WILDFIRE RETURN INTERVALS: IMPACTS OF DIMINISHING FIRE RETURN INTERVALS ON BOREAL PEATLANDS USING COMBINED FIELD/LIDAR APPROACHES

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DEDICATION

To my parents.

ABSTRACT

This thesis investigates the influence of increasing fire frequency on the soil characteristics, vegetation structural characteristics, and rate of growth, in peatlands in Single Burn, SB (2011 fire areas) and Double Burn, DB (1956 + 2011 fires) areas within the Boreal Plains Ecozones of Canada. Field data were acquired in July 2019 at the Utikuma Region Study Area (URSA) and were used in combination with a multi-temporal lidar dataset that spanned 17 years. Soil characteristics, such as bulk density, pH, and depth of burn were found to be significantly different in DBs than in SB areas. DB areas were found to have greater heights, and greater rates of growth than SB Peatlands. Results suggest increasing shrub encroachment into peatlands under conditions of increased burning which has implications on the trajectory of peatland succession, and changes to fire fuels, which may impact future peatland resiliency.

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LIST OF SYMBOLS & ABBREVIATIONS

DB	Double Burn
SB	Single Burn, usually in reference to the 2011 SB
SB11	2011 Single Burn Sites, also displayed as 2011SB
SB56	1956 Single Burn Sites, also displayed as 1956SB
GIS	Geographic information system
P:A	Perimeter to Area Ratio
BPE	Boreal Plains ecozone
URSA	Utikuma Region Study Area
Lidar	Light detection and Ranging
DEM	Digital Elevation Model
CHM	Canopy Height Model
DSM	Digital Surface model
P99, p75, p50, p^{nth}	Height at the nth percentile

CHAPTER 1: INTRODUCTION

1.1 INTRODUCTION

The boreal region is a circumpolar biome accounting for approximately one-third of the Earth's forested area (Arneth et al., 2002), and plays a critical role in maintaining global climatic regulation through carbon sequestration, and energy and moisture exchanges (Wieder et al., 2006; Komers & Stanojevic, 2013). Peatlands are a dominant feature of the boreal biome and are characterized primarily as bog and fen wetlands with deep organic deposits associated with the storage of large amounts of net carbon C (Vitt, 1990). Peatlands form deep organic layers because of an imbalance between the rates of production in comparison to plant decomposition (Benscoter, 2006; Wieder et al., 2006; Benscoter & Vitt, 2008). Within Canada, peatlands are estimated to store more than half of the country's soil C deposits, despite only making up a comparatively small portion of the land-area, approximately 12 % (Tarnocai, 2006). Boreal peatlands typically act as net carbon sinks, but with increasing air temperatures disproportionally affecting northern latitude ecosystems, these ecosystems are becoming increasingly vulnerable to drying (Thompson et al., 2017b; Nelson et al., 2021). With drying conditions, peatlands also have the capacity to release large amounts of atmospheric carbon dioxide (CO_2) into the atmosphere, feeding into the positive feedback loop of climate change (Tarnocai, 2006; Thompson et al., 2017b)

Within the Canadian Boreal Plain ecozone (BPE), peatlands exist at the dry-limit of peatland formation, in sub-humid climates where potential evapotranspiration is equal to or slightly exceeds precipitation over long-periods of time (Petrone et al., 2007; Yu, 2012). They are formed on heterogenous glacial landforms and are the dominant wetland type in the BPE (Ireson et al., 2015). Peatland water-balance budget is maintained by precipitation, surface water inputs, and sometimes groundwater sources (Benscoter & Vitt, 2008; Devito et al., 2012). As such, peatlands of the BPE, especially those in central Alberta can be sensitive to climatic changes. Based on climate models, the BPE is predicted to become increasingly drier, opening peatlands up to increasing disturbance and shifts in vegetation composition (Gorham, 1991; Ireson et al., 2015; Thompson et al., 2017b).

Fire is the dominant landscape disturbance in the BPE (Turetsky et al., 2004; Wieder et al., 2009). Driving landscape formation and playing a critical role in ecosystem succession (Stocks, 1993; Harden et al., 2000; Turetsky et al., 2002), fire is essential for maintaining healthy ecosystems. However, climate-induced alterations to weather conditions, such as warming air temperatures (Chapin et al., 2005) and increasing drought (Petrone et al., 2005), can exacerbate fire conditions. It is widely acknowledged that fire activity within the BPE has been increasing in recent decades (Flannigan et al., 1998; de Groot et al., 2013; Whitman et al., 2019). Further, climate-driven increases in fire frequency, extent, and severity are expected to have widespread impacts on peatlands as well as transitional and forested ecosystems of the BPE (Flannigan et al., 2009).

Fire behavior is strongly influenced by 1. Climate and weather conditions, such as prolonged warm conditions (Bradshaw, 1984), dry cycles (Kettridge et al., 2014; Devito et al., 2016), wind speed and direction, all of which can impact the spread and severity of fire (Rowe & Scotter, 1973; Rowe et al., 2017; Thompson et al., 2017b). 2. Fuels, in composition and abundance, where the available fuel load can become a limiting factor to fire spread (Flannigan et al., 2013). Peat is an ideal form of fire fuel. Because peatland soils are composed of a dense, often deep layer of organic material, a smoldering peat fire can persist from days to months, and even years, propagating by spreading downward and latterly through the peat column (Frandsen, 1991, 1997; Johnston et al., 2015). Peatlands typically maintain resilience to fire due to their relatively high moisture contents, and high-water tables (Johnston et al., 2015; Turetsky et al., 2015; Waddington et al., 2015). Resilience to fire can be over-come in instances of severe-fire or peatland drying, from climate-mediated aridity or anthropogenic disturbance (Turetsky et al., 2002; Kettridge et al., 2015). Additionally, fuel-compositions are directly influenced by previous fire-conditions. For example, serotinous seeds, such as pine, require fire to open (Schneider et al., 2009; Dalponte et al., 2019). 3. The spatial variability of fuel and topographic positioning of the landscape can be indicative of local topographic moisture characteristics, which can influence organic layers and nearness to the water table (Carlson et al., 2011; Hokanson et al., 2016; Nelson et al., 2021).

Boreal peatlands are characterized by deep organic layers and sub-humid climates that are sensitive to climate-mediated changes, such as increasing atmospheric aridity. Under conditions of increasing air temperatures and aridity, peatland drying can enhance the susceptibility of peatlands to increasing fire frequency. In accordance with these predictions, instances of fires returning at decreased return intervals have been observed within the boreal region (Kasischke & Turetsky, 2006; Flannigan et al., 2009; Brown & Johnstone, 2012; Whitman et al., 2019). Here fire return intervals indicate the number of years between fires. Therefore, as fire return intervals decrease, ecosystems burn more frequently. Brown and Johnstone (2012) demonstrated that increased fire frequency has led to a reduction in the viable seed availability, number of total seedlings, and had increased the mortality of young seedlings in black spruce stands in forested sites in northern Yukon, Canada. Whitman et al. (2019) found that short-return interval fires within boreal forested uplands and wetlands resulted in reduction of conifer density. They also found post-fire increases in broadleaf stem density, and the reduction of organic soil depth and content (percentage of soil composition that is organic) compared to paired long-return interval fire sites. While these are excellent examples of the impacts of increased fire frequency/decreased return intervals on boreal ecosystems, the impacts of diminishing fire return intervals on peatlands remain largely unknown.

Within the last decade, research has been focused on the impacts of double disturbances on fire severity, e.g., via peatland draining and harvest (Sherwood et al., 2013), however, to our knowledge, there are few to no studies that have examined the impacts of short return fire intervals on Boreal peatlands in Canada. This thesis aims to address this gap. As climate-mediated air temperatures continue to increase in northern/central Boreal ecosystems, it is critical to understand how peatlands are changing in response to increases in fire frequency. This is important because they are a critical component of the global carbon-climate feedback system due to the broad area they cover, ability for carbon sequestration and local climate mediation of the climate by cooling due to ET loses (Helbig et al., 2017). Kettridge et al. (2015), reviewed in Nelson et al. (2021), have predicted that shortening of the return interval within peatlands is likely to impact the soil characteristics and regenerating vegetation composition. This corroborates work by Johnstone et al. (2016), who hypothesized reduction of tree recruitment from increased fire frequency, thus predicting changes to vegetation composition and black spruce distribution not specific to peatlands. Here-in we aim to explore the impacts of increased fire frequency on peatlands, to address these gaps in the literature.

The Utikuma Region Study Area (URSA), located in the Boreal Plains Ecozone approximately 120 km north of Slave Lake Alberta, Canada, is a mosaic of forested uplands and wetlands, including a large proportion of peatlands. During the summers of 1956 and 2011 large portions of this study area burned (Devito et al., 2016; Alberta Wildfire, 2018). The burn scars of these areas overlapped in a 12 km^2 area, providing a unique opportunity to study impacts of a reduced fire return interval on multiple peatlands within the study area. The 55-year return interval is considered a short to medium length return interval, however, it is important to note that typical fire return intervals for peatlands are estimated to be approximately 100-120 years or more (Turetsky et al., 2004; Wieder et al., 2009). Therefore, peatlands in this overlapping area were burned within less than half of their expected time frame.

In this thesis, we compare the post fire species and structural characteristics of regenerating vegetation within a fire that burned peatlands in 1956; a more recent fire in 2011; and the area where the two fires overlapped (1956 and 2011), using a combination of field measurements and time series airborne lidar data. This project takes advantage of the increasing availability of remote-sensing data and lidar, in conjunction with field studies, to address the research gap of the impact of reducing fire return intervals on peatland regeneration within the boreal plains.

1.2 Key Definitions

1.2.1. Peatlands and Transition Zones

Peatlands are one of the two broad groups of wetlands defined in the Alberta Wetland Classification System and encompass bog and fen wetland 'classes' (Government of Alberta, 2015). Like all wetlands, they are characterized by 1) the presence of shallow-water tables, as surface water or water within the root zones, 2) unique soil characteristics, in this case the accumulation of deep, partially decomposed organic matter, and 3) the presence of hydrophytic vegetation, often bryophytes such as *Sphagnum* moss species, which make up a large component of the decomposing organic material, or vascular hydrophilic plants such as black spruce (Wieder et al., 2006; Mitsch & G Gosselink, 2015). Peatland ecosystems are most simply defined as terrestrial ecosystems where the net rate of production exceeds that of decomposition (Wieder et al., 2006), allowing peatlands to accumulate deep deposits of decomposing organic material over long periods of time. The presence and depth of 40cm or more of peat, a term for relatively undecomposed or partially decomposed organic soil, is the primary diagnostic tool for differentiating peatlands from mineral wetlands, which are also common in the boreal region (Government of Alberta, 2015; Mitsch & G Gosselink, 2015).

The two primary classes of peatlands, bogs, and fens, include more than 90% of the peatlands found in Canada (Tarnocai et al., 2005), and are primarily differentiated by their water inputs. Bogs are ombrotrophic, with acidic, nutrient poor soil conditions, which tend to limit vegetation species communities that establish within them, making them relatively homogenous (Vitt, 1990; Benscoter & Vitt, 2008; Government of Alberta, 2015). As their primary water input is precipitation, they typically maintain drier conditions than fen peatland ecosystems (Price & Maloney, 1994), which is one of the reasons why bogs are often more susceptible to wildland fire (Bourgeau-Chavez et al., 2020). Fens are characterized as minerotrophic, or groundwater fed, making their soils generally more saturated and with higher water tables than bogs. Fens also have a greater range of pH and are typically more alkaline than bogs (Vitt & Chee, 1990; Vitt, 2006). As such, fens have an overall greater degree of ecosystem heterogeneity and diversity than bogs, especially in terms of species richness of bryophytes and herbs (Warner & Asada, 2006). However, these two classes often overlap in vegetation and species composition, and can resemble each other, making the two difficult to distinguish.

Peatlands in the boreal are often surrounded by forested uplands which have vastly different soil characteristics and composition, different local hydrology, and differences in vegetation species composition, distribution, and structure, compared with peatlands. The area between peatlands and uplands or the adjacent forested area is termed "transition zones" (Hartshorn et al., 2003). These transition zones often have soil profiles, water tables, and vegetation, that exist within the range of the characteristics of both peatlands and upland forested areas (Yarrow & Marín, 2007; Dimitrov et al., 2014; Lukenbach et al., 2015). This makes it difficult to distinguish where the transition zone ends, and the peatlands begin. Typically, they are loosely defined by their vegetation, soil, and/or topographic characteristics *in situ* (Bhatti et al., 2006; Dimitrov et al., 2014).

1.3. Single Burns and Double Burns

In 1956 and 2011, URSA underwent two major fires, overlapping in a 12 km^2 area (Figure 1.1). Throughout this thesis, the overlapping area, is referred to as "Double Burn" or DB, which indicates that this area had the overlapping fire scars of two separate burns determined from the Alberta Spatial Wildfire database historical records (Alberta

Wildfire, 2018). As the peatlands studied within the DB areas had a return-burn within 55 years, less than half of the expected 100-120 years that is typically found in peatlands (Turetsky et al., 2004; Wieder et al., 2009), they are considered short-interval burns. Thus, double burns and short-return interval burns are here-in used synonymously.

The areas of the 2011 and 1956 wildfires that did not overlap, are individually referred to as “Single Burn” or SB, meaning that these areas did not have overlapping burn scars with any other fires in the historical records from the Government of Alberta database, which has fire records dating back to 1931 (Alberta Wildfire, 2018). The use of the symbology “SB11” or “2011SB” are in reference to areas which had burned in the 2011 Utikuma complex wildfires and had no overlap with other burn scars in the spatial data base. This is used interchangeably throughout with the term “long-return interval” burns or fires. Areas burnt in 1956 with no overlapping previous or post-fire scars in the spatial database are referred to as “SB56” and “1956SB”. Examples of peatlands measured in each of these areas are illustrated in Figure 1.1.

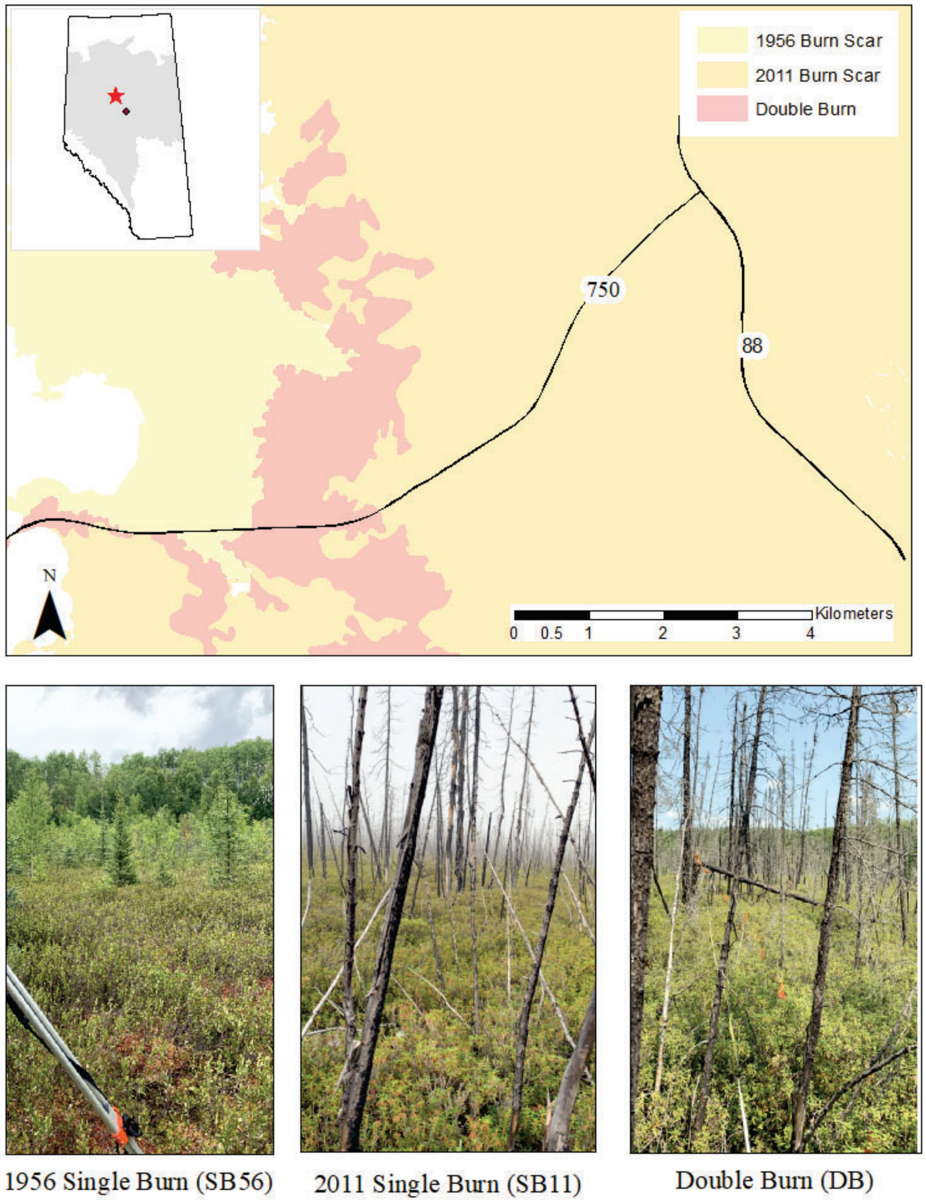

Figure 1.1. Map of burn areas, including Single Burn 1956, Single Burns 2011, and the overlapping DB area, polygon boundaries were obtained from the Spatial Wildfire Database (Alberta Wildfire, 2018). Alberta highways 750 and 88 run through the study area and are denoted on this map. Inset uses a red star to identify the location of URSA within the BPE in reference to Slave Lake (black dot) in Alberta, Canada. The photos were taken during Field collection in July – August 2019, and are examples of the collection sites in each of the three areas, SB56, SB11, and DB.

1.4 Study Objectives

The over-arching aim of this thesis is to identify vegetation structural changes in post-fire successional landscapes and to quantify the spatial variations of regeneration in transitional zones and peatlands. Broadly, the goal is to understand how shortening fire return intervals will alter the regeneration of vegetation within peatlands in the years post-fire. In addition, multi-temporal lidar data are used to determine whether or not field observations also occur across the broader region of mixed peatlands and transitional land cover types.

The thesis is divided into two main objectives:

1) Using field data collected in 19 peatlands, compare soil characteristics and the structural attributes of vegetation within adjacent double burn and single burn peatland and transition zones to:

> a) Assess if differences exist between soil characteristics, such as pH, bulk density, and depth of organic soil within peatlands (bogs and fens) and transition zones, and across different burn regimes (2011 single burn, SB) and 1956 + 2011 double burn (DB).
>
> b) Determine if differences exist between the density and variability of regenerating vegetation species and height within SB and DB peatlands.
>
> c) Determine if these differences exist between SB and DB peatlands on a broader landscape, representing the trends seen *in situ* at field sites using airborne lidar data acquired coincident with field data.

2) Quantify the rate of early post-wildfire regeneration of vegetation within peatlands and transition zones in SB and DB burn scars across the broader region using multiple lidar datasets collected since the Slave Lake Complex Wildfire in 2011 to:

a) Determine the rate of growth, as measured by lidar-based height metrics, within SB and DB peatlands, divided into class and form.

b) Compare the differences between SB and DB vegetation regeneration, specifically vegetation height, to determine if regrowth is occurring at different rates between fire regimes.

c) Based on variable rates of regeneration, examine how fragmentation in the landscape (represented by the ratio of the perimeter of individual peatlands to their area) might influence rates of vegetation growth.

1.5 Thesis Organisation

This thesis is divided into five chapters. Chapter 2 provides the contextual background for boreal peatlands, regeneration of vegetation following wildfire, and the use of multi-temporal lidar for change detection of vegetation regeneration and rates of growth/change/mortality, relevant to all subsequent chapters. The two chapters following are prepared as standalone manuscripts. Chapter 3 addresses objective 1 and presents a research study examining the impacts of reduced fire-return intervals on peatland and transition zone soil characteristics, vegetation species composition and characteristic changes using *in situ* field data and remote sensing to determine patterns across the broader landscape. It also briefly examines vegetation height characteristics in response to peatland fragmentation. Chapter 4 presents an analysis of the rate of regrowth post wildfire using multi-temporal lidar remote sensing over the broader region, where multiple lidar datasets exist following the Utikuma Complex Wildfires, to determine if the rate of regeneration in peatlands is impacted by shortened wildfire return intervals.

Chapter 5 concludes the thesis by presenting a summary of the findings and limitations of research. It also provides potential directions for future research.

2. BACKGROUND: INFLUENCE OF FIRE IN BOREAL PEATLANDS AND REMOTE SENSING APPLICATIONS

2.1. INTRODUCTION

Air temperatures are predicted to increase globally over the next century, and with these, fire activity (Flannigan et al., 2013). Fire, under normal circumstances, plays an important role in landscape formation, ecosystem maintenance, and successional processes (Heinselman, 1981; Turetsky et al., 2002). However, if fire becomes too frequent or damaging to an ecosystem, it can negatively impact the ecosystems' health and ability to re-establish pre-fire conditions (Heinselman, 1981; McCullough et al., 1998; Brown & Johnstone, 2012).

