# ECOHYDROLOGICAL THRESHOLDS TO HIGH PEAT BURN SEVERITY

# ECOHYDROLOGICAL THRESHOLDS TO HIGH PEAT BURN SEVERITY: IMPLICATIONS FOR PEATLAND WILDFIRE MANAGEMENT

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**ABSTRACT**

Northern peatlands represent a globally significant carbon stock, equating to almost one-third of the world's organic soil carbon. The largest areal disturbance to northern peatlands is wildfire where carbon loss, through peat smouldering combustion, is highly variable. The tightly-coupled ecohydrological nature of peatlands results in autogenic feedbacks and the occurrence of threshold behaviour. High depth of burn has been evidenced in black spruce dominated peatlands in the sub-humid Boreal Plains ecozone of Alberta, Canada so this was chosen as the area of study. A landscape-scale assessment of peat hydrophysical properties found that peat smouldering combustion vulnerability was greatest at stand-age > 80 years, in coarse/heterogeneous hydrogeological settings, and in peatland margins compared to peatland middles. In combination, and when exposed to a climatic water deficit, we found that these drivers of cross-scale variability could lead to high peat burn severity. Assessment of a partially-drained and burned peatland enabled the identification of a black spruce basal diameter threshold that corresponded to the occurrence of high peat burn severity. We suggest that the above-ground fuel load threshold could occur due to the initiation of a self-reinforcing feedback by anthropogenic disturbance or climate change. Moreover, surpassing a peat burn severity threshold can cause the breakdown of an important feedback that limits evaporation losses post-fire, likely leading to further carbon losses through increased decomposition rates and/or ecosystem regime shift. We found that although peat moisture content was increased by fuel modification treatment, combustion carbon losses were greater in fuel-treated areas compared to the control because of the addition of mulch (wood) to the surface. Hence, peatland wildfire management that

integrates the modification of above- and below-ground fuels, considers ecohydrological thresholds, and drivers of cross scale variability, is required to effectively reduce the risk of high peat burn severity in black spruce dominated peatlands.

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**PREFACE**

This Ph.D. dissertation is composed of six chapters. Chapter 1 provides a literature review of the thesis topic and highlights key research needs to be addressed by the thesis. Chapters 2, 3, 4 and 5 constitute the main body of the dissertation; each of these four chapters is written as a journal article for peer-review. Chapter 2 identifies drivers of cross-scale variability in peat burn severity across black spruce dominated peatlands in the Boreal Plains ecozone of Alberta, Canada. Chapter 3 uses a partially-drained, burned peatland to investigate the relations between black spruce stand characteristics and peat burn severity. Chapter 4 assesses the effects of peat burn severity on peatland ecosystem resilience (post-fire recovery) and the stability of remnant peat carbon stocks. Chapter 5 tests novel wildfire management, by fuel modification, in a black spruce dominated peatland, and evaluates the potential to reduce peat burn severity using an experimental fire. Finally, Chapter 6 provides a summary of the key findings from the thesis and suggests areas of focus for future research.

## DECLARATION OF ACADEMIC ACHIEVEMENT

In addition to the work presented here I have also had the pleasure of being involved in the publication of two other peer-reviewed journal articles that are relevant to this project.

During the 2016 field season I worked with MSc. students Kristyn Mayner and Rebekah Ingram to collaboratively collect field data from a chronosequence of peatlands across the Boreal Plains ecozone of Alberta. I contributed substantially to the writing of each manuscript which led to co-authorship on the following articles:

Mayner, K. M., Moore, P. A., Wilkinson, S. L., Petrone, R. M., & Waddington, J. M. (2018). Delineating boreal plains bog margin ecotones across hydrogeological settings for wildfire risk management. *Wetlands Ecology and Management*, *26*(6), 1037-1046. https://doi.org/10.1007/s11273-018-9636-5

Ingram, R. C., Moore, P. A., Wilkinson, S. L., Petrone, R. M., & Waddington, J. M. (2019). Post-fire soil carbon accumulation does not recover boreal peatland combustion loss in some hydrogeological settings. *Journal of Geophysical Research: Biogeosciences, 124(4)*, 775-788. https://doi.org/10.1029/2018JG004716

The main body of this thesis is contained in Chapters 2, 3, 4 and 5 which each represent a journal article that is published, or submitted, to a peer-reviewed academic journal.

***Chapter 2:***

Wilkinson, S. L., Moore, P. A. & Waddington, J. M. Assessing drivers of cross-scale variability in peat smouldering combustion vulnerability in forested boreal peatlands. *Frontiers in Forests and Global Change*. *Accepted.*

Sophie Wilkinson (dissertation author) is the main researcher, first author and corresponding author of this paper. Sophie Wilkinson collected and processed the field data and samples with assistance from Kristyn Mayner, Rebekah Ingram and Cameron McCann. Data analysis and writing was undertaken by Sophie Wilkinson with insight and guidance from Dr. Moore and Dr. Waddington.

***Chapter 3:***

Wilkinson, S. L., Moore, P. A., Flannigan, M. D., Wotton, B. M., & Waddington, J. M. (2018). Did enhanced afforestation cause high severity peat burn in the Fort McMurray Horse River wildfire? *Environmental Research Letters*, *13*(1), 014018.

Sophie Wilkinson (dissertation author) is the main researcher, first and corresponding author of this paper. Sophie Wilkinson and Dr. Waddington scouted the research area and designed the research. Sophie Wilkinson collected field data with assistance from Cameron McCann. Data analysis was conducted by Sophie Wilkinson with assistance from Dr. Moore. Dr. Flannigan, Dr. Wotton, Dr. Moore and Dr. Waddington edited the manuscript and guided the discussion.

***Chapter 4:***

Wilkinson, S. L., Verkaik, G. J., Moore, P. A. & Waddington, J. M. (2019). Threshold peat burn severity breaks evaporation-limiting feedback. *Ecohydrology*.

Sophie Wilkinson (dissertation author) is the first and corresponding author of this paper and led the research design. Field sampling was conducted by Sophie Wilkinson, Greg Verkaik and Patrick Deane. Laboratory experiments were conducted by Greg Verkaik and Ryan Threndyle. Data analysis was undertaken by Sophie Wilkinson and Greg Verkaik, and the manuscript was written by Sophie Wilkinson with guidance from Dr. Moore and Dr. Waddington.

***Chapter 5:***

Wilkinson, S. L., Moore, P. A., Thompson, D. K., Wotton, B. M., Hvenegaard, S., Schroeder, D., & Waddington, J. M. (2018). The effects of black spruce fuel management on surface fuel condition and peat burn severity in an experimental fire. *Canadian Journal of Forest Research*, *48*(12), 1433-1440.

Sophie Wilkinson (dissertation author) is the first and corresponding author of this paper. Research was designed collectively by Dave Schroeder, Steven Hvenegaard, Dr. Thompson and Dr. Waddington. Field data were collected by Kristyn Mayner and Rebekah

Ingram, as well as Max Lukenbach, Nick Kettridge and members of the Alberta Wildfire Fuel Inventory crews. Sophie Wilkinson conducted data analysis and wrote the manuscript with guidance and assistance from Dr. Moore, Dr. Wotton and Dr. Waddington.

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| | |
|---|---|
| Basal diameter | BD |
| Boreal Plains | BP |
| Carbon | C |
| Diameter at breast height | DBH |
| Depth of burn | DOB |
| Gravimetric water content | GWC |
| Heavily-drained | HD |
| Hydrogeological setting | HS |
| Hydrophobicity-evaporation feedback | HEF |
| Moderately-drained | MD |
| Ring width | RW |
| Specific yield | Sy |
| Time-since-fire | TSF |
| Water drop penetration time | WDPT |
| Water table depth | WTD |
| Wildland-industry interface | WII |
| Wildland-urban interface | WUI |
| Volumetric water content | VWC |
| Undrained | UD |

# CHAPTER 1

## INTRODUCTION

### 1.1 Literature Review

The following literature review highlights the state of the science of peat burn severity in northern peatlands, in particular focussing on black spruce dominated peatlands in the Boreal Plains ecozone of Alberta, Canada. The review highlights several research needs; many of which are addressed within this thesis.

#### 1.1.1 Northern peatlands

Peatlands, wetlands that have accumulated at least 0.4 m of organic matter (National Wetlands Working Group, 1997), develop when saturated ground is colonised by peat-forming mosses or sedges, and primary production exceeds losses such as decomposition and combustion, over long periods of time. Peatlands are most abundant in northern latitudes (above 45°N), where they have developed following deglaciation of the region (Halsey et al., 1998; Kuhry and Turunen, 2006), and are sustained by a suite of autogenic hydrological feedbacks that act to maintain high (shallow) water table positions (Waddington et al., 2015). Stable, high water tables limit decomposition rates (Clymo, 1984), vascular vegetation growth (Belyea and Baird, 2006) and combustion vulnerability (Benscoter and Wieder, 2003; Lukenbach et al., 2015). Currently, northern peatlands store approximately 455-547 Pg C (Yu et al., 2010) which equates to almost one-third of the world's organic soil carbon pool (Gorham, 1991), and is an amount similar to atmospheric carbon. Hence, the stability of the northern peatland carbon stock has global significance.

Peatlands are complex adaptive systems with tightly coupled ecological, biological and hydrological processes (Belyea and Baird, 2006; Waddington et al., 2015). This results in the prevalence of non-linear responses to internal and external forcings (*e.g.* Belyea and Malmer, 2004) and the occurrence of threshold behaviour (*e.g.* Granath et al., 2010). Therefore, the concepts of ecohydrological feedbacks and thresholds are considered throughout this thesis. Due to a multitude of negative (self-regulating) feedbacks, peatlands generally exhibit strong ecosystem resilience, persisting in landscapes that undergo inter-annual and long-term climatic variability (Kuhry and Turunen, 2006), cyclical disturbance (Kuhry et al., 1993) and anthropogenic disturbance (Turetsky et al., 2002). However, more recently, the ever-increasing and compounding pressures of anthropogenic disturbances, climate change and intensifying natural disturbance regimes, have led to concern over the resilience of northern peatland ecosystems (Ise et al., 2008; Turetsky et al., 2011a; Kettridge et al., 2015).

### 1.1.2 Peatland wildfire disturbance

Wildfire represents the largest areal disturbance of peatlands in western boreal Canada (Turetsky et al., 2002). Dependent on the connectivity to large-scale groundwater systems, water chemistry, and hydroclimate, different types of peatlands can co-exist or dominate a region (Zoltai and Vitt, 1990). In the Boreal Plains (BP) ecozone of Alberta, Canada (Figure 1.1), treed/forested peatlands (classified as bogs or fens) are the most common peatland type (AMBI, 2016). Specifically, bogs cover ~20 % of the land area (Turetsky et al., 2004) and are characterised by a black spruce (*Picea mariana* Mill. BSP) stand with a *Sphagnum* and feather moss dominated ground cover (Beckingham and Archibald, 1996), and they tend to have limited connectivity to large-scale groundwater systems. Black spruce dominated peatlands (bogs) are highly-susceptible to wildfire disturbance due to substantial above-ground fuel loads (Wieder et al. 2009; Johnston et al., 2015) that do not differ significantly from upland forest fuel loads (Thompson et al., 2019). Hence, they are affected by wildfire disturbance at a similar return interval to upland forests in the BP; on average once every 120 years (Turetsky et al., 2004). Moreover, the BP has a sub-humid climate whereby potential evapotranspiration generally exceeds precipitation on an annual basis and peatlands exist at the edge of their climatic limit, often under a water deficit (Woo and Winter, 2003; Devito et al., 2012). In light of their high areal coverage and vulnerability to wildfire disturbance, this thesis focuses on black spruce dominated peatlands in the BP ecozone. To capture the variability of black spruce dominated peatlands in the BP, 26 sites were studied in detail (Figure 1.1), and two other sites near Fort McMurray and Red Earth Creek, respectively, were also studied for this thesis project.

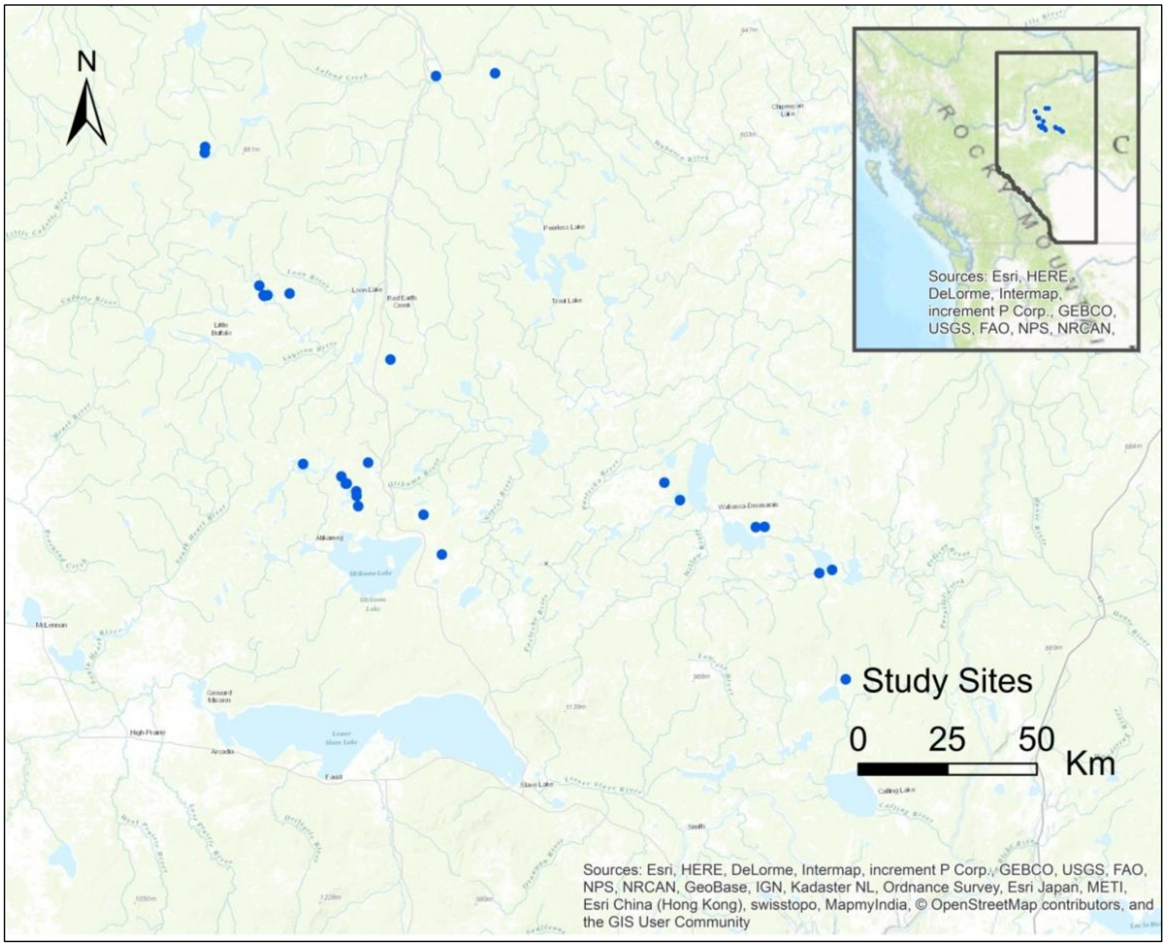


**Figure 1.1** Overview of study area in Alberta, Canada showing the 26 sites used for the landscape-scale assessment (blue polygons).

