, vegetation type (zonal forest, bog forest, bog woodland and blanket bog)				
Step 3: Choose an appropriate distribution	Poisson distribution for count data				
Step 4: Graphically check the variances in the data, outliers, etc.	No apparent issues and no assumptions violated				
Step 5: Fit the beyond optimal (global) GLM and undertake model selection with AIC _c	Elevation, slope, aspect, distance to habitation, distance to shoreline and vegetation type. AIC _c 119.3. Model uncertainty exists; there are several models within the 95% confidence set of models. Model averaging is warranted.				
Step 6: Apply model averaging and 95% confidence	Parameter	Estimate	Unconditional Standard Error	95% Confidence intervals	Relative importance

intervals	Intercept	1.678	0.270	(1.094, 2.244)	
	Elevation	-0.003	0.006	(-1.664, 0.125)	0.15
	Distance to shoreline	0.001	0.0007	(1.971, 0.002)	0.50
	Distance to habitation	-0.002	0.0006	(-3.532, - 0.001)	1.00
	Slope	0.002	0.011	(-2.657, 0.026)	0.18
	Aspect	0.001	0.0008	(-0.002, 2.508)	0.76
	Bog forest	0.716	0.317	(-6.479, 1.315)	0.54
	Bog woodland	0.030	0.365	(- 7.084,0.792)	0.44
	Zonal forest	0.367	0.366	(-3.633, 1.130)	0.52
	Blanket bog	0.425	0.331	(-1.002, 0.013)	0.44
Step 7: Select the most parsimonious model from the selection with AIC _c	Parameters	Log-Likelihood	AIC _c	Delta	Weight
	Distance to habitation, aspect, bog forest type	-56.31	119.54	0.00	0.30
	Distance to habitation, aspect	-52.32	120.29	0.75	0.20
	Distance to habitation	-54.67	121.85	2.31	0.09
	Distance to habitation, aspect, bog forest type, zonal forest type, distance to shoreline	-56.23	122.07	2.53	0.08
	Distance to habitation, elevation, slope, distance to shoreline	-56.30	122.21	2.67	0.08
	Distance to habitation, bog forest type	-58.93	122.30	2.76	0.07

	slope				
	Distance to habitation, elevation, bog forest type	-52.01	123.12	3.58	0.05
Step 8: Select the most parsimonious model	The most parsimonious model contained the parameters distance to habitation, aspect, and bog forest type. Distance to habitation site is the most important predictor variable followed by aspect and bog forest vegetation type.				
Step 9: Assess model adequacy	I used a range of graphical methods to assess model adequacy and I found no evidence of overdispersion or departures from model assumptions. Distance from habitation, aspect, and bog forest vegetation type explained 59% of the variation in the frequency of fire events (<i>pseudo R</i> ²).				

Table B1.5 G-test of goodness of fit of observed versus expected frequencies of fire occurrence and single and multiple phase interactions of the PDSI, ENSO, PDO, and AO climate indexes. Abbreviations are the El Niño-Southern Oscillation (ENSO), the Pacific Decadal Oscillation (PDO), the Palmer Drought Severity Index (PDSI), and the Arctic Oscillation (AO). All single and phase combinations were nonsignificant.

Climate Index	G-test of goodness of fit	X ² Degrees of freedom	P-value
PDSI	$G = 3.1916$	1	0.07402
ENSO	$G = 0.5576$	1	0.4552
PDO	$G = 0.603$	1	0.4374
AO	$G = 0.6254$	1	0.4291
ENSO x PDO	$G = 0.3286$	3	0.9546
ENSO x AO	$G = 1.714$	3	0.6338
PDO x AO	$G = 1.686$	3	0.64
ENSO x PDO x AO	$G = 3.2029$	7	0.8656

Appendix B2: Tree ring reconstructions and bivariate event analyses

Figure B2.1 Years of fire occurrence in the study area on Hecate Island, British Columbia, Canada are indicated with red diamond symbols. Tree-ring reconstructions of

the Palmer Drought Severity Index (PDSI (green), El Niño-Southern Oscillation (ENSO) (orange), Pacific Decadal Oscillation (PDO) (blue), and the Arctic Oscillation (AO) (pink) indexes are plotted with their associated warm/dry and cool/wet phases during the analysis period AD 1300-1900. References for tree-ring reconstructions are included in Appendix B1: Table B1.1.

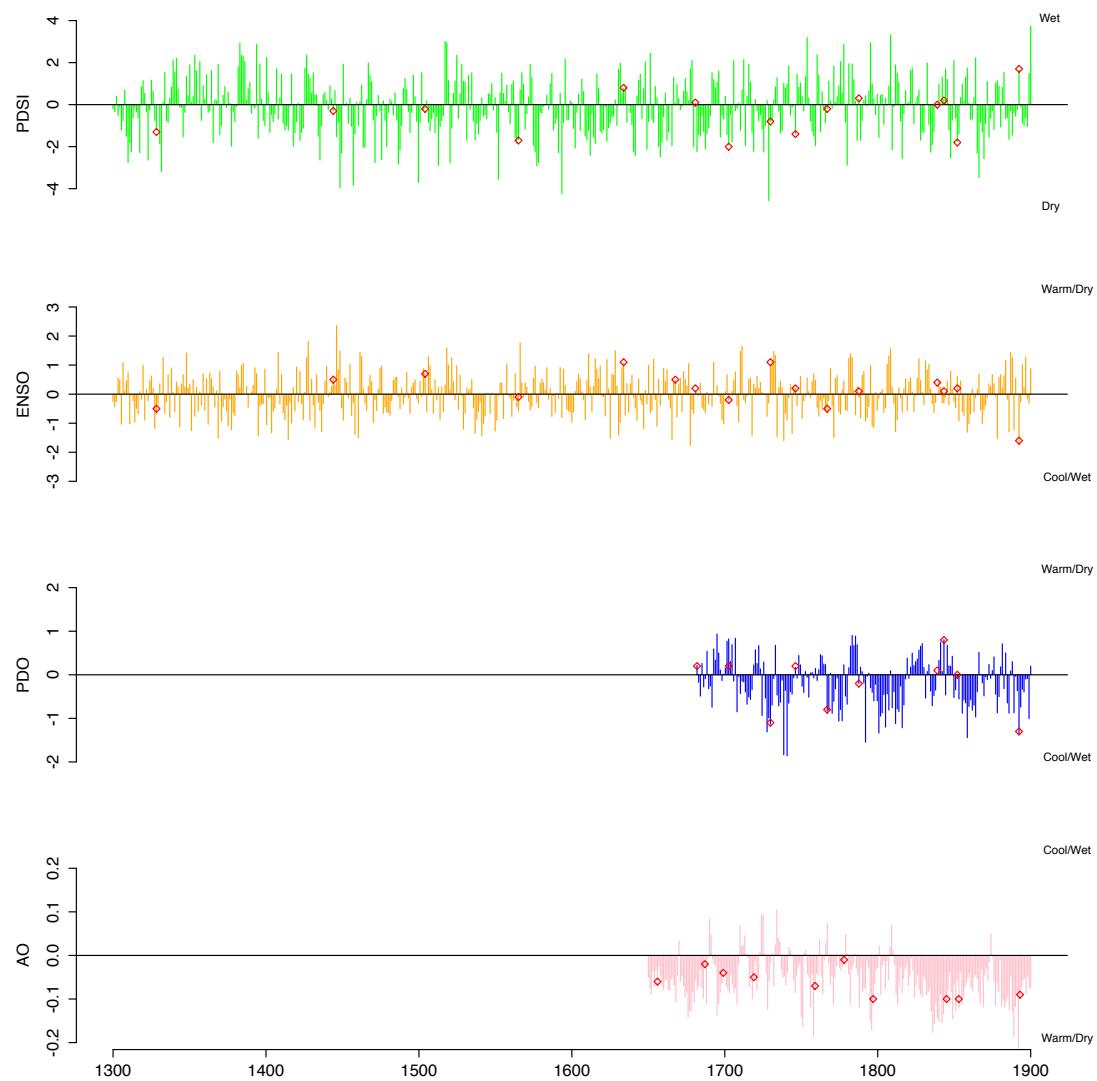


Figure B2.2: Relative frequencies of fire events. Observed (black bars) versus expected (grey bars) of fire events and climate occurrences for each of the two-way (a-c) and three-

way (d) combined phases of the El Niño-Southern Oscillation (ENSO), Pacific Decadal Oscillation (PDO), and Arctic Oscillation (AO) climate indexes (see methods: single index and multiple index combinations). Positive phases are indicated with the (+) symbols and negative phases with the (-) symbols. Significant departures from expected fire occurrence were evaluated with the G-test of goodness of fit for all single, two-way, and three-way interactions.