Boreal peatlands, especially those found at URSA and across the western part of Boreal Plains ecozone, are formed in sub-humid climates, where potential evapotranspiration equals or exceeds precipitation, making them more vulnerable to shifts in climate (Brown et al., 2010). Evapotranspiration is driven by a vapor pressure deficit, which in warming climates, is expected to increase (Helbig et al., 2020). As evapotranspiration is often the primary method of water loss within the peatlands (Devito et al., 2005; Petrone et al., 2007; Brown et al., 2010), this is expected to continue to increase over the next century associated with climate-mediated warming (Helbig et al., 2020). Therefore, peatlands will become increasingly susceptible to drying, making them more vulnerable to wildland fire. Peatlands provide essential services including animal habitat, water and climate regulation, and carbon (C) sequestration (Turetsky & Louis, 2006). Peatlands store disproportionately large amounts of C in comparison to their land masses (Tarnocai, 2006). However, they also have the capacity to release large amounts of carbon into the atmosphere, more than most other ecosystems, due to changes in moisture regimes, which enhances microbial decomposition of organic matter, and also fire, which can burn deep into peatlands (Tarnocai et al., 2005; Tarnocai, 2006; Thompson et al., 2017b).

Boreal peatlands have been largely resilient to fire, however, with these recent warming and drying trends, peatlands have been burning more severely and more frequently. Burn severity is defined as the effects from fire on the environment, or how the environment's soil, hydrology, and vegetation is impacted by fire, including variables such as the biomass lost from an ecosystem (Keeley, 2009; Whitman et al., 2018). Fire frequency is defined as the number of times that fires burn an area over a given period, whereas the return interval is defined as the number of years between fires. As increasing fire frequency in a peatland setting could 1. Release large amounts of carbon dioxide into the atmosphere, and 2. Shift the balance of vegetation species and possibly, the successional trajectories of these peatlands. Therefore, it is important to understand peatland recovery in scenarios of shortened-return fire return intervals.

The following chapter provides a brief background on 1. The boreal plains ecozone and how it is changing; 2. How fires impact peatlands and peatland recovery post-fire; 3. The current state of the literature on fire return intervals within the context of boreal peatlands and similar ecosystems; and 4. How remote sensing can be used to observe post-fire recovery and changes to ecosystems.

2.2. THE BOREAL PLAINS ECOZONE & HOW IT IS CHANGING

2.2.1. The changing climate and hydrological conditions

The boreal plains (BP) ecozone extends from northern Alberta to southern Manitoba and is bordered on the north by the Taiga Plains and Taiga Shield, and on the east by the boreal shield (Ireson et al., 2015). Large portions of the ecozone are post-glacial, with a surface geology largely made up of heterogenous glacial deposits (Ireson et al., 2015); the three dominant landscapes being a coarse-textured outwash, a fine-textured hummocky moraine, and lacustrine clay plains (Thompson et al., 2017b). The heterogeneity of the surface geology influences the hydrological characteristics of the region (Devito et al., 2016), and the deep glacial sediments allow for a large soil storage capacity, which influences the water cycling within the region (Redding & Devito, 2008).

Since 1948, the annual temperature across the Boreal Plain ecozone has risen by 1-3^{O} C – more than double the global average (Hengeveld et al., 2005; Lemmen et al., 2008). Increasing aridity and prolonged dry periods in parts of this region have resulted in widespread decline in forest health, increased forest mortality, and unknown rates of wetland loss through shrub encroachment and succession (Beck & Goetz, 2011; Flannigan et al., 2013; Chasmer et al., 2017). For example, upland forests within prairie parkland areas were found to be sensitive to increased evapotranspiration (ET) losses, which resulted in widespread drought-induced aspen mortality over a period of four years (Michaelian et al., 2011). In the boreal region, aridity-based changes in plant physiology could increase susceptibility to disease, fire, and pests (Thompson. et al., 2017). Dry conditions can also reduce water levels in small lakes and ponds, thereby increasing the potential for terrestrialization, or the accretion and deposition of clastic and/or organic material in wetlands limiting water storage capacity such that it leads to the colonization of aquaphobic species, and forest encroachment within the wetland areas (Waddington et al., 2015; Thompson. et al., 2017). Such broad area changes could lead to boreal biome shifts, including predicted northward movement of grassland and parkland areas bordering the southern boreal region, which modelling estimates have predicted could displace boreal vegetation by 2050 (Schneider et al., 2009).

2.2.2. How fire activity is changing withing the boreal plains

Fire is an important ecological process within the Boreal Plains as it maintains natural stand structures and successional patterns (Finney et al., 2011). Fire regimes are generally characterized by frequency, intensity, and severity of fire, as well as burn season, fire type, and fire size (Weber & Flannigan, 1997; Gill & Allan, 2009; de Groot et al., 2013). Climate change scenarios and fire activity modeling have predicted both fire behavior and extent of area burned will likely increase within this century (de Groot et al., 2013). Predictive climate models used in Flannigan et al. (2013) also suggest that fire season in the Boreal Plains is likely to increase by up to 30 days by 2100. Changing wildfire regimes are a cause for concern as post-fire regeneration can result in large species shifts, potentially causing large area landcover changes and/or total biome shifts (de Groot et al., 2013).

Increasing fire activity within the boreal plains could result in a preference for deciduous tree species regeneration (de Groot et al., 2013), similar to what was seen in Whitman et al. (2019). Whitman et al. (2019) observed that forested areas undergoing increased fire frequency intervals had an increased presence of deciduous vegetation, with a simultaneous decline in conifer stem density. Whitman et al. (2019) postulated that this shift may have been the result of changes to the soil substrate drivers (temperature and degree of combustion) in short interval return sites. This may have resulted in a favoring of resprouting and nitrogen-fixing species, such as aspen and alder species (upland and transitional species), over seed propagating species (such as black spruce). Shifting vegetation could also have implications for future fire types by potentially reducing crown fires, as deciduous species are less liable to burn due to typically wider spacing between trees, higher bark and leaf moisture content, and a high crown height (Alberta, 2012). It is possible that a shift to increased deciduous tree composition and distribution could result in a future ecosystem that undergoes less severe fires and is overall more resilient to fire. Wang (2002), for example, found stands with an increased density of coniferous trees fell within the most severely burned class, where high deciduous components often were less severely burned. This may hold true for peatlands as well, as Kettridge et al. (2015) predicted that with increased fire frequency, the carbon stores of peatlands may be reduced such that the mineral soil underlying the ecosystems may become exposed over time, providing an opportunity for shifts in the composition of the seed banks and rapid broad leaf growth. While the results of increased fire activity have been studied in upland forested areas, relatively little is known about how peatlands may respond to these changes.

2.3. PEATLANDS: DRY CONDITIONS, BURNING, AND RECOVERY

Peatlands are a dominant wetland type within the Boreal biome (Petrone et al., 2007; Government of Alberta, 2015). They store approximately 56% of Canada's soil C which accumulates over long periods of time by continual C uptake and low decomposition rates (Turetsky et al., 2002). Under normal conditions peatland decomposition releases small amounts of soil C and dissolved organic carbon (DOC) in the form of carbon dioxide gas (CO_2), but the release is offset by rates of carbon storage from production, such that peatlands act as a gradual net carbon sink (Wieder et al., 2009). Decomposition is highly sensitive to changes in water balance, thus warming and drying would increase decomposition rates, thereby increasing the amounts of C emission to the atmosphere (Kettridge et al., 2015). Further, widespread drying of peatlands under climatic warming conditions will also enhance fuels within peatlands, especially associated with the drying of accumulated carbon within deep organic peat soil (Tarnocai, 2006; Petrone et al., 2007), thereby further reducing peatland function as a significant, broad area carbon sink.

2.3.1. Peat Water-Retention and Drying

The mosses that make up the surface of peatlands play a key role in modulating surface moisture and water storage, as well as maintaining their resilience to wildfire. *Sphagnum* mosses are the dominant genus in peatlands, making up to 70% of peatland surface vegetation (Benscoter & Vitt, 2008). As peatlands are subject to the climatic seasonal and decadal wet- and dry-cycles that occur across the boreal plains (Devito et al., 2012), mosses must be adaptable to a certain degree of changing water availability. *Sphagnum* mosses have dead hyaline cells that act as water storage reservoirs giving them high water holding capacities and providing them with some drought tolerance

(Dickinson & Maggs, 1974; Bu et al., 2013), while simultaneously reducing water loss via evapotranspiration (Kettridge et al., 2014). These mosses, along with variations in microtopography within peatlands, topographic positioning, and surficial geology, contribute to the maintenance of high moisture contents of peat and the high-water tables that are characteristic of boreal peatlands (Devito et al., 2012; Kettridge et al., 2014). The combination of these provides enhanced protection from fire (Thompson & Waddington, 2013). In addition to mosses, many peatlands are dominated by coniferous trees, such as black spruce trees (*Picea mariana*) which have low stomatal conductance. This helps to maintain low rates of evapotranspiration (Dang et al., 1997), further reducing water loss in peatlands.

Prolonged periods of drying can increase evapotranspiration such that the moisture conditions of peatlands become reduced over time (including water storage in organic peat), causing enhanced drying of surface soils and organic matter, increasing fuel availability and vulnerability to fire (Turetsky et al., 2002; Lukenbach et al., 2016). Miller et al. (2015) demonstrated that increases in air temperature, such as those predicted by global climate models (GCMs) used in Flannigan et al. (2013), are likely to promote further drying in peatlands such as fens, and bogs (Miller et al., 2015). Further, multi-decadal atmospheric drying conditions are likely to promote shifts in vegetation composition, including increased tree and shrub biomass and the movement of these ecosystems towards a more terrestrial landscape, particularly in shrub-dominated peatlands. In addition, shifts from *Sphagnum* moss composition to feather mosses (Miller et al., 2015) may also occur (Weltzin et al., 2003; Beck & Goetz, 2011). Feather mosses do not have the same capacity for water retention or for pulling water up to the capillary fringe as *Sphagnum* mosses do (Carleton & Dunham, 2003). Thus, a shift from *Sphagnum* dominated to feather moss dominated ecosystems creates a drier upper surface layer in the peatland (Lukenbach et al., 2015), increasing its capacity as a fuel source.

2.3.2. Peatland Burning

Fire is the most prevalent disturbance type on peatlands within the boreal region (Turetsky et al., 2004; Turetsky & Louis, 2006). The severity of peat fires largely depends on the moisture conditions and the fuel load (Turetsky & Louis, 2006). While peatlands can experience limited damage and largely recover from light surface fires, severe, deep fires can burn through protective moss layers (Kettridge et al., 2019). Peat can also by consumed by smouldering fires, which move latterly and downwards through the peat column, and can last for up to months to a year or more in severe cases, thereby releasing large amounts of carbon to the atmosphere (Benscoter & Wieder, 2003; Benscoter et al., 2011).

In addition to moisture conditions, peat bulk density, defined as the density of organic material to air (or water) spaces within a given volume, is an important controlling factor in both the depth of burn and the intensity of combustion in peatlands (Boby et al., 2010; Benscoter et al., 2011; Hokanson et al., 2016; Lukenbach et al., 2017). Bulk density is a useful indicator of organic soil conditions, soil moisture, and water retention capacity (Hokanson et al., 2016). Areas with higher bulk density result in greater depth of burn, whereas areas with lower bulk density tend to experience lower burn depths (Hokanson et al., 2016). Hokanson et al. (2016) and Lukenbach et al. (2017) illustrate variable burning in the margins versus centers of peatlands in response to variations in peat bulk density. Bulk density variations can also be observed in the humocks and hollows of peatland terrain surfaces, which also have spatial implications for variable burn severity and species regeneration (Benscoter et al., 2015). From these observations, spatial variations in the severity of burns is an important indicator of post-fire regeneration species composition (Benscoter et al., 2005; Johnston et al., 2015).

2.3.3. Peatland Recovery

Pre-fire conditions, burn severity, and time since fire, are examples of key influences on post-fire peatland recovery. Pre-fire conditions encompass many variables, such as: 1. Vegetation distribution prior to fire, and in turn the available seed-bank for re-establishment (Benscoter & Vitt, 2008; Brown & Johnstone, 2012; Whitman et al., 2019); 2. Moisture conditions, such as soil moisture, which can determine the extent of burn (Rein et al., 2008); and 3. Presence of prior disturbances, such as harvesting or peatland draining, which can impact post-fire water availability and soil characteristics (Sherwood et al., 2013). The intensity at which the peatland was burned (defined as the energy that is released during the fire as well as the fire behaviour; Keeley, 2009) can be used to determine post-fire conditions such as the abundance of surviving peat, post-fire water loss, the amount of water lost from evapotranspiration (Lukenbach et al., 2016; Kettridge et al., 2019), and the remaining post-fire microtopography of the peatland (Lukenbach et al., 2016).

Time since fire allows the peatland to recover, however, this may be interrupted by other disturbances, thereby altering peatland condition and ecosystem service. Benscoter and Vitt (2008) used a chronosequence method of observation to approximate the trajectory of bog succession and recovery post-fire. They found that some species of *Spagnum* mosses begin to develop approximately 3-4 years after fire, but continue to diversify for another 80 years or more (Benscoter & Vitt, 2008). Black spruce (*Picea mariana*), which are often a dominant tree species in boreal peatlands, increase in stem density for up to approximately 80-90 years post-fire (Sirois & Payette, 1989; Wieder et al., 2009). Post-fire peatland recovery begins to reach a steady-state approximately 100-125 years post-fire, with successional vegetation community change becoming less variable over time (Odum, 1969; Benscoter & Vitt, 2008). Increasing temperatures are expected to result in increasing fire frequency (Flannigan et al., 2013), which could impact the ability of ecosystems to recover pre-fire levels of productivity and succession.

2.4. DIMINISHING FIRE RETURN INTERVALS

Incidents of increasing fire frequency within Canadian ecosystems have already been observed (Kasischke & Turetsky, 2006; Brown & Johnstone, 2012; Whitman et al., 2019), supporting numerous predictions of increased fire frequency due to climate change over the next century (Flannigan et al., 2003; Flannigan et al., 2006). The age distribution of species and observed/typical fire frequency for ecosystems within the boreal region is highly variable in space and through time. Therefore, the time frame that is considered a “long” or “short” return interval between fires, is largely dependent on the ecosystems’ fire history. Peatlands for example, typically have a fire return interval of approximately every 100-120 years (Turetsky et al., 2004). An ecosystem that experiences double burn, for example, the study area of the thesis, described in the first chapter, being 55 years between fires, would be considered relatively short. In Whitman et al. (2019), who examined the influence of short-interval fire returns on primarily forested upland areas, classified short interval returns as less than or equal to 17 years post fire, and long-interval returns greater than 30 years, as the typical return interval for fires within boreal forests is approximately 30 years. Brown and Johnstone (2012), in comparison, defined short return interval fire in black spruce stands located in the Yukon at approximately 14-15 years between fires (depending on the site).

With regards to forest structure, which can increase or reduce the fire return interval, Whitman et al. (2019) found that conifer stem density, along with total stem density, was significantly lower in short interval fires occurring in upland and wetland areas, in comparison to long-interval fire sites (Whitman et al., 2019). Additionally, they found that the overall proportion of conifer stems was significantly lower in short-interval return sites than in long-interval return areas. This indicates a greater prevalence of deciduous trees and shrubs in the period following fire, such that fewer conifers are established before the next fire. Brown and Johnstone (2012), in comparison, found greater mortality of black spruce seedlings in short-return interval fire sites than in longer-return interval fire sites. Whitman et al. (2019) also found a greater proportion of broadleaf trees and less organic material in the soils in short-interval sites.

Other than Whitman et al. (2019), there is little research on the impacts of shortening return intervals on wetlands, and none, to our knowledge, that specifically address boreal peatlands. However, leaning on the examples of peatlands that underwent multiple disturbances, can provide some insight into how peatlands may respond. Sherwood et al. (2013) demonstrated the response of peatlands to draining and wildfire by examining changes in hydro-physical properties. Their research indicates that combination of two separate disturbance events on a single peatland area can significantly alter peat hydro-physical properties beyond that caused by an individual disturbance (Sherwood et al., 2013). Ecosystem changes following combined disturbance (of draining and fire) included lower specific yields, indicating a deeper water table position that is more susceptible to variable hydrologic conditions. Peat soils were more likely to have rooting systems grown in over time since disturbance (Sherwood et al., 2013). Wildfire severity within their study was minimal to moderate, and the double disturbance mechanism was representative of first a wildfire, followed by human disturbance (Sherwood et al., 2013).

Based on the combined disturbance of draining and fire, it may by hypothesized that a double burn may have similar outcomes on ecosystems, however, this is an area which needs further research. The evidence from Whitman et al. (2019) and Brown and Johnstone (2012) generally support the predictions of Kettridge et al. (2015) and conclusions from Nelson et al. (2021) that increased repeat fire cycles are likely to impact soil characteristics and regenerating vegetation composition. Though the evidence provided was not specifically addressed for peatlands as Kettridge (2015a) and Nelson et al. (2021) predict, many vegetation species examined in upland forests were also common in peatlands and could potentially behave similarly. Considering the importance of peatlands' role in carbon sequestration, and their ability to release large amounts of C into the atmosphere during fire events, due to climatic-mediated warming (Vitt, 1990; Wieder et al., 2006; Komers & Stanojevic, 2013), understanding how peatlands recover and the rate of regeneration during periods of increased fire frequency is of globally relevant importance, considering the area of peatland coverage and the amount of C stored. Despite this, understanding of peatland C and fire influences tends to be limited to highly localized studies. In the next section, we discuss the use of remote sensing for determining post-fire changes in peatlands.

2.5. REMOTE SENSING APPLICATIONS IN CHANGE DETECTION

Boreal peatlands are often situated in remote and difficult to access locations. Changes in these ecosystems, such as the regeneration of peatland vegetation post-fire, can vary both spatially and temporally. However, changes are often slow and difficult to characterize over short periods of time (e.g., 5 years) using field methods (Næsset et al., 2013). This makes remote sensing an excellent method for examining change over a broader area and sometimes over longer time periods than what could be feasibly done using traditional *in situ* field methods (Gomez et al., 2016). While field sampling produces invaluable information and insight into the processes and conditions associated with ecosystem change, it can be logistically, fiscally, and technically challenging (Mahdavi et al., 2018). Remote sensing methodologies can also provide a greater number of data acquisitions at time intervals that could otherwise prove prohibitive using field methods (Mahdavi et al., 2018). Collecting data remotely has also become increasingly feasible through time as computing powers and data availability increase, while costs decrease (Yu et al., 2004; Zhao et al., 2018). A uniting of both field and remote sensing observations is an ideal methodology, as this combines the insights from field data collection with the power of spatial and temporal remote sensing data.

2.5.1. Remote Sensing and Change Detection

Change detection encompasses a broad category of research objectives, such as: estimating canopy growth (Yu et al., 2006), detecting increases or decreases in above ground carbon (Næsset et al., 2013; Zhao et al., 2018), monitoring changes in wetland land classifications (Chasmer et al., 2020), or classifying forest change trajectories (Noordermeer et al., 2019). As change most simply defined is a difference over time, this necessitates a multi-temporal approach. As remote sensing products, such as satellite imagery, RADAR, and lidar, become increasingly available, so do the opportunities for change detection (Yu et al., 2004; Gomez et al., 2016). After Landsat 8 satellite imagery became freely and publicly available in 2008, it became the standard product for land cover change detection (Wulder et al., 2012; Gomez et al., 2016). The availability of temporally and spatially repeating satellite imagery allowed for time series change detection by looking at spectral variation over multiple time periods, on a much larger scale than could be done with aerial photography (Kennedy et al., 2014; Gómez et al., 2015; Gomez et al., 2016).

The applications of optical remote sensing time-series analysis extend beyond an individual field site and can extend to a region, or even nationwide scale. For example, White et al. (2017) demonstrated the use of optical remote sensing for determining recovery intervals in vegetation within the Boreal Plains ecozone post-fire and post-harvesting. Using a comparison of Normalized Difference Vegetation Indices (NDVI), they found that vegetation recovers more slowly from wildfire (8-9 years) than it does from forest harvest (6-7 years), largely due to the variability of severity and magnitude of fire in comparison to forest harvest (White et al., 2017). Multi-temporal change detection using optical imagery has proven to be sufficient at detecting landcover change (White et al., 2011), above-ground carbon change (Gómez et al., 2012), and disturbance detection (Vogelmann et al., 2009; Schroeder et al., 2011; Gómez et al., 2015). There are, however, limitations to optical remote sensing. Users of Landsat imagery must contend with persistent cloud and/or smoke cover, especially in applications such as identifying fire burn scars or calculating burn severity indices (Gomez et al., 2016). Additionally, some high radiometric and spatial resolution optical data, such as hyper- and multi-spectral optical imagery, do not contain 3-dimensional information of the structural characterization of a forest (Marinelli et al., 2018). Lidar Detection and Ranging, or lidar, can be used to obtain 3-dimensional structural measurements, and with increasingly available datasets and decreasing costs, is well suited for applications in forest change detection (Marinelli et al., 2018; Noordermeer et al., 2018; Noordermeer et al., 2019).