Natural stand-replacing wildfire is crucial for forest renewal across boreal regions and peatlands tend to be generally resistant to high peat burn severity when this occurs. Average peat carbon loss is estimated at 2-5 kg C $m^{-2}$ (Benscoter and Wieder, 2003), and is generally recovered within the first 40 years following wildfire (Wieder et al., 2009). However, recent research has highlighted examples of high peat burn severity (Turetsky et al., 2011b; Hokanson et al., 2016) where carbon loss estimates are up to 85 kg C $m^{-2}$, equating to thousands of years of carbon sequestration (Lukenbach et al., 2015). The variability in the severity of peat combustion is a leading source of uncertainty in boreal wildfire carbon emissions (de Groot et al., 2009) and a major challenge for regional wildfire management (Flannigan et al., 2013; 2016). Given that the variability of peat burn severity also impacts the severity of smoke and particulate matter emission associated with hazardous air quality (Shaposhnikov et al., 2014) there is a growing research need to assess the drivers of this variability.

### 1.1.3 Peat smouldering combustion

Peat typically burns by smouldering combustion (Frandsen et al., 1987); a flameless form of combustion that can be sustained under low oxygen conditions (Rein et al., 2008). The hydrophysical properties of peat are fundamental to the thermodynamic reaction of smouldering combustion (Frandsen, 1987; 1997). Specifically, the ratio of energy sink to fuel source in peat can be approximated by the ratio of volumetric water content to peat bulk density (*i.e.* gravimetric water content (GWC) g $g^{-1}$) (Benscoter et al., 2011). When the energy output from fuel combustion outweighs the energy required to burn off moisture and raise peat to combustion temperature (~300°C) smouldering combustion can propagate downwards through the peat profile (Benscoter et al., 2011), hence lower water content and/or high density are associated with higher peat burn severity and both relations exhibit threshold behaviour (Rein et al., 2008; Benscoter et al., 2011).

Peat burn severity is measured as depth of burn (DOB); the difference between the pre-fire and post-fire surface, where the pre-fire surface can be directly measured or estimated using adventitious roots (Kasischke and Johnstone, 2005) and unburned indicators (Mack et al., 2011). High peat burn severity has been documented in disturbed (Turetsky et al., 2011b) and undisturbed (natural) peatlands (Hokanson et al., 2016), including DOB measurements of over 1 m (Lukenbach et al., 2015). Comparatively, typical, low peat burn severity corresponds to 0.01 – 0.05 m DOB (Benscoter and Wieder, 2003). As such, the variability in peat carbon loss from black spruce dominated boreal peatlands has been shown to span two orders of magnitude, from <1 kg C $m^{-2}$ to 85 kg C $m^{-2}$ (Turetsky et al., 2011b; Lukenbach et al., 2015; Hokanson et al., 2016). More research is required to better understand the key processes controlling peat burn severity, and enable rapid assessments of peat smouldering vulnerability across the landscape.

### 1.1.4 Variability in peat burn severity

Peat burn severity is influenced by a multitude of factors, such as those that directly impact peat hydrophysical properties, *e.g.* decomposition/densification (Frandsen, 1997; Benscoter et al., 2011), those that affect peat drying rates such as species-dependent moisture retention (Benscoter and Wieder, 2003; Shetler et al., 2008), and also ecosystem and landscape-scale factors that control the exposure to water table drawdown (Lukenbach et al., 2015, Hokanson et al., 2016; 2018) and increased evapotranspiration losses (Kettridge et al., 2013). Hence, peat smouldering variability is a truly cross-scale issue, requiring both detailed small-scale analysis and a landscape-scale perspective to fulfil the research need. Consideration should also be made of the interactions between drivers of variability, and their interaction with climate change, which could cause the initiation of positive feedbacks (Ise et al., 2008) leading to threshold behaviour, overwhelming the capacity of ecosystem resilience (*i.e.* the ability of an ecosystem to recover pre-disturbance function) (Scheffer and Carpenter, 2003).

### 1.1.5 High peat burn severity and ecosystem resilience

Although peatlands generally show ecosystem resilience to high peat burn severity, recovering pre-fire vegetation and peat carbon combustion losses within the average fire return interval (Benscoter and Vitt, 2008; Wieder et al., 2009), high-extreme peat burn severity may be a detriment in ecosystem recovery (Kettridge et al., 2015). Smouldering combustion removes the upper moss/peat layer and exposes higher density peat (Sherwood et al., 2013), which is more sensitive to water table change (Thompson and Waddington, 2013) and low soil-water pressures (< -100 mb) that limit the recolonization of keystone peat-forming *Sphagnum* mosses (Price and Whittington, 2001). Kettridge et al. (2015) found that a black spruce dominated peatland that underwent drainage and wildfire disturbance experienced high peat burn severity (mean DOB = 0.25 m), and pre-fire ecosystem function was not recovered. Dominant vegetation type changed from moss-dominated to shrub/grass-dominated and this constituted an ecosystem regime shift to a non-carbon accumulating ecosystem with increased wildfire frequency compared to the pre-fire ecosystem (Kettridge et al., 2015). An increased fire frequency would further degrade the remaining carbon stock, similarly, post-fire drying, through increased evaporation could further degrade remnant carbon stocks (Kettridge et al., 2017).

Peat, like other soils, tends to develop water repellency due to fire (DeBano, 2000; Moore et al., 2017), although this is likely detrimental for moss recolonization, the hydrophobic layer has been found to act as evaporative cap and reduce water losses in post-fire peatlands (Kettridge et al., 2014; 2017). This is a critical feedback for ecosystem resilience, especially in sub-humid regions where evaporative demand is high (Kettridge et al., 2017). However, there is concern that this feedback is dependent on peat burn severity, and that exposure of underlying *Sphagnum* peat may increase evaporative losses, exposing the remnant carbon stock to degradation and increasing the likelihood of ecosystem regime shift (Kettridge et al., 2019). Moreover, since climate change is predicted to increase the severity and duration of droughts, increase evaporative demand and, in turn, increase the length and intensity of the boreal wildfire regime (Flannigan et al., 2013; Wang et al., 2015), the limits of boreal peatland ecosystem resilience will likely be tested in the coming century. Hence, there is a research need to test thresholds of peat burn severity and the ability of peatlands to recover from such disturbance.

### 1.1.6 Peatland wildfire management

At a regional scale, the wildland-urban interface (WUI) and wildland-industry interface (WII) are affected by high peat burn severity due to smoke pollution, hazardous air quality, and over-wintering fires that cause considerable challenges for wildfire managers (*e.g.* Shaposhnikov et al., 2014), not to mention the potential loss of critical habitat, biodiversity and ecosystem services. Because of these detrimental impacts, and the expansion of the WUI and WII across the boreal, there is an immediate need to develop and test wildfire management techniques to limit peat burn severity in these areas. Fuel treatment is one component of FireSmart Canada's approach to community wildfire risk mitigation. The primary aim of fuel treatment is to reduce forest fuel ignition potential, reduce fire intensity, and aid in the efficacy of fire suppression efforts, whilst minimizing the socioeconomic impacts of wildfires (Hirsch et al., 2001). In Canada, the testing and implementation of fuel modification treatments is usually limited to upland forest stands (Agee and Skinner, 2005; Hudak et al., 2011; Stephens et al., 2012), therefore, there is little research into the implementation or efficacy of such techniques in fire-prone black spruce dominated peatlands.

## 1.2 Objectives

The literature review has revealed that black spruce dominated peatlands in the BP ecozone of Alberta, Canada, are vulnerable to wildfire disturbance that may result in high severity peat burn. The review has identified that high severity peat burn through the propagation of smouldering combustion causes extensive carbon loss and smoke pollution, is a substantial resource-draw on wildfire management, and can affect the long-term stability of northern peatland ecosystems. Moreover, high peat burn severity increases risks to communities at the WUI and WII. In light of the aforementioned research gaps surrounding the controls, identification, and management of high severity peat burn in black spruce dominated peatlands, this thesis has the following objectives:

1) Assess the cross-scale drivers of variability in peat smouldering combustion across the BP ecozone
2) Identify stand-level thresholds of high peat burn severity
3) Evaluate burn severity thresholds of hydrological feedbacks that affect peatland resilience
4) Test peatland fuel modification treatments to assess their efficacy of reducing peat burn severity

# CHAPTER 2

## ASSESSING DRIVERS OF CROSS-SCALE VARIABILITY IN PEAT SMOULDERING COMBUSTION VULNERABILITY IN FORESTED BOREAL PEATLANDS


## ABSTRACT

Wildfire represents the largest areal disturbance of forested boreal peatlands and the spatial variability in the severity of these peat fires is both a leading source of uncertainty in boreal wildfire carbon emissions and a major challenge for regional wildfire management. Peat smouldering can emit large quantities of carbon and smoke to the atmosphere, and therefore can contribute to hazardous air quality. The wildland-industry interface and wildland-urban interface are both extensive across the sub-humid boreal plains (BP) ecozone where one-third of the area is covered by peatlands. As such, there is a growing research need to identify drivers of variability in smouldering combustion. This study uses hydrophysical peat properties to assess the drivers of cross-scale variability in peat smouldering combustion in forested peatlands across the BP. Using a space-for-time chronosequence across the 120-year fire return interval and three main hydrogeological settings, and by incorporating hummock, hollow and margin locations, cross-scale variability is studied. We find that, based on peat properties such as specific yield (Sy) and gravimetric water content, forested peatland margins represent areas of high peat smouldering vulnerability, and that this is exacerbated with an increasing time-since-fire (stand-age). Although increasing Sy with time-since-fire in peatland middles may buffer water table drawdown,

when accounting for increases in canopy fuel load, transpiration and feather moss dominance forested peatland middles also become more vulnerable to smouldering combustion with time-since-fire. Moreover, the interaction of peatland margins with coarse- and heterogeneous-grained hydrogeological settings leads to lower Sy and higher density margin peat than in fine-grained settings, further increasing smouldering vulnerability. We estimate that forested peatland margins are vulnerable to combustion throughout their entire profile *i.e.* burn-out, under moderate-high water deficits in the BP. Furthermore, we identify peatland margin: total area ratio as a driver of smouldering vulnerability where small peatlands that are periodically disconnected from regional groundwater systems are the most vulnerable to high total peat carbon loss. We suggest that these drivers of cross-scale variability should be incorporated into peatland and wildfire management strategies, especially in areas near the wildland-industry and wildland-urban interface.

## 2.1 INTRODUCTION

Peatland ecosystems store approximately one-third of the world's organic soil carbon (C) pool (Gorham, 1991) and they are most abundant in northern latitudes (above 45°N) where they store approximately 455-547 Pg C (Yu et al., 2010). Carbon is stored in organic soil (peat) when long term production exceeds losses, where losses are primarily through decomposition and combustion. Forested peatlands also accumulate substantial above-ground fuels, consisting mainly of black spruce in the Canadian boreal, making them highly susceptible to wildfire (Johnston et al., 2015; Thompson et al., 2017; Thompson et al., 2019). In fact, wildfire represents the largest disturbance of forested boreal peatlands (Turetsky et al., 2002) and the spatial variability in the severity of these peat fires (*i.e.* depth of burn (DOB)) is both a leading source of uncertainty in boreal wildfire carbon emissions (de Groot et al., 2009) and a major challenge for regional wildfire management (Flannigan et al., 2013; 2016). For example, during wildfire disturbance, the ignition of surface moss/peat can lead to minimal DOB (Shetler et al., 2008), can undergo smouldering combustion to depths up to 1 m (Lukenbach et al., 2015; Walker et al., 2019), or can consume the entire peat profile exposing the underlying mineral soil (Wilkinson et al., 2018). As such, the variability in peat carbon loss from forested boreal peatlands has been shown to span two orders of magnitude, from <1 kg C $m^{-2}$ to 85 kg C $m^{-2}$ (Turetsky et al., 2011; Lukenbach et al., 2015; Hokanson et al., 2016). Given that this smouldering combustion variability also impacts the severity of smoke and particulate matter emission associated with hazardous air quality (Shaposhnikov et al., 2014) there is a growing research need to assess the drivers of this spatial variability. By characterising peat hydrophysical properties, two important factors that influence peat smouldering vulnerability are evaluated; 1) the response of the water table to a water deficit *i.e.* specific yield (Sy), and 2) the ability of peat to retain water under a change in water table depth *i.e.* moisture retention. Using these vulnerability metrics we address the research need by assessing cross-scale variability in peat smouldering combustion vulnerability in forested peatlands across the boreal plains (BP) ecozone of northern Alberta.

Peat typically burns via smouldering combustion rather than flaming combustion (Frandsen, 1987; Rein et al., 2008). The vertical propagation of smouldering and the ability to be sustained in oxygen-depleted conditions causes considerable challenges for wildfire managers as smouldering can persist for several months to years (Rein et al., 2008). The hydrophysical properties of peat are fundamental to the thermodynamic reaction of smouldering combustion (Frandsen, 1987; 1997) and are controlling factors on DOB (Miyanishi and Johnson 2002; Benscoter et al., 2011). The propagation of smouldering combustion is controlled by the ratio of energy sink to fuel source, where for peat this can be approximated by the ratio of volumetric water content to peat bulk density (*i.e.* gravimetric water content (GWC) g $g^{-1}$) (Benscoter et al., 2011). However, bulk density is not often linearly related to GWC because bulk density has been shown to be positively correlated to water retention on a volumetric basis (Sherwood et al., 2013; Moore et al., 2014). Because of the complex interactions between increasing moisture retention and increasing fuel density, threshold values of combustion-critical GWC have been shown to increase with bulk density (*e.g.* Frandsen, 1997; Rein et al., 2008). Hence, high density peat can smoulder at a GWC of up to 2.95 g $g^{-1}$ (Benscoter et al., 2011) whereas low density peat has a lower combustion-critical GWC (~1.5 g $g^{-1}$) (Frandsen, 1997; Benscoter et al., 2011).

Peat with a lower Sy has a more "flashy" water table, where a water deficit causes a larger increase in water table depth (WTD) and a lower soil-water pressure compared to peat with high Sy (Price et al., 1996; Lukenbach et al., 2015). For a given soil-water pressure peat with poorer moisture retention and/or higher bulk density will have a lower GWC and consequently be more vulnerable to smouldering combustion. Therefore, estimations of soil-water pressure are useful to better assess peat smouldering variability across the landscape. WTD is commonly used to approximate soil-water pressure in the unsaturated zone by assuming hydrostatic equilibrium (*e.g.* Thompson and Waddington, 2013). Although equilibrium conditions are a simplification of real-world processes, linear equations have been shown to accurately predict soil-water pressure using WTD (Lindholm and Markkula, 1984), where the slope and intercept of these relationships differ with bulk density (and/or microform and species) (Lukenbach et al., 2015).

The variation in peat hydrophysical properties, in particular bulk density, between hummock and hollow microforms that make up the interior of peatlands (middles) has been well evidenced (Branham and Strack, 2014). However, peatland margins have only recently been objectively identified as both a common and distinct feature of forested boreal peatlands (Mayner et al., 2018). In the BP, peat at the edge of peatlands (margins) has been shown to have higher bulk density compared to peatland middles (Ingram et al., 2019) and margins tend to experience greater DOB (Lukenbach et al., 2015; Hokanson et al., 2016). These margins tend to be dominated by a black spruce - deciduous swamp species mix, and leaf litter and feather moss (*e.g. Pluerozium schreberi*) at the ground surface (Dimitrov et al., 2014; Housman, 2017). The mix of vegetation inputs into the peat profile and higher bulk density, suggest that margin peat will have lower Sy compared to hummock and hollow peat, since bulk density correlates to the approximated pore-size distribution (Boelter, 1968).