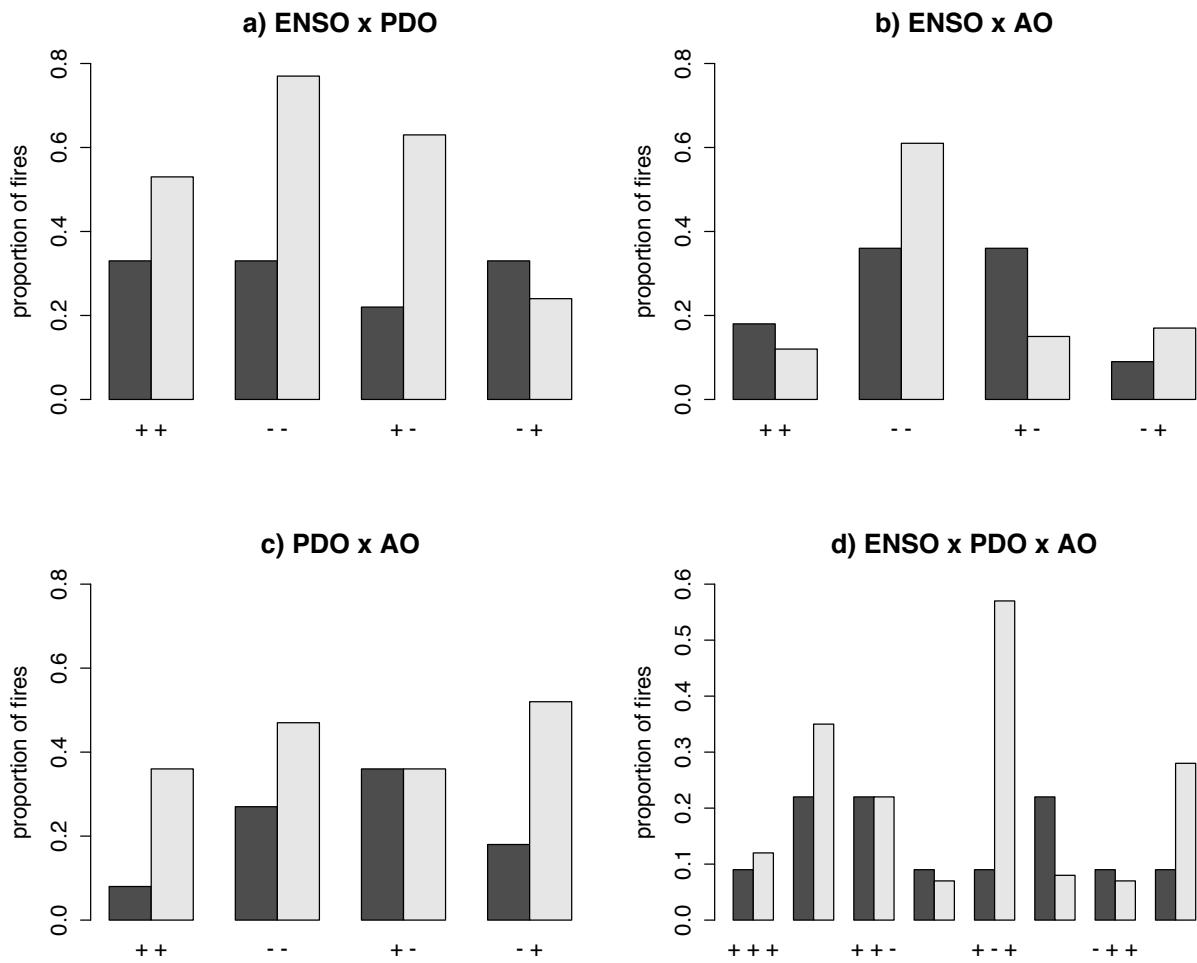
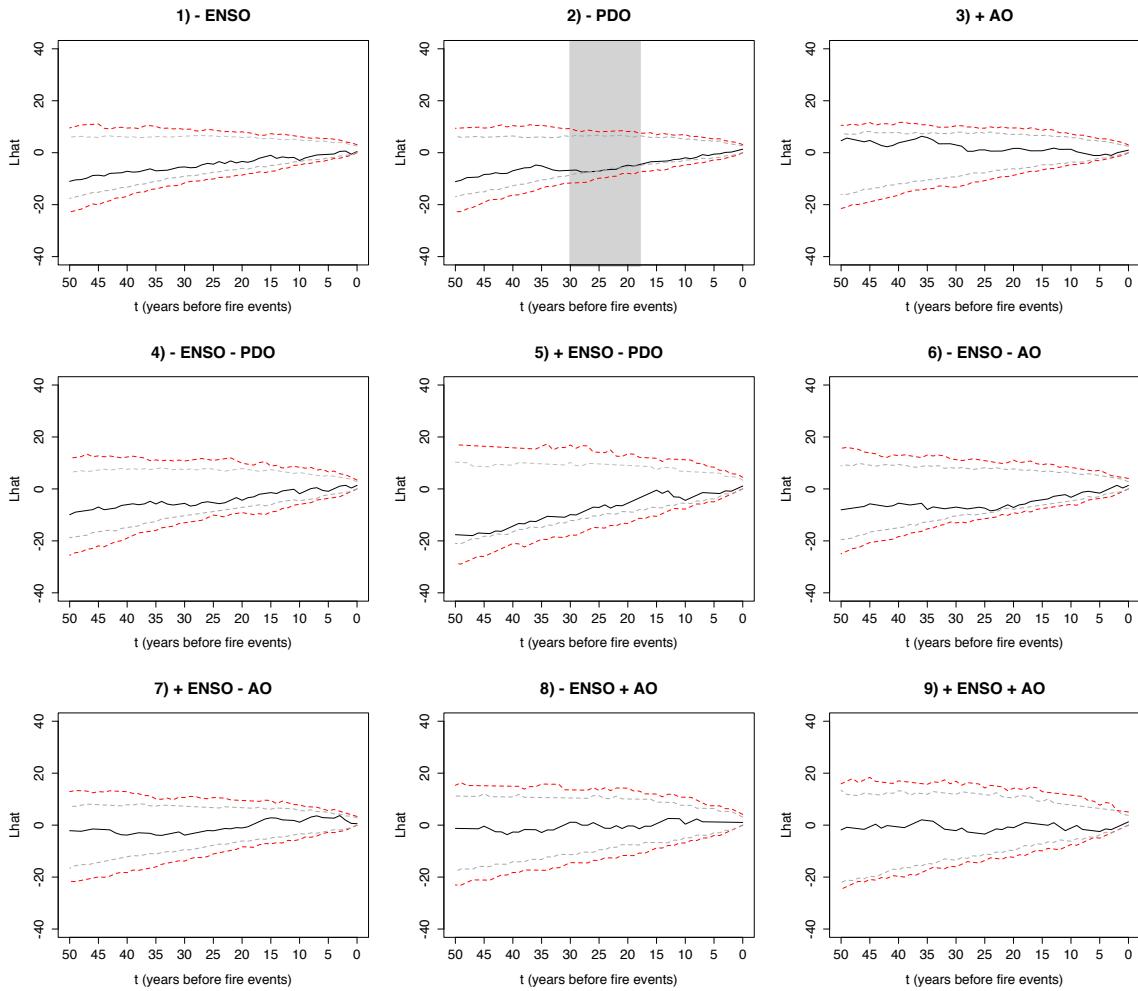


Figure B2.3 Fifteen Bivariate Event Analyses (BEA) of the temporal associations between fire years ($n = 16$) and extreme climate events (single indexes $n = 50$, combined

indices $n = 100$). Black lines above the dotted red (99% confidence envelopes) and the dotted grey (95% confidence envelopes) lines indicate synchrony between the two events (events occurred more often than expected, t years prior to fire events) and black lines below lower confidence envelopes indicate asynchrony (extreme events occur less often than expected t years prior to fire events). Black lines between confidence envelopes indicate independence between events. Confidence envelopes are based on 1000 Monte Carlo simulations and years of significant asynchrony are shaded in grey.