2.5.2. Lidar Time Series Change Detection

Lidar systems, most basically explained, are laser systems that calculate 3D coordinates of a target object by emitting laser pulses towards the Earth. These pulses are reflected by objects on the surface and are used to measure ranges between the system and the target (Akay et al., 2009). The system uses the time it takes the reflected light to travel back (the return time) to the instrument to calculate the distance to the target (Lim et al., 2003; Akay et al., 2009). Some of the earliest instances of use in a research context, have been for forestry applications (Means et al., 2000; Akay et al., 2009), measuring canopy height. With increased lidar availability, as well as development of new systems with greater capacities, these applications have expanded, as have the opportunities for 3-dimensional change analysis (Cao et al., 2016; Noordermeer et al., 2019). Some examples of time-series lidar change detection that have already been demonstrated include: 1) changes in above ground biomass (AGB) (Meyer et al., 2013; Andersen et al., 2014; Srinivasan et al., 2014; Hopkinson et al., 2016b), 2) changes to canopy gaps and closure (St-Onge & Vepakomma, 2004; Marinelli et al., 2018), and 3) detecting changes in vertical, lateral, and volume growth (Yu et al., 2006; Vepakomma et al., 2011; Marinelli et al., 2018).

Most lidar change analysis done to date have been using bitemporal (or two) data sets (Yu et al., 2004; Yu et al., 2008). Yu et al. (2004) first demonstrated the use of bitemporal lidar data sets in estimated tree growth. They produced digital surface models (DSM), and then differenced canopy height models (CHM)s to detect vertical growth in forested stand in Finland (Yu et al., 2004). While this differencing method did underestimate tree-growth, the error remained lower than field-based tachometer measurements (0.5 – 1m) (Yu et al., 2006). With the development of more sophisticated software and analysis, some algorithms have managed to increase accuracy in individual tree detection to 97%, allowing for detection of single tree 3-dimensional changes (Marinelli et al., 2018). Solberg et al. (2006) used three laser scans obtained in summer 2005 and related derived structural characteristics to field-based measurements of leaf area index (LAI) to estimate LAI using lidar and detect foliage health which is useful for long-term monitoring. Hopkinson et al. (2008) demonstrated the use of four lidar acquisitions (2000, 2002, 2004, and 2005) in estimating rates of growth at annual time-steps in conifer plantations within southern Ontario. Zhao et al. (2018) also considered four repeat lidar surveys (2002, 2006, 2008, and 2012) to estimate forest growth trajectories at the individual tree and the grid level. Multi-temporal lidar can effectively characterize changes in vegetation distribution, height, biomass, volume, canopy cover, and basal area (Tompalski et al., 2018). While the cost of lidar still remains relatively high, especially in comparison to freely available Landsat options, increasing availability makes 3-dimensional change detection across broad landscapes more feasible.

3.0 HYDRO-ECOLOGICAL IMPACTS OF SHORTENING FIRE RETURN INTERVALS ON REGENERATING BOREAL PEATLANDS AND TRANSITION ZONES USING INTEGRATED IN SITU SAMPLING AND LIDAR APPROACHES

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Abstract

Aridity associated with rising air temperatures in northern latitudes are expected to contribute to increased frequency of wildland fires. Here we test several hypotheses on vegetation regenerating following short-(double burn, DB) return interval fire (56 years post-fire) compared to long- (single burn, SB11) return interval fire (>80 years post-fire) in boreal peatlands and their adjacent transitional areas. The objectives of this study were to quantify if differences exist between 1. peatland and transitional soil characteristics in SB vs. DB areas; 2. regenerating vegetation species, structural characteristics, and diversity. We then used lidar data to determine if patterns of vegetation structural characteristics observed at the field sampling sites occur across the broader landscape using airborne lidar.

The study area is located in central Alberta, Canada, in the Utikuma Region Study Area (URSA). Transects with between 15 and 30 understory plots and 1 - 2 tree plots were deployed in 19 peatlands, divided into those that were burned once in recent history (in 2011) and those that were burned twice (1956 and 2011). Field data were collected coincident with an airborne lidar survey of the broader region, where 120 peatlands in short and long- fire return intervals were identified. We found that DB transitional zones had significantly deeper organic soil deposits than those found in SB11 transition zones ($p < 0.000$), while having significantly lower peat bulk densities. Peatlands also had significantly greater pH in DBs than in SB11s ($p < 0.006$). Proportions of regenerating species, such as *Sphagnum*, black spruce (*Picea mariana*), and willow (*Salix spp.*) differed significantly between peatlands and transitional areas in DB vs SB. Greater proportion of coniferous species were observed in SB. Deciduous upland species, such as paper birch *(Betula papyrifera*), were more commonly found at DB sites. Also, significantly greater heights were observed in vegetation both in peatland and transitional areas that had experienced DB. This agreed with the results of our lidar analyses, which demonstrated taller vegetation in DB areas, particularly in bogs. Our results suggest that short interval return fires may result in enhanced fragmentation and the transition of boreal peatlands into ecosystems that have characteristics of transitional and upland forests. We present a conceptual model of a potential feedback schematic representing peatland responses to shortened fire return intervals.

Key Words: Remote Sensing, Wetlands, Regeneration, Ecosystem Succession

3.1 INTRODUCTION

Boreal peatlands are globally significant carbon sinks, yet in northern regions, peatlands are increasingly impacted by climate change (Benscoter & Vitt, 2008). Within the western part of the Boreal Plains Ecozone, Canada, peatlands exist at the dry limit of peatland formation, where potential evapotranspiration is equal to or slightly exceeds precipitation inputs during most years (Yu et al. 2012; Devito et al., 2016). Peatlands in this sub-humid region are responsive to relatively small shifts in water balance and are especially sensitive to climatic changes. While it is recognised that average air temperatures in the northern latitudes are predicted to increase by 1-4 degrees Celsius within the next 100 years (Thompson et al., 2017), if there is little to no corresponding increase in precipitation, overall water loss will be exacerbated (Thompson et al., 2017). Vegetation moisture declines, species shift, and changes to soil composition that can occur from long-term drying could lead to more vulnerable and fire-susceptible ecosystems (Flannigan et al., 2013; Thompson et al., 2017; Kettridge et al., 2019).

Boreal peatlands are composed of thick layers of insulating moss vegetation, which extend to depths greater than 0.4 m due to gradual processes of carbon sequestration, biomass accumulation and decomposition (Benscoter et al., 2011). As such, peatlands have largely been an important refuge from wildfire for longer time periods compared with surrounding upland forests (Turetsky et al., 2004; Whitman et al., 2019), however, this may change as boreal peatlands become drier (Flannigan et al., 2013).

Predicted climatic warming and drying in western Boreal Canada is expected to have important implications to the fire regimes of forests and peatlands. Drier conditions may mean that fires burn more frequently. The period between fires is described as a "fire return interval". Estimates of wildfire return intervals range 100-130 years between peatland fires (Turetsky et al., 2004). However, changes in the severity and frequency of wildland fire (Flannigan et al., 2013), could further decrease return intervals for peatlands in the future (Whitman et al., 2019). Long return intervals are necessary for a complete re-establishment of some peatland vegetation species. *Sphagnum* moss for example, can take decades to re-establish to conditions prior to fire following a relatively long return interval (Benscoter & Vitt, 2008). Black spruce (*Picea mariana)*, which are often a dominant tree species in boreal peatlands, have been shown to increase in stem density for up to approximately 80-90 years post-fire (Benscoter & Vitt, 2008). Post-fire peatland recovery begins to reach a steady-state approximately 100-125 years post-fire, with successional vegetation community change becoming less variable over time (Benscoter & Vitt, 2008).

In addition to peatland succession, long-term post-fire hydro-ecological feedbacks can maintain resilience to fire, resulting in maintenance of moisture conditions (reviewed in Nelson et al. 2021). Examples of resilient features of peatlands reviewed in Nelson et al. (2021) include: 1) hydrological connectivity to groundwater or larger peatland complexes, which results in greater stability of the water table and soil moisture regimes (Lukenbach et al., 2015); 2) high proportions of *Sphagnum* moss regeneration, which have greater moisture retention during dry periods and are often found on hummocks, which provide enhanced variability of post-wildfire microtopography (Thompson & Waddington, 2013); and 3) greater dominance of black spruce trees, which have low stomatal conductance, helping maintain low rates of evapotranspiration (Dang et al., 1997; Whitman et al., 2019).

Climatic drying, in combination with previous low fire severity conditions, and anthropogenic fire prevention over the past decades, could increase the density of fuels found within peatlands, thereby reducing peatland resilience to fire by predisposing them to more intense fires and greater burn severity than might otherwise occur in the future (Flannigan et al., 2013; Kettridge et al., 2019). For example, Kohlenberg et al. (2018) tested the burn severity of peat by comparing samples from non-disturbed peatlands with those from drained peatlands where drained peatlands represent a proxy for prolonged aridity.

They found that samples from drained peatlands experienced greater rates of consumption than those from undrained peatlands (Kohlenberg et al., 2018). The combination of peatland draining and severe burning damages water-repellent peat layers that restore peat moisture following fire, subsequently increasing soil drying (Kettridge et al., 2019). Additionally, soil bulk density is greater in burned peatlands than in unburned peatlands (Thompson & Waddington, 2013), which increases water table variability reviewed in Nelson et al., (2021). The combination of increased fuel bulk density, evaporative water losses from atmospheric warming, and post-fire drying associated with damage to the peat layer, result in enhanced water table fluctuations. This can create a positive high fire severity feedback, which can further expose dense, high carbon organic soils as a potential fuel for fires (Kettridge et al., 2019; Nelson et al., 2021). This positive feedback could contribute to increased peatland vulnerability to wildland fire at potentially shorter intervals than previously experienced.

Alternatively, increased fire frequency could create deep burning in the margins, increasing soil moisture, and reducing the depth to water table (Lukenbach et al., 2015; Hokanson et al., 2016), allowing flooding to occur. Under the condition of a shallow water table, the non-tolerant deciduous species would be less likely to establish within these areas, thereby increasing available resources for hydrophilic species, allowing peatland and water-tolerant species to thrive. While several studies have examined the post-fire effects of "normal" return-interval wildland fires on peatlands (up to ~150 years), particularly in terms of carbon loss (Turetsky et al., 2004; Kettridge et al., 2014), relatively little is known about the implications of shortening return intervals on the post-fire boreal peatland environment.

In the late spring prior to leaf out of 1956 and 2011, the area north of Utikuma Lake in Central Alberta, Canada was disturbed by wildland fire. The fire scars overlap in a 12 km^2 area (Alberta Wildfire, 2018). This provided a unique opportunity to examine the impacts of a shortened return interval fire of 55 years (herein "double burn" (DB), where burn scars overlap) and longer-return interval fire (the area that burned in 2011, herein "single-burn" (SB) because no other fires were observed in the past recorded history of >80 years). The overall aim of this study was to examine potential indicators of sensitivity of peatlands and transitional areas to shortened fire return interval by quantifying differences in post-fire soil characteristics and vegetation regeneration. Specific questions include: a) Do differences in soil characteristics (bulk density, pH, and soil depth) exist between long-return interval fire (SB) and short- return interval fire (DB)?; b) Do these differences vary between peatlands and transitional areas? c) Were there differences in the density and variability of regenerating vegetation species and height (representing growth rate), within SB and DB peatlands and transition zones? ; And d) If differences in rate of regeneration exist between SB and DB peatlands and transitional areas measured in the field, do we also observe similar trends across the broader landscape?

Based on the current understanding of fire processes in peatlands, we develop and test the following hypothesis. Typically, in lower severity fires, the partially living, drier acrotelm (living upper aerobic layer of peat) is burned, while the catotelm (water-logged and compact lower anerobic layer of peat), which has higher bulk density, remains intact, allowing peat to accumulate over long time periods (Benscoter & Vitt, 2008; Benscoter et al., 2011). In higher severity fires and in drained peatlands, fire removes the top layer and consumes peat at deeper depths to expose denser layers (Thompson & Waddington, 2013;

Nelson et al., 2021). Additionally, Thompson and Waddington (2013) found that burnt peat had greater surface soil bulk density than that of unburnt peat. Increased evapotranspiration, in combination with the burning of acrotelm layers exposes denser peat layers (Thompson & Waddington, 2013; Nelson et al., 2021). Therefore, increasing frequency of fire could increase soil bulk density. We hypothesize that soil bulk density found within recent DB peatlands will be greater than those experiencing a longer return interval (SB sites). Hokanson et al. (2018) and Lukenbach et al. (2015) measured greater peat bulk densities within margin (transition) zones of peatlands, in comparison to peat samples taken from the peatland centers, due to greater water table fluctuations and compaction within the margins. From this, we would expect that within our peatlands, which are located within the same study area, would similarly have greater soil bulk densities within the transition zones, in comparison to soil samples from the peatlands. This could make peatland transitional areas more vulnerable to fire, especially as drying and changes in bulk density extends towards peatland centres.

In addition to the above hypotheses, re-establishment of moss species regains nearly full ground cover approximately 20-25 years post-fire but continues to diversify for approximately 80 or more years (Sirois & Payette, 1989; Benscoter & Vitt, 2008). Therefore, during the early regeneration phase, we would not expect significant variability of returning mosses. Additionally, increase in bulk density proposed above can impact hydrology, such that soils with greater bulk density and lower specific yield (SY) can result in greater variability of depth to water table with given precipitation or evaporative removal, thereby potentially reducing *Sphagnum* regeneration (Thompson & Waddington, 2013; Nelson et al., 2021).

Whitman *et al.* (2019) suggest that short-interval fires in boreal forests have less coniferous recruitment, and lower organic soil content. Additionally, they found an increased presence of deciduous species (Whitman et al., 2019). While they do not specify the wetland class, the wetlands described in their study are characterized by deep organic layers and coniferous vegetation (Whitman et al., 2019), similar to that found in peatlands within our study area. We therefore hypothesize that post-fire species in double burns will have similar regeneration trajectories as Whitman *et al.* (2019), with greater deciduous regrowth. In the context of a peatland, which is often dominated by stunted black spruce trees, we expect that under a shortened return interval, there might be a shift to greater density of upland broadleaf tree or shrub species encroachment (hypothesized by Nelson *et al.* 2021 and Kettridge et al. (2015)), which regenerate more rapidly than black spruce. Therefore, we expect to observe greater overall vegetation growth in DB sites, as indicated by taller vegetation heights.

We also expect that greater fragmentation of peatlands post-fire (determined by comparing SB and DB peatlands) impact hydrology and therefore may also contribute to differences in vegetation regeneration. We hypothesize that peatlands experiencing DB will have greater shape complexity as determined using the Perimeter:Area (P:A) ratio than those that have had SB. Transition zones surrounding the peatland perimeters, are known to be more susceptible to severe burning, due to their characteristic fluctuating water tables, dense organic materials, and increased rates of evapotranspiration (Lukenbach et al., 2015; Lukenbach et al., 2017; Hokanson et al., 2018). Since increasing fragmentation increases transitional zones while reducing peatland area, it is possible that increasing perimeter:area ratios will result in increasingly fire-susceptible peatlands. Based on these hypotheses, this research will quantify important post-fire ecosystem characteristics within the first 10 years of fire following a shortened vs. a longer return interval and will provide baseline understanding of how peatland successional trajectories may alter in the future with more frequent and more severe fires in boreal environments.

3.2. Methods

3.2.1. Study Area

Post-fire peatlands are located in the Utikuma Region Study Area (URSA), located approximately 400 km north of Edmonton, Alberta Canada and within the Boreal Plain Ecozone (56.107^{O}N 115.561^{O}W) (Figure 3.1). The climate in this region is characterized as sub-humid with cold, long winters, and cool dry summers. Mean annual temperature and precipitation is 1ºC and 483 mm, respectively, and annual ET averages 517-519 mm (Devito et al., 2016). The study area is characterized by low relief with three dominant glacio-derived geologic landforms, which influenced the formation and proportion of well drained mineral forests and organic rich wetlands typical of the Boreal Plains (Devito et al. 2017). Peatlands are the dominant wetland class (bogs and fens) and represent 30-40% coverage in low lying regions of coarse textured glacial fluvial outwash landforms, 20-30% coverage as isolated or loosely connected peatlands on fine textured hummock moraine landforms, and greater than 60% coverage extensive peatlands over lacustrine plains (Devito et al. 2016, 2017). Wetlands are dominated by black spruce (*Picea mariana*), with tamarak (Larix sp) in fens, bog birch (*Betula glandulosa*) and Labrador tea (*Rhododendron groenlandicum*) as well as *Sphagnum* moss species (Devito et al., 2016). Vegetation within the region consist of predominantly mixed-wood upland forests and large expanses of wetland complexes. Broadleaf deciduous trees, including aspen (*Populus tremuloides*) and balsam poplars (*Populus balsamifera*), with minor stands of coniferous white spruce (*Picea glauca*), dominate forest canopy on fine-textured hummocky moraines or transition glacial-fulvial areas with considerable silts; while Jack Pine (*Pinus banksiana*) dominate forest canopy on coarse textured soils (Redding and Devito, 2011; Bridge and Johnson, 2000).

During the summer of 1956, a large wildfire burned approximately 5700 ha within the study region (Alberta Wildfire, 2018) and 55 years later, in May 2011, the Utikuma Complex Wildfires burned approximately 120000 ha (Devito et al., 2016; Alberta Wildfire, 2018) (Figure 3.1). The 1956 fire burned east over a fine-textured hummocky moraine, partially burning several kilometers into the adjacent peatland dominated clay plain. In contrast the 2011 fire originated in peatland dominated clay plain and burned to the west coming to rest after briefly entering the hummocky moraine. Details of burn severity from the 1956 wildfire were not available and therefore burn severity is not considered in the subsequent analysis. Figure 3.1 illustrates fire overlap (1956 + 2011), the SB area of 2011, and the location of field transects and a coincident lidar survey. Wetlands within the study area were identified using a wetland/upland classification of Chasmer et al. (2016) and were classified into wetland class (bog/fen) using manual interpretation and field validation (Government of Alberta, 2015). A large portion of this landscape is dominated by forested uplands, open water ponds/lakes, swamps, and marshes (some of which can be seen in Figure 3.1), which were not analyzed in this study.

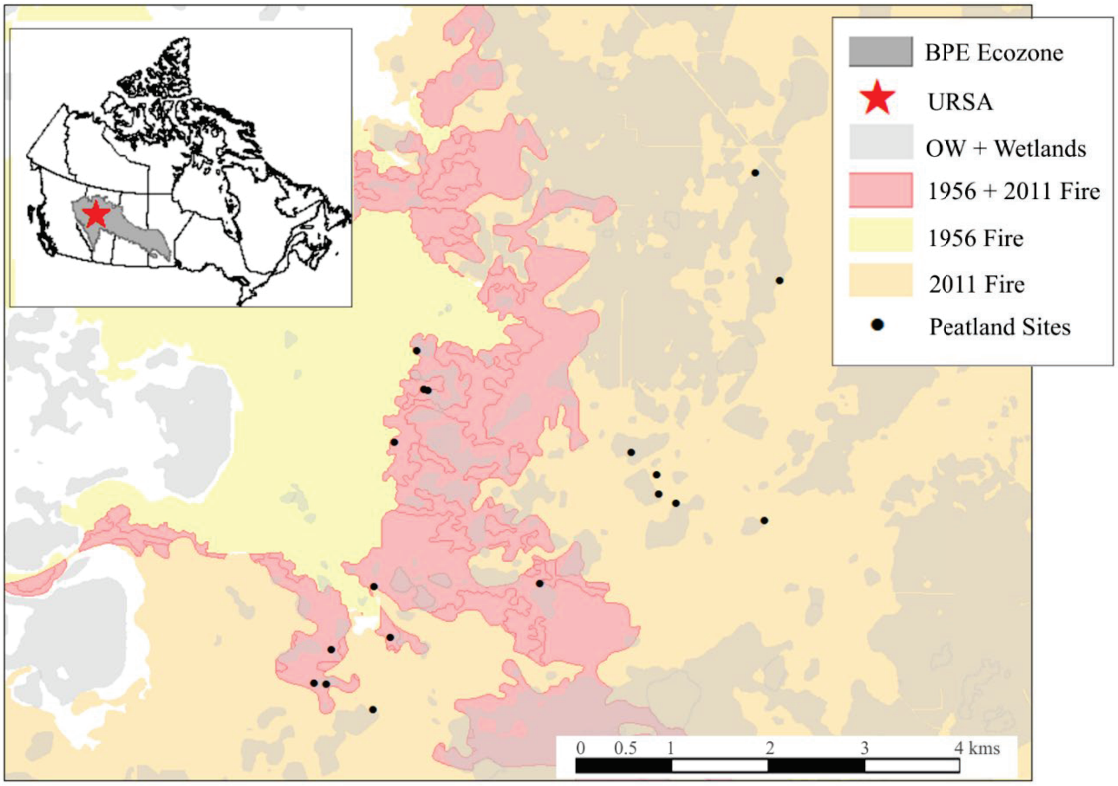

Figure 3.1. The study area is found within the Boreal Plains ecozone (inset). Here, polygons indicate marsh (OW) and wetlands (mostly peatlands) found within the areas burned by fire in 1956, 2011 (SB) and in the area of DB (1956 and reburned in 2011). Peatland sites represent areas of field data collections, including small plots of 1 m^2 along transects.