Forested boreal peatlands become more susceptible to high intensity active canopy fire as fuel load increases with time-since-fire (TSF) (Johnston et al., 2015). Thompson et al. (2015) examined the variability of crown fire heat transfer to the peat surface and determined that peatland surface ignition varied by soil water deficit and peat hydrophysical properties, however, the interaction of TSF and peat hydrophysical properties has not previously been assessed. Peat properties in the near-surface may change due to the succession of moss species from *Sphagnum*- dominated to feather moss dominated (Benscoter and Vitt, 2008), as well as through peat and surface fuel accumulation (Wieder et al., 2009; Ingram et al., 2019). Forested peatlands in the BP have developed in a range of hydrogeological settings (HS) (*e.g.* glaciofluvial/coarse, heterogeneous fine-grained/moraine, glaciolacustrine/fine) resulting from the deposition of deep glacial sediment after deglaciation (Fenton et al., 2013). The interaction between peatland ecosystems and HS results in varying degrees of connectivity with regional-scale groundwater systems (Devito et al., 2005; Devito et al., 2012; Hokanson et al., 2018). For example, peatlands in glaciofluvial and heterogeneous fine-grained HS are often (ephemerally) perched *i.e.* experience (periodic) isolation from larger groundwater systems (Hokanson et al., 2016; James, 2017). In such circumstances margins can undergo large water table flucatuations and have been associated with extreme smouldering combustion ‘hotspots’ (> 1 m DOB) (Hokanson et al., 2016).

There has been much research into the influence of individual factors on peat smouldering combustion (Frandsen, 1987; 1997; Rein et al., 2008; Bencoter et al., 2011; Lukenbach et al., 2015; Hokanson et al., 2016). However, the spatial variability across the BP cannot be adequately explained without assessing these factors throughout the landscape and considering their interactions across scales. By characterising key peat hydrophysical properties and evaluating chosen smouldering vulnerability metrics we assess cross-scale variability in peat smouldering combustion in forest peatlands of the BP, with particular focus on the interaction between within-peatland location, TSF and HS. We hypothesise that peat smouldering vulnerability will be greatest; 1) in margins due to higher bulk density and lower Sy peat, 2) in near-surface peat under increasing TSF, and 3) in peatlands in glaciofluvial and heterogeneous HS that are subject to repeated water table drawdown.

## 2.2 METHODS

### 2.2.1 Study sites and research design

The BP ecozone has a sub-humid climate whereby potential evapotranspiration generally exceeds precipitation, with long term mean values of ~520 mm and 480 mm, respectively (Devito et al., 2012). As such, peatlands exist at the edge of their climatic limit where the region often experiences long- and short-term water deficits resulting in annual water deficits of up to ~ 200 mm (Devito et al., 2012). Approximately one-third of the BP land area is covered by peatlands (Vitt et al., 2000), where the majority of peatlands are classified as treed or forested and black spruce (*Picea mariana* Mill. BSP) is a dominant stand species (AMBI, 2016). As such, BP peatlands are susceptible to stand-replacing wildfire, with a current average fire return interval of 120 years (Turetsky et al., 2004; Wieder et al., 2009) resulting in a mosaic of peatlands of differing stand-ages between 0 and 120+ years on the landscape.

Twenty-six black spruce forested peatlands were selected in a space-for-time chronosequence, spanning the current average fire return interval for BP peatlands (~120 years, Turetsky et al., 2004). TSF is separated into three categories based on the early-, mid- and late-successional stages identified by Benscoter and Wieder (2008). Peatlands in the Central Mixedwood Natural Subregion of north central Alberta, Canada, were mapped using Ducks Unlimited Wetland Inventories. The Canadian National Fire Database (Canadian Forest Service, 2011) and Alberta Surficial Geology map (Fenton et al. 2013) were used to shortlist peatlands across a range of fire years and HS. Aerial photographs (1940-2016) (Government of Alberta, 2016) were used to aid the assessment of peatland type (black spruce dominated; bog) and confirm disturbance by wildfire (see Mayner et al., 2018). Peatlands were finally selected for detailed study based on their level of accessibility. Selected peatlands were preliminarily visited in May-June 2016 to confirm HS classification by soil texturing (see Ingram et al., 2019). HS are grouped into glaciofluvial/coarse, heterogeneous fine-grained/moraine, and glaciolacustrine/fine. The selected peatlands were found to follow the general vegetation recovery trajectory outlined by Benscoter and Vitt (2008) (Housman, 2017).

2.2.2 Peat hydrophysical properties

At each of the twenty-six peatlands, peat cores were taken from a representative hummock, hollow and margin location. The peatland margin ecotone (the transition from peatland proper to upland) was delineated in the field based on vegetation community and the core was taken from a central position. Cores (0.1 m diameter PVC) were taken to 0.6 m depth in the middle of the peatland and to mineral soil in the margin ecotone where peat depths were usually < 0.6 m (Mayner et al., 2018). Peat cores were then frozen and sub-sectioned into 0.05 m increments using a band saw. Peat samples were enclosed at one end using cheese cloth, thawed, and then slowly saturated from below with deaired water for 48 hours to prevent entrapped gas. Moisture retention was measured by placing samples on a saturated Soil Moisture Equipment Corp (Goleta, CA) ceramic plate with an air-entry pressure of 1 bar. The wet surface of the ceramic plate and the cheese cloth form an uninterrupted connection between the sample and plate (Klute, 1986), to which a negative pressure was applied using a central vacuum. Pressure steps of -10, -20, -50 and -200 hPa were used to measure water retention. Samples were kept at a pressure step for ~24 hours or until mass was unchanging, and sample volume changes were accounted for in volumetric water content calculations.

Specific yield, which is the amount of water required to raise or lower the water table by one unit length (Freeze and Cherry, 1979), was estimated using the water yield from the first pressure step ($\psi$ = -10 hPa) and was calculated for each 0.05 m peat core increment according to:

$$Sy = \theta_{sat} - \theta_{\psi=-10\,hPa} \quad [1]$$

where $\theta_{sat}$ is the volume of water at saturation and $\theta_{\psi=-10\,hPa}$ is the volume of water at the first pressure step. A value closer to zero represents a water table that is more sensitive to the addition or removal of water, and a value closer to one represents a more stable water table. Porosity of the peat samples was calculated using a particle density of 1.48 g $cm^{-3}$ for *Sphagnum* peat (Redding and Devito, 2006), and saturated water content was calculated as equal to porosity, assuming no entrapped gas. Bulk density was calculated from sample dry weight after oven drying at 65 °C for 48 hours or until mass was unchanging.

### 2.2.3 Water table drawdown

The response of the water table in the middle of the peatland is regulated by the average response of hummock and hollow peat in the saturated zone, and the spatial proportion of such microforms. Hence, for water deficit analyses, hummock and hollow peat profiles were combined, accounting for an average hummock height of 30 cm and a 1:1 ratio of hummock: hollow microforms (*data not shown*). A linear regression of the natural log of mean Sy with depth was done for each peat profile, where "profile" refers to the specific within-peatland location and TSF category combination, to determine potential water table drawdown under water deficit. Water deficit values were constrained based on literature-derived values where water deficit can be up to ~200 mm in the study region (Devito et al., 2012).

2.2.4 Peat smouldering vulnerability metrics

The ratio of volumetric water content (VWC) to peat bulk density (*i.e.* gravimetric water content (GWC) g $g^{-1}$), was used to describe the ratio of energy sink to fuel source and hence the smouldering combustion vulnerability of peat. GWC and VWC at a soil-water pressure of -200 hPa, hereafter referred to as $GWC_{-200}$ and $VWC_{-200}$, were chosen as common metrics to compare between peat profiles. Soil-water pressures of < -100 hPa lead to the draining of the hollow hyaline cells in key peat-forming *Sphagnum* mosses (Hayward and Clymo, 1982) consequently, we used -200 hPa to represent peat under "dry" conditions (once hyaline cells had begun to drain). Moreover, we used the modelled WTD to calculate soil-water pressure in the near-surface using WTD- soil-water pressure relationships from Lindholm and Markkula (1984) for higher density peat. Linear relationships were found to adequately represent the changes in soil-water pressure at WTD between 0 and 0.8 m ($R^2$ = 0.77). Similarly, Lukenbach et al. (2015) measured near-equilibrium soil-water pressure for undisturbed peat to WTD of 0.7 m. At greater WTD non-linear behaviour was observed, suggesting that our estimates of soil-water pressure are conservative. We calculated the necessary WTD required to reach the combustion-critical GWC associated with vulnerability to smouldering combustion for each peat profile (Benscoter et al., 2011). Combining the water table drawdown values, the WTD required to reach combustion-critical GWC, and the average margin depths across the BP (Mayner et al., 2018), we estimate water deficit amounts that correspond to peat smouldering combustion vulnerability in the near-surface, and entire average margin peat profiles across all HS.

2.2.5 Statistical analyses

Non-parametric statistics are used as the data do not tend to follow a normal distribution. A series of un-paired non-parametric t-tests (Mann Whitney), and ANOVA (Kruskal Wallis) tests were performed in Matlab 2017b to test for significant differences in bulk density, Sy, VWC and GWC for within-peatland location, time-since-fire category and their interaction. We used the same methodology to compare depth-integrated peat properties between location – HS combinations. Post-hoc tests were conducted using non-parametric Tukey's HSD (Steel-Dwass) tests where alpha was set at 0.05 unless otherwise stated. We report arithmetic means and standard deviations unless otherwise stated.

## 2.3 RESULTS

2.3.1 Peat hydrophysical property variability

*2.3.1.1. Within peatland location*

Mean bulk density, specific yield (Sy) and $VWC_{-200}$ were significantly different for within-peatland locations where bulk density followed the trend hummock < hollow < margin, and Sy and $VWC_{-200}$ followed the opposite trend (Appendix A). Bulk density, Sy and $VWC_{-200}$ showed strong depth-dependency (Figure 1). Sy decreased with depth for all within-peatland locations where values were 0.73 ± 0.13, 0.68 ± 0.18 and 0.67 ± 0.16 at the surface (top 0.05 m) of hummocks, hollows and margins, respectively, and 0.24 ± 0.15, 0.05 ± 0.06 and 0.06 ± 0.05 at the deepest points measured (0.50-0.55 m) (Figure 1b). Conversely, bulk density and $VWC_{-200}$ increased with depth for all locations (Figure 1a and 1c). Bulk density was a good predictor of Sy ($r_2$ = 0.62, 0.45, 0.70 for hummocks, hollows and margins, respectively, (linear fit, F test $p < 0.01$)), and also a good predictor of $VWC_{-200}$ ($r^2 = 0.66$, 0.49, 0.78 for hummocks, hollows and margins, respectively, (linear fit, F test $p < 0.01$)).

#### *2.3.1.2. Hydrogeological setting*

For margins, depth-integrated bulk density was higher and Sy was significantly lower in heterogeneous hydrogeological setting (HS) compared to glaciofluvial and glaciolacustrine HS ($F = 2.92$, $p < 0.1$) (Figure 2). $VWC_{-200}$ was significantly higher in margins in heterogeneous HS compared to glaciolacustrine ($F = 5.99$, $p < 0.1$) while $VWC_{-200}$ for margins in glaciofluvial HS was not significantly different to any other HS (Appendix A).

#### *2.3.1.3 Time-since-fire*

Analyses of depth profiles by TSF category found that Sy in hummocks tended to shift towards increasing values with TSF (Figure 3a), where there was a significant difference between near-surface (upper 0.25 m) Sy in the 0-20 and 81-120 year TSF categories (Table 1). Hollows and margins also shifted to higher values of Sy (Figures 3b and 3c), however, this difference was largest between 0-20 and 21-80 years since fire, and the difference was confined to the upper 0.15 m of the peat profile (Table 1).

#### *2.3.1.4 Sensitivity to water table drawdown*

Using Sy depth relationships for the TSF category peat profiles for middle (hummock and hollow combined) and margin (Appendix A), we calculated potential water table drawdown values as water deficit increases (Figure 4). Based on average measured peat properties, WTD in peatland middles is decreasingly responsive to water deficit with TSF (less change in WTD for a given water deficit) whereas margin WTD is most responsive in early- and late-successional phases, 0-20 and 81-120+ years since fire, respectively. In isolation from other water inputs, on average, margin peat would require > 80 mm of water deficit to reach a WTD of 0.4 m at 21-80 years since fire, whereas < 50 mm would be required at > 80 years since fire (initial WTD of 0.05 m). Comparatively, middle peat would require > 150 mm water deficit in all TSF categories (initial WTD of 0.1 m). Margin WTD is more sensitive to water deficit than middle WTD across all TSF categories and margins tend to show an increase in expected water table response at around 0.4 m WTD, whereas the middle only becomes more responsive at WTD > 0.7 m (Figure 4).

*2.3.1.5 Combined effects on peat smouldering combustion*

When accounting for bulk density in moisture retention analysis, gravimetric water content (GWC; g $g^{-1}$) at a given soil-water pressure follows the trend hummocks > hollows > margins (Figure 5). Median GWC with decreasing soil-water pressure shows that margin GWC crosses the critical threshold of 2.95 g $g^{-1}$ at ~ -200 hPa whereas hollow and hummock median GWC is ~5 and 6 g $g^{-1}$ respectively (Figure 5). The relationship between mean Sy and mean $GWC_{-200}$ differed between within-peatland locations (Figure 6). In hummock profiles $GWC_{-200}$ stayed near constant with depth (4.6 – 5.5 g $g^{-1}$), and mean Sy tended to be higher than in hollows and margins. In hollows, $GWC_{-200}$ ranged from 6.0 to 3.5 g $g^{-1}$ across the range of mean Sy, where lower $GWC_{-200}$ values corresponded with higher Sy in the near-surface. Although Sy values were similar for a given depth in margins and hollows, margins retain relatively less water on a gravimetric basis. Margin $GWC_{-200}$ was lower than hummock and hollow $GWC_{-200}$ regardless of mean Sy and values were close to the critical 2.95 g $g^{-1}$ threshold over the entire depth and range of mean Sy.

Soil-water pressure of -200 hPa corresponded to the critical GWC in margin peat. Using this benchmark, we estimated the average WTD for each peat profile where soil-water pressure would fall below -200 hPa. We found that a WTD in excess of the calibrated range (*i.e.* > 2 m) would be required to reach critical GWC in middle peat profiles, whereas WTD of ~1.1 m was required in margins (Appendix A). Applying this to the estimated water table drawdown values, we approximate the water deficit required to lower GWC to critical values in the near-surface (lower water deficit) and entire margin peat profile (upper water deficit; Figure 7). This suggests that margins are vulnerable to smouldering combustion throughout their entire peat profile, hereafter referred to as "burn-out", when water deficit exceeds ~130 mm and 100 mm in 21-80 and 81-120+ years since fire, respectively (Figure 7).

## 2.4 DISCUSSION

### 2.4.1 Variability in peat hydrophysical properties

Peat hydrophysical properties were found to vary significantly depending on location within the peatland, depth below ground, time-since-fire (TSF) and hydrogeological setting (HS). Hummocks, hollows and margins showed distinct profiles of bulk density, specific yield (Sy) and moisture retention (Figure 1) likely due to their composition of different moss species (*e.g. Sphagnum fuscum* dominates hummocks, and *Sphagnum angustifolium* dominates hollows; Housman, 2017). Moss species has been shown to affect peat recalcitrance, whereby *S. fuscum* is resistant to decomposition, often having lower bulk density than other species under similar environmental conditions (Turetsky et al., 2008).