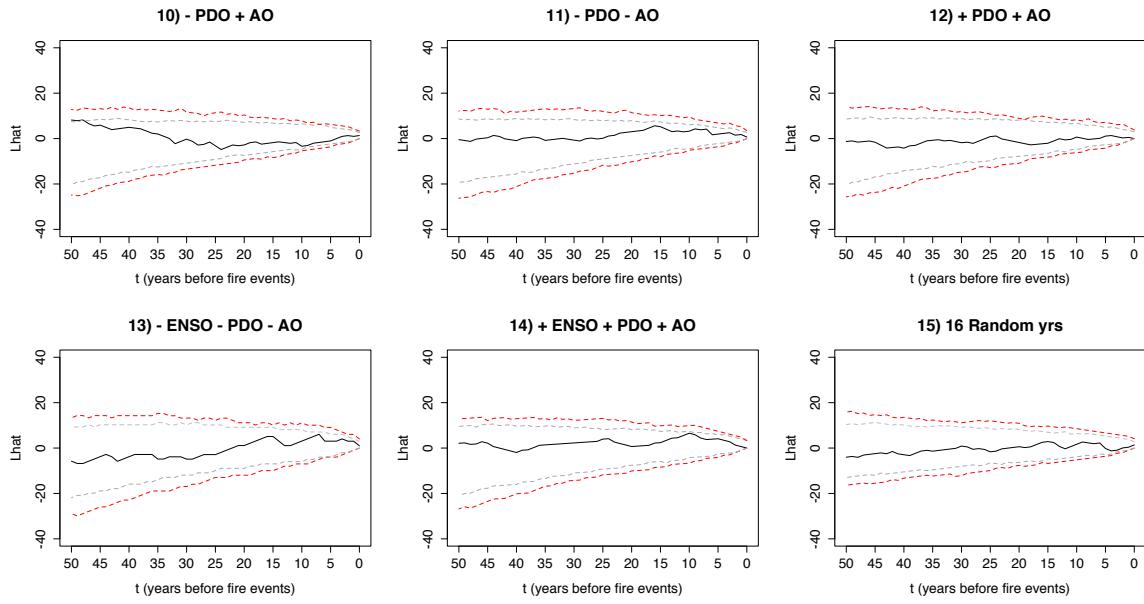
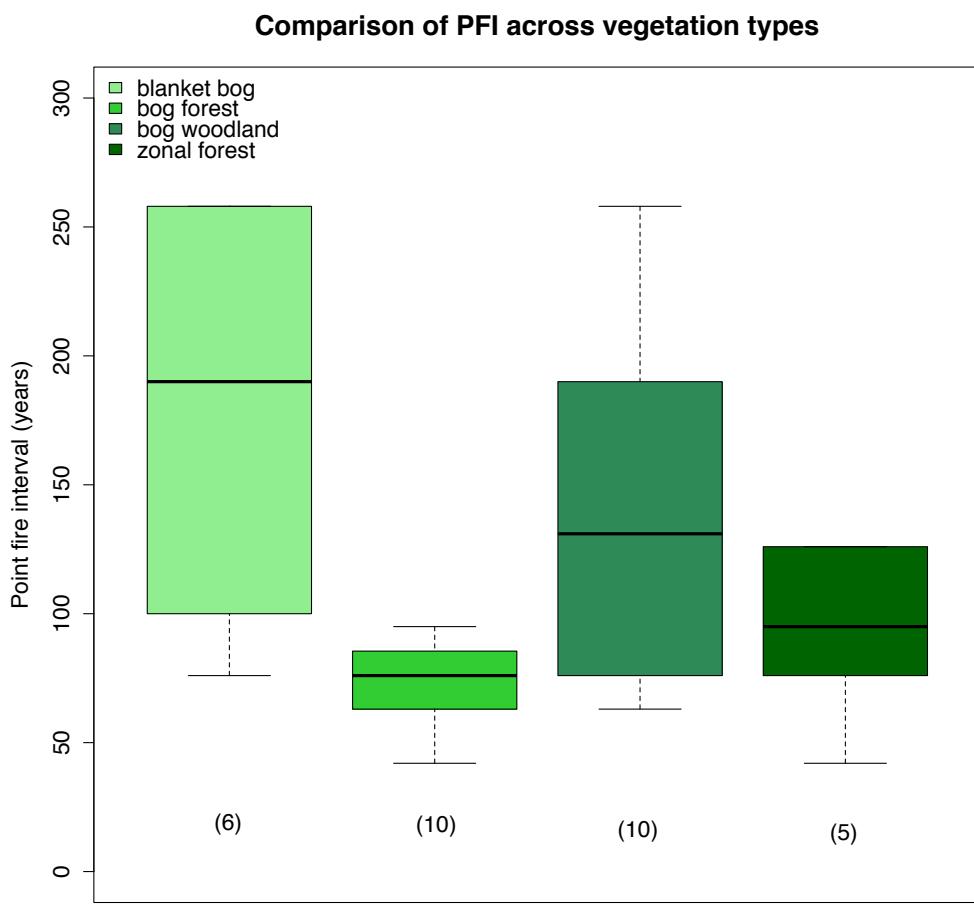


Figure B2.4 Point fire intervals (PFI [mean fire interval obtained from all trees within one-hectare plots]) were pooled by vegetation type (four types differentiated by different shades of green) during the analysis period AD 1376-1893. The number of plots assessed is given in parentheses. The boxes enclose the 25th to 75th percentiles, and whiskers enclose the 10th to 90th percentiles. The horizontal lines across each box indicate the median. No pairwise comparisons in median PFI were statistically significant.



Appendix C1: Additional figures and tables describing plant associations with fire activity

Table C1.1 The list of species found in the 30 fire history plots on Hecate Island and in the 20 transects in former habitation and control sites. * Signifies that the species has been associated with historic anthropogenic burning by First Nations groups in British Columbia (see reference column). Direct and indirect uses of anthropogenic fire are described in Table C1.2.