3.2.2. Field Data Collection

To quantify influences of post-fire environments on vegetation regeneration in SB and DB boreal peatlands and transition zones, we installed 19 transects each perpendicular to the approximate edge of individual peatlands. Transects were installed between 27 July 2019 and 4 August 2019, which were defined *in situ* using the Alberta Wetland Classification System. Within these 19 transects, nine were located within the limits of the 2011 SB and 10 were installed in the area of overlap, DB. In the SB peatlands, we identified and installed transects in three bogs and five fens. Within DB peatlands, we identified and installed transects in six bogs and four fens. These were limited by accessibility and similar environmental characteristics, including size, shape and orientation. All peatlands included an organic peat layer of greater than 0.4 m, with characteristic vegetation and hydrologic properties typical of peatlands. Transition zones were identified *in situ* by locating the edge of the peatland, with the primary identifier being sudden or gradual transitions in vegetation and peat depth. The transition zones were differentiated from surrounding uplands by their vegetation, which was a mixed peatland- and upland- dominant vegetation, and by differences in observed surface vegetation and soil. The transects were visually approximated in the field by identifying the approximate area where peatland vegetation was considered the dominant feature, and where peat depth was consistently greater than 40cm. From this approximate point, the transition zone was captured by walking approximately 10 m towards the upland (away from the peatland and into the transition), to capture a portion of the transitional area. The approximate 10 m stopping point from peatland edge was considered the start (0 m) of the transect. From the identified peatland edge to point zero was defined as "transition zone".

Transects included approximately 10 m forest to peatland transition and extended to distances of up to 20 m within peatlands. Each transect was accurately located using Global Navigation Satellite Systems (GNSS), with a base station and rover placed at the start (0 m, within forest/peatland ecotone) and at the end (in peatland) of each transect. GNSS base and rover stations were operated for a minimum of 1 hour, resolving the location of the transect to centimeter accuracy. A tape measure was placed along the transect to locate the position of 1 m x 1 m vegetation plots every 2 m along each transect with bearing recorded. To determine elevation, survey level measurements were recorded at 1 m intervals, (including within vegetation plots) along the length of the transect. Vegetation measurements within each 1 m x 1 m plot included: dominant and sub-dominant vegetation overstory, understory and ground cover species, visual and photographic estimates of percent cover, and understory vegetation height at three locations within each plot (averaged). Percent cover was estimated for each species/cover type. Depth of peat was also measured at 2 m intervals along the transect by inserting a 2 m steel probe into the peat until it intercepted a mineral lens to a maximum of 1.5 m.

Along each transect, we also collected three soil samples within the transition zone and three soil samples within the peatland zone (at the start and end of the transects) at depths of 0-10 cm and 10-20 cm (12 samples in total). Deeper samples were not considered due to the position of the water table. Soil samples were extracted using a steel "cookie cutter" and a knife, to ensure that the peat/organic layer was not compressed in the extraction process. For the deeper layer, we cut a small area on one side of the sample so that the layer (10-20 cm) could be removed. Measurements were taken *in situ* for calculation of soil volume using a ruler. Upon return from the field, all soil samples were weighed and then placed into industrial grade-ovens. Samples were dried at a temperature of $105^{O}C$, for a minimum of 72 hours or until dry weight remained consistent (maximum eight days). Samples were weighed approximately every 12 hours during this period and a final dry weight was measured and recorded. Dry weight and volume were then used to determine soil bulk density. Variations in bulk density can be indicators of intrinsic organic and moisture characteristics of the soil at the location of measurements (Gregorich & Carter, 2007). In addition to these measurements, dried samples were also used to measure pH using a 1:10 ratio of peat to deionized water via the Vernier Go Direct Tris-Compatible

Flat pH sensor (Vernier, Oregon, USA). Soil pH controls many soils chemical and biological reactions, such as the availability of soil phosphorous and nitrogen, and is a determining factor in differentiating bogs and fens (Gregorich & Carter, 2007). These methods follow those described in Gregorich and Carter (2007).

3.2.3. Lidar Data Collection and Processing

An airborne lidar survey was acquired coincident to the collection of field data on the 29th of July 2019, using a Teledyne Optech Inc., (Ontario, Canada) Airborne Laser Terrain Mapper "Titan" multi-spectral lidar. This survey was planned and operated by the ARTeMIS Lab at the University of Lethbridge. The Titan includes three laser emission wavelengths at 1550 nm (shortwave infrared, Channel 1), 1064 nm (near infrared, Channel 2), and 532 nm (green, Channel 3), though the multi-spectral capability of the sensor was not utilised for this study. The use of airborne lidar systems to retrieve vegetation structural parameters has been well documented in literature, including canopy height and canopy cover (Dubayah & Drake, 2000). We used the lidar data to compare differences in vegetation height associated with rate of growth since fire between SB and DB in peatlands and transition zones that were not measured *in situ*. This provides an opportunity to quantify spatial patterns of vegetation regeneration across a broader range of peatland characteristics than those sampled using limited field data.

Post-processing of lidar data included classification of ground and non-ground returns, quality control and removal of spurious points using TerraScan (TerraSolid Inc., Finland). Interpolation routines for deriving lidar metrics were performed using LasTools (Rapid Lasso GmbH, Germany) and Surfer (Golden Software, Inc., USA) and derivatives were gridded to a cell resolution of 2 m. A Digital Elevation Model (DEM) was derived using all ground-classified returns from channel 2 (1064 nm) by generating a triangulated irregular network and rasterizing to a DEM (Las2DEM, Rapid Lasso GmbH, Germany). A canopy height model based on the difference between the digital surface model and the DEM was used to determine vegetation height since the fire across the broader region.

3.2.4. Spatial Expansion and Fragmentation Analysis

Peatlands examined using lidar were identified using a combination of spatial data freely available from the Alberta Biodiversity Monitoring Institute (Hird et al., 2017), a higher spatial resolution landcover classification (Chasmer et al. 2016) and confirmed visual assessment of pre-fire satellite imagery from Google Earth. Buffer tools and visual editing segregated areas as peatlands and transition zones in ArcMap (ESRI, USA). Transition zones were limited to a maximum of 10 m from peatland transitions to remain consistent with field data measurements. Using the random points generator in ArcGIS, a total of ~ 4000 random points were distributed across the four main strata: SB-Bogs, SB-Fens, DB-Bogs, and DB-Fens. Each strata received approximately 1000 points, which had a minimum distance of 5 m between each point. Random points were used to extract lidar-derived wetland vegetation height at each point location, similar to extracting information from within measurement plots. To determine if field observations of regenerating structural characteristics in peatlands and transitional zones are representative across the broader region, we used lidar-derived height applied to peatlands and transitional zones in SB and DB areas. Using these methods, we sampled 60 peatlands (30 bogs, 30 fens) in the broader area that has experienced SB and 56 peatlands (30 bogs, 26 fens) in the area of DB, with a total of 116 peatlands identified. The use of lidar provides greater understanding of the variability of post-fire vegetation regeneration associated with fire regime and a broad range of possible environmental drivers. By using a sampling approach, as opposed to using all grid cells available, we limited the potential for spatial autocorrelation between proximal sites, thereby reducing exaggeration of statistical significance associated with high sample numbers.

Perimeter to area ratio (P:A) was used to estimate complex shapes, possibly associated with fragmentation of peatlands within SB and DB areas. Larger (P:A) values represent greater peatland shape complexity, relative to area of each individual peatland. Both perimeter and area were calculated using the geometry calculator in ArcGIS. Ratios were calculated and analyzed in R.

3.2.5. Statistical Analysis

To measure differences between 1) soil characteristics, and 2) regenerating vegetation structures within SB and DB in peatland and transitional environments, and to test hypotheses, a number of statistical tests were performed. Except for depth of organic matter, all measured data variables were non-parametric using a Shapiro-Wilks Normality test. Depth of organic matter, due to its normally distributed data curve, was tested for significance of difference using a parametric T-test, and a Bartlett's test for analyzing the normality of variance. All other variables, including but not limited to, bulk density, pH, vegetation height, and canopy cover, were found to be non-normally distributed, and therefore, were tested for significance of difference between SB and DB sites using a two-sample Kolmogorov-Smirnov tests. Variances between the non-parametric sample populations were tested for significance of differences using the Fligner-Killeen test.

3.3. Results

3.3.1. Do post-fire soils vary between long (SB) vs. short (DB) return intervals?

Shortening of the return interval had significant impacts on some aspects of the soil characteristics within peatlands and transitional areas. As expected, depth of peat within transitional areas was less than that in peatlands and became greater as measurements approached the peatland centres (Figure 3.2). We found no significant differences in depths of organics between peatlands at SB and DB sites ($p = 0.5$, $n = 271$). However, for transition area sites the depth of organics within DB were significantly deeper ($p < 0.004$, $n = 166$) than within SB sites (Figure 3.2).

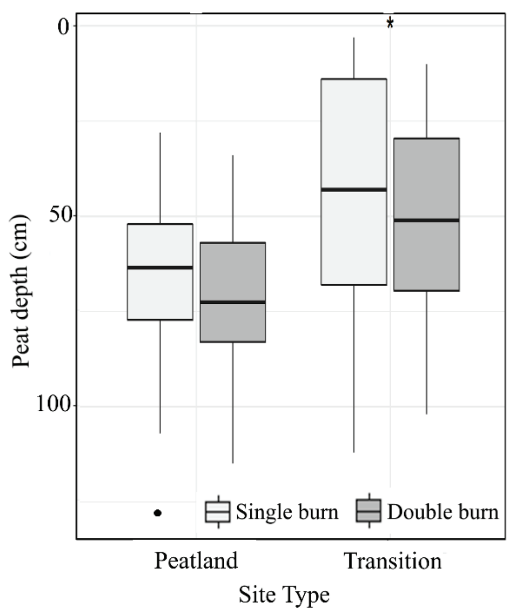

Figure 3.2. Depth of Peat (cm) in SB and DB in peatlands and transition zones. Boxplots represent the interquartile range, where the center line is the median of the data and whiskers are 1.5*IQR. The asterix (*) signifies significant differences double burn and single burn transitions. X-axis is reversed to indicate depth of peat below ground surface (defined as 0cm).

We hypothesized that soil bulk density in DB peatlands would be greater than SB peatlands which had experienced longer return intervals. Bulk density increased with depth but were not significantly different between SB and DB peatlands (Figure 3.3 a, b, $p = 0.15$, $n = 271$). Transition areas demonstrated significant variations in bulk density between SB and DB areas. Soil in SB transition zones had significantly greater ($p = 0.03$, $n = 166$) mineral proportions (average = 0.176 g cm^{-3}, stdev ± 0.240 g cm^{-3}) than DB transition zones (average = 0.122 g cm^{-3}, stdev ± 0.138 cm^{-3}) (Figure 3a, b).

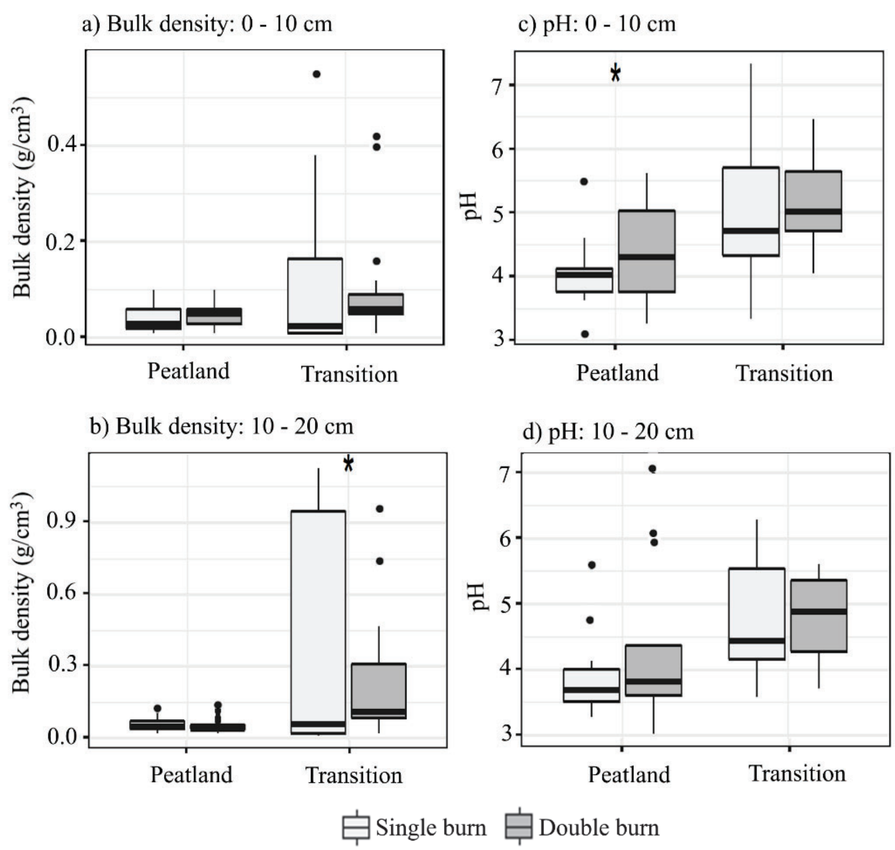

Figure 3.3. Bulk density and pH of soil samples from peatlands and transition zones in SB and DB sites at a depth of 0-10cm (a, c), and 10-20cm (b, d).

The difference in pH between SB and DB peatlands was significant only in the top 0-10cm ($p < 0.006$), where DB peatlands had higher pH than SB. Despite these differences, we found no significant differences in pH between bogs and fens. There was no significant difference in pH in SB and DB transitional areas sites.

3.3.2 Do variations in vegetation species occur following long (SB) vs short (DB) fire return intervals?

Vegetation species observed within peatland SB and DB sites vary from those found within transition zones. Of the total number of plots within our sites, 28% of plots were entirely burnt organic surface material (denoted as "Bare ground" in Figure 3.4). with 38% of plots in SB and 21% in DB peatland sites consisting of no living vegetation (sig. different, $p = 0.0003$) (Figure 3.4). In transitional areas >99% plots had vegetation cover (Figure 3.4). In peatlands, we also observed a greater proportion of scorched (dead) *Sphagnum* hummocks in SB areas (46% of plots) compared with DB (21% of plots), which had fewer regenerating and/or surviving *Sphagnum* species (Figure 3.4). Feather moss species experienced minimal regeneration, appearing only in a few of (2%) DB plots.

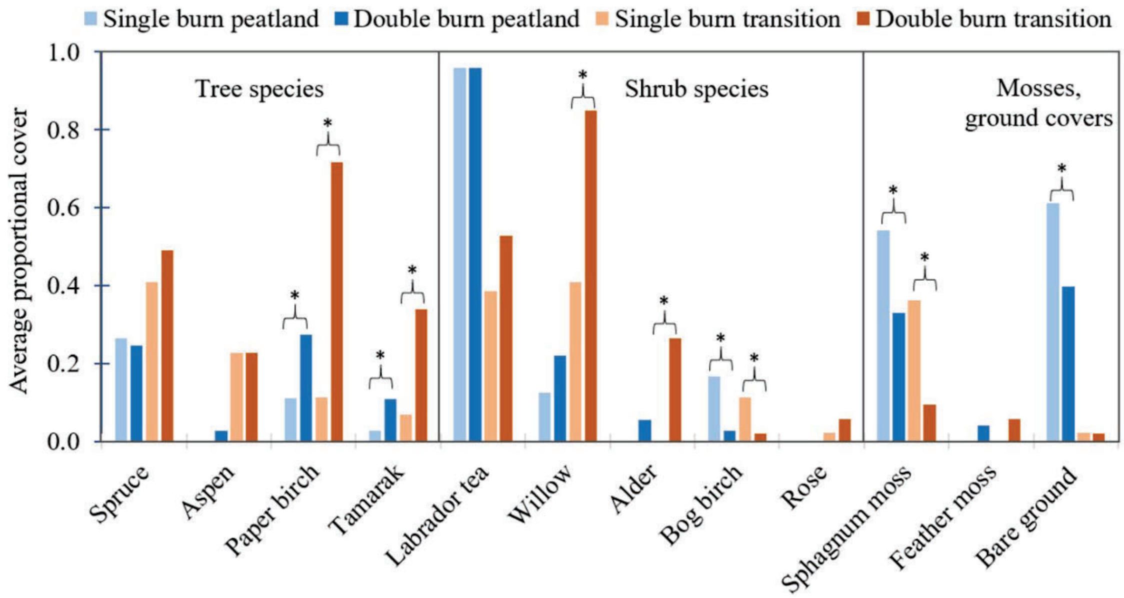

Figure 3.4. Bar plot of vegetation species distribution within SB and DB peatlands and SB and DB transitions determined from field measurements. Percent cover was individually estimated for each species/cover type.

In peatland SB sites the largest proportions of shrub species were those common to peatlands (e.g bog birch (*Betula glandulosa*) (16% of plots), $p = 0.0001$), whereas species more commonly found in uplands or transitional areas were found at greater proportions in peatland DB sites. For example, alder species (*Alnus tenufolia* and *Alnus Crispa*) common to transitional areas were found in 5% of DB peatland plots and 27% DB transition zone plots, while no alders were found in SB peatland plots ($p = 0.05$, and $p < 0.0000$, respectively). Additionally, DB peatlands had greater proportions of plots in which the dominant regenerating species were shrubs (all species; 46% of all DB plots) when compared with SB sites (8% of all SB plots) ($p < 0.0000$). Prevalence of willow species (*Salix spp.*) found in higher proportions in transitional areas was greater in DB than SB sites in both transitional areas and peatlands. Stunted black spruce were present in both SB and DB peatlands, however, we found that SB peatland sites, regardless of being fen vs bog, had higher proportions of black spruce than in DB sites (Figure 4). In contrast, DB sites had higher proportions of paper birch (*Betula papyifera*, $p = 0.0003$) and balsam poplar (*Populus balsamifera*, $p = 0.01$), which are species that are typically more prevalent in upland forests.

3.3.3. Do differences in vegetation height/cover occur in long (SB) vs. short (DB) fire return intervals?

Variations in post-fire vegetation structural characteristics, such as height, were found between SB and DB areas, often corresponding to differences in regenerating vegetation species composition (e.g., conifer vs. broadleaf trees/shrubs, Figure 4). The average height of regenerating vegetation in DB peatlands, as measured *in situ*, was much taller than in SB peatlands ($p < 0.000$, n =) (Figure 3.5). Vegetation heights were also more variable (indicated by height variance) in DB peatlands when compared with SB ($p < 0.000$). This indicates greater complexity of vegetation structures in DB peatlands (Figure 5), while field-estimated total ground cover within peatland vegetation plots was greater in SB (average 76% cover) compared with DB (average = 53% cover) ($p < 0.000$).

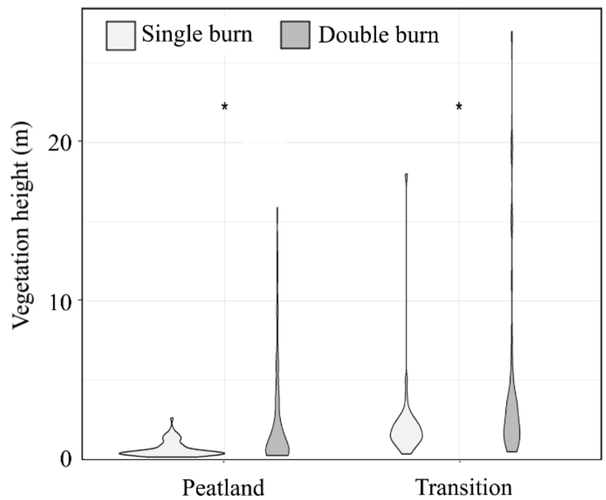

Figure 3.5. Field measured height of the dominant tree or shrub in each vegetation plot at SB and DB peatlands and transition zones. Violin plots show stretch in the y-direction to indicate the range of maximum heights measured and horizontal stretch in the x direction represents percent frequency.

Measured regenerating vegetation height within transitional zones also varied, where average vegetation heights were shorter in SB compared with DB ($p = 0.004$) (Figure 3.5). Shrubs were taller in DB (average = 2.92 m, stdev = 3.8 m) compared with SB (average = 1.24 m, stdev = 0.7 m). Despite height differences of shrubs, tree heights were not significantly different, though were taller in DB sites (Figure 3.5). We also found similar patterns of height variance and vegetation cover in transitional areas, including greater variability in DB areas than SB ($p < 0.000$). Greater average vegetation cover was also found in SB plots (64%) relative to DB plots (52%) ($p < 0.000$), similar to peatlands. Similarly, trees also had greater average canopy cover in SB (62%) compared with DB sites (44%) ($p = 0.04$), while there were no significant differences found between SB and DB shrub cover.

3.3.4. Differentiating post-fire variations in vegetation height in SB and DB bogs and fens

While differences in regenerating vegetation between bog and fen peatland classes may contribute to the variation, we found no significant differences between SB bog vs. SB fen vegetation heights ($p = 0.08$), or between DB bog vs. DB fen ($p = 0.83$). We suggest that: 1. Different sample sizes of bogs and fens between DB and SB areas did not contribute significantly to weighting of vegetation height within samples, and 2. Differences observed when testing peatland types between fire regimes (SB vs. DB) were more likely to be the result of the fire regime than peatland type.

Within peatland sites only (not transition zones), both SB bogs and fens had significantly shorter vegetation heights compared with their paired DB bogs and fens ($p < 0.000$ and $p = 0.0001$, respectively). With in peatland sites of bogs the variance of vegetation heights within DB sites was significantly greater ($p < 0.000$) than those found in SB sites (Figure 3.6). Vegetation heights in fens in SBs were significantly shorter ($p < 0.000$, average = 1.65 m) than those of DBs (average = 3.73 m) (Figure 3.6).