Moreover, using bulk density as a proxy for pore-size distribution (Boelter, 1968), it corresponds that hummocks have a greater Sy than hollows and margins. Although Sy and volumetric water content (VWC) are similar with depth in margins and hollows, margin bulk density generally exceeds hollows, which can be attributed to the different peat composition in the margins *i.e.* the input of deciduous leaf litter and a higher quantity of woody roots (Housman, 2017; Ingram et al., 2019).

Time-since-fire was found to have a significant effect on Sy in the uppermost 0.15 – 0.2 m of the peat profiles (Figure 3; Table 1). In black spruce dominated peatlands, hummocks are rarely inundated, and the water table resides at or below the surface of hollows for most of the growing season (Lukenbach 2015b; 2017). Hence the change in Sy will have relatively less impact on the water table depth (WTD) response in the middle, compared to in the margins which frequently experience flooding (Lukenbach et al., 2017). Margin WTD response is far greater than middles (generally twice as responsive (Figure 4)), especially at low TSF, when the uppermost, least decomposed, peat is burned away exposing denser peat at the surface (Sherwood et al., 2013; Thompson et al., 2015; Ingram et al., 2019). Margin WTD becomes more responsive to water deficit between mid- and late-succession (21-80 and 81-120+ years) which supports our original hypothesis. Comparatively, middle WTD becomes slightly less responsive due to increasing Sy (Figure 4).

The trends found between within-peatland locations were consistent across hydrogeological settings, however, the differences between margins and middles are amplified in glaciofluvial and heterogenous HS compared to fine HS due to greater bulk density and lower Sy (Figure 2). This supports our hypothesis that the large WTD fluctuations experienced in glaciofluvial and heterogenous peatland margins (Hokanson et al., 2018) causes increased decomposition and lower Sy. In addition to the large-scale hydrological control of HS, the WTD-Sy feedback also impacts margin peat properties (Waddington et al., 2015); a positive feedback when initiated, whereby peat with low Sy more frequently experiences water table fluctuation leading to increased rates of decomposition and increasingly lower Sy. However, this positive feedback on decomposition is likely regulated by the build-up of decomposition by-products and decreasing sources of labile carbon (Moore and Basilko, 2006). Moreover, high density margin peat may also act to buffer WTD response in the peatland middle by reducing lateral water loss (Lapen et al., 2008), as peat bulk density is negatively correlated with saturated hydraulic conductivity (Branham & Strack, 2014).

### 2.4.2 Implications for peat smouldering vulnerability

Margins and hollows have lower Sy than hummocks, such that WTD will be more responsive to water deficit, and more peat will be exposed to drying. However, WTD in peatland middles is buffered by the response of combined hummock and hollow peat due to small hydraulic gradients between microforms (Malhotra et al., 2016), leading to less severe water table drawdown (Figure 4). Under decreasing soil-water pressure (*i.e.* increasing WTD) GWC is lowest in margins, and therefore smouldering combustion vulnerability is highest, followed by hollows and hummocks (Figure 5). Median GWC falls below the combustion-critical GWC (2.95 g $g^{-1}$) at much higher (less negative) soil-water pressure in margins than in hummocks or hollows (corresponding to shallower WTD). Although mean Sy with depth is similar between margin and hollow peat (Figure 1b), the greater bulk density of margins (Figure 1a), and the buffering of middle WTD (Figure 4) result in margins being vulnerable to smouldering combustion throughout their entire profile at soil-water pressures ~ - 200 hPa (Figure 6). Comparatively, hollows have their lowest GWC values in the surface peat (upper 0.1 m) where Sy is highest (Figure 6). This supports actual depth of burn (DOB) measurements in the BP where margin DOB was approximately four times greater than middles (mean margin DOB = 0.25 m and mean middle DOB = 0.06 m; Lukenbach et al., 2015) and middle DOB was dominated by hollows because hummock DOB is generally minimal (Benscoter and Weider, 2003; Hokanson et al., 2016).

### 2.4.3 Applying ecosystem – landscape scale context

The abundance and species of both above-ground and surface vegetation also have an effect on vulnerability to wildfire (Johnston et al., 2015) and peat smouldering (Benscoter et al., 2015; Thompson et al., 2015; 2019). The potential for the propagation of surface and crown fire with TSF (analogous to stand-age) has been documented for BP black spruce forested peatlands (Johnston et al., 2015). Low intensity surface fire can be supported after just 10 years following wildfire, whereas the canopy fuel load necessary for high-intensity crown fire requires a stand-age of ~80 years (Johnston et al., 2015). For margins specifically, above-ground biomass has been found to accumulate quickly post-fire, and biomass generally exceeds that of black spruce forested peatland middles (Housman, 2017), although some above-ground biomass in margins comprises deciduous species that are generally less susceptible to wildfire (Hély et al., 2001). Overall, peat smouldering vulnerability increases with TSF due to the accumulation of above-ground fuels. Moreover, increased canopy cover contributes to increased rates of transpiration, that, in some cases, can outweigh reductions in surface evaporation and increase peatland water deficit (Kettridge et al., 2014). Increased canopy cover increases shading of the ground surface, reducing evaporation as mentioned, but also increasing the competitive advantage of feather moss species over *Sphagnum* species (Bisbee et al., 2001). Feather mosses are more susceptible to combustion than *Sphagnum* and begin to dominate margin and middle ground cover at ~60 (Housman, 2017) and 80 years post-fire (Benscoter and Vitt, 2008), respectively. In combination with an increasingly responsive WTD in the margin with TSF, we highlight forested peatlands margins at > 60 years since fire as areas extremely vulnerable to peat smouldering combustion. In fact, since mean margin depths are only 0.4 – 0.6 m across the range of HS in the BP, we expect that they are vulnerable to smouldering combustion throughout their entire profile *i.e.* "burn-out" under water deficits of 100-130 mm and above (Figure 7).

We find that, in general, peat smouldering vulnerability is much greater in forested peatland margins than middles, and since groundwater connectivity (Hokanson et al., 2018) and the ratio of margin to total area (Mayner et al., 2018; Ingram et al., 2019), varies by HS, we expect differences in total peat smouldering vulnerability to vary by HS as presented in a conceptual model (Figure 8). In conjunction with topographic position (relative elevation in the landscape), HS determines interactions with regional groundwater systems in the BP ecozone (Devito et al., 2005). Peatlands located in glaciolacustrine HS (on the clay plain) tend to be more expansive (Mayner et al., 2018) and can create their own larger scale flow regimes (Devito et al., 2012). Conversely, peatlands located in glaciofluvial/coarse and heterogenous/moraine HS at mid-upper topographic position can be periodically (or fully) disconnected from regional groundwater systems, resulting in hydrological regimes that are susceptible to climatic water deficits (Hokanson et al., 2016; James, 2017; Hokanson et al., 2018). Because these peatlands tend to be smaller in areal extent, margins constitute a larger proportion of their total carbon store (Housman, 2017) and thus have greater total peat smouldering vulnerability. Consequently, we further identify small peatlands in the mid-upper topographic reaches of glaciofluvial and heterogeneous HS as areas of the greatest peat smouldering vulnerability, within our conceptual model (Figure 8).

## 2.5 CONCLUSIONS

Peat hydrophysical properties show considerable variability across the BP leading to different responses to water deficit. In combination with ecosystem – landscape scale variability in the potential for wildfire disturbance and exposure to climatic water deficits, we find that peat hydrophysical properties can act to buffer or exacerbate peat smouldering combustion (Figure 8). A slight decrease in WTD responsiveness with TSF may decrease peat smouldering vulnerability in peatland middles. However, this is likely outweighed by increases in feather moss dominance (Bisbee et al., 2001), transpiration (Kettridge et al., 2014), above-ground fuel density (Johnston et al., 2015), and associated energy transfer during crown fire (Thompson et al., 2015). Furthermore, an increasing sensitivity of margin WTD likely exacerbates the increase in peat smouldering combustion vulnerability with

TSF (Figure 8). Margins are expected to be more vulnerable than middles across all HS, however, less connectivity with regional groundwater (Hokanson et al., 2018), greater margin area (Mayner et al., 2018) and greater margin WTD sensitivity in glaciofluvial and heterogeneous HS, will amplify peat smouldering vulnerability in these areas when subjected to climatic water deficit (Figure 8).

In light of these findings and the increasing severity of the boreal wildfire regime (Flannigan et al., 2013; 2016, Wang et al., 2015; Wotton et al., 2017) we suggest that future research should focus on developing adaptive wildfire management strategies for black spruce forested peatlands, especially those at the wildland-industry-interface and/or wildland-urban-interface. Wildfire management strategies should consider time-since-fire as well as hydrogeological setting whilst taking a multi-faceted approach, incorporating above-ground, surface, and below-ground fuels, to manage the detrimental effects of peat smouldering combustion across the boreal plains ecozone of northern Alberta.

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**Author contributions statement**

Funding was secured by JMW. Research was designed by SLW, PAM and JMW, data analysis was undertaken by SLW and assisted by PAM and writing was conducted by SLW, PAM and JMW.

**Conflicts of interest**

The authors report no conflicts of interest.

**Data availability statement**

The data supporting the conclusions of this manuscript will be made available by the authors, without undue reservation, to any qualified researcher.

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**List of tables**

**Table 2.1** Subset of significant differences of within-peatland location depth-increments by time-since-fire category (TSFcat); 0-20 years, 21-80 years and 81-120 years (*, **, *** corresponds to $p \leq 0.1$, $p \leq 0.05$, $p \leq 0.01$, respectively). *Non-significant differences not shown.*

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smouldering vulnerability correspond to: low (white), near-surface (light grey), and full profile "burn-out" (black).

**Figure 2.8** Conceptual model of total peat smouldering vulnerability with increasing water deficit; where increasing groundwater connectivity is depicted by a → b → c, increasing time-since-fire is depicted by d → e, and hydrogeological setting as a proxy for margin : middle ratio is shown by f → g → h. Darker green represents denser peat, and yellow to red represents increasing peat burn severity.

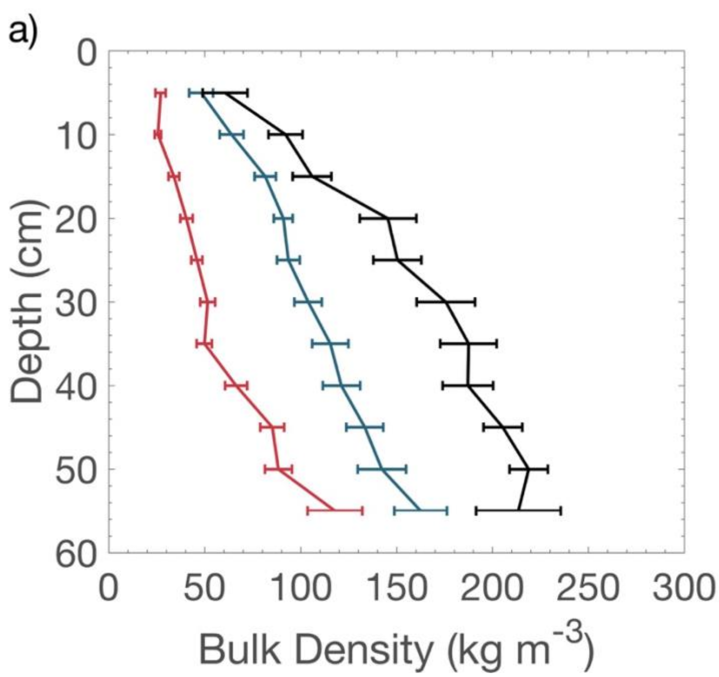
**Tables**

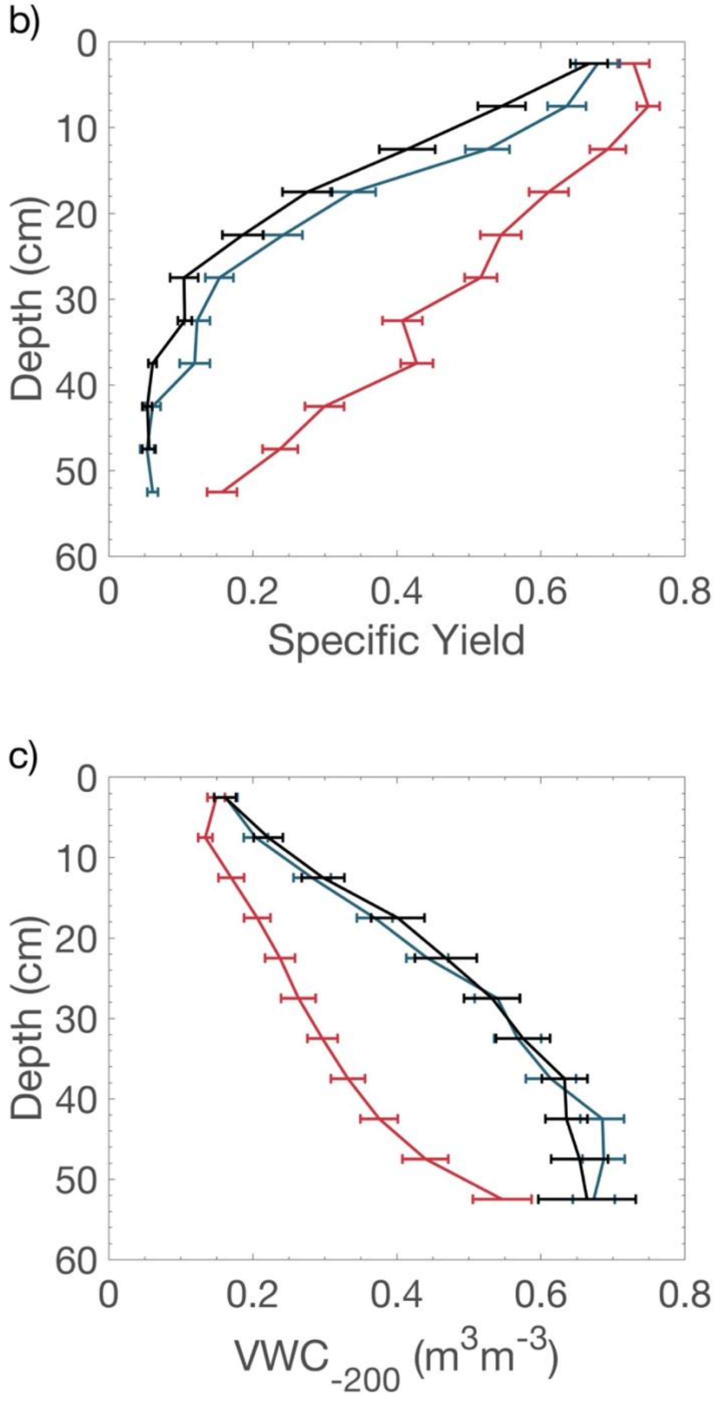
| **Location** | **Depth Mid-point (cm)** | **TSFcat 1** | **TSFcat 2** | **Specific Yield p-Value** | **Specific Yield Significance** |
|---|---|---|---|---|---|
| **Hummock** | 2.5 | 0-20 | 81-120 | 0.025 | ** |
| **Hummock** | 7.5 | 0-20 | 81-120 | 0.025 | ** |
| **Hummock** | 12.5 | 0-20 | 81-120 | 0.030 | ** |
| **Hummock** | 17.5 | 0-20 | 81-120 | 0.050 | ** |
| **Hummock** | 22.5 | 0-20 | 81-120 | 0.023 | ** |
| **Hollow** | 2.5 | 0-20 | 81-120 | 0.049 | ** |
| **Hollow** | 2.5 | 0-20 | 21-80 | 0.039 | ** |
| **Hollow** | 7.5 | 0-20 | 21-80 | 0.083 | * |
| **Hollow** | 12.5 | 0-20 | 81-120 | 0.050 | ** |

| | | | | | |
|---|---|---|---|---|---|
| **Margin** | 2.5 | 0-20 | 81-120 | 0.074 | * |
| **Margin** | 2.5 | 0-20 | 21-80 | 0.038 | ** |
| **Margin** | 7.5 | 0-20 | 81-120 | 0.017 | ** |
| **Margin** | 7.5 | 0-20 | 21-80 | 0.021 | ** |
| **Margin** | 12.5 | 0-20 | 21-80 | 0.039 | ** |