Ground layer		* References
Common name	Latin name	
Step moss	<i>Hylocomium splendens</i>	
Kindbergia spp.	<i>Kindbergia spp.</i> (various)	
Clear moss	<i>Hookeria lucens</i>	
Lanky moss	<i>Rhytidadelphus loreus</i>	
Yellow moss	<i>Homalothecium fulgescens</i>	
Coastal leafy moss	<i>Plagiomnium insigne</i>	
Flat-moss	<i>Plagiothecium undulatum</i>	
Common fold-leaf liverwort	<i>Diplophyllum albicans</i>	
Golden shortcapsuled moss	<i>Brachythecium frigidum</i>	
Sphagnum spp.	<i>Sphagnum spp.</i> (various)	
Large leafy moss	<i>Rhizomnium glabrescens</i>	
Comb liverwort	<i>Riccardia multifida</i>	
Three-toothed whip liverwort	<i>Bazzania tricrenata</i>	
Cedar-shake liverwort	<i>Plagiochila porelloides</i>	
Bazzania trilobata	<i>Bazzania trilobata</i>	
Common scissor-leaf liverwort	<i>Herbertus aduncus</i>	
Shiny liverwort	<i>Pellia neesiana</i>	
Peltigera spp.	<i>Peltigera spp.</i> (various)	
Maritime reindeer	<i>Cladina portentosa</i>	
Dicranum spp.	<i>Dicranum spp.</i> (various)	
Blunt-leaved moss	<i>Scleropodium obtusifolium</i>	
Herb layer		
Common name	Latin name	
Labrador tea	<i>Rhododendron groenlandicum*</i>	(Turner, 1999; Wray and Anderson, 2003; Wray, 2009; Turner, 2014)

Deer fern	<i>Blechnum spicant</i>	
Spiny wood fern	<i>Dryopteris expansa</i>	
Northern maidenhair	<i>Adiantum aleuticum</i>	
Licorice fern	<i>Polypodium glycyrrhiza</i>	
Bracken fern	<i>Pteridium aquilinum*</i>	(Norton, 1979; Wray and Anderson, 2003; Wray, 2009; Turner, 2014)
Heart-leaved twayblade	<i>Listera cordata</i>	
Bunchberry	<i>Cornus spp.</i>	
Alaska rein-orchid	<i>Piperia unalascensis</i>	
Skunk cabbage	<i>Lysichiton americanus</i>	
Three-leaved foamflower	<i>Tiarella trifoliata</i>	
False lily-of-the valley	<i>Maianthemum dilatatum*</i>	(Turner, 2014)
Spleenwort-leaved goldthread	<i>Coptis aspleniifolia</i>	
Three-leaved goldthread	<i>Coptis trifolia</i>	
Indian hellebore	<i>Veratrum viride</i>	
Northern rice-root	<i>Fritillaria camschatcensis</i>	
Rattlesnake-plantain	<i>Goodyera oblongifolia</i>	
Running club-moss	<i>Lycopodium clavatum</i>	
Crowberry	<i>Empetrum nigrum</i>	
Clasping twistedstalk	<i>Streptopus amplexifolius</i>	
Round-leaved sundew	<i>Drosera rotundifolia</i>	
Deer-cabbage	<i>Fauria crista-galli</i>	
Twinflower	<i>Linnaea borealis</i>	
Swamp gentian	<i>Gentiana douglasiana</i>	
Pacific hemlock parsley	<i>Conioselinum gmelinii</i>	
Five-leaved bramble	<i>Rubus pedatus</i>	
Pacific reedgrass	<i>Calamagrostis nutkaensis</i>	
Bluejoint reedgrass	<i>Calamagrostis canadensis</i>	
Carex sp.	<i>Carex sp.</i>	
Sticky false asphodel	<i>Triantha glutinosa</i>	
Apargidium	<i>Microseris borealis</i>	
Common butterwort	<i>Pinguicula vulgaris</i>	
King gentian	<i>Gentiana sceptrum</i>	
Northern starflower	<i>Trientalis europaea ssp. <i>Arctica</i></i>	
Slender bog orchid	<i>Platanthera stricta</i>	
White bog-orchid	<i>Platanthera dilatata</i>	
Hooded ladies' tresses	<i>Spiranthes romanzoffiana</i>	
Fairy slipper	<i>Calypso bulbosa</i>	
Tall mountain shootingstar	<i>Dodecatheon jeffreyi</i>	
Caltha-leaved avens	<i>Geum calthifolium</i>	
Great burnet	<i>Sanguisorba officinalis</i>	
Partridgefoot	<i>Luetkea pectinata</i>	
Tufted hairgrass	<i>Deschampsia cespitosa</i>	

White beak-rush	<i>Rhynchospora alba</i>	
Narrow-leaved cotton-grass	<i>Eriophorum angustifolium</i>	
Tufted clubbrush	<i>Trichophorum cespitosum</i>	
Ground-pine	<i>Lycopodium dendroideum</i>	
Running clubmoss	<i>Lycopodium clavatum</i>	
Fir clubmoss	<i>Lycopodium selago</i>	
Great sundew	<i>Drosera anglica</i>	
Shrub layer		
Common name	Latin name	
Pacific crab apple	<i>Malus fusca</i>	
Sitka spruce	<i>Picea sitchensis</i>	
Shore pine	<i>Pinus contorta var. contorta*</i>	(Turner, 2014)
Pacific yew	<i>Taxus brevifolia</i>	
Western redcedar	<i>Thuja plicata*</i>	(Turner, 1999; Turner, 2014)
Western hemlock	<i>Tsuga heterophylla*</i>	(Turner, 2014)
Yellow-cedar	<i>Cupressus nootkatensis *</i>	(Turner, 2014)
Mountain hemlock	<i>Tsuga mertensiana</i>	
Red alder	<i>Alnus rubra*</i>	(Turner, 2014)
Salal	<i>Gaultheria shallon*</i>	(Turner, 1999; Wray, 2009; Turner 2014)
False azalea	<i>Menziesia ferruginea</i>	
Red huckleberry	<i>Vaccinium parvifolium*</i>	(Turner, 1999; Turner 2014)
Kinnikinnick	<i>Arctostaphylos uva-ursi</i>	
Bog cranberry	<i>Vaccinium oxycoccus*</i>	(Wray, 2009; Turner, 2014)
Oval-leaved blueberry	<i>Vaccinium ovalifolium*</i>	(Turner, 2014)
Stink currant	<i>Ribes bracteosum</i>	
Thimble berry	<i>Rubus parviflorus*</i>	(Wray, 2009; Turner, 2014)
Black gooseberry	<i>Ribes lacustre*</i>	(Turner, 2014)
Salmonberry	<i>Rubus spectabilis*</i>	(Wray, 2009; Turner, 2014)
Sweetgale	<i>Myrica gale</i>	
Common juniper	<i>Juniperus communis</i>	
Bog-rosemary	<i>Andromeda polifolia</i>	
Alpine-azalea	<i>Loiseleuria procumbens</i>	
Copperbush	<i>Elliottia pyroliflora</i>	
Bog blueberry	<i>Vaccinium uliginosum*</i>	(Wray, 2009; Turner, 2014)
Dwarf blueberry	<i>Vaccinium caespitosum*</i>	(Turner, 1999; Turner 2014)
Lingonberry	<i>Vaccinium vitis-idaea*</i>	(Turner, 2014)

Table C1.2 Documented ethnobotanical evidence of direct and indirect uses of fire as a tool for resource management by First Nations groups in coastal British Columbia and