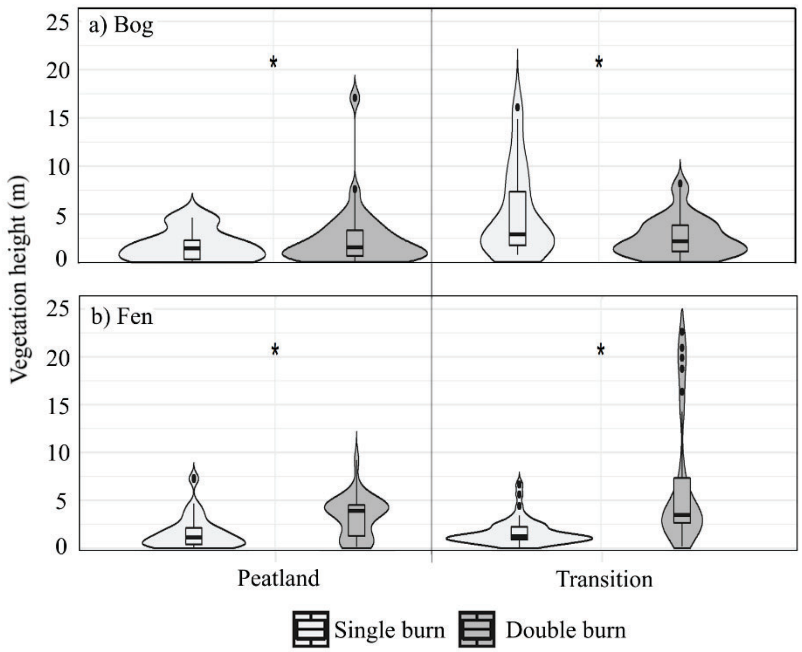

Figure 3.6. Lidar-derived height (m) of vegetation found within single burn and double burn measurement plots and per peatland class for the 19 field wetland sites. The Asterix denotes significant differences (alpha = 0.05).

3.3.5. Trends in post-fire vegetation regeneration in SB and DB exist across the broader region

Using lidar data acquired across the broader region, we observed slightly different results from our field/lidar-based measurements. Peatlands that had undergone DB had slightly greater proportion of shrubs and trees > 3 m in height (32% shrub/tree proportion than that found in SB peatlands (29%, p = 0.04)), similar to that observed in measured vegetation plots. Within fens in SB and DB areas, regeneration vegetation heights in DB areas were found to be significantly greater across the broader region (p = 0.016), averaging 2.29 m in DB fens (stdev = 1.6 m, n = 1003) and 1.93 m in SB fens (stdev = 1.3 m, n = 924) (Figure 3.7). Regeneration heights in bogs were found to be significantly greater (p =

0.04) in DB areas, averaging 2.5 m (stdev = 2.1) than in SB areas, which average 1.5 (stdev = 0.9).

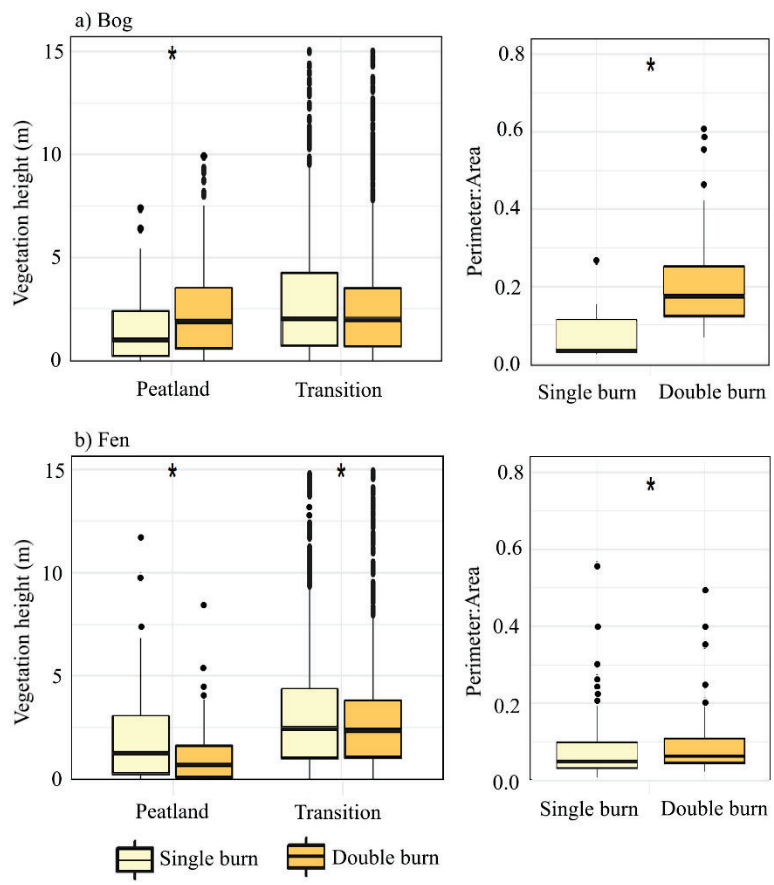

Figure 3.7. Vegetation height at the 95th percentile in m extracted from lidar data in SB and DB sites on the left and the perimeter to area ratio illustrating shape complexity of bogs and fens, right, where increasing P:A indicates increasing complexity, possibly associated with fragmentation or less 'rounded' shapes.

We hypothesized that peatlands with greater fragmentation, as represented by the ratio of the ecosystem area relative to the surrounding area of the surrounding ecosystem (P:A), would be differentially affected by fire regime. Bogs located in the DB areas had significantly greater P:A, indicating that they are characterized by greater fragmentation, and have more complex shapes (average = 0.20, stdev ± 0.12) than those found in SB areas (average = 0.07, stdev ± 0.05; $p < 0.000$) (Figure 3.7). Fens in DB areas also had greater P:A (average = 0.10, stdev ± 0.09) compared with those of SB areas (average = 0.08, stdev ± 0.09).

3.4. Discussion

In this study, both long return interval 'SB' and shorter return interval 'DB' peatlands and transitional areas experienced wildfire in 2011 and have had the same length of time to regenerate post-fire, yet differences are apparent between the regimes. Here we found that DB fire in these land covers demonstrate differences in soil characteristics (Figures 3.2, 3.3), tree/shrub/moss species composition and proportions (Figure 3.4), as well as heights of shrubs and trees regenerating post-fire relative to sites that experienced longer fire return interval (SB) (Figures 3.5-7).

3.4.1 Differences in soil characteristics between SB and DB peatlands and transition zones

Whitman et al. (2019), found that mean depth of the residual organic soil layers was similar between long interval return areas and short interval return areas in wetlands. Also, they found that the variability of organic soil depth was greater in short interval return areas, which are analogous to DBs. Maintenance of deeper organic soil layers was attributed to the resilience of wetlands to wildfire due to moisture conditions and overall lower fuel consumption (Whitman et al., 2019). In our study, we found no significant differences in peat-depth within peatlands that had experienced shorter or longer return intervals (Figure 2). Peatlands within our study typically maintained deep organic/peat soils. Within transitional areas between peatlands and forests, the depth of organic layer in DB was significantly greater than areas found in SB, with less variability of depth. In contrast, Whitman et al. (2019) found that residual organic soil depths were greater in long-interval (SB) return compared with short-interval (DB) return in boreal forests and peatlands. Reduced depth of organic soils in transitional areas compared with peatlands confirmed in other studies, have shown that organic soils in these areas are more sensitive to deeper burning due to local hydrology (Lukenbach et al., 2015). One possible explanation for our results is that the DB transition zones may have had some *a priori* resilience to the 2011 fire, and therefore did not burn as deeply. Another possible explanation is that mosses were highly productive in the period following the 1956 fire due to nutrient and water availability. Since mosses have long post-fire establishment periods (Benscoter & Vitt, 2008) it is possible that the peat had not yet reached an equilibrium between "production" and decomposition phases, observed in older peatlands. This is supported by Ward et al. (2014), who found that organic layer thickness builds for up to 64 years post fire, where it there-after reaches a stage of stability. Additionally, using logarithmic models based on field measurements, Simard et al. (2009) predicted that organic layer depth would increase in peatlands most quickly in the first 150 years post fire, especially in those peatlands that had low- relief elevations, similar to those found at URSA.

We hypothesized that bulk densities would be greater in DB compared with SB peatlands, assuming that burning of the acrotelm exposes layers of higher density peat (Thompson & Waddington, 2013; reviewed in Nelson et al., 2021). Additonally, burnt peatlands have been observed to have greater bulk densities in comparison to non-burnt peatlands (Thompson & Waddington, 2013), which suggests that under scenarios of multiple burns, peat bulk density would be greater. Contrary to what was hypothesized, we found that bulk density did not vary significantly between SB and DB peatlands. Similar to results found in Lukenbach et al. (2015) and Hokanson et al. (2018) we found that transition zones had higher bulk densities due a greater mineral soil content as well as more densely packed organic materials. Greater bulk densities in transition zones, in addition to variability in water-table depth have been demonstrated to burn more severely in comparison to adjacent peatlands (Lukenbach et al., 2015; Hokanson et al., 2018), which suggests that the transition zones may have been more vulnerable to fires. Our findings, in terms of fire regime however, contradicted our expectations that transition zones in DB would have greater bulk densities than SB transition zones. We found that transition zones in SB had, on average, significantly greater amounts of mineral soil, higher bulk densities, and greater variability of bulk density than those found in the DB area (Figure 3.3 a,b). DB transition zones had greater depths of organic material, and lower mineral soil contents within soil samples than that of SB transition zones, which could suggest that the transition zones of short interval return sites may have had some resilience to fire consumption. Lukenbach et al. (2017) observed flooding at peatland transition zones, where severe burns had exposed the water table. Long term flooding could mitigate severe fires within these zones, providing some resiliency to the peatland margins against short interval reburns. If the DB sites were burnt less severely in 2011 than the SB11 sites, this could explain why bulk density was higher in SB11 sites than in DB sites. Further studies would be required to determine the impacts of fire severity in reburn sites in comparison to single burn peatlands and transition zones.

Soil pH was found to be not significantly different between transitional areas in DB compared to SB (Figure 3.3c and d). We did, however, find that pH was significantly greater in DB than in SB peatlands. Post-fire increases in pH have been reported and are often attributed to the increases in carbonates from ash (Dikici & Yilmaz, 2006). Peatlands in DB areas may have increased pH due to the prevalence of (more recent) ash layers in the upper layers of the peat from multiple (recent) fires.

3.4.2 Differences in Regenerating Vegetation Species and Structural Characteristics

Longer return intervals represented by SB areas in peatlands had significantly larger proportions of scorched, non-productive (bare) ground than that found in DB (Figure 3.4). Large areas of scorched ground are not unusual in regenerating peatlands, as many species common to peatlands including *Sphagnum* mosses, black spruce trees and ericaceous shrubs, can take up to 20 years to re-establish to a significant seeding community (Benscoter & Vitt, 2008). Additionally, under conditions of severe fire peat can burn via smoldering, which can last more than a year, which could contribute to making regeneration slower than in mineral dominant soils (Benscoter et al., 2011; Hokanson et al., 2018). We expected greater proportions of feather moss species, which are more successful in drier conditions (Lukenbach et al., 2015). Feather moss were found in a low proportion of plots (2% of all vegetation plots measured), and only within the DB areas. This finding alone, while interesting, is not conclusive enough to establish differences in hydrological conditions between DB and SBs and would require surface and ground-water measurement.

The composition of regenerating tree and shrub species differed significantly between fire regimes. Shortened return intervals between wildland fires (DB) led to greater proportions of regenerating species more typical in upland and transitional areas than those of peatlands (Figure 4). Additionally, DB peatland areas experienced a reduction in proportion of returning conifer species, such as black spruce (*Picea mariana*), similar to that observed in other Boreal studies (Whitman et al., 2019). Recruitment of broad-leaf deciduous species, such as aspen (*Populus* tremuloides), balsam poplars (*Populus balsamifera*), and paper birch (*Betula papyifera*) were more prevalent in double burn peatlands and transitional areas, supporting the suggestion that short-disturbance intervals may shift conifer dominant ecosystems to broad-leaf species (Whitman et al., 2019). Further, ecological pioneer species appear to be increasing in prevalence within shorter return interval fires. This finding has significant implications to both: 1) the "shrubificiation" or terrestrialization of peatlands, wherein upland species competing for resources may reduce water levels, further drying already susceptible peatlands to transitional land cover types and reducing the overall area of peatlands (Thompson et al., 2017); and 2) changes to returning fuel structures, from greater conifer re-establishment to shrub and broad-leaf deciduous re-establishment, which have lower levels of resilience to future fires (Whitman et al., 2019; Nelson et al., 2021).

Shortened return interval sites (DBs) also included significant differences in heights of regenerating vegetation compared with longer fire return intervals (SBs). Vegetation within peatlands that had experienced DB in the last ~65 years were characterized by significantly taller shrubs within peatlands than those found in corresponding single burn peatland areas (Figure 6). Additionally, we found that heights in DB peatlands and transitional areas also had significantly greater variability of height than those of SB, supporting our hypothesis that DB had greater shrub/tree height variability in part due to differences in regenerating species composition. Changes in roughness associated with spatial variability of shrubs could alter the evaporative demand of regenerating peatlands, especially where roughness enhances turbulence and increases evapotranspiration rates (e.g. Green et al. 2021). Lidar-based findings across a broader area of peatlands and transitional areas indicated that heights of regenerating short-vegetation (less than 3 m in height) were taller in DB peatlands than in SB areas (Figure 3.7). Additionally, the proportions of short vegetation in DB were found to be significantly greater than those of SB.

Tree and shrub vegetation greater than 3 m in height in DB peatlands were also significantly taller than those of SB, supporting our field-measured observations across a broader range of peatlands. As there were relatively similar species composition of shrubs between SB and DB sites, shrub height differences could be a result of enhanced productivity at shorter fire return interval sites. This is supported by Ward et al. (2014) who, by measuring variables such as above ground biomass, leaf area index (LAI), organic layer thickness (cm), and carbon/nitrogen ratio, demonstrated that fire increases productivity within boreal stands post-fire for approximately 80 years post fire. Greater structural variability and species biodiversity in DB areas may be related to changes in hydrology, peat mineralization, and nutrient cycling, which occurs with expansion of the aerobic zone (Strack et al., 2006). Further, Lukenbach et al. (2017) found that peatlands dominated by local groundwater movement were prone to fire and had more dynamic hydrological conditions post-fire, resulting in the regeneration of bryophytes found in mineral uplands.

3.4.3 Peatland fragmentation and increased fire return interval

We found that peatlands within DB areas had higher P:A ratios, indicating greater fragmentation and/or smaller peatlands in comparison to SBs. Increasing the length of a peatland edge, relative to the peatland ecosystem, exposes the peatland to greater potential for edge effects from the surrounding ecosystems (ie. surrounding uplands), including further fire disturbance, as peatland margins are susceptible to burns (Baltzer et al., 2014; Lukenbach et al., 2017). With each subsequent burn, it is possible that the peatland area is reduced by the progressive removal of peat with each short-interval burn, which was predicted by Kettridge et al. (2019), in combination with the shrub encroachment that can occur post-fire (Thompson & Waddington, 2013). This reduction in peatland area would be reflected in an increased P:A ratio.

While increased fire frequency may have some impact on peatland shape-complexity, it is unlikely that it is only the cause of the larger P:A ratios in DB areas in comparison to SBs. One probable explanation for the difference could be due to differences geomorphological conditions between DBs and SB areas. While the study design here-in includes DB and SB field sites within both the lacustrine clay plain and the hummocky moraine regions, a greater proportion of the DB overlap area was located within the heterogenous hummocky moraine than in the clay plains. Therefore, it is probable that the DB peatlands within this study are more bounded by a heterogenous landscape, thus increasing its perimeter:area ratio. However, with greater fragmentation, there is more exposure to surrounding ecosystems, potentially allowing for increased interaction with- and competition from deciduous upland species. Our findings of vegetation height increases, as well as greater proportions of deciduous vegetation regrowth support the hypothesis of changing fuel types in peatlands under future fire regime changes.

Responding hydrological conditions at peatland margins post-wildfire are in part, impacted by depth of burn, severity of burn, and climatic conditions (Lukenbach et al., 2017). Lukenbach et al. (2017) found that post-fire, margins were prone to rapid flooding, as the depth to the water table had been reduced through the burning of the over laying organic layers, which can impact moss-recolonization (Lukenbach et al., 2017). Lowered water table levels, which can occur through climatic drying common in dry cycles, transitioned the flooding marginal zones rapidly to dry conditions (Lukenbach et al., 2017). These rapid changes in hydrological conditions in marginal zones may not be conducive to peat-forming vegetation return, which rely on stable environmental conditions (Potvin et al., 2015; Lukenbach et al., 2017), potentially providing an opening for new species encroachment. Potvin et al. (2015) found that ericaceous shrubs (such as Labrador tea, which was found extensively in our study areas), increase in abundance and biomass with lowered water tables. Depante et al. (2019) demonstrate greater numbers of aspen suckers and seedlings in peatland margins, which had undergone severe burns, and were likely to have lower moisture than adjacent peatlands. Given these suggested changes to the hydrological conditions which influence returning vegetation, along with findings of this study, it is possible then that areas with increasing P:A and fragmentation (thus greater marginal zones), are more susceptible to encroaching upland vegetation (summarized in Figure 3.8).

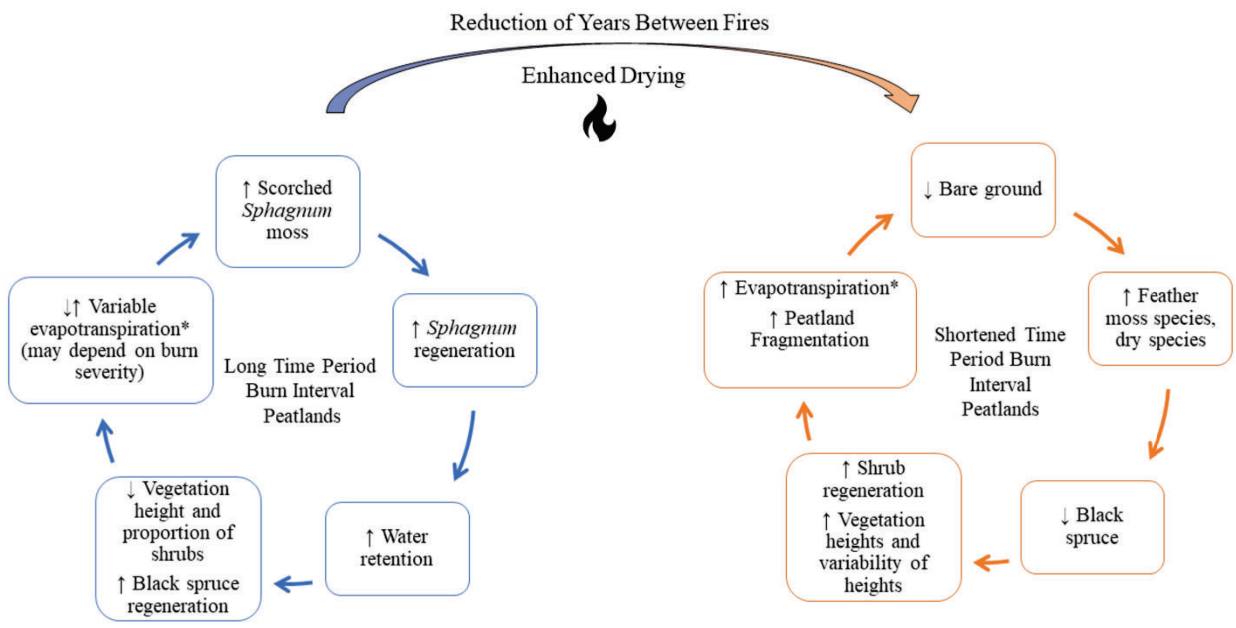

Figure 3.8. Feedback schematic based on the findings observed here using field and lidar methods. Notes marked with a * are predicted conclusions based on known literature. This schematic supports hypotheses in Kettridge et al. (2019), and Nelson et al. (2021) observed for a) minimal sites; and b) within a review of the literature.

3.5 Conclusions

This study determined that peatland and transitional zone soil and vegetation regeneration characteristics associated with shortened return-interval fire differ in comparison to a typical (longer) return-interval fire. This study was limited in terms of historical spatial data of wildfires beyond the ~60-year mark (Alberta Wildfire, 2018).