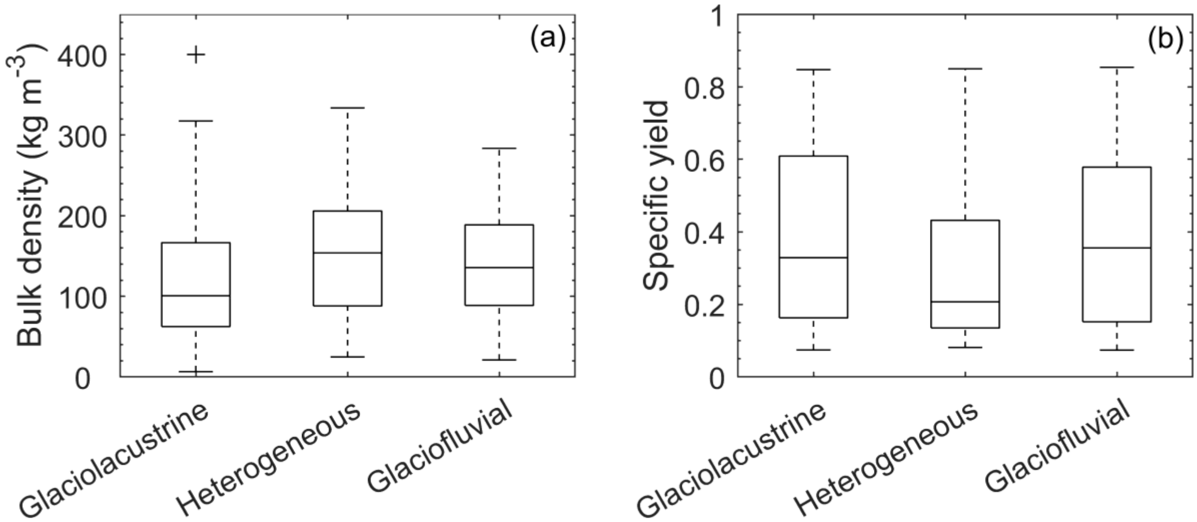
[Table 2.1]

**Figures**

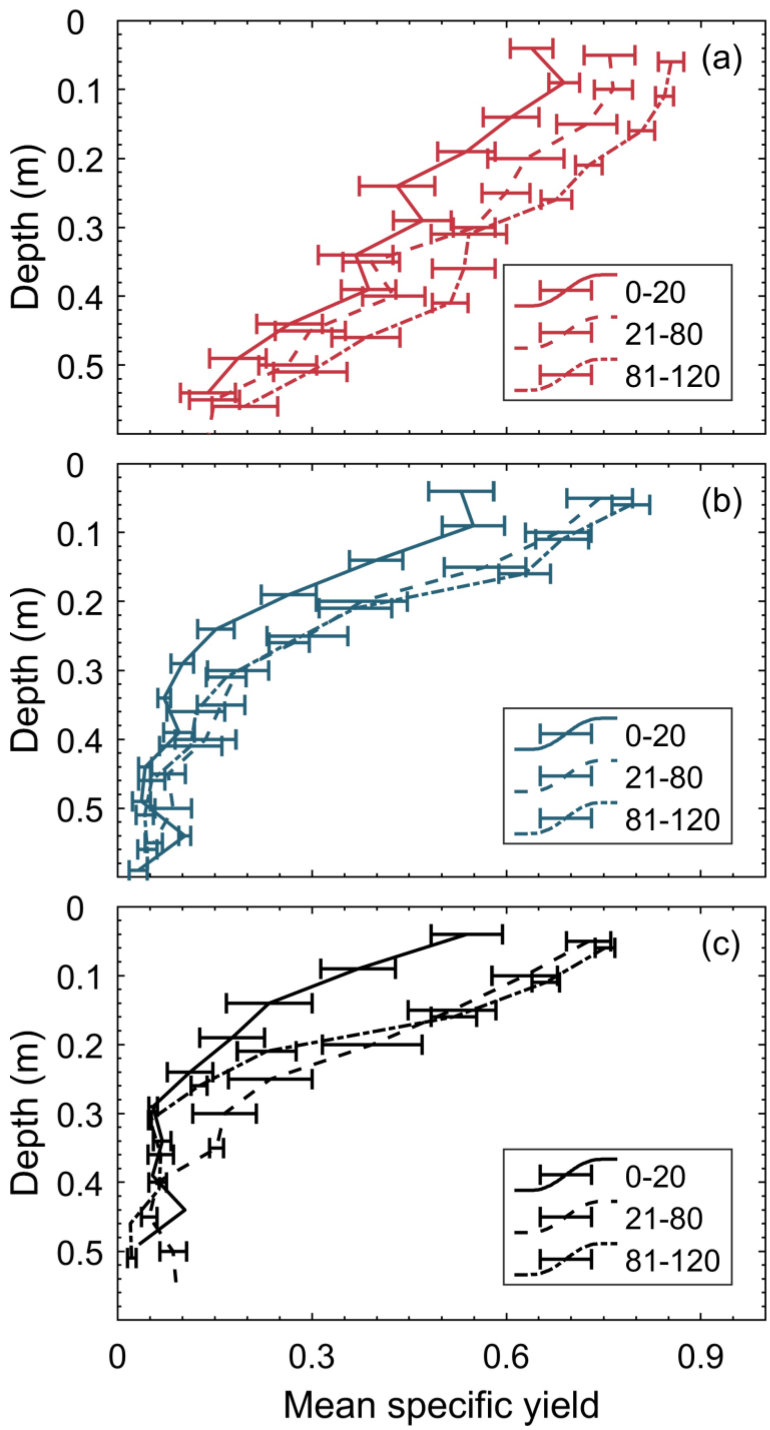



b)
Depth (cm)
0
10
20
30
40
50
60
0
0.2
0.4
0.6
0.8
Specific Yield
c)
Depth (cm)
0
10
20
30
40
50
60
0
0.2
0.4
0.6
0.8
VWC -200 (m3m-3)

[Figure 2.1]

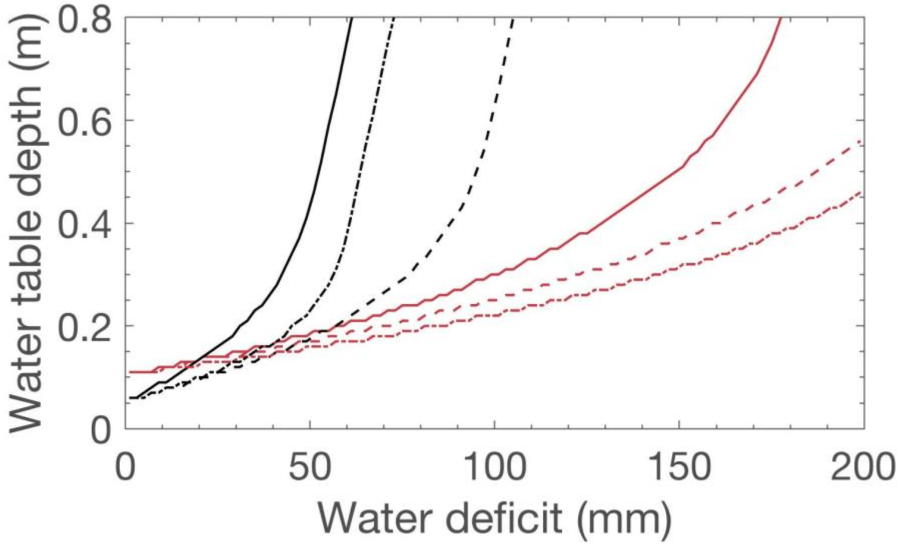

(a)
Bulk density (kg m-3)
0
100
200
300
400
Glaciolacustrine
Heterogeneous
Glaciofluvial
(b)
Specific yield
0
0.2
0.4
0.6
0.8
1
Glaciolacustrine
Heterogeneous
Glaciofluvial

[Figure 2.2]

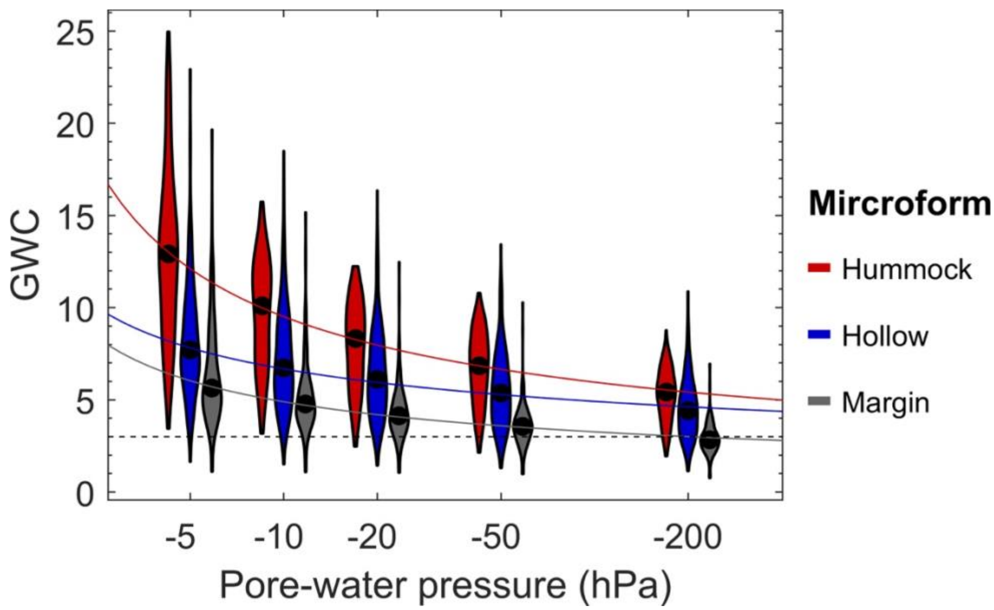


(a)
(b)
(c)
0-20
21-80
81-120
Depth (m)
Mean specific yield
0
0.1
0.2
0.3
0.4
0.5
0.6
0.9

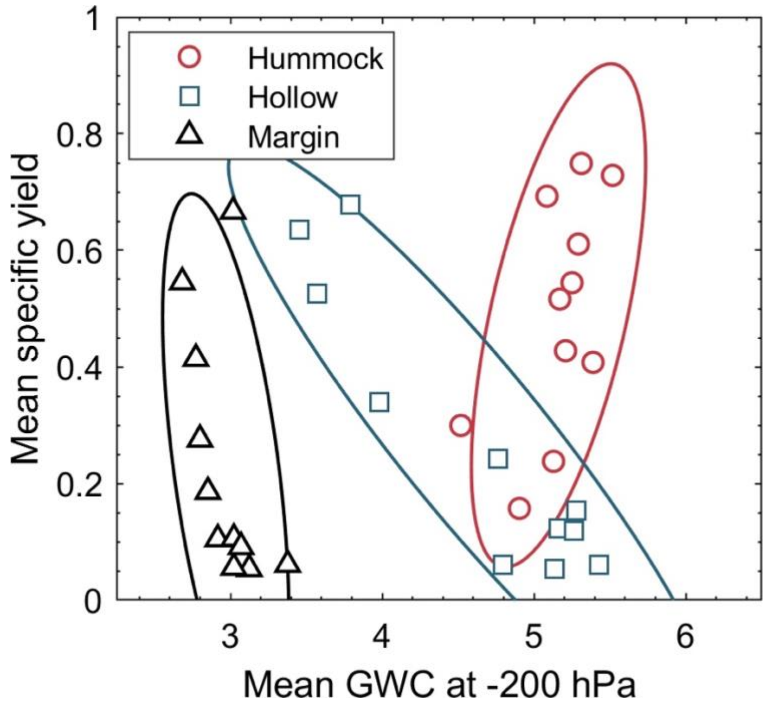

[Figure 2.3]


Water table depth (m)
0.8
0.6
0.4
0.2
0
0
50
100
150
200
Water deficit (mm)

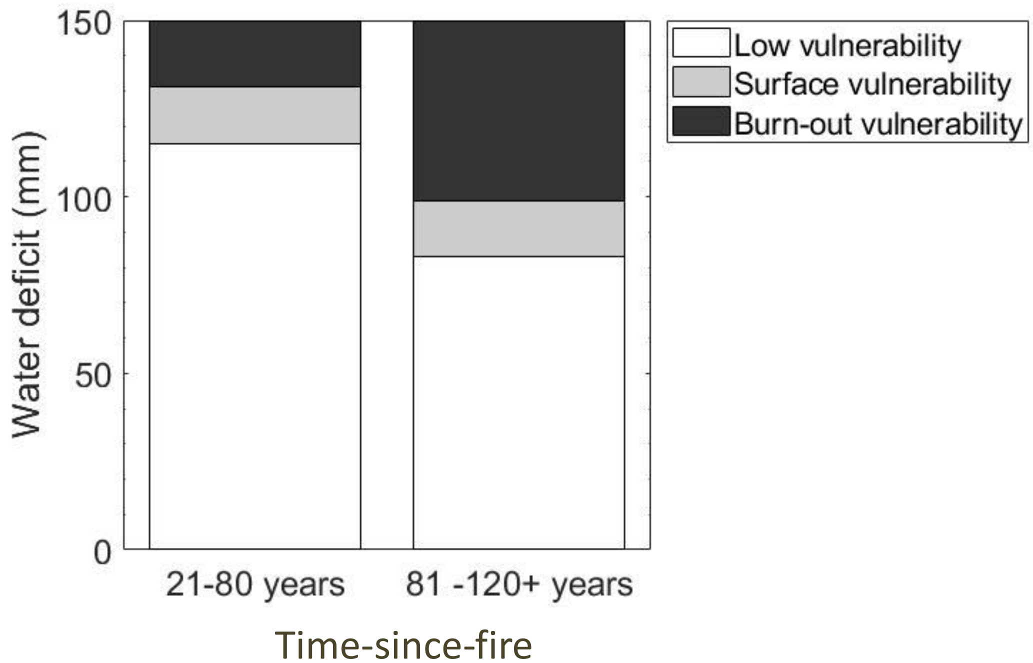
[Figure 2.4]