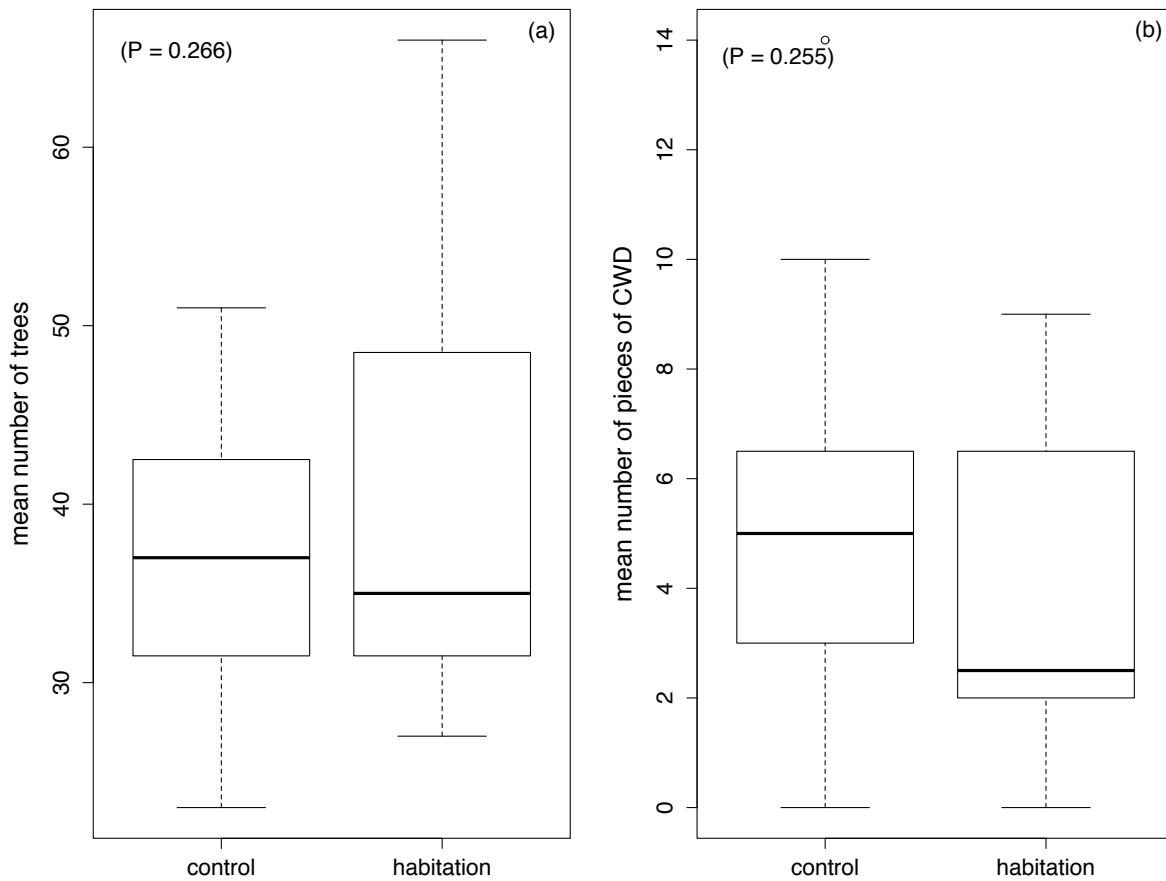
Western Washington. Much of the information from this table has been adapted from extensive ethnographic research on anthropogenic burning by Dr. Nancy Turner.

Management type	Burning season	Direct effects	Indirect effects	Genus or species affected	References
Hunt	Fall	Corral game, force into open meadows/bogs, corridors or waterways	Increase fodder, create more accessible and predictable hunting locations	Deer (<i>Odocoileus hemionus</i>), rabbit (<i>Sylvilagus spp.</i>), geese (<i>Chenm Branta spp.</i>), ducks (<i>Anas spp.</i>)	(Turner, 1999; Turner, 2014)
Individual trees	Summer	Create fire wood, fall trees, activate pitch, house construction, canoes	Clear brush around trees, mark territory	Western redcedar (<i>Thuja plicata</i>), shore pine (<i>Pinus contorta var. contorta</i>), Sitka spruce (<i>Picea sitkachensis</i>), western hemlock (<i>Tsuga heterophylla</i>)	(Turner, 1975; Turner, 1999; Turner, 2014)
Fruiting shrubs	Early spring, late summer	Stimulate growth of woody perennials, remove underbrush to increase the productivity of food plants, create fuel for firewood	Increase berry/seed production, remove underbrush, clear areas surrounding habitation sites, reduce the effects of accidental fires	Blueberry (<i>Vaccinium spp.</i>), salal (<i>Gaultheria shallon</i>), gooseberry, black current, (<i>Ribes spp.</i>), thimbleberry, salmonberry (<i>Rubus spp.</i>)	(Turner, 1975; Turner, 1999; Deur and Turner, 2002; Lepofsky et al. 2003; McDonald, 2005; Lepofsky and Lertzman, 2008; Trusler and Johnson, 2008; Turner, 2014)
Clear land, maintain/expand natural forest openings	Spring, late summer, and early fall	Increase the productivity of herbaceous food plants, select culturally important tree species (i.e. western redcedar)	Alter successional stages of forests, create a diversity of habitats, keep areas clear surrounding habitations, create fodder for game/hunting areas, easier travel through forests	Bracken fern (<i>Pteridium aquilinum</i>), labrador tea (<i>Rhododendron groenlandicum</i>), blueberry, (<i>Vaccinium spp.</i>), false lily-of-the-valley (<i>Maianthemum dilatatum</i>)	(Norton, 1979; Turner, 1999; Gottesfeld, 1994; Wray and Anderson, 2003; Storm and Shebitz, 2006)
Spiritual and defense	During ceremony (any season)	Increase rains during drought, bring back salmon, maintain open/safe corridors, communicate	Decrease enemy hiding spots, replenish soils	Several species	(Turner, 2014)

Table C1.3 The fire history chronology reconstructed from fire scars sampled in the 287 ha fire on Hecate Island and validated with tree-rings sampled from forests with no aboveground evidence of fire activity in the broader study area.

Number of fire scars	99
Total years with fire	16
Number of fire events	15
Total years of record	16,290
Average years of growth/scar	164
Composite mean fire interval (MFI) (1376-1893)	39
Point mean fire interval (PFI) (1376-1893)	95
Pre-contact composite mean fire interval (MFI) (1376-1893)	39
Post-contact composite mean fire interval (MFI)	Not applicable, no fire events recorded

Figure C1.4 Box and whisker plots of the presence of fire evidence in forests surrounding former habitation and control sites ($n = 20$) located on 15 islands throughout the region (see Fig. 4.1). Boxes represent the second and third quartile ranges and the centreline is the median. Circles represent outliers. There was no significant difference in the number of trees and the average number of pieces of coarse woody debris (CWD) between transects in former habitation and control sites.



Appendix C2: Model steps, fit and validation for two response variables predicted using a Generalized Linear Mixed-effects Model (GLMM)

Table C2.1 Candidate GLMM models were selected with Akaike Information Criterion (AIC_c) for two response variables: 1) Abundance of traditional food plants, and; 2) Density of western redcedar with the parameters (elevation, slope, aspect, distance to habitation site in metres, and distance to shoreline in metres). Vegetation type was included as the random model effect due to hierarchical clustering in the data set. The explained deviance (*pseudo R*²) is the quality of fit achieved by maximum likelihood for the model.

a) Density of western redcedar in fire history plots	
Step 1: Assess collinearity in predictor variables	All variables have Variance Inflation Factor (VIF < 3). No evidence of correlation among variables
Step 2: Specify fixed and random effects to include in the model	Fixed model effects: (explanatory variables): aspect, elevation, slope, distance to shoreline, distance to habitation, and mean fire interval (MFI). Random model effect: vegetation type
Step 3: Choose an error distribution and link function	Poisson distribution and log link function for count data
Step 4: Graphically check the variances in the data, outliers, etc.	Evidence of overdispersion, scale parameter = 12. Adding covariates and interaction terms were unsuccessful. Attempt to use an alternate distribution and refit model: return to Step: 3
Step 3: Choose an error distribution and link function	Negative binomial distribution and log link function
Step 4: Graphically check the variances in the data, outliers, etc.	No apparent issues
Step 5: Fit the beyond optimal (global) GLMM and undertake model	MFI, aspect, elevation, slope, distance to shoreline, distance to habitation, and vegetation type. AIC _c 266.8 Model uncertainty exists; there are several models within the