Our findings suggested that short interval return wildfires (DBs) have greater variability of tree and shrub species composition and proportions, as well as taller vegetation and greater proportional cover of trees/shrubs compared with SB longer return interval fire. Enhanced structural variability within DB may also further influence aerodynamic roughness and evaporative losses. The use of lidar remote sensing across 120 peatlands provides additional confidence in field measured results, which confirmed that a broader range of peatlands that had experienced DB were susceptible to increased shrubification. This could have significant implications for drying in areas where climate-mediated changes result in a shortening of the fire return interval. Further, this could create a positive drying feedback, resulting in the expansion of transitional areas into peatlands, reducing overall peatland areas experiencing shortened return intervals in the future. We also observed that greater proportions of peatlands had more complex shapes in the DB areas determined from the ratio of the peatland perimeter to area prior to the 2011 fire. Shape complexity and fragmentation could further reduce the connectivity of peatlands, which is important for peatland maintenance (Thompson et al., 2019; and reviewed in Nelson et al., 2021). The terrestrialization of wetlands, wherein upland species competing for resources can reduce water levels, may enhance further drying of already fire-susceptible ecosystems. Additionally, our findings suggest that shortened return intervals may influence changes to returning fire fuel structures by transitioning these ecosystems to more broadleaf vegetation species (Kettridge et al., 2019).

Acknowledgements

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4.0 RATES OF RECOVERY IN SHORTENED-FIRE RETURN INTERVAL PEATLANDS USING MULTI-TEMPORAL LIDAR IN ALBERTA, CANADA

Keywords: remote sensing, ecosystem succession, growth rate, boreal

4.1 Introduction

Fire is the dominant disturbance type found within the boreal region (Turetsky et al., 2002), and is expected to increase in frequency and extent over the coming decades, due to increasing temperatures and aridity associated with anthropogenic climate change (Flannigan et al., 2009; de Groot et al., 2013). The frequency and severity of wildland fire disturbance is key in influencing ecosystem succession and trajectory (Weber & Flannigan, 1997), vegetation productivity and recruitment (Sirois & Payette, 1989), and the net carbon budget (Amiro et al., 2001; Wieder et al., 2009). Reducing time intervals or years between fires can produce changes in the environment, such as soil structure and composition, which may influence regenerating vegetation structures and community patterns (Chapter 3; (Whitman et al., 2019). This is a potentially significant concern for the global carbon-climate cycle as boreal peatlands, which have historically maintained resilience to fire and act as long-term carbon sinks, may shift to becoming large sources of carbon (C). Losses of C to the atmosphere occur during combustion, while longer-term release into the atmosphere can occur associated with decomposition for more than ten years post wildfire (Turetsky et al., 2002; Wieder et al., 2009).

Estimates of fire return frequencies within boreal peatlands have ranged from decades to hundreds of years (Sirois & Payette, 1989; de Groot et al., 2013) but more recent estimates indicate that peatland reburn occurs every 100-120 years (Wieder et al., 2009). However, changes in the severity and frequency of wildland fire (Flannigan et al., 2013), could further decrease return intervals for peatlands in the future.

Boreal peatlands are characterized by deep organic deposits which develop over long periods of time due to increased saturation and reduced microbial decomposition (Nordbakken, 2001; Benscoter & Vitt, 2008), making woody vegetation successional processes comparatively slow. While adjacent upland tree stands may recover to their former stem density, height and cover within several decades, dominant tree species within peatlands, such as black spruce *(Picea mariana*) can take up to 90 years to develop pre-fire stem density (Wieder et al., 2009). Reducing the time-interval between fires could therefore interrupt the ecosystem's ability to recover to pre-fire conditions, thereby reducing stand age, increased competition from reduction of seed availability, and an increased demand on nutrient resources (Brown & Johnstone, 2012). Additionally, severe fire-events have been shown to increase peatland evapotranspiration post-fire (Kettridge et al., 2019). Spatial variations in post-fire evapotranspiration can also increase the variability of the depth of the water table, improving conditions for shrub-encroachment (Sherwood et al., 2013; Kettridge et al., 2015; Lukenbach et al., 2015).

Relatively little is known about the impacts of reduced fire intervals on boreal peatland vegetation regeneration and succession, and the potential implications for peatland resilience vs. change. Whitman et al. (2019) examined the impact of reduced fire intervals within the context of boreal forested uplands, and low-relief wetlands. They found that short fire return-interval sites were more prone to reduced canopy density, altered soil chemistry and organic content, and vegetation species compositional changes. Brown and Johnstone (2012) studied the impacts of shortened fire return intervals on seed availability in black spruce stands in Yukon Territory, Canada and found a significant decline in viable seeds compared with sites that had experienced longer-fire return intervals. In Chapter 3, we demonstrated that reducing the interval between fires in peatlands results in significant differences in soil conditions, vegetation composition, and structures. Our results showed that peatlands with reduced fire intervals had greater regrowth across the broader landscape, and a shift in returning vegetation more typical of transitional areas (chapter 3). While disturbance is essential for the maintenance and health of an ecosystem, increasing fire frequency beyond the peatlands ability to sustain resilience could impact the future trajectory of the ecosystem (Thompson. et al., 2017).

The overall goal of this study was to quantify the rate of early post-fire vegetation growth within peatlands and their adjacent transition zones in short-interval (DB) and long-interval return (SB) fire areas using multiple lidar data collections since the fire. Airborne lidar systems and data are well-established within the literature for their ability to measure canopy height with relatively high accuracy (Dubayah & Drake, 2000; Hopkinson et al., 2005; Chasmer et al., 2016; Hopkinson et al., 2016b). Specific questions include: a) At what rate of growth, as measured by the height of returning vegetation, are peatlands regenerating within SB and DB peatland class and forms and transitional zones? b) Comparing between SB and DB regeneration, are there differences in the rates of regrowth? And c) Does the shape of peatlands (including round vs. complex shapes) contribute to differences in rates of regeneration? How do these vary with class and form?

We aim to answer these questions using a multi-temporal lidar remote sensing approach. Remote sensing is an appropriate method for quantifying changes in vegetation structures and cover over a broader scale and over multiple timeframes than what could be achieved using field data collection, alone. The use of airborne lidar for the direct measurement of vegetation structural characteristics has been well documented in the literature (Dubayah & Drake, 2000). These datasets also have distinct advantages over optical remote sensing: specifically, the ability to measure three-dimensional vegetation structures at the top of, within, and under the overstory canopy as well as ground surface elevation (Dubayah & Drake, 2000; Yu et al., 2003; Yu et al., 2006).

4.2 Methods

4.2.1 Study Area

The Utikuma Region Study Area (URSA) is located in the Boreal Plains ecozone, approximately 400 km north of Edmonton, Alberta Canada (Figure 4.1a). Established in 1998 as a long-term hydrological and land-disturbance monitoring site (Devito et al., 2016), there is an extensive network of hydrological and meteorological measurements, vegetative data, and remote sensing data associated with it. URSA is characterized by low relief in a glacio-derived geologic landform area, with a mosaic of forested uplands, wet lowlands, and transitional zones. The dominant wetland class is peatlands, accounting for more than 60% coverage over the lacustrine plains (Devito et al., 2016; Devito et al., 2017). The study area underwent two major fires in recent (scientifically recorded) history, the first in 1956 and a second fire in 2011 (denoted here-in as 1956SB and 2011SB, respectively, where SB refers to Single Burn areas that were burned once since records began). The two burn scars overlapped in an area spanning approximately 12 km^2 (denoted as DB, or areas that experienced Double Burn, Figure 4.1b) (Alberta Wildfire, 2018). Wetlands, including peatlands (bogs and fens) were identified using an enhanced wetland classification map modified from Chasmer et al. (2016) to include class and form, specific to peatlands.

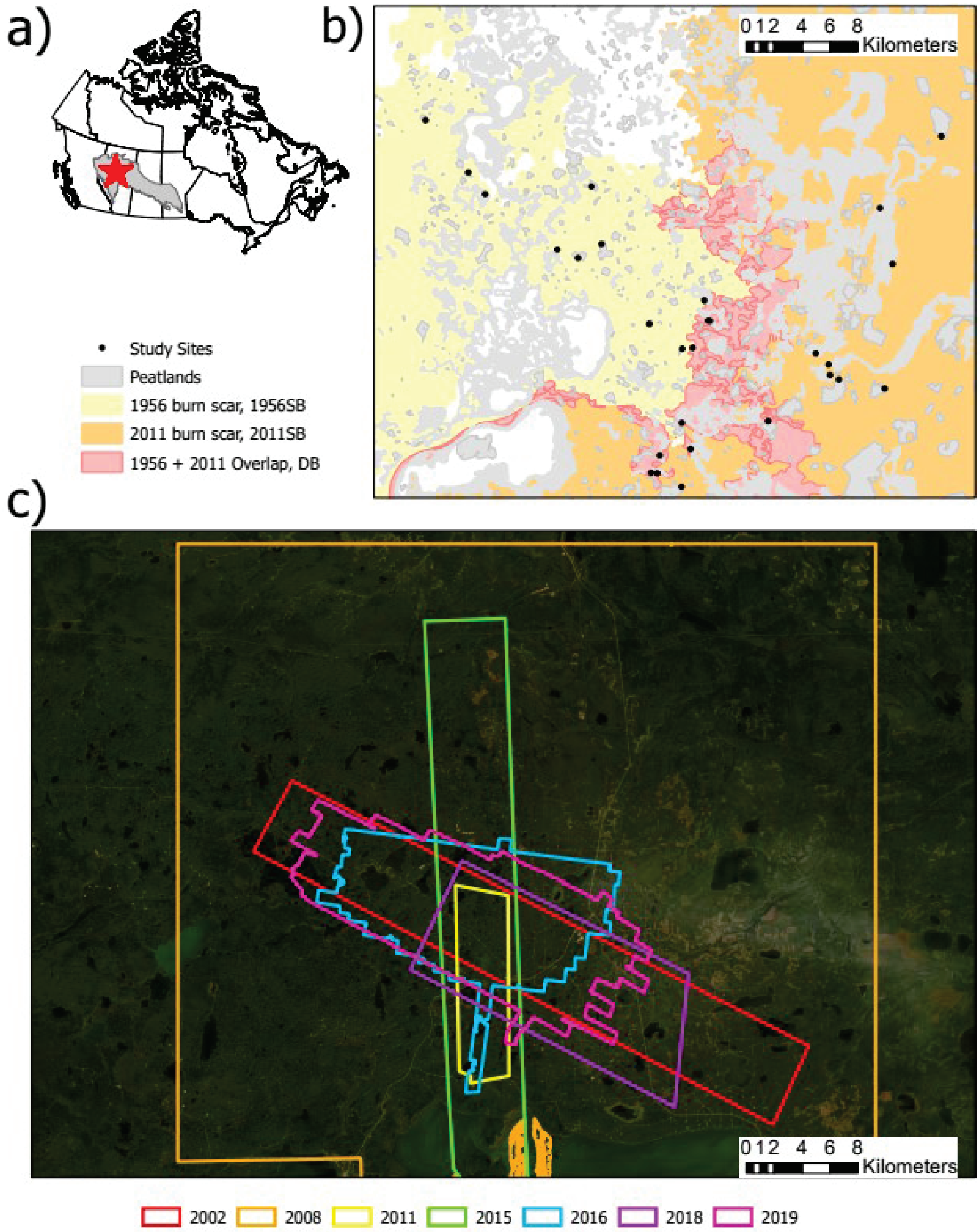

Figure 4.1 a) The study area located in the Boreal Plains ecozone, Alberta Canada. Polygons in (b) indicate peatlands located within the areas burned by fire in 1956 (1956SB), 2011 (2011SB), and in the area of overlap, DB (1956 and reburned in 2011). Peatland sites in (b) represent areas of field data collection and lidar validation survey transects. In (c) the outlines represent the flight swaths of the seven lidar data sets included in this study.

4.2.2 Field Data Collection for Lidar Validation

Field data including vegetation height, cover, and species were collected coincident to lidar data collection in late July 2019 (described in Chapter 3). To validate 2019 lidar-derived height measurements a total of 28 transects were installed into the same number of peatlands, traversing from upland forest into the peatland (described in Chapter 3). Of these, nine were located within the limits of the 1956 SB area that had not burnt in 2011, nine were located within the limits of the 2011 SB and 10 were installed in the overlapping DB area. Within this 1956 burn area bogs and fens were identified in the field using Alberta Wetland Classification System (Government of Alberta, 2015). Each transect extended perpendicularly into each peatland and ranged in length to a distance of 30 m, with 10 m in forest to transitional area. Understory and overstory vegetation were measured within 1m x 1m plots along each transect. Transects were located using GNSS, and the location of each plot was determined using a tape measure. Each plot location was surveyed, as was described in chapter 3. For this study, height was measured at 3 locations in each plot, left of center, center, and right of center, 50 cm into the plot. Other measurements not used in this study are described in chapter 3. Average height measurements were directly compared with height measurements from lidar to determine understory and short vegetation heights, which are representative of a post-fire environment.

In addition to field measurements, a post-processed kinematic survey (PPK), serving as a lidar elevation control, was completed by installing the GNSS on top of the roof of the vehicle (with measurement to ground surface) and driving along Highways 88 and 750 on the 31st of July 2019 within the extent of our study area.

4.2.3 Lidar Data and Processing

Spanning a period of 17 years, seven airborne lidar surveys were collected over URSA. Lidar data were collected in 2002 (Hopkinson et al., 2005), 2008 (Chasmer et al., 2016; Montgomery et al., 2019), and in 2011, 2015, 2018-2020 following approaches described in Hopkinson et al. 2013, 2016.. The details of these flights and lidar systems can be found in Table 4.1. The 2019 airborne lidar survey was acquired coincident with the collection of field data on the 29th of July 2019.

Table 4.1. Specifications of lidar flight acquisitions used in study.

Year	System Used	Flight Altitude (m above ground level)	Repetition Rate (kHz)	Scanner Frequency (Hz)	Scan Half Angle (Deg)	Average Point Density (points per m^2)	Original Vertical Projection
2002	Lidar ALTM 2050	<1200	50	36	±16	2.20	Ellipsoidal
2008	Optec 3100	1400	70	22	±25	1.81	Orthometric
2011	Lidar ALTM 3100	1800	50	23	±23	1.27	Ellipsoidal
2015	Aquaris	700	33	34	±20	1.06	Ellipsoidal
2016	Titan	1100	75	32	±32	3.06	Ellipsoidal
2018	Titan	1000	300	32	±25	4.39	Ellipsoidal
2019	Titan	1000	300	32	±25	5.9	Ellipsoidal

Post-processing of the lidar data included classification of ground and non-ground returns, quality control and removal of isolated points using TerraScan (TerraSolid Inc., Finland), on each dataset. Interpolation routines to derive lidar metrics for each data set were performed using LasTools (Rapid Lasso GmbH, Germany). However, prior to completing vegetation derivatives, any vertical and horizontal offsets that may occur between datasets were determined so as to remove errors in vegetation height derivatives. To do this, the digital elevation models (DEMs) of each data set were compared to the post-processed kinematic survey of the road surface elevation completed in 2019. Any offsets were then applied to the lidar point-clouds by either adding or subtracting the offset from each point cloud. All derivatives were first gridded to a cell resolution of 2 m, and then the mean was determined across a lower resolution 5m cells. Lidar derivatives used in this study to determine change in vegetation height included height at the 95th, and 99th percentile (m), and a digital surface model (DSM). The 2016 DEM was derived using ground-classified returns from all three lidar channels. The 2016 DEM was chosen as it was the earliest post-fire data set with sufficient overlap in all data flight years. Canopy height models (CHMs) for each respective year were based on the difference between DSM and the 2016 DEM. To remove potential influence from standing burn stems, in all post-fire data sets, heights of vegetation within the enhanced wetland classification polygons were removed if greater than 7 m. This height was chosen as within our field observations, the average maximum regeneration height of living species in the transition zones (where greatest heights of vegetation were recorded) was 7 m.

4.2.4 Quantifying Multi-temporal Height Metrics and Rate of Change Analysis

Peatlands were identified using an enhanced wetland classification map modified from Chasmer et al. (2016) to include class and form, as seen in Figure 4.2. Class and form were identified visually prior to 2011, using a hillshade of the 2008 DEM surface and historical high resolution optical imagery from the SPOT satellite. Bogs were identified as being visually disconnected with slightly upraised peat, while fens did not have the same upraised elevation and were connected often with elongated shape. Form was determined for open, shrub, and trees determined from the 2008 CHM. This enhanced classification was limited to the 2019 flight area to ensure overlapping coverage with field-validated lidar and the identified burn scar areas. The classification included 221 fens and 558 small sized bogs which could be solely identified using lidar/SPOT data, resulting in identification of 779 peatlands. Within this study area, proportional coverage of peatland area was estimated to be approximately 18% in SB56, 31% in SB11, and 36% in DB (Figure 4.2). These estimates may underestimate overall peatland area, as they were estimated using the enhanced wetland classification.

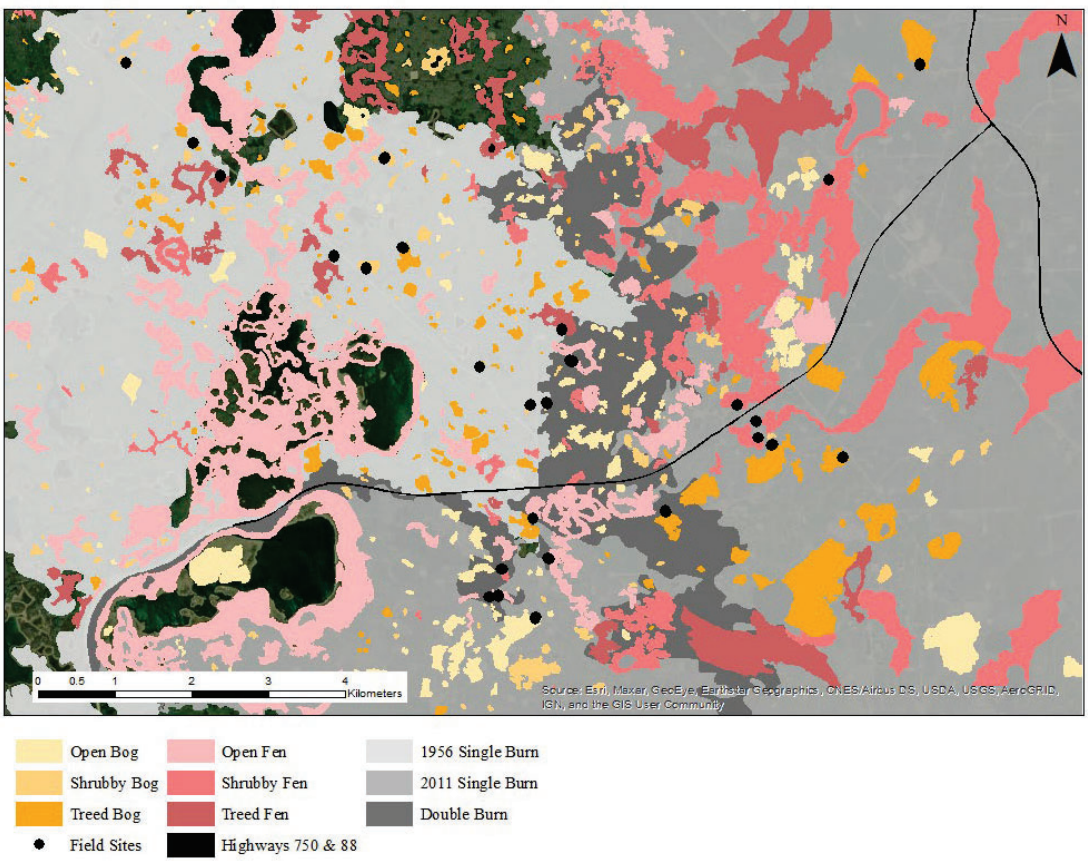

Figure 4.2. Enhanced peatland classification map, derived from Chasmer et al. (2016).

Transition zones were identified using a buffer tool and visual editing in ArcMap (ESRI, USA), limiting to 10m to maintain consistency with the methodology in Chapter 3. To first quantify the height statistics over time, the random points generator in ArcMap was used to generate ~6000 randomly distributed points across 6 strata: SB56-Bogs, SB56-Fens, SB11-Bogs, SB11-Fens, DB-Bogs, DB-Fens. Each strata received approximately 1000 points, with a minimum distance of 5 m between points. These points were used to extract lidar-derived wetland vegetation mean of the 99th height percentile at each location and were used to quantify the height metrics of vegetation within each separate data set.

Difference maps of the mean of the 99th height percentiles with cell resolution of 5 m were produced between each post-wildfire data to determine the change between each individual year that the lidar data were collected, on a pixel-by-pixel basis. The difference between years of surveys was performed on SB11 and DB sites only, as they were in early-stage regeneration post-fire. Given that the 1956 peatlands burned more than 50 years ago and based on average return interval of peatland fires (Turetsky & Louis, 2006; Wieder et al., 2009), SB56 sites were assumed to represent a mid-or intermediate stage of regeneration, and thus were not used in this analysis. The rate of change, focused on the years between 2011-2019 and 2016-2019, was calculated using the Curve Fit tool from USGS (Fox Tools, USGS, USA). The 2015 data set was excluded from this analysis as the wavelengths used in the sensor was not comparable to other data sets, such that it consistently underestimated heights. The periods 2011 to 2019 was chosen because it represented the greatest time between fire, and the greatest change period. The 2016 to 2019 were also chosen because the 2016 data set had the greatest overlap, allowing for a larger study area. The datasets were then fitted to a linear model, assuming a linear rate of growth, in the Curve Fit tool as the peatlands were still in early growth stages and were not at the stage of a sigmoidal growth curve (Hopkinson et al., 2016a). The average rate of change of vegetation per peatland was derived from the slope of the linear vegetation change raster produced from the Curve Fit tool using the Zonal Statistics function in ArcMap.