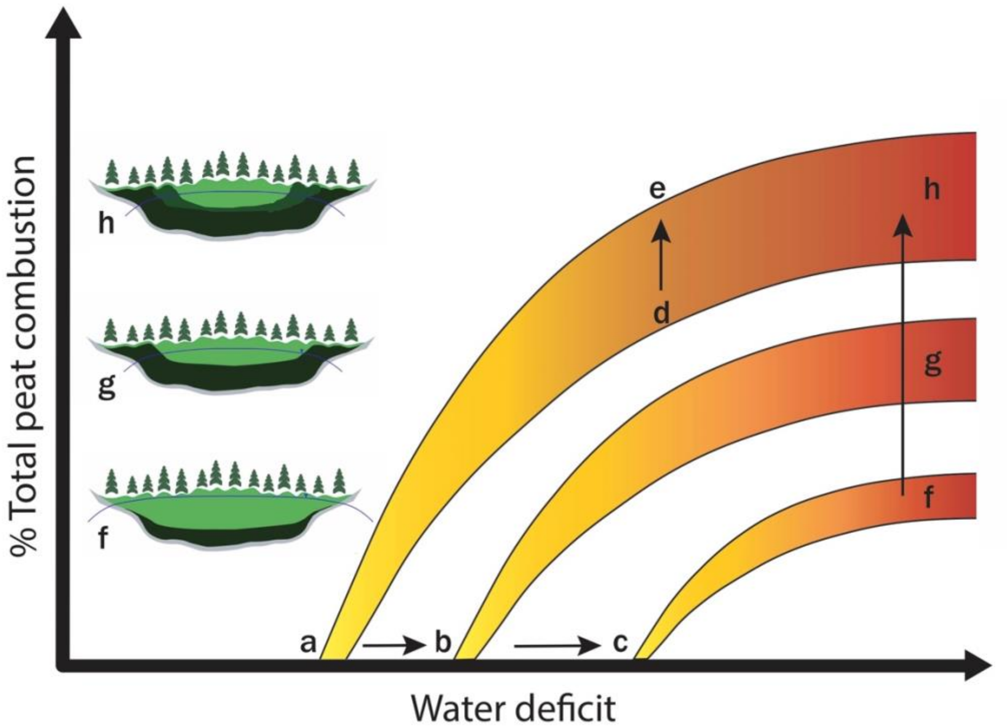

GWC
25
20
15
10
5
0
-5
-10
-20
-50
-200
Pore-water pressure (hPa)
Mircroform
Hummock
Hollow
Margin

[Figure 2.5]


Hummock
Hollow
Margin
Mean specific yield
0
0.2
0.4
0.6
0.8
1
3
4
5
6
Mean GWC at -200 hPa

[Figure 2.6]


Low vulnerability
Surface vulnerability
Burn-out vulnerability
Water deficit (mm)
0
50
100
150
21-80 years
81 -120+ years
Time-since-fire




[Figure 2.8]

# CHAPTER 3

# DID ENHANCED AFFORESTATION CAUSE HIGH SEVERITY PEAT BURN IN THE FORT MCMURRAY HORSE RIVER WILDFIRE?

## ABSTRACT

Climate change mediated drying of boreal peatlands is expected to enhance peatland afforestation and wildfire vulnerability. The water table depth - afforestation feedback represents a positive feedback that can enhance peat drying and consolidation and thereby increase peat burn severity; exacerbating the challenges and costs of wildfire suppression efforts and potentially shifting the peatland to a persistent source of atmospheric carbon. To address this wildfire management challenge, we examined burn severity across a gradient of drying in a 14 ha peatland that was partially drained in 1975-1980 and burned in the 2016 Fort McMurray Horse River wildfire. We found that post-drainage black spruce annual ring width increased substantially with intense drainage. Average (±SD) basal diameter was 2.6 ± 1.2 cm, 3.2 ± 2.0 cm and 7.9 ± 4.7 cm in undrained (UD), moderately drained (MD) and heavily drained (HD) treatments, respectively. Depth of burn was significantly different between treatments ($p<0.001$) and averaged (±SD) 2.5 ± 3.5 cm, 6.4 ± 5.0 cm and 36.9 ± 29.6 cm for the UD, MD and HD treatments, respectively. The high burn severity in the HD treatment included 38% of the treatment that experienced combustion of the entire peat profile, and we estimate that overall 51% of the HD pre-burn peat carbon stock was lost. We argue that the HD treatment surpassed an ecohydrological tipping point to high severity peat burn that may be identified using black spruce stand

characteristics in boreal plains bogs. While further studies are needed, we believe that quantifying this threshold will aid in developing effective adaptive management techniques and protecting boreal peatland carbon stocks.

## 3.1 INTRODUCTION

Boreal peatlands represent a globally important long-term carbon sink with the majority of the carbon stock residing in peat where primary production has exceeded losses from decomposition and combustion throughout the Holocene (Vitt *et al.*, 2000). These boreal peatlands also represent a large wildfire fuel source on the landscape in boreal sub-humid regions (*e.g.* Canada's Boreal Plains ecozone (BP)). BP peatlands generally experience low severity peat burn during wildfire, with depth of burn (DOB) ranging from 5 to 10 cm and releasing 2–3 kg C $m^{-2}$ (*e.g.* Hokanson *et al*., 2016). Black spruce (*Picea mariana*) dominated peatlands, common to the BP landscape, are generally resilient to low burn severity wildfire, returning to an annual net carbon sink within ~20 years post-fire (Wieder *et al*., 2009). However, with enhanced drying, black spruce dominated peatlands in the BP can experience severe smouldering combustion with high DOB (>20 cm) releasing 10–85 kg C $m^{-2}$ (Turetsky *et al.*, 2011; Lukenbach *et al.*, 2015). These high burn severity peat fires are costly and challenging for fire suppression operations and often cause potentially hazardous air quality (Flannigan *et al*., 2009; Shaposhnikov *et al*., 2014). These fires also demand extra resources due to prolonged smouldering and the subsequent "mop-up", exemplified by the Fort McMurray Horse River wildfire that was not considered extinguished until 456 days after ignition due to such smouldering (Alberta Agriculture and

Forestry, 2017). Moreover, these fires can trigger an ecosystem regime shift causing the loss of keystone *Sphagnum* mosses and recruitment of vascular vegetation, resulting in a long-term change in peatland ecohydrological structure and function. This shift is sustained by a low intensity, high frequency wildfire regime that leads to further degradation of the peat reserve (Kettridge *et al*., 2015). Given that the areal extent, frequency and severity of peatland drying (Granath *et al*., 2016) and boreal wildfires (Flannigan *et al*., 2005; 2013) are predicted to increase due to climate change, there is an urgent need to gain a better understanding of the processes controlling high severity peat burns, including the influence of peatland drying and the associated enhanced afforestation.

Previous research suggests that the loss of peatland ecohydrological resilience due to high severity peat burns likely occurs when an ecosystem structure and function threshold (*i.e.* tipping point, see Scheffer, 2009) is exceeded (Kettridge *et al*., 2015). Tipping points, known as catastrophic bifurcations in ecological theory, have been identified in a number of important ecosystems (Scheffer, 2009), and while they have received little attention in peatland studies (*c.f.* Hilbert *et al*., 2000), peatland ecosystems have tightly-coupled ecological and hydrological processes that are precursors of threshold behaviour (Scheffer, 2009). As such, the response of peatlands to wildfire is the result of both pre-fire ecohydrological conditions and numerous ecohydrological feedbacks (Thompson *et al*., 2015; Waddington *et al*., 2015). The majority of these are negative feedbacks which are centred around key traits of the peat-forming moss genus, *Sphagnum* (Waddington *et al*., 2015). Undecomposed or partially decomposed *Sphagnum* mosses have high porosity, providing a high specific yield which regulates water table (WT) fluctuations (Waddington *et al*., 2015). Low moss bulk density (fuel density), together with high surface moisture content, enables *Sphagnum* to act as an energy sink during wildfire (*e.g.* Shetler *et al.*, 2008). However, positive feedbacks can alter peatland ecohydrological conditions and increase wildfire vulnerability.

One such feedback is the water table depth (WTD) – afforestation feedback, which can exacerbate drying and negatively impact the peatland water balance (Waddington *et al.*, 2015). As WTD increases (due to drying or peatland drainage), black spruce net above-ground productivity increases resulting in greater tree heights, basal diameters, and stand density (Lieffers and Rothwell, 1986), and a concomitant increase in transpiration and rainfall interception (Price *et al.*, 1997), further increasing WTD (Waddington *et al*., 2015). This increase in above-ground fuel load also increases the potential for sustaining high-intensity crown fires (Johnston *et al*., 2015). Moreover, because feather moss has been shown to out-compete *Sphagnum* under low light conditions as afforestation increases (Bisbee *et al.,* 2001), and tends to be drier than *Sphagnum* under field conditions (Lukenbach *et al.,* 2015), peatland afforestation may also increase smouldering ignition potential and peat burn severity (Thompson *et al*., 2015). Because enhanced afforestation has been associated with deep burning in temperate peatlands (Davies *et al*., 2013), we suggest that quantifying stand characteristics may provide an opportunity to identify peatlands at high risk of exceeding an ecohydrological tipping point and thereby potentially help reduce wildfire management challenges and costs.

As a first step towards identifying a deep burning tipping point through drying and enhanced afforestation this study capitalises on a multi-decadal peatland drainage experiment that burned in the 2016 Horse River wildfire. We use a gradient of peatland drainage as a proxy for climate-mediated drying with measurements of depth of burn to assess peat burn severity.

## 3.2 METHODOLOGY

### 3.2.1 Study site

The research site is a 14 ha black spruce dominated (>95%) peatland located 11 km south of Fort McMurray, Alberta (56.732°N, 111.376°W) that burned in the 602,000 ha Horse River wildfire (MWF-009) in 2016. As part of a silviculture experiment, a portion of the peatland was drained between 1975 and 1980 (Hillman, 1987). Drainage was initiated by clearing and scarification of the black spruce canopy along a ditch network in 1975-1976, and in 1979-1980 the drainage ditch network was expanded with 0.76–1.06 m deep, 3 m wide, ditches spaced 9 m or 18 m apart (Hillman, 1987). The southern portion of the peatland remained undrained, with regional flow being roughly south to north. We classified the peatland into three treatments along a pre-fire ecohydrological gradient based on drainage ditch density: i) undrained (UD) being >30 m from drainage ditches; ii) moderately-drained (MD) with ditch spacing every 18 m; and iii) heavily-drained (HD) with ditch spacing every 9 m. Three 50 $m^2$ plots were randomly located in each treatment and used to assess tree productivity pre- and post-drainage, stand characteristics, as well as peat burn severity.

The peatland experienced a crown fire between the 5th and 6th of May 2016, with below-ground smouldering continuing from this date (pers. comm. Mark Newman, Fire Manager). The Drought Code (DC), calculated using the Canadian Fire Weather Index (FWI) system, represents the moisture content of mesic and humic organic layers (Van Wagner, 1987). On the days of the crown fire the DC value averaged 452 which is greater than 88% of the DC values during the fire season (May-Oct) over the last 50 years. Fire-fighting efforts were required to control and extinguish peat smouldering in some areas of the HD treatment due to the proximity of the peatland to important transportation infrastructure. Hence, our plots were chosen to avoid these heavily disturbed fire suppression areas.

### 3.2.2 Peat burn severity

Peat burn severity, was estimated by making 900 DOB measurements using the adventitious root method (see Kasischke *et al.,* 2008) five months post fire. DOB was estimated as the vertical distance between the burned surface and the datum provided by the adventitious roots between tree pairs. Average DOB per tree pair was based on five equally spaced measurements. In each 50 $m^2$ plot (three per treatment), average DOB was estimated for five clusters of four tree pairs (*i.e.* 15 clusters/treatment). DOB could not be assessed in an area within the HD treatment using the adventitious root method due to the complete smouldering consumption of the peat profile, resulting in exposure of mineral soil, and complete tree fall. In the burned-to-mineral portion of HD, we took DOB to be equal to the estimated pre-fire peat depth. The average and standard deviation of DOB for the entire HD treatment was derived from a weighted random resampling of measured DOB and estimated residual peat depth, with weighting based on the proportional cover of the two areas within the treatment. Measurements of post-fire peat depth were taken at nine random locations in each 50 $m^2$ plot by auguring to mineral soil. Pre-fire peat depths in each treatment were estimated to be the sum of DOB and post-fire (residual) peat depth. Mean and standard deviation of pre-fire peat depth were derived by random resampling of the measured DOB and residual peat depth (see supplementary material). Post-fire ground-cover was assessed using 15 randomly located 0.6 x 0.6 m quadrats in one plot of each treatment.

Carbon loss from peat smouldering was estimated using DOB at each measurement location, depth-dependent average bulk density and average ash content from the Zoltai database (Zoltai *et al*., 2000). As a lower and upper estimate of average depth-dependent bulk density, we used values for *Sphagnum* and sylvic peat, respectively. Average ash content for *Sphagnum* and sylvic peat were taken to be 5% and 12%, respectively, and organic matter was assumed to have an organic carbon content of 51.7% (Gorham, 1991) (i.e. peat C-content of ~49% and 46%). Estimated carbon loss in the burned-to-mineral section of the HD treatment used estimated pre-fire peat depth (see supplementary material) and average bulk density for the corresponding depth from the Zoltai et al. (2000) database. The same approach was used to estimate total pre-fire peat carbon content.

### 3.2.3 Stand characteristics

Stand characteristics were assessed by measuring the basal diameter (BD), diameter at breast height (for trees > 1.3 m), and tree species for all trees in each plot. Stand biomass and carbon/fuel loadings were then calculated using standard allometric equations (*e.g.*

Bond-Lamberty *et al*., 2002; Johnston *et al.,* 2015). Canopy closure was estimated using the relationship defined in Housman (2017) based on total above-ground stand biomass in black spruce dominated BP peatlands (supplementary material). In each plot, five trees were randomly chosen and 2-3 cm thick discs of the tree trunk were cut just above the root collar, hereafter referred to as "tree cookies". Tree cookies were used to measure annual tree ring widths (RW) in order to estimate annual above-ground tree net productivity. Prior to measuring RWs, tree cookies were smoothed with sandpaper of progressively finer grit until all annual rings were clearly visible. Tree cookies were digitized using a flatbed scanner at 1200 dpi. RW were subsequently measured using the R package *digitizeR* (Poisot, 2011). To account for non-uniform radial growth of the tree trunk, RW was measured in four quadrats of each tree cookie, and averaged on an annual basis.

### 3.2.4 Statistical analyses

All statistical analyses were conducted using R (R Core Team, 2013) and results presented are means and standard deviation unless stated otherwise. DOB measurements were rank transformed due to being non-normally distributed based on the Shapiro-Wilk test (*shaprio.test* function – R). A 1-way ANOVA (*aov* function – R), followed by a Tukey-HSD post-hoc test was used to determine significant differences in DOB and BD with treatment. A Spearman rank correlation test (*cor.test* function - R) was used to assess correlation between DOB and treatment level stand characteristics. A linear mixed effects model (*lmer* function – R) was used to evaluate treatment differences in annual RW.

## 3.3 RESULTS

3.3.1 Peat burn severity

DOB was significantly different between treatments (*$F = 439.2, p < 0.001$*) (Figure 1). DOB was 2.5 ± 3.5, 6.4 ± 5.0, and 16.0 ± 10.2 cm for UD, MD, and HD treatments, respectively. Measurements from the HD treatment (Figure 1) exclude the burned-to-mineral portion (38%) of the HD treatment. Given that the estimated pre-fire peat depth in the HD treatment (see supplementary material) was 70.9 ± 16.4 cm (median = 70 cm), average DOB across the HD treatment was calculated to be 36.9 ± 29.6 cm.