selection with AIC _c		95% confidence set of models. Model averaging is warranted.			
Step 6: Apply model averaging with 95% confidence intervals	Parameter	Estimate	Unconditional standard error	95% Confidence Intervals	Relative importance
	Intercept	3.593	0.623	(2.350, 4.837)	
	MFI	-0.005	0.002	(-0.011, -0.000)	0.68
	Aspect	-0.001	0.001	(-0.004, 0.002)	0.22
	Elevation	-0.003	0.007	(-0.018, 0.011)	0.20
	Slope	0.018	0.023	(-0.030, 0.066)	0.22
	Distance to shoreline	0.000	0.000	(-0.001, 0.001)	0.14
	Distance to habitation	0.000	0.001	(-0.00, 0.001)	0.28
	Bog forest	1.241	0.508	(0.197, 2.285)	0.15
	Bog woodland	0.886	0.478	(-0.093, 1.866)	0.15
	Zonal forest	1.179	0.603	(-0.003, 2.363)	0.15
	Blanket bog	0.675	0.367	(-0.001, 2.312)	0.15
Step 7: Select the most parsimonious model from the selection with AIC _c	Parameters	Log-Likelihood	AIC_c	Delta	Weight
	MFI	-128.62	264.16	0.00	0.26
	MFI, distance to habitation	-130.78	266.00	1.84	0.10
	MFI, slope	-128.35	266.30	2.14	0.10
	MFI, aspect	-128.43	266.46	2.30	0.08
	MFI, elevation	-128.61	266.81	2.65	0.07
	MFI, distance to habitation, slope	-127.34	267.18	3.02	0.06
	MFI, Distance to habitation, aspect, vegetation type	-127.69	267.87	3.71	0.05
Step 8: Select the most parsimonious model	Although the most parsimonious model contained the parameter MFI only and MFI was the most important predictor variable, I found that this model was not much better than the full (global) model. I used AIC _c together with hypothesis testing to select the best model which contained the parameters MFI, distance from habitation, aspect, elevation, and vegetation type				
Step 9: Assess model adequacy	Use a range of graphical methods to assess model adequacy. No evidence of overdispersion or departures of model				

	assumptions.				
b) Abundance of traditional food plants					
Step 1. Assess collinearity in predictor variables	All variables have Variance Inflation Factor (VIF < 3). No evidence of correlation among variables				
Step 2. Specify fixed and random effects to include in the model	Fixed model effects: (explanatory variables): elevation, slope, aspect, distance to habitation, distance to shoreline, and Mean Fire Interval (MFI). Random model effect: vegetation type				
Step 3: Choose an error distribution and link function	Poisson distribution and log link function for count data				
Step 4: Graphically check the variances in the data, outliers, etc.	No apparent issues				
Step 5: Fit the beyond optimal (global) GLMM and undertake model selection with AIC _c	Elevation, slope, aspect, distance to habitation, distance to shoreline, MFI, ~1 vegetation type. AIC _c : 134.6 Model uncertainty exists; there are several models within the 95% confidence set of models. Model averaging is warranted.				
Step 6: Apply model averaging with 95% confidence intervals	Parameter	Estimate	Unconditional Standard Error	95% Confidence intervals	Relative importance
	Intercept	3.441	0.300	(2.853, 4.029)	
	Elevation	-0.015	0.004	(-0.024, -0.006)	0.87
	MFI	-0.001	0.001	(-0.005, 0.001)	0.38
	Distance to shoreline	-0.000	0.005	(-0.002, -2.045)	0.21
	Distance to habitation	-0.004	0.000	(-0.001, -0.000)	1.00
	Slope	0.015	0.008	(-0.001, 0.032)	0.59
Step 7: Select the most parsimonious model from the selection with AIC _c	Parameters	Log-Likelihood	AIC _c	Delta	Weight
	Distance to habitation, elevation	-60.31	130.23	0.00	0.20
	Distance to habitation, MFI, elevation, slope	-58.88	130.26	0.03	0.20
	Distance to habitation,	-59.11	130.72	0.49	0.16

	elevation, aspect				
	Distance to habitation, elevation, aspect, slope, MFI	-57.75	131.14	0.91	0.13
	Distance to habitation, elevation, MFI	-58.19	132.03	1.80	0.08
Step 7: Select the most parsimonious model		Model averaging indicated that the most parsimonious model contained the predictor variables distance to habitation and elevation. This model was not much improved from the full (global) model; therefore, I used AIC _c together with hypothesis testing to select the best model which contained the parameters distance from habitation, elevation, aspect, slope, and MFI.			
Step 8: Assess model adequacy		Use a range of graphical methods to assess model adequacy. No evidence of overdispersion, or departures of model assumptions.			

Appendix D1: Supporting light detection and ranging (LiDAR) and field-based ecological analyses and figures.

Figure D1.1 Regeneration of seedlings (<7.5 cm diameter at breast height [dbh]) was sampled on the downhill slope in five replicate subplots (1 m wide x 4 m long) contained within every 11.28 m plot (400 m²).

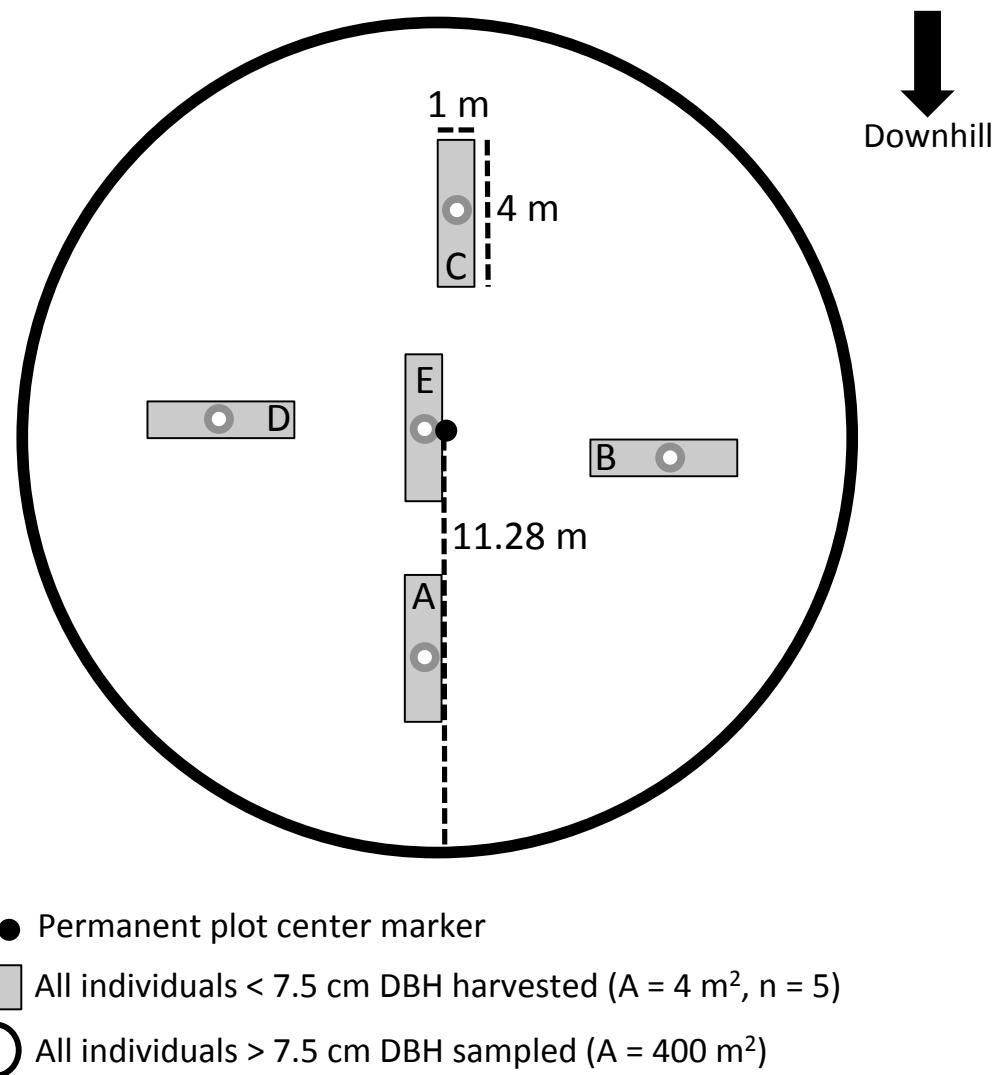


Figure D1.2. Box and whisker plots describe the difference in diameter at breast height (DBH) >7.5 cm between burned (27) and unburned (11) plots aggregated across four vegetation types (zonal forest, bog forest, bog woodland, and blanket bog) and modelled with a nested analysis of variance (ANOVA). Boxes represent the second and third quartile ranges, and the centreline is the median. I found no significant difference in DBH between burned and unburned plots.