To determine if the shape of the peatland (class and form) has an influence on the rate of regeneration, perimeter and area was calculated for each wetland complex shape using the geometry calculator function in ArcMap. The ratio of perimeter to area (P:A) was calculated in R. Lower P:A values represent more compact shapes with higher perimeter relative to the areas, so there is more edge adjacent to another land cover type (often forests) than peatlands with higher P:A. A larger P:A is indicative of a more complex shape, or greater area, and may indicate other attributes, such as hydrological connectivity. A linear regression analysis was performed to compare the P:A ratios with the average rate of change per peatland. All summary statistics and regression analyses were performed in R.

4.3 Results

4.3.1 Correlating field data with lidar data

. To quantify rate of change of shrub and immature tree heights, the accuracy of lidar-derived vegetation was compared with measured vegetation height from field plot data (Figure 4.3). We show that the 1956 SB areas had the greatest correlation with an R^2 value of 0.62. DB areas and SB11 areas only have marginally lower correlations with R^2 values of 0.61 and 0.60, respectively.

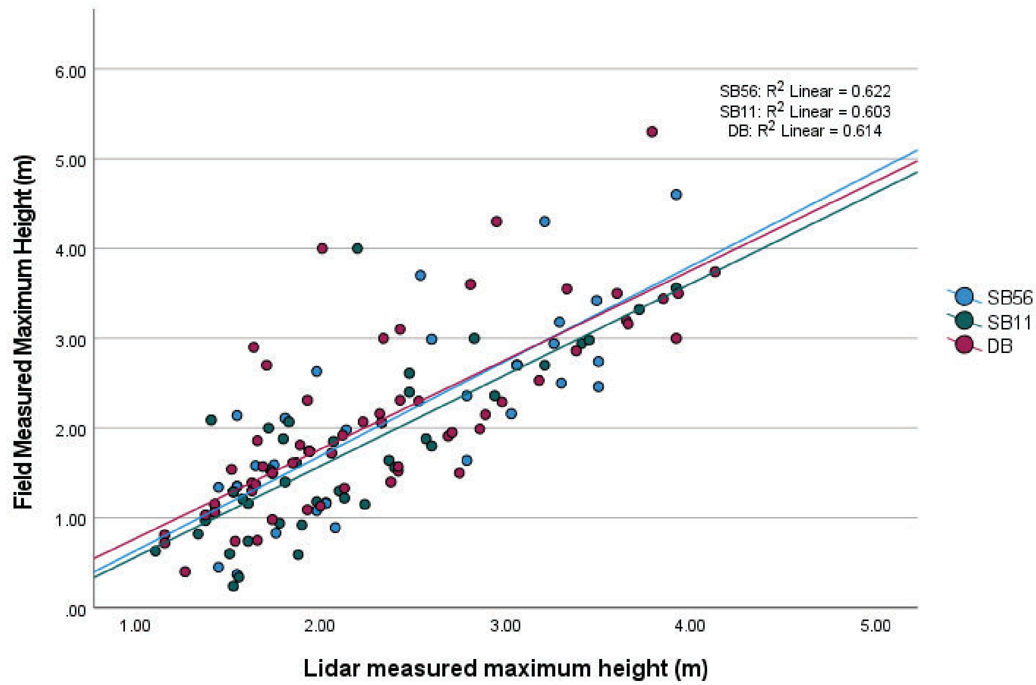

Figure 4.3. Correlation of field heights and the means of the maximum lidar measured height (derived from the 2m CHM).

4.3.2 Heights of Regenerating Vegetation Pre- and Post-fire

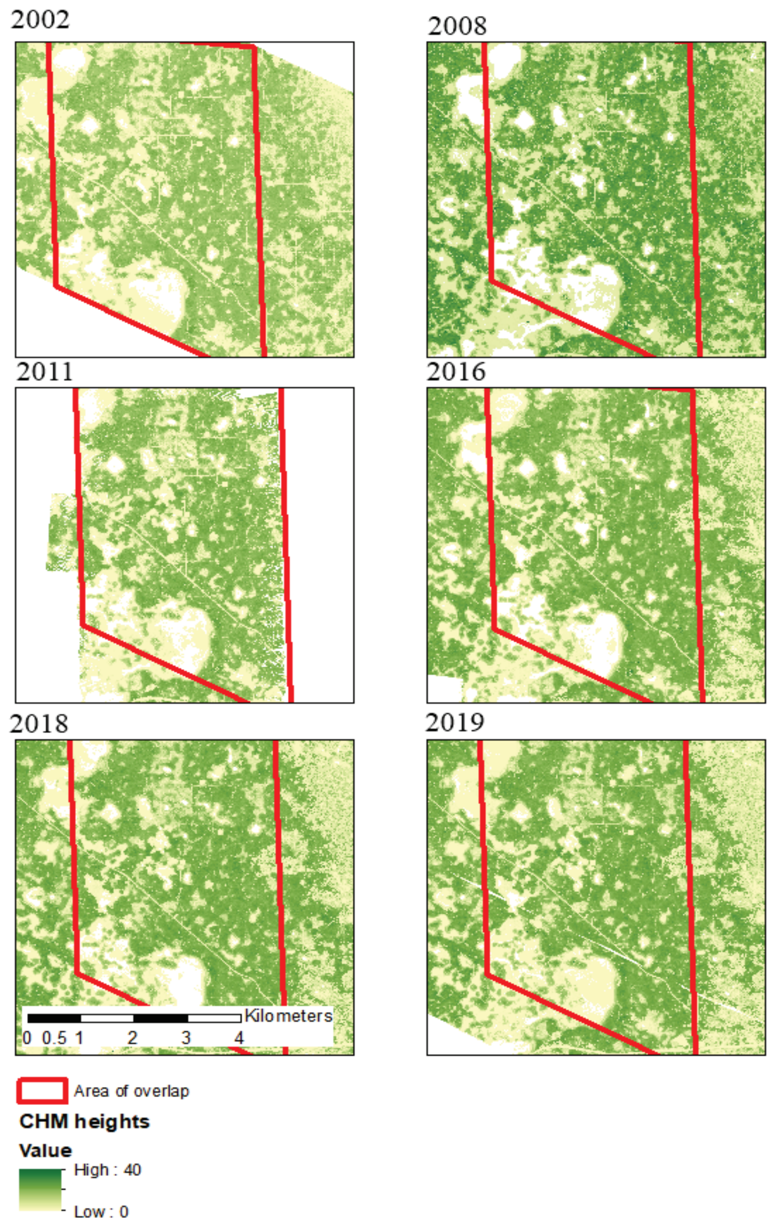

Figure 4.4. Canopy Height models per year. As each data swath had variable extents, each data set includes polygon of the area in which all data sets overlap, outlined in Red. Scale is constant between figures.

The average height of vegetation within bogs that were not burnt in 2011 (i.e., SB56 bogs and fens), was 2.6 m (SD ± 0.8), compared to fens, which averaged 2.1 m (SD

$\pm$ 1.1). We found that within bogs, the heights of returning vegetation in DB areas were significantly greater for each year post-fire than those of SB areas ($p < 0.001$ for all). Returning vegetation heights within SB-fens showed little difference to those heights in DB-fens (Figure 4.4, Figure 4.5). Mean height, as derived from the CHM (seen in figure 4.4), in SB56 areas in 2002 and 2008 was 3.0 m and 2.9 m, respectively. Old growth areas which were burnt in 2011 (becoming the 2011SB and DB sites), averaged 3.7m prior to the fire.

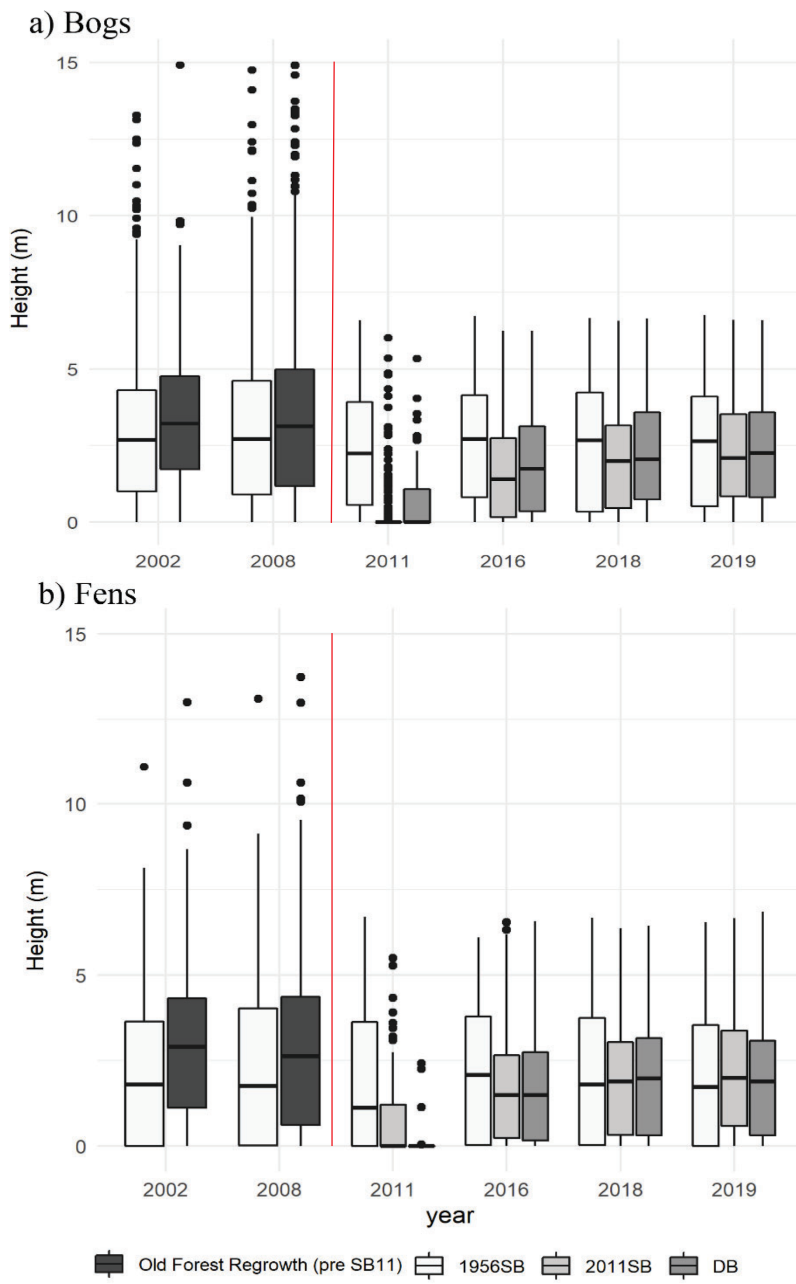

Figure 4.5. Height at the 99th percentile within a) bogs and b) fens. Old forest regrowth areas were burnt in the 2011 fire, becoming the SB11 and DB areas. Red line indicates fire event in 2011, which occurred prior to data collection in 2011.

Transitional areas within DBs were not significantly different from SB areas for each year post-fire across the broader region, corresponding with observations (Figure 3.7) in Chapter 3.

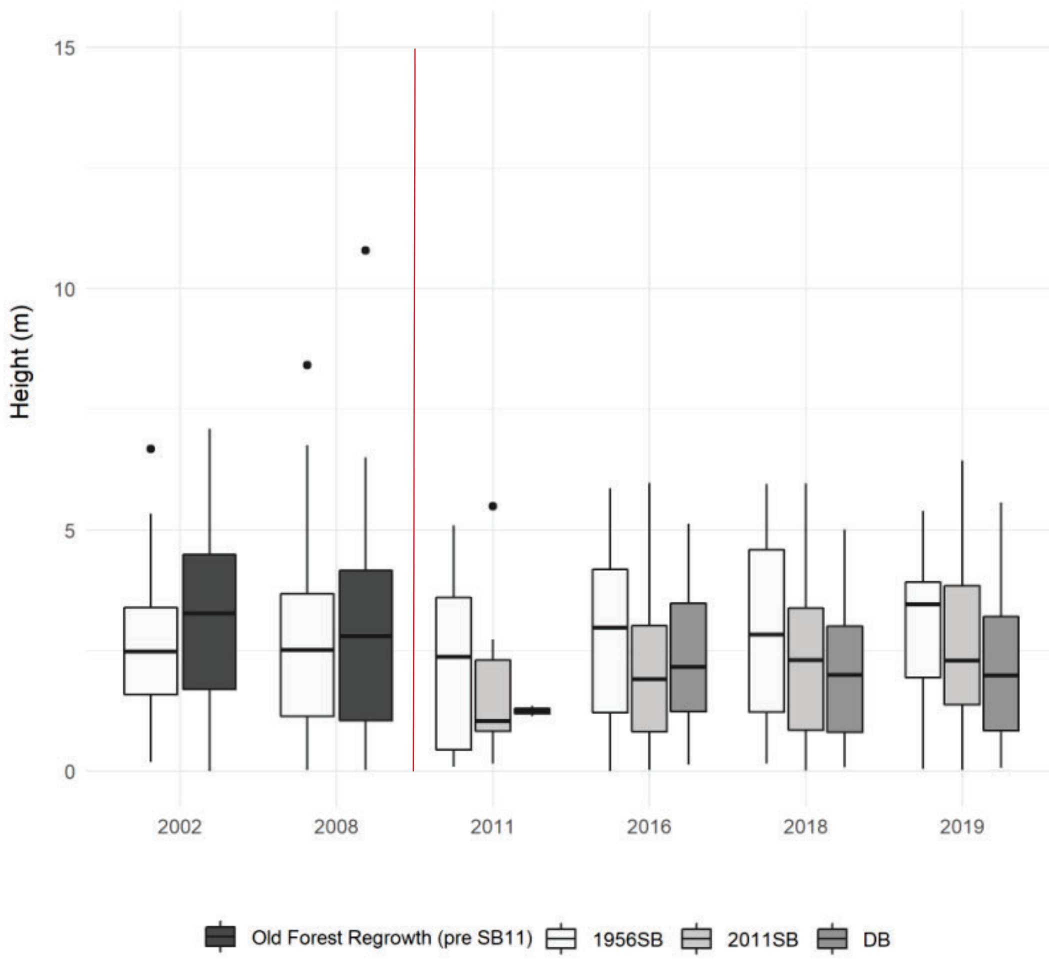

Figure 4.6. Height at the 99th percentile within transition zones across the study area. Old forest regrowth areas were burnt in the 2011 fire, becoming the SB11 and DB areas. Red line indicates fire event in 2011, which occurred prior to data collection in 2011.

4.3.3. Height differences

Vegetation heights were found to have greater absolute differences in DB areas between time increments (2011-2016, 2016-2018, 2016-2019), than SB11s, at 0.85

m/year and 0.79 m/year, respectively. In both DB and SB11 peatlands the largest differences appear between the 2011 – 2016 period averaging a total difference of 0.93 m (standard error, SE = 0.26 m) and 0.90 m (SE = 0.12 m), respectively. Assuming a linear growth between these, we would estimate the rate of growth between 2011 and 2016 to have been approximately 19 cm/year in DB and 18 cm/year in SB11 peatlands. These rates of changes were not significantly different between burn regimes. Overall differences in height between 2011 and 2019, as well as the differences in height between 2016 and 2019 are illustrated in Figure 4.7a, b.

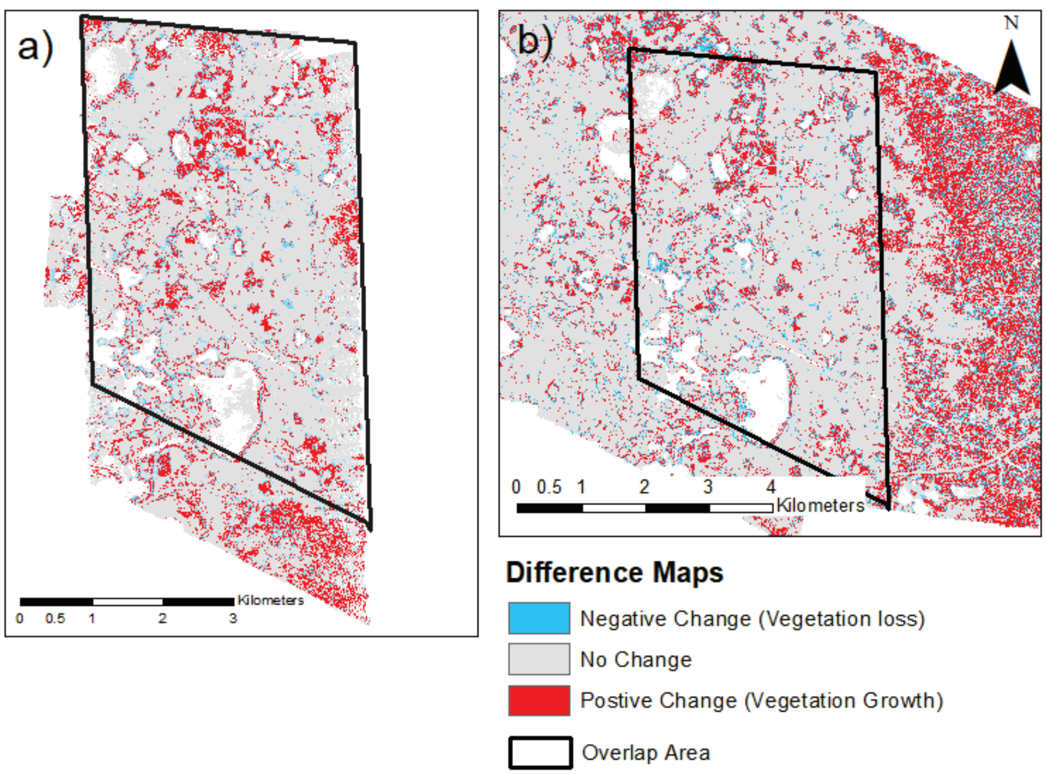

Figure 4.7 a) Difference map of heights (m) between 2011 – 2019. b) Difference map of heights (m) between 2016 – 2019.

Bogs within DB areas had greater average vegetation growth between 2011 and 2019 than those of SB-Bogs, with a mean growth of 1.6 m (SE = 0.27 m) and 1.0 m (SE = 0.28 m) respectively. In the DB especially, bogs that had been characterized as open or shrubby pre-fire, were observed to have greater growth than treed bogs. SB-bogs characterized as shrubby pre-fire tended to have the greater growth between years in comparison to treed or open bogs, though they were not significantly different. Fens within SB areas showed greater increases in height over a longer period on average than those of DB-fens, with a mean growth between 2011 and 2019 of 1.2 m (SE = 0.38 m) and 1.4 m (SE = 0.54 m). As with SB-bogs, the SB-fens typically had greater growth characterized as shrubby than those of open or treed fens, where DB-fens had significantly greater growth in open and treed fens than shrubby ($p < 0.01$) between 2011 and 2019. Adjacent transitional zones showed no significant difference between the DB and SB fire regimes in terms of returning vegetation height.

4.3.3 Growth Rates of Returning Vegetation

Using the slope of linear equation from the curve-fit tool, average rate of growth between 2011 and 2019 was calculated to be 0.16 m/year (SD ± 0.10 m/year) in DB peatlands and 0.13 m/year (SD ± 0.06 m/year) in SB peatlands (Figure 4.7a). The rate of growth in both SB and DB fens was not significantly different. For reference, peatlands within the mid regeneration stage (>60 years post-fire) underwent relatively little change, with vegetation growing at an average rate of 0.05 m/year (SD ± 0.14) in SB56 areas between 2011 and 2019. Some of these areas have net losses in growth between years, potentially indicating tree mortality.

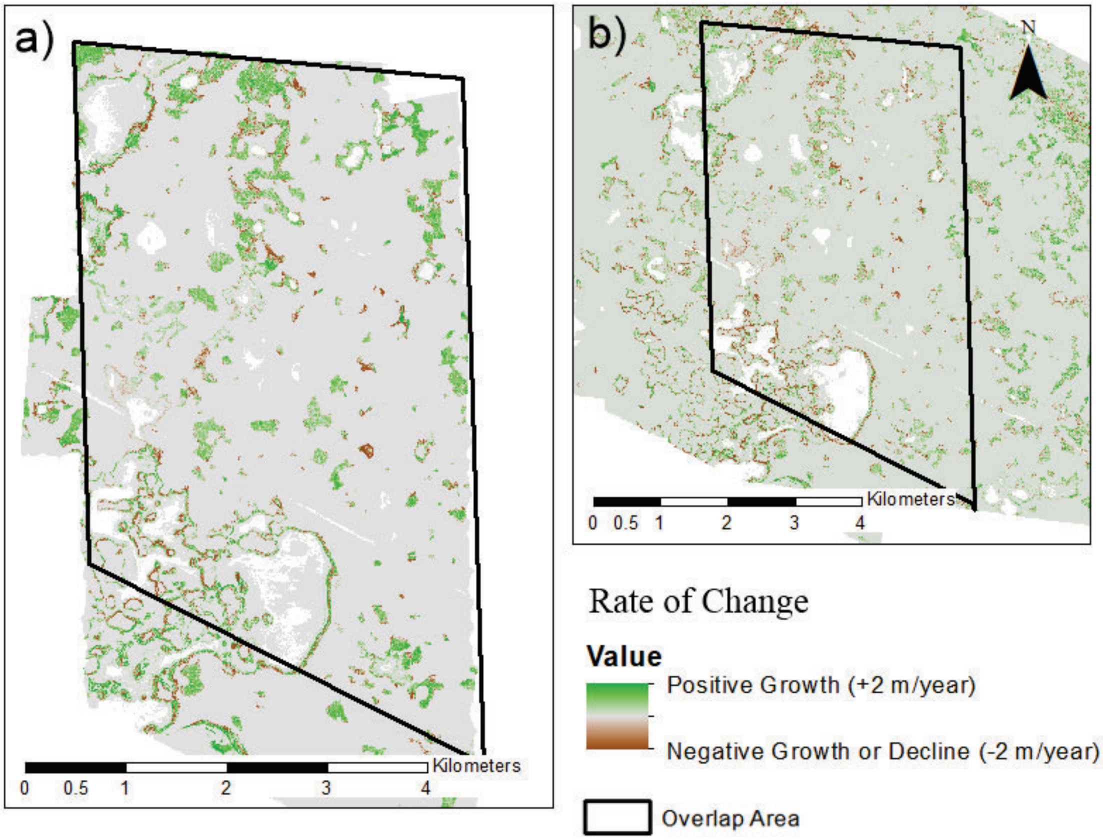

Figure 4.8 a) Rate of change (m/year) between 2011 – 2019. b) Rate of change (m/year) between 2016 – 2019. Scale is consistent between a) and b).