Negligible DOB (≤0.5 cm) occurred in 46% and 14% of the UD and MD treatment plots, respectively, indicating ground cover was unburned or singed. In contrast, the HD treatment had no areas of negligible DOB recorded. Correspondingly, spatial surveys of ground cover showed that singed *Sphagnum* hummocks were present in both the UD and MD treatments but not in the HD treatment (supplementary material). Peat carbon loss from the three treatment areas was estimated to be greatest from the HD treatment, followed by MD, and UD (Table 1). When assessed as a percent of estimated pre-fire peat carbon stock, this loss equates to 2.8%, 5.7% and 20.4% (50.6% when burned-to-mineral included) in the UD, MD and HD treatments, respectively (Table 1).

3.3.2 Pre-fire peatland stand characteristics

The apparent increase in tree productivity post-drainage relative to the UD baseline, was much greater at the HD versus MD treatment, based on average annual measured ring width (RW) (Figure 2). A linear mixed effects model was used to evaluate average annual RW, with drainage treatment and tree sample as fixed and random effects, respectively. Drainage treatment was shown to have a significant effect on average annual RW ($F = 87.86$, $p<0.001$) where post-drainage UD, MD and HD RW were 0.22 ± 0.07, 0.32 ± 0.07 and 0.84 ± 0.17 mm, respectively (Figure 2). Maximum average annual ring width was 1.22 mm for the HD treatment, 0.45 mm for the MD and 0.43 mm for the UD treatment. Peak annual RW occurs after a three-year time lag since drainage in the MD treatment compared to nine years in the HD treatment (Figure 2). Differences in tree productivity result in treatment stands with significantly different basal diameters ($F = 106.9$, $p<0.001$). Stem density was greatest in the MD treatment compared to the UD treatment and HD treatment. However, due to the proportionally larger basal diameters, basal area was greatest in the HD treatment, followed by the MD and UD treatments. Correspondingly, crown fuel load, total stand biomass and canopy closure follow the trend HD > MD > UD (Table 2).

An ANOVA showed that BD varied significantly with treatment ($F_{2,6} = 41.83$, $p=<0.001$) with a linear model showing a significant effect of local drainage density (ditch area $ha^{-1}$) on plot level BD ($F_{1,7} = 14.65$, $p=0.006$). The corresponding average ditch spacing for MD and HD treatments are 16.5 m and 9.5 m on centre, respectively. Conversely, within treatment, a two-way ANOVA with treatment, distance to ditch, and their interaction as factors, shows that distance to ditch has no significant effect on the BD of individual trees ($F_2=1.85$, $p=0.158$). A correlation matrix containing treatment average DOB, stem density, basal area and drainage density shows that all pairwise combinations excluding stem density have a Spearman rank correlation equal to one. Pearson correlations are similarly high ($r>0.86$), but with only three treatments, the correlations are generally not considered significant ($p>0.05$). Finally, using all treatments together, there was a strong linear correlation between the average DOB measured at tree clusters (n=15 per treatment – see methods), and the median basal diameter of the tree cluster (Figure 3).

## 3.4 DISCUSSION

### 3.4.1 WTD – afforestation feedback and peat burn severity

Our results demonstrate that experimental drainage substantially increased above-ground tree productivity at the HD treatment compared to MD treatment (Figure 2) where HD average annual ring width (RW) was approximately double that of MD and UD 20 years after drainage. We suggest that above-ground tree productivity at the MD and HD treatments was affected by post-drainage enhancement of the WTD-afforestation feedback (Waddington *et al*., 2015). With higher above-ground biomass, not only is canopy fuel load higher, but there has likely been a decrease in *Sphagnum* moss cover (Bisbee *et al*., 2001) and near-surface peat moisture content (Lukenbach *et al.,* 2015) at MD and HD treatments, resulting in enhanced peat burn severity during the wildfire.

The enhanced afforestation increased canopy fuel loads at both the MD and HD treatments (approximately 2x and 5x higher than the UD treatment, respectively; Table 2), which increases the capability and likelihood of sustaining a high-intensity crown fire and the probability of widespread surface ignition and potential smouldering (Johnston *et al.,* 2015). The burning of greater crown fuel loads provides more energy to supply the downward propagation of smouldering combustion (Thompson *et al.,* 2015). While there are many complexities to the ignition and propagation of smouldering peat fire (Benscoter *et al.,* 2011), it is worth noting that the total stand biomass estimate in both the MD and HD treatments is greater than measurements from an undisturbed BP peatland 108 years since fire (Johnston *et al.,* 2015) despite maximum tree age being <64 years.

Differences in above-ground tree productivity corresponded with canopy development, resulting in canopy closure estimates of 20, 30, and 70% for the UD, MD and HD treatments, respectively. As canopy closure (and shading) increases, the competitive advantage of *Sphagnum* moss declines (Bisbee *et al.,* 2001) and shade-tolerant feather moss becomes the dominant moss cover, usually after 60-80 years post-fire (Benscoter and Vitt, 2008; Housman, 2017). The importance of moss moisture content as an energy sink means that *Sphagnum* mosses can limit carbon losses from peat fires given their superior moisture retention traits (Shetler *et al.,* 2008; Thompson *et al.,* 2015). The poor water retention properties of feather moss exacerbate low surface moisture conditions and is likely responsible for the greater DOB associated with its ground cover (Thompson *et al.,* 2015). Indeed, DOB was greatest where feather moss was likely the dominant moss cover, in the HD treatment (with highest canopy closure estimate) followed by the MD treatment, and DOB was smallest in the UD, which contained a much higher proportion of *Sphagnum* cover compared to the other treatments (supplementary material).

Stand density and leaf area index are the primary predictors of the bulk rates of transpiration from peatlands (Waddington *et al.,* 2015) indicating that transpiration water losses increase with afforestation. However, Kettridge et al. (2013) suggest that changes in evapotranspiration are insensitive to afforestation until very high foliage densities (as observed at the HD treatment). Nevertheless, this positive feedback is amplified further by the increased levels of interception (Price *et al.,* 1997) with higher foliage density. Water intercepted by the canopy is lost directly via evaporation, reducing the net input of water to the peatland and decreasing surface moisture content, an important variable for smouldering potential (Thompson *et al.,* 2015). The complex interactions of the WTD-afforestation feedback likely progressed the HD treatment to exceed a tipping point resulting in high peat burn severity.

### 3.4.2 Exceeding a tipping point to high peat burn severity

Our results suggest the exceedance of an ecohydrological tipping point to high peat burn severity in the HD treatment of the study site as the HD and MD treatments experienced significantly different peat burn severity (Figure 1). While average DOB in the UD treatment (2.5 ± 3.5 cm) is comparable to the shallow peat burns common to BP peatlands (*e.g.* Hokanson *et al.,* 2016), we attribute the increased DOB at the MD (6.4 ± 5.0 cm) and HD (36.9 ± 29.6 cm – includes area burned-to-mineral) treatments to drainage and enhanced afforestation, similar to other northern and temperate peatlands (Turetsky *et al.,* 2011; Davies *et al.,* 2013). By defining the tipping point as carbon loss in excess of the product of long-term carbon accumulation rate and average fire return interval we find that the HD treatment has surpassed the tipping point. Moreover, the HD treatment was the greatest resource draw on fire suppression efforts (pers. comm. Mark Newman, Fire Manager), and we speculate that the high depth of burn and partial exposure of mineral soil may increase the recruitment of vascular vegetation, potentially leading to a regime shift. In the case of a shift to shrub/grassland, the new vegetation community is likely to be sustained by higher frequency, low intensity fires, resulting in the degradation of residual carbon stocks (Kettridge *et al.,* 2015).

Our estimated peat carbon loss of ~5–7 kg C $m^{-2}$ in the HD treatment (excluding the area burned-to-mineral) is double that of a typical peat fire in this region (2-3 kg C $m^{-2}$) (*e.g.* Hokanson *et al.,* 2016). However, when the burned-to-mineral portion of the HD treatment is included, peat carbon loss (~12–17 kg C $m^{-2}$) is an order of magnitude greater than UD, and also exceeds carbon loses of other drained BP peatlands (Turetsky *et al.,* 2011; Kettridge *et al.,* 2015). We suggest that this is due to the relatively high drainage density in the HD treatment and the initiation of the WTD-afforestation feedback, allowing for prolonged drying-enhanced tree growth (Figure 2) (Waddington *et al.,* 2015). Moreover, this is supported by a strong linear correlation between basal diameter and DOB (Figure 3). Of greatest concern is the percent of peat carbon lost due to smouldering combustion; this equates to 20% in the HD treatment (51% including burned-to-mineral area) but only 6% in the MD treatment, and 3% in the UD treatment. With an average carbon accumulation rate of continental western Canadian peatlands over the last 1000 years of 0.0194 kg C $m^{-2}$ $yr^{-1}$ (Vitt *et al.,* 2000), the extensive carbon loss from the HD treatment equates to ~240–350 years of carbon accumulation (~600–860 years when the area burned-to-mineral is included). It is unlikely that enough carbon will be accumulated within a typical fire return interval (100–120 years) to retain a carbon sink status (Turetsky *et al.,* 2002), hence we argue the tipping point as previously defined has been surpassed.

Conversely, a loss of 6% and 3% of peat carbon at the MD and UD treatments, represents ~80–120 and 30–50 years worth of average carbon accumulation, respectively. Given the current fire return interval and residual peat depths of 68.9 ± 11.3 cm and 83.5 ± 13.5 cm in the UD and MD treatments respectively, it appears that moderate drainage may not impact long-term carbon storage. We suggest that the original function is maintained in the UD and MD treatments primarily by the presence of *Sphagnum* moss, associated with singed ground cover and negligible DOB, because it is the keystone moss species that promotes fast recovery and the re-initiation of carbon accumulation (Shetler *et al.,* 2008; Waddington *et al.,* 2015). In contrast, there is no evidence of low burn severity *Sphagnum* in the HD treatment. *Sphagnum* moss promotes the redevelopment of peatland negative feedbacks such as the WTD-moss productivity feedback and WTD-moss surface resistance feedback (see Waddington *et al.,* 2015). With natural post-fire recovery and establishment of *Sphagnum*, peatland ecohydrological conditions return to a state which promotes moss productivity and carbon accumulation (Waddington *et al.,* 2015).

### 3.4.3 Implications for peatland and wildfire management

Average tree basal diameter and stand basal area may provide easily measurable indices of proximity to the ecohydrological tipping point surpassed in the HD treatment. The tipping point identified in this study is bounded between the MD and the HD basal diameters of 3.2 ± 2.0 and 7.9 ± 4.7 cm, and basal area estimates of 16.5 and 60.3 $m^2$ $ha^{-1}$, respectively. Although there are many confounding variables that influence fire severity and energy input to the peat surface (Thompson *et al.,* 2015), we suggest that the identification of this bounded tipping point is a useful and practical guide to identify peatlands that are vulnerable to high severity peat burns in moderate-extreme fire weather. This is especially valuable as fire management in the sub-humid region of Canada's boreal is approaching a critical threshold of effectiveness, and enhancement of the fire regime due to climate change will only add stress to the system (Flannigan *et al.,* 2009).

Climate change is predicted to increase the incidence and areal extent of high/extreme fire weather across central western Canada (Flannigan *et al.,* 2005) with longer drought periods and fire weather index values, such as the Drought Code, likely to increase (Collins *et al.,* 2013; Flannigan *et al.,* 2016). The drying of northern peatlands leading to WT-drawdown will enhance the effects of the WTD-afforestation feedback, increase peat burn severity (Flannigan *et al.,* 2013), and potentially increase the likelihood of peatlands exceeding ecohydrological tipping points to high severity peat burn. Although there is much research needed to quantify more specific effects of afforestation on peat burn severity, we suggest that the concept of ecohydrological tipping points to high severity peat burn should be incorporated into fire and land management techniques. By managing peatlands to remain below ecohydrological tipping points through fuel load management and potential *Sphagnum* moss propagation, fire management challenges and costs could be reduced and the carbon stock of boreal peatlands further sustained.

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Tables

**Table 3.1** Estimated peat carbon (C) loss (mean ± SD) based on measured depth of burn, depth-dependent estimates of average peat bulk density, and estimated C-content for *Sphagnum* and sylvic peat in western boreal Canada from the Zoltai database (Zoltai *et al*., 2000). *Sphagnum* and sylvic peat are used as rough analogues for undrained and drained peat bulk density, respectively.

| **Treatment** | **Peat Depth** | **Peat Carbon Loss** | | | |
|---|---|---|---|---|---|
| | (cm) | (kg C $m^{-2}$) | | (% of pre-fire peat carbon stock) | |
| | Pre-fire | *Sph. peat* | Sylvic peat | *Sph.* peat | Sylvic peat |
| UD | 68.9 ± 11.3 | 0.63 ± 0.93 | 0.92 ± 0.34 | 2.8 % | 2.9 % |
| MD | 83.5 ± 13.5 | 1.65 ± 1.42 | 2.40 ± 2.01 | 5.7 % | 6.1% |
| HD | 70.9 ± 16.4 | 4.71 ± 3.63 | 6.74 ± 5.21 | 20.4 % | 20.4% |
| HD* | -- | 11.70 | 16.75 | 50.6 % | 50.6% |

* - weighted average C-loss including 38% of HD site which burned to mineral soil.

**Table 3.2** Treatment level black spruce stand characteristics. Crown fuel load and total stand biomass calculated using empirical equations from Bond-Lamberty et al. (2002) and Johnston et al. (2015). Values include ± one standard deviation.

| **Treatment Stand Characteristic** | **UD** | **MD** | **HD** |
|---|---|---|---|
| Average Basal Diameter (cm) | 2.6 ± 1.2 | 3.2 ± 2.0 | 7.9 ± 4.7 |
| Stem Density (stems $ha^{-1}$) | 16,100 | 20,300 | 9000 |
| Basal Area ($m^2$ $ha^{-1}$) | 10.0 | 16.5 | 60.3 |
| Crown Fuel Load (kg $ha^{-1}$) | 6668 | 13,778 | 32,269 |
| Total Stand Biomass (kg $ha^{-1}$) | 12,025 | 31,554 | 110,903 |
| Canopy Closure (%) | 20 | 30 | 70 |

**Figures**

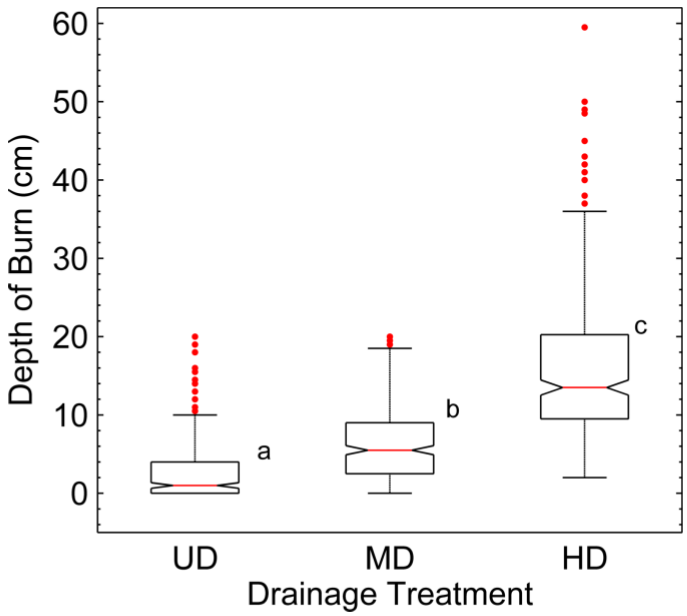


**Figure 3.1** Measured depth of burn (DOB) across the undrained (UD), moderately drained (MD), and heavily drained (HD) treatments. Median DOB and 95% confidence interval of the median is represented by the horizontal red line and error bars, respectively. Outliers are presented as red dots. Letters indicate a significant difference in DOB between treatments, using a significance level of 0.05.