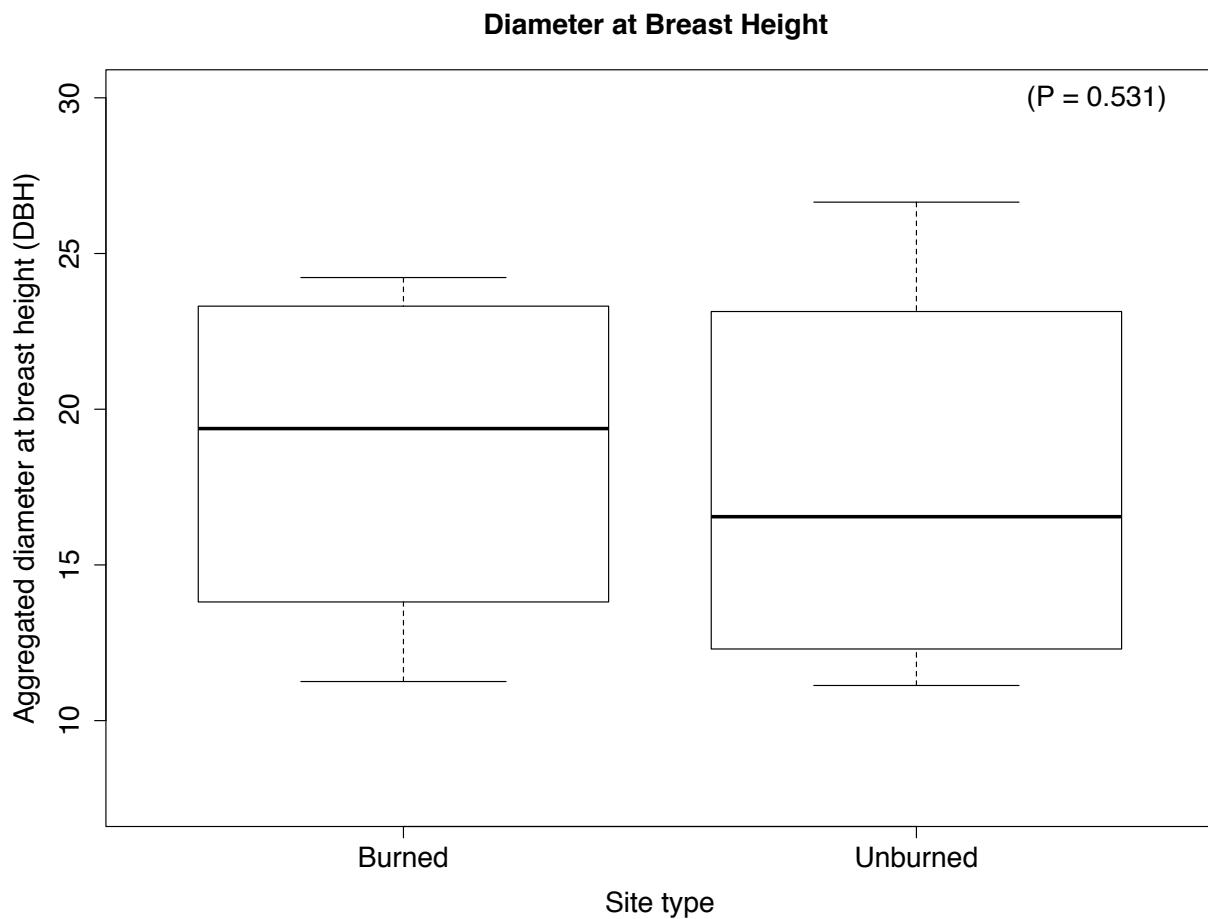


Figure D1.3 Box and whisker plots describe the forest metric Normalized Difference Vegetation Index (NDVI) between burned (27) and unburned (11) plots aggregated across four vegetation types (zonal forest, bog forest, bog woodland, and blanket bog) and modelled with a nested analysis of variance (ANOVA). Boxes represent the second and third quartile ranges, and the centreline is the median. I found no significant difference in NDVI between burned and unburned plots.

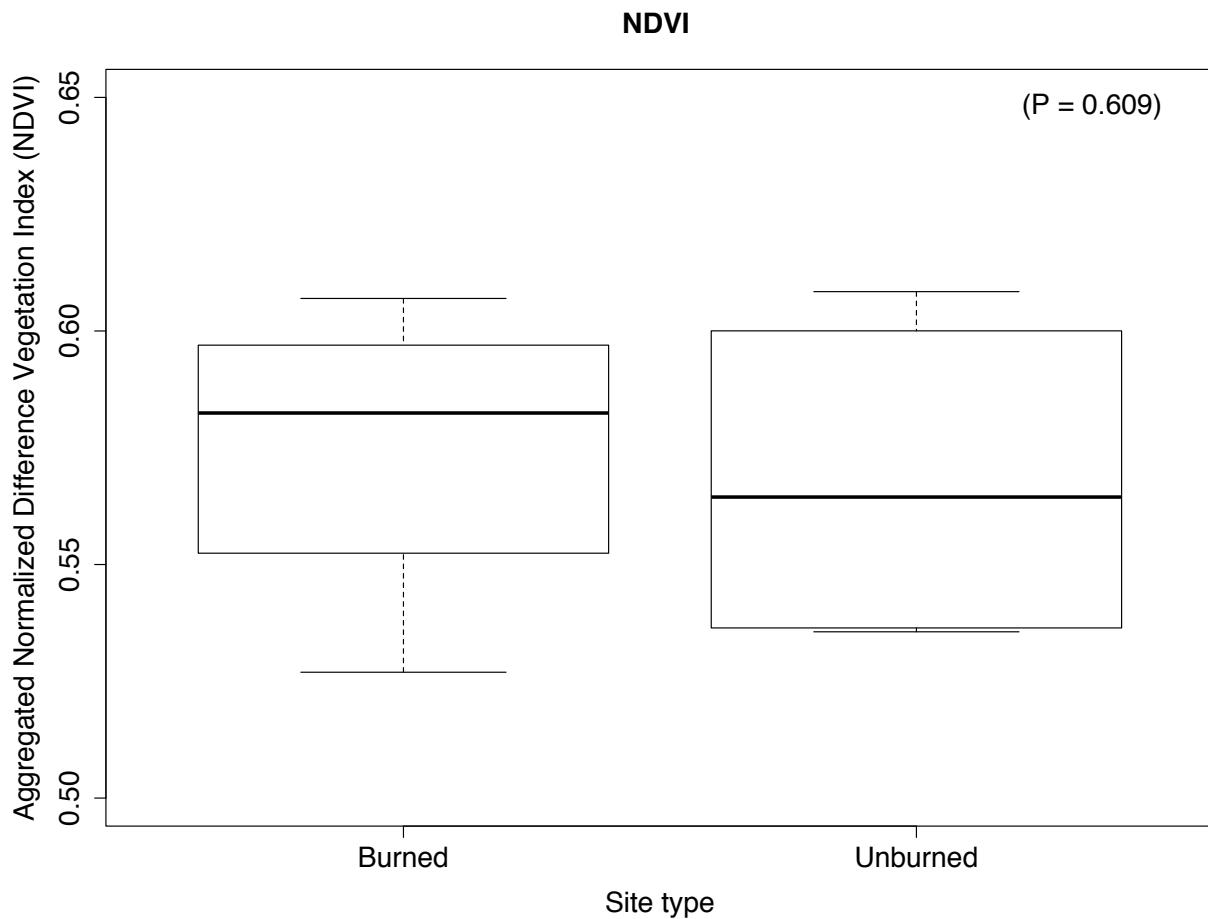
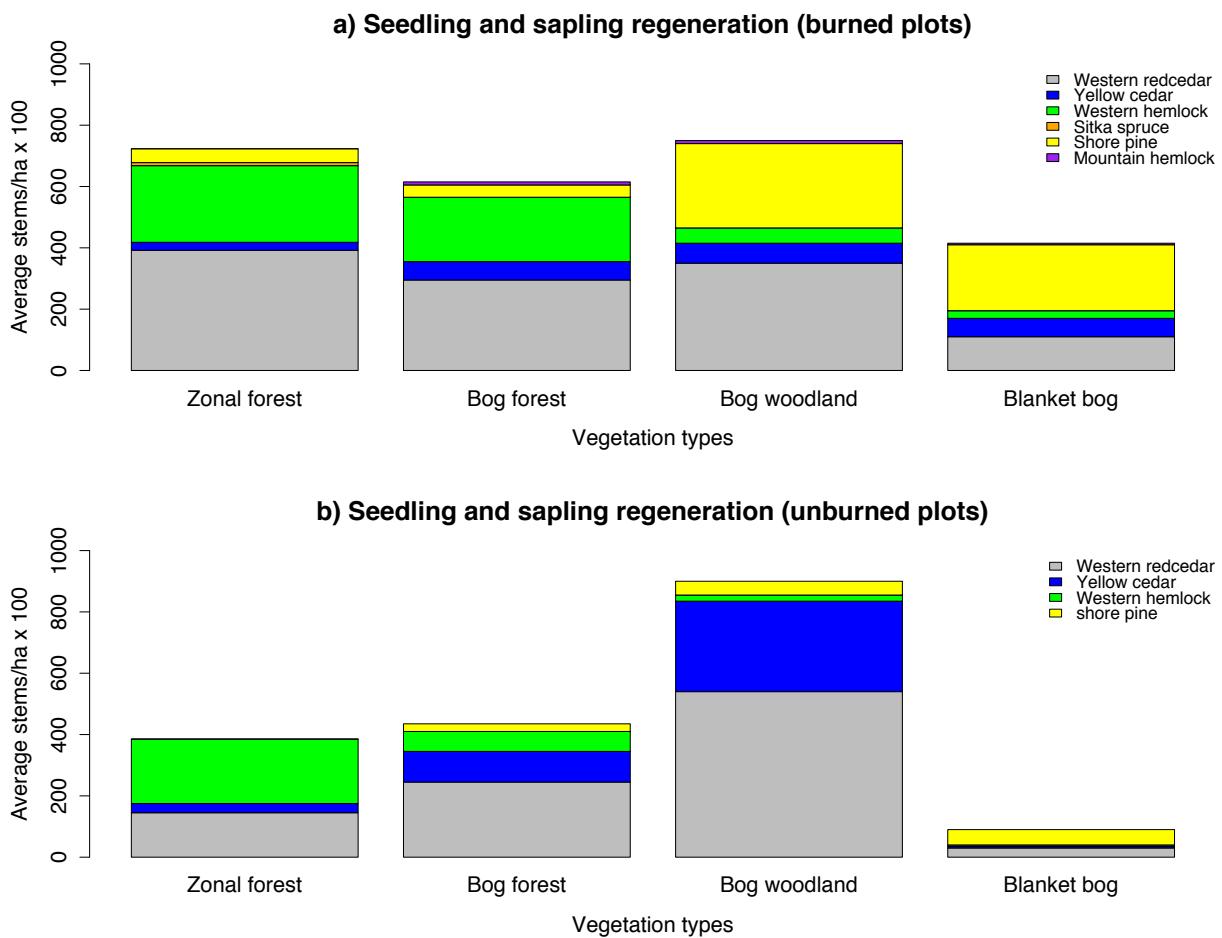