Rate of change was similarly calculated between 2016 and 2019 (Figure 4.8b) and was found to be similar between SB and DB peatlands, at 0.11 m/year (SD ± 0.10 m/year) and 0.15 m/year (SD ± 0.11 m/year) respectively. In both DB and SB peatlands, growth rates were greater in bogs than in fens, especially in shrubby and open peatlands. Rates of growth were greatest in the period between 2011 and 2016 for bogs and began to reduce in the period between 2016-2018 and 2018-2019 (Figure 4.9a). Within fens, average rate of growth per year was greatest between 2016 and 2018 (Figure 4.9b). On average, fens in DB areas had greater growth rates than single burns, where the opposite is true of bogs.

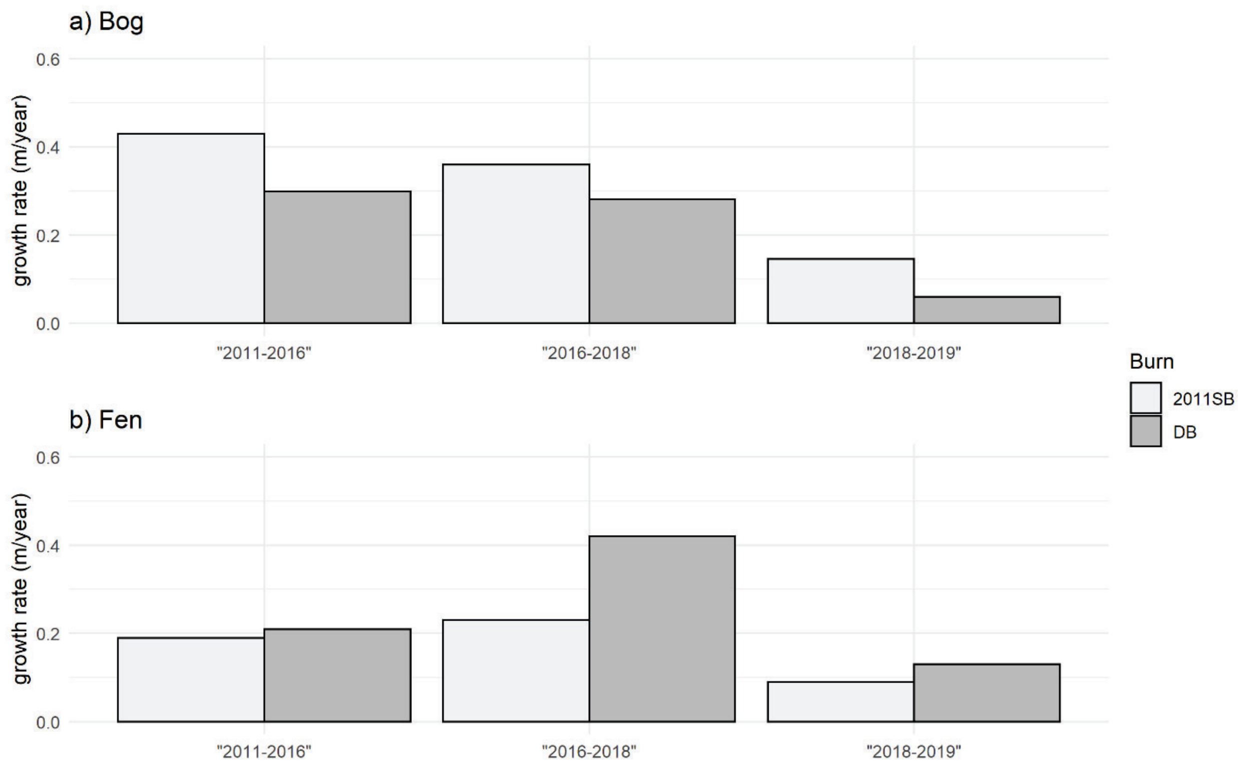

Figure 4.9 Average rate of growth per year of mean 99th height percentile at a) bogs, and b) fens.

4.3.4 Peatland Fragmentation and Growth Rates

We found no significant correlation between the growth rates and peatland fragmentation as represented by P:A ratio (r = -0.21). As peatland shape became more complex and with greater area relative to the peatland perimeter (increased P:A), rates of vegetation growth were reduced. SB-bogs and fens were found on average to have greater P:A ratios (Figure 4.10) than DB-bogs and fens. While these are not significantly different, greater variability in P:A, indicating both compact and elongated/complex peatlands may pre-dispose some of these to greater rates of vegetation growth, especially observed in fens. Additionally, SB areas were found to have a significantly greater variability (p = 0.03) in P:A ratio than DBs.

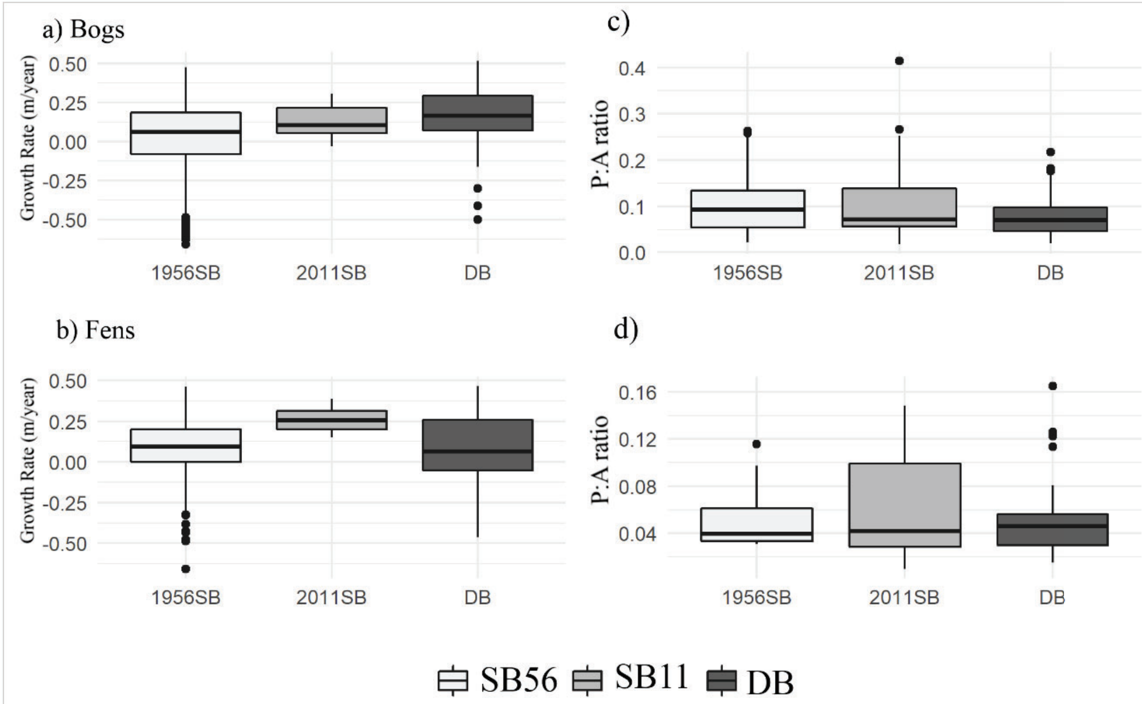

Figure 4.10. Growth rate in m/year of the 99th height percentile between 2011-2019 in a) bogs and b) fens. Perimeter:Area (P:A) ratios of c) bogs and d) fens.

4.4 Discussion

4.4.1 Growth and Rates of Change

Our results indicate that growth rates differ between DB and SB areas, though not to the magnitude we may have expected. Short interval (DB) burns on average we found to have greater differences in height per year, particularly in open and shrubby bog areas (Figure 4.4, 4.9), but there were no significant differences between their rates of growth. Fens were more productive in SB areas than DB areas, showing greater differences in height per year, but were also not found to have significantly different growth rates between fire regimes (Figure 4.4, Figure 4.5). Regeneration is a complex process that occurs over decades, especially within peatlands, so differences in rates of change may not yet be apparent, even eight years post fire.

Post-fire succession of peatlands is highly dependent on 1. Pre-fire conditions, such as the availability of a viable seed bank (Brown et al., 2010; Brown & Johnstone, 2012), recruitment potential (Whitman et al., 2019), soil conditions (Frandsen, 1997), and water-availability (Bradshaw, 1984; Benscoter, 2006; Benscoter & Vitt, 2008); and 2. Fire conditions, such as the severity of the burn, which in turn impacts the depth and extent of the burn (Rein et al., 2008). Kettridge et al. (2019) observed that with greater depths of burn, the potential for evapotranspiration loss was higher for peatlands. The DB sites in our study experienced greater growth than SB on average (Figure 4.8) which could indicate: A) drier conditions which allow for recruitment of fast-growing shrub species. These dry conditions could result from increased evapotranspiration with each subsequent fire as is seen in Kettridge et al. (2019). Alternatively, a secondary explanation could be that the areas delineated in the enhanced wetland classification used in this study, which were based on pre-fire conditions, could represent drier upland conditions. The peatland could have subsequently contracted in size, which has been shown to occur in wetlands post-fire through processes of losses of organic matter through combustion as seen in Benscoter and Wieder (2003) and as predicted under scenarios of future burn in Kettridge et al. (2015), or through enhanced shrubification into previously anaerobic wet areas as is suggested in Thompson and Waddington (2013). Peatlands in Indonesia have been demonstrated to drastically reduce in extent after fire-disturbance, through peat loss and through changes in landcover species dominance (Hoscilo et al., 2011). In such a scenario, our pre-fire peatland class may be capturing an area that post-fire no longer exists as a peatland, but rather as a drier transition zone or upland area.

Alternatively, B) Greater growth in DB peatlands could indicate less severe burns, suggesting a more intact-seed bank. There was some evidence to support an increased resilience in double burns, especially within transition zones, namely the deeper peat and decreased bulk density found within double burn transition zones in chapter 3, in comparison to SB transition zones. It is not however the intention of this thesis to specifically conclude if increasing burn frequency is likely to increase or decrease resilience and further studies would be required to form a conclusion on this matter. In chapter 3, we found that DB areas had greater proportions of shrubby and transitional upland vegetation species, suggesting that scenario "a" (that double burn peatlands are drier and therefore allow for greater variability in growth and more productive species establishment in peatlands post-fire) may be more probable. Further hydrological analysis on a larger scale, with a focus on peatland response to multiple burns in shortened time periods, would be needed to form a firm conclusion, however. SB11 peatlands were not significantly different in terms of rate of regeneration in comparison to DB, and the slight variations in growth rates could be accounted for simply by spatial variability of species distribution.

Greater rates of growth between the years of 2011-2016, in comparison to 2016-2019, were observed in both DB peatlands and SB peatlands, which is likely to be largely accounting for the recruitment of new species and re-establishment of pre-fire vegetation. Immediately post-fire vascular plants can dominate the peatlands, allowing traditionally upland species and ericaceous shrubs begin to encroach on nutrient-rich burnt wetlands (Benscoter & Vitt, 2008; Potvin et al., 2015; Depante et al., 2019), which may have accounted for the comparatively greater growth rates in 2011 – 2016 (Figure 4.8). The rate of growth appeared to slow in 2016 – 2019, where the seedlings surviving the initial recruitment stage could grow at a less rapid, but equally steady pace, as is seen in Benscoter and Vitt (2008).

Open and shrubby bogs, in both SB11 and DB sites, were found to have greater rates of growth than treed bogs. Shading from surrounding trees aids in maintaining low rates of evapotranspiration, often making open bogs comparatively drier than treed bogs, despite the greater demands on water tables from vegetation (Wieder et al., 2009). Additionally, in both SB11 and DB sites the growth rates in bogs were greater than in corresponding fens. This follows what was observed in chapter 3 where bogs were found to have taller vegetation on average fens, especially in DB areas.

4.4.2. Limitations

Time-series lidar change analysis, such as examining rates of growth over time using multiple datasets, has a unique set of challenges. As it is attempting an analysis on a 3-dimensional point cloud, difficulties can arise in multiple ways, including 1) laser pulse densities between any two point-clouds can be significantly different making direct comparisons difficult, 2) tree crowns are irregular in structure, decreasing the probability that a laser pulse can hit the same portion of a crown in different acquisitions, and 3) different flight conditions and sampling rates can create complicated inconsistencies (Hirata, 2004; Næsset, 2009; Marinelli et al., 2018; Zhao et al., 2018). The 2015 lidar data set, for example, was acquired with the Aquaris lidar system, which as a bathymetric system, can under-estimate heights, as the pulses tend to go lower into the canopy. In Figure 4.7, the 2015 data set was removed from the analysis as the results were not as directly comparable to the other lidar datasets. Offsets were adjusted for in our study, and averaging heights to a lower resolution may have helped mitigate some of these difficulties, but these inherent difficulties are limitations to using time-series lidar as a remote sensing tool. In addition to the aforementioned difficulties with time-series lidar, lacking coincident ground data can make calibrating the lidar data sets infeasible (Zhao et al., 2018). The 2019 lidar survey was conducted coincident with a field campaign, and a kinematic survey of a paved road (fixed structure) did help calibrate the data sets used in this study, however, errors may have propagated through.

While all seven of the datasets used in this study did have some degree of overlap, the 2011 data set, which was surveyed immediately post-fire, had the least about of overlap with any of the other lidar surveys. This can be seen in Figures 4.1.c and is the basis for performing the change and growth curve analysis in Figure 4.7 and 4.8 over multiple timeframes. To some degree the study is also limited by the time frame between fires, and the ecological timeframe of wildfire return. Peatlands have long periods between fires, making long-term monitoring a challenge, especially considering how relatively new of a technology lidar, and most remote-sensing products are. Since the 2011 burn was relatively recent, within the context of the 100-120 year expected return interval (Wieder et al., 2009), it is possible that it is simply too early in the peatlands recovery stages to see true impacts of a reduced return interval on the rates of change. Additionally, our lidar datasets were only analyzed over the polygons of the peatlands, and a 10 m buffer surrounding, in the wetland classification. Thus, it is possible that the edges were slightly under- or over-estimated. In cases where the edges were over-estimated, averaging to a lower resolution may have resulted in edge effects. As the raster pixels were aligned in all the lidar metric outputs, the edge effects could have produced a consistent bias. While it is possible that this analysis over-estimated the area of the peatlands and transition zones, it is more likely that it is an under-estimate of the peatlands within the area. Peatlands vary greatly in size, and smaller peatlands, especially treed peatlands which spectrally and structurally can be similar to forested areas were more likely to have been missed.

4.5 Conclusions

We succeeded in quantifying the rate of early post-fire vegetation growth within peatlands and their adjacent transition zones in short-interval (DB) and long-interval return (SB) fire areas using multiple lidar data collections since the fire. Double Burn areas were found to grow at slightly greater rates than SB areas on average and appeared to have more impact on the increase growth rates of fens than of bogs. However, bogs overall were have found to have greater growth between 2011 and 2019. This supports what we found in chapter 3 and may indicate a shift in the vegetation composition of bogs. The perimeter to area ratio did not have any meaningful correlation with rate of growth, however, we did observe that fens generally had greater variability in SBs than DBs. Overall, our findings suggest that the reduced fire-intervals have some effects on the growth of peatlands, but it may be too early to see the full magnitude of these effects.

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5. CONCLUSIONS

5.1. Summary of Research

This research examined the impacts of increased fire frequency on boreal peatlands in Alberta, Canada, using a combination of field data and lidar remote sensing methods. The study was motivated by an interest in understanding if and how peatlands would recover differently in areas where fires had occurred in a shorter than expected time frame.

In chapter three we combined methodologies of *in situ* field sampling and lidar remote sensing and time-series analysis, which helped address the first research objective, in determining if there were differences in soil and vegetation characteristics in DBs vs SB peatlands and transition zones. Our findings suggested that reducing the interval between wildfire return had impacted soil bulk density, depth of organic matter and soil pH, all of which are key influences on hydrological activity and vegetation composition within peatlands (Thompson & Waddington, 2013; Thompson et al., 2017a). We also found greater variability of tree and shrub species composition and proportions, as well as taller vegetation and greater proportional cover of trees/shrubs in short interval return areas compared with longer return interval fire. The use of lidar remote sensing across 120 peatlands provided additional confidence in field measured results and confirmed that a broader range of peatlands that had experienced DB were susceptible to increased shrubification. The key take-away from this analysis is that short-return interval peatlands (DBs) are effectively losing vegetation attributes and moving towards drier upland and transitional characteristics. This work gives an initial insight into the future trajectory of peatlands in response to increasing fire frequencies due to climate change.

Chapter four addressed the second research objective, in determining differential rates of growth post-fire between SB and DB peatlands and transition zones. Using seven airborne lidar data sets, we examined the changes in peatlands and transition zones over a 17-year period, with a focus on the post-2011 fire recovery. We found that while DB areas had slightly greater rates of growth than SB areas, theses results were not significant. Double Burn areas were found to grow at slightly greater rates than SB areas on average and appeared to have more impact on the increase growth rates of fens than of bogs. Bogs, overall, were found to have greater growth between 2011 and 2019. Overall, our findings suggest that the reduced fire-intervals did have some effects on the rate of growth of peatlands in DB vs SB areas, but it may be too early to see the full magnitude of these effects.

We had hypothesized that the increasing fire frequency would impact the growth and returning vegetation of peatlands. In both chapters three and four, we found that peatlands and transition zones. DBs had different characteristics than peatlands and transition zones in SB areas. Our findings suggest that increasing fire frequency does impact the soil characteristics, vegetation structure and species distribution, as well as the rates of growth in peatlands within the Boreal Plain Ecozones. Vegetation within double burn peatlands were seen to have greater increases in height than single burn peatlands, supporting our findings in chapter three. Increased growth rates in DB peatlands could indicate either 1) drier conditions, increasing the growth productivity within the peatlands, or 2) shifts towards taller, more upland dominant deciduous vegetation within the peatlands. The later of these possibilities is supported by our findings in chapter three and corroborates what was found in Whitman et al. (2019) and predicted in Kettridge et al. (2019), in that there are shifts in returning vegetation to increased deciduous upland and transitional dominant species. Shifts in vegetation composition within peatland ecosystems is an altering of the fire fuel type, which has implications on the resiliency of the peatland to future fire regimes. Further studies are recommended to specifically address how peatland resiliency is impacted by increased fire frequency within peatlands.

5.2 Limitations and Future Directions

There are some limitations to this research. First, our data relies on the accuracy of historical spatial fire polygons. We assume that the areas delineated in the Alberta Wildfire (2018) data sets are correct and do in fact encompass the appropriate burn areas. However, in some instances, field observations indicated that the chosen peatland sites had been burned less severely or at slightly different locations than were suggested in the spatial data set. Additionally, this research does not address burn severity, which can greatly impact the trajectory of recovery in an ecosystem.

This thesis examines a relatively short period of fire recovery, and may not fully capture the scope of impacts from reduced return intervals. Long-term monitoring of peatlands that have undergone multiple fires would be beneficial, especially considering the important role of peatlands. As climate-mediated air temperatures continue to increase in northern/central boreal ecosystems, it is critical to understand how peatlands are changing in response to increases in fire frequency. This is important because they are a critical component of the global carbon-climate feedback system due to broad area, ability for carbon sequestration, and their role in climate-mediation by cooling due to ET loses (Helbig et al., 2017). Additionally, performing an analysis of the impacts of burn severity within the context of increasing fire frequency, on peatland recovery post-fire would provide insights that more fully address the predicted increases in fire frequency, severity, and extents.

5.3. Concluding statement

Air temperatures are predicted to increase globally over the next century, and with it fire activity (Flannigan et al., 2013). The purpose of this thesis was to determine if increased fire frequency impacted peatland characteristics and growth in a post-fire context. The results of this study suggested that reduced fire return intervals within peatlands can shift the ecosystem’s characteristics. Presented here-in was evidence that double burn peatlands show increased shrubification across a broader landscape, and increased growth rates post-fire in comparison to single-burn areas. This research provides a unique insight into the consequences of climate-induced increases in fire frequency, and how peatland succession may change in the future.

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