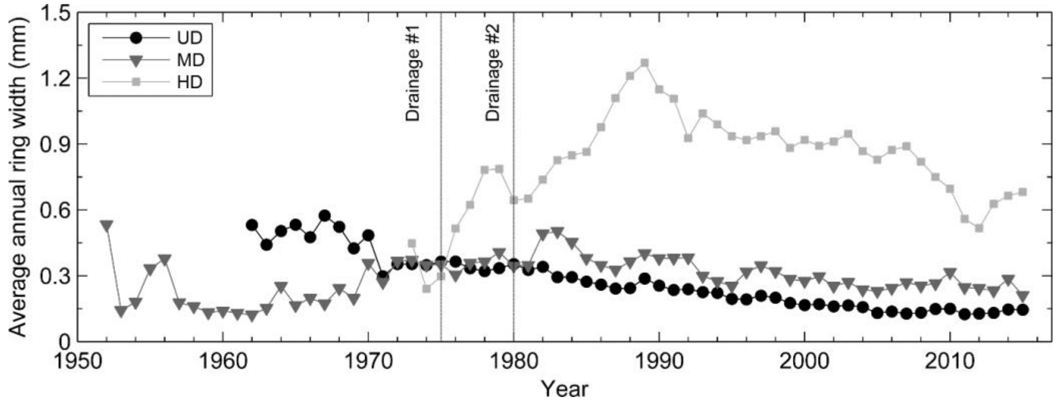


**Figure 3.2** Variation in average annual ring width with time for the undrained (UD), moderately drained (MD), and heavily drained (HD) treatments. Vertical lines show the two instances of peatland drainage at the MD and HD treatments.

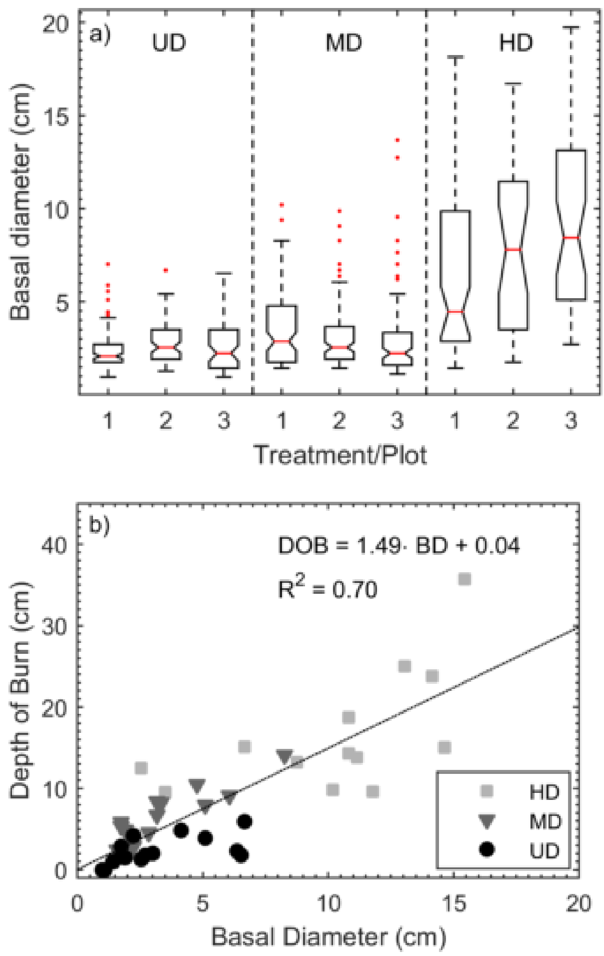


**Figure 3.3** Boxplots for basal diameter measured in each 50 m2 plot per treatment **(a)**. Depth of burn (cm) as a function of black spruce basal diameter (cm) of each tree cluster for the undrained (UD – black circle), moderately drained (MD – dark grey triangle), and heavily drained (HD – light grey square) treatments **(b)**.

# CHAPTER 4

## THRESHOLD PEAT BURN SEVERITY BREAKS EVAPORATION-LIMITING FEEDBACK

## ABSTRACT

A suite of autogenic ecohydrological feedbacks and moss traits are important for protecting vast peatland carbon stocks following wildfire disturbance. Here we examine how peat burn severity and water table depth (WTD) affect the strength of one such feedback - the hydrophobicity-evaporation feedback (HEF). The HEF is an evaporation-limiting feedback known to minimize water loss following wildfire. The peatland surface becomes hydrophobic creating an evaporative cap and thereby reducing post-fire evaporation, however, recent studies hypothesize that this is dependent on peat burn severity. To test this hypothesis, we studied plots along a peat burn severity gradient in a partially-drained black spruce peatland that burned during the 2016 Fort McMurray Horse River wildfire. Evaporation rates were significantly lower in plots where hydrophobicity was present. Hydrophobicity was lowest in the severely burned area, and the average instantaneous evaporation rate (2.75 mm day$^{-1}$) was significantly higher compared to moderately and lightly burned areas (0.82 and 1.64 mm day$^{-1}$, respectively). Based on lab results, increasing WTD affected hydrophobicity within lightly-burned (singed) feather moss samples but not in heavily-burned feather moss, showing the importance of post-fire ground cover and in-situ WTD. Our results provide evidence of a burn severity threshold where increased depth of burn removes the feather moss evaporative cap which causes the HEF to break down.

We argue that this threshold has important implications for boreal peatlands, which are predicted to undergo climate-mediated pre-fire drying and increasing burn severities, potentially leading to further carbon losses due to enhanced post-fire drying and concomitant decomposition.

## 4.1 INTRODUCTION

For millennia northern peatlands have exhibited strong ecosystem resilience, persisting in landscapes that undergo inter-annual and long-term climatic changes (Kuhry and Turunen, 2006). More recently, the ever-increasing pressures of anthropogenic disturbance and climate change (Turetsky et al., 2002), such as a doubling in the areal extent of North American boreal wildfires during the last century (Kasischke and Turetsky, 2006), has led to concern over the future resilience of northern peatland ecosystems. Given that northern peatlands store an estimated 455-547 Gt of organic carbon (C) (Yu et al., 2010), an amount similar to atmosphere C, it is imperative to understand the processes controlling ecosystem resilience in order to protect these globally significant organic carbon stocks. This is especially important in the fire-prone boreal forest of western Canada where peatlands cover approximately one-third of the land area (Vitt et al., 2000) and are exposed to wildfire disturbance on average once every 120 years (Turetsky et al., 2004). In this region, key peat-forming species (*e.g. Sphagnum* moss sp.) recolonize black spruce dominated peatlands and recover the carbon lost due to peat combustion within ~40 years following fire (Wieder et al., 2009; Ingram et. al., 2019), allowing for around 80 years of positive carbon accumulation in some hydrogeological settings (*e.g.* glaciolacustrine – expansive clay plain). With rising temperatures in northern latitudes, evapotranspiration losses are expected to increase from peatlands and could cause ecosystem drying (Thompson et al., 2017) leading to enhanced tree productivity (*e.g.* Lieffers and McDonald, 1990; Wilkinson et al., 2018). In combination with an increase in fire weather indices and fire intensity (Flannigan et al., 2013) enhanced drying and tree growth may lead to increasing and unprecedented peat burn severity in northern peatlands (*e.g.* Wilkinson et al., 2018). Therefore, it is imperative to investigate the potential thresholds of feedbacks that contribute to peatland resilience to wildfire disturbance.

Peatland resilience to wildfire is, in part, controlled by a number of ecohydrological feedbacks that act to maintain a high (shallow) peatland water table position, restricting both tree growth and peat drying (Waddington et al., 2015), therefore limiting burn severity. However, recent research has also highlighted the importance of post-fire ecohydrological feedbacks to the preservation of the remnant carbon stock and the recovery of carbon lost to fire (Kettridge et al., 2019). Peat, like other soils, and moss have been found to develop fire-induced hydrophobicity (DeBano, 2000; Moore et al., 2017). The formation of a hydrophobic layer due to the condensation of volatile compounds during a fire (Savage, 1974) tends to be concentrated just below the ground surface (1-5 cm depth) (Kettridge et al., 2014), and this layer can act as a barrier to the upward transfer of water for evaporation (Diamantopoulos et al., 2013; Kettridge et al., 2017). Since evapotranspiration tends to dominate water losses from northern peatlands (Lafleur et al., 2005; Petrone et al., 2007), and energy for evaporation increases post-fire (Thompson et al., 2015), this feedback has been argued to be crucial to limiting post-fire drying and promoting ecosystem recovery (Kettridge et al., 2017; 2019).

The degree of peat hydrophobicity is dependent on moss species, water content, and burn severity (Moore et al., 2017), where moss species and water content are also controlling factors on burn severity (Benscoter et al., 2011). Black spruce dominated peatlands have ground cover that tends to be dominated by *Sphagnum* moss species for the first 60-80 years following fire (Benscoter and Vitt, 2008; Housman, 2017), before infilling of the tree canopy leads to the dominance of feather moss species (*e.g. Pleurozium schreberi*), that are more shade tolerant (Bisbee et al., 2001). Using the simple water drop penetration time test (WDPT; Letey, 1969; Dekker et al., 2000), Moore et al. (2017) found that water repellency (a proxy for hydrophobicity) increased as moisture content decreased, and that feather moss has high water repellency under low moisture contents. Given the high moisture retention properties of *Sphagnum* species, especially hummock-forming species such as *Sphagnum fuscum* (McCarter and Price, 2014), moisture content tends to be relatively high and burn severity relatively low (Shetler et al., 2008; Hokanson et al., 2016), leading to low to no water repellency (Moore et al., 2017). Burning increased the degree of water repellency in feather moss samples, where fire-impacted feather moss often exhibited "extreme" hydrophobicity (WDPT category 5) (Moore et al., 2017). This suggests that in later-successional feather moss dominated peatlands, that tend to be impacted by wildfire due to relatively high above-ground fuel loading, the hydrophobicity-evaporation feedback will be strong. However, due to the nature of peatland succession and the layering of feather moss atop *Sphagnum* peat, there is likely a burn severity threshold whereby the moist

*Sphagnum* peat below the water-repellent feather moss peat will be exposed. Kettridge et al. (2019) found that evaporation was up to 410% higher in moderately and severely burned feather moss plots compared to low burn severity feather moss, leaving remnant peat carbon stocks vulnerable to post-fire drying. Persistently high evaporation, particularly in ombrotrophic peatlands, would be detrimental for the preservation of slow anoxic decomposition rates in peat that is usually saturated (Moore and Basiliko, 2006) and the recolonization of key moss species that require a shallow and stable water table (Price and Whitehead, 2001; Lukenbach et al., 2017).

To gain insight into the potential breakdown of this important post-fire feedback we tested the hydrophobicity-evaporation feedback in a burned, partially-drained peatland where a range of peat burn severities, from low to severe depth of burn, had previously been quantified (Wilkinson et al., 2018). We supplemented this with two lab experiments to gain a better understanding of the processes controlling peat hydrophobicity. Our objectives therefore were to quantify: 1) surface and near-surface hydrophobicity across a burn severity gradient; and 2) evaporation rates across ground cover types and burn severities. We hypothesized that moss/peat surface evaporation would be negatively related to hydrophobicity independent of site drainage treatment, and where hydrophobicity would be significantly affected by site drainage treatment.

## 4.2 METHODS

### 4.2.1 Study site

Field measurements occurred two years after the onset of fire, on 13 May 2018, at a 14 hectare black spruce dominated peatland located 11 km south of Fort McMurray, Alberta (56.56056°N, 111.318282°W). The site was divided into three separate drainage treatments in the late 1970s as part of a horticultural experiment (see Wilkinson et al., 2018 for details). One portion of the peatland was left undrained (UD), one was moderately drained (MD) with drainage ditches approximately 18 m apart, and the third portion was heavily drained (HD) with drainage ditches approximately 9 m apart. In 2016 the site was burned during the Fort McMurray Horse River wildfire. The UD and MD treatments experienced relatively low and moderate depth of burn (2.5 ± 3.5 cm, 6.4 ± 5.0 cm, respectively; Wilkinson et al., 2018), similar to natural peatlands in the region (Hokanson et al., 2016). Whereas the HD treatment experienced relatively high depth of burn (36.9 ± 29.6 cm; Wilkinson et al., 2018).

### 4.2.2 Ground cover and burn severity classification

Site surface cover was visually assessed using a 50 cm x 50 cm quadrat at 15 randomly selected plots within each of the three treatments using a classification system that accounts for moss burn severity and ground cover species (see Lukenbach et al., 2015 for full details). This process was used to determine representative ground cover classes across each of the treatments. Briefly, if moss capitula/moss structure is intact but there is evidence of burning, the area is classified as lightly-burned (singed); if capitula/structure have been damaged considerably the area is classified as heavily-burned (burned). Hence, the two dominant moss genera in the peatland are separated into singed *Sphagnum* (SSph) and burned *Sphagnum* (BSph), and singed feather moss (SFM) and burned feather moss (BFM). Live *Sphagnum* (LSph), bare peat (BP), fire moss (*Ceratodon purpureus*)/liverwort (FrM), pool and litter ground cover were also included. Litter surface consisted of mainly wood horsetail (*Equisetum sylvaticum*) detritus and pool ground cover type refers to areas of the ground surface with current (or evidence of) ponded water. Since mosses tended to be impacted by fire, identification past genus (*Sphagnum* and feather moss) level was not possible, however, it is noted that *Sphagnum* mosses were almost exclusively hummock-forming, and feather moss generally resided in the intermediate and hollow areas.

4.2.3 Hydrophobicity, evaporation rate and volumetric water content

Three to five plots were randomly selected for each of the ground cover classes determined in each treatment (Table 1). At each plot water drop penetration time (WDPT; Letey, 1969; Dekker et al., 2000) tests and evaporation measurements were made, along with a measure of the volumetric water content (VWC – see *Water table manipulation lab experiment*) in the top 6 cm of the moss/peat. Evaporation measurements were made using a small (15.2 cm tall by 15.2 cm diameter) cylindrical plexiglass mobile chamber. The chamber housed a small fan for circulation, powered by a 12-volt battery, and a Hygrochron™ iButton® DS1923 by Maxim Integrated™ which measured and logged relative humidity (RH) and temperature every 2 seconds (accurate to 0.6 % RH, 0.5°C). The chamber was placed on each plot and imbedded within the surface moss for a two-minute measurement cycle. If the measurement was on ponded water the chamber was held in contact with the water surface for two minutes. Between plot measurements, the chamber was left open towards the prevailing wind for a minimum of one minute in order to allow chamber head space to return to ambient conditions. RH and temperature values were then used to determine vapour pressure and instantaneous evaporation rates for each plot. Due to the short measurement cycle and relatively small temperature changes during measurements (range, max or average), we assumed chamber air pressure equaled ambient atmospheric pressure. Mean evaporation rates for ground cover types within each treatment area were then multiplied by the mean ground cover fraction to determine a contribution of evaporation for each ground cover type within each treatment (UD, MD and HD).

The WDPT test, which is a visual assessment based on the time for infiltration of water drops on the surface, has been used extensively for assessing the