Figure D1.4 The amount of regenerating seedlings in each of the four vegetation types (western redcedar (grey), yellow cedar (blue), western hemlock (green), Sitka spruce (orange), shore pine (yellow), and mountain hemlock (purple) in a) 27 burned plots and b) 11 plots with no aboveground evidence of fire activity. The average number of stems per hectare multiplied by 100 is presented on the y-axis.



Appendix D2: Supporting forest mensuration datasets and statistical analyses

Table D2.1 Model steps, fit and validation for a two factor nested analysis of variance (ANOVA) with unequal sample sizes. The site type (burned and unburned plots) represents a random factor (Factor A) at the top of the hierarchy and the sample type (four vegetation types: zonal forest, bog forest, bog woodland, and blanket bog) are the nesting random factor (Factor B).

Four vegetation sample types nested within burned and unburned site types	
Step 1: Assess assumptions of normality and homogeneity of variance for each null hypothesis.	Factor A: (site type- random factor). The site type included 27 burned, 11 unburned and 31 additional plots included in the larger LiDAR analysis. Factor B: (vegetation type-random factor). Replicates of the four vegetation types were nested under site type and an aggregated dataset was used to assess the means of forest variables measured (tree height, diameter at breast height, tree age, canopy height, canopy cover, canopy complexity and normalized difference vegetation index). Boxplots were aggregated using the entire data set. There was no evidence of non-normality and no evidence of a relationship between the mean and variance.
Step 2: Assess whether the design is balanced	The design is not balanced (there are a different number of replicates of site types and replicates of sample types. This design is best modelled using linear mixed effects procedures.
Step 3: Fit one or more linear models to test the null hypothesis that there is no added variation due to treatment type and no added variation due to forest types within treatment types.	I used a linear mixed effects model to assess the effects of site type against each aggregated variable (tree height, diameter at breast height, tree age, canopy height, canopy cover, canopy complexity, and normalized difference vegetation index) for each vegetation type.
Step 4: Calculate the 95% confidence intervals of the random effects (based on Markov chain Monte Carlo sampling).	The 95% confidence interval for the random effect treatment does not include 0; therefore, I rejected the null hypothesis and concluded that site type (burned vs. unburned plots) has a significant impact on trees found within the four vegetation types. I found no significant impact of vegetation type on the variables measured.

Table D2.2 The average density of trees per plot (11.28 m radius [0.04-ha]) and per hectare along with the median age in years, diameter at breast height (DBH) >7.5 cm and median height in metres in A) 27 burned and B) 11 unburned plots across the four dominant vegetation types (zonal forest, bog forest, bog woodland, and blanket bog).

A) Burned plots (n = 27)					
Vegetation type	Average # trees/plot	Density trees/ha	Median age	Median DBH	Median height
Zonal forest (n = 5)	34	850	180	18	12
Bog forest (n = 8)	40	1000	150	18	11
Bog woodland (n = 10)	42	1050	102	13	6
Blanket bog (n = 4)	11	275	99	10	4
B) Unburned plots (n = 11)					
Vegetation type	Average # trees/plot	Density of trees/ha	Median age	Median DBH	Median height
Zonal forest (n = 3)	80	2000	201	18	9
Bog forest (n = 3)	93	2325	224	18	7
Bog woodland (n = 2)	95	2375	230	12	5
Blanket bog (n = 3)	10	250	123	9	3

Table D2.3 The number of trees in each plot binned by six age classes (0-50, 51-100, 101-150, 151-200, 200-250 and >250 years) in a) 27 burned plots and b) 11 unburned plots across the four dominant vegetation types (zonal forest, bog forest, bog woodland, and blanket bog).

A) Burned plots (n = 27)						
Vegetation type	0-50 yrs	51-100 yrs	101-150 yrs	151-200 yrs	200-250 yrs	>250 yrs
Zonal forest (n = 5)	--	2	11	8	7	9
Bog forest (n = 8)	--	6	13	7	6	9
Bog woodland (n = 10)	--	13	20	4	3	4
Blanket bog (n = 4)	--	5	8	1	1	--
B) Unburned plots (n = 11)						
Vegetation type	0-50 yrs	51-100 yrs	101-150 yrs	151-200 yrs	200-250 yrs	>250 yrs
Zonal forest (n = 3)	1	3	16	20	9	32
Bog forest (n = 3)	1	2	13	19	15	29
Bog woodland (n = 2)	3	3	9	17	21	42
Blanket bog (n = 3)	--	1	6	3	1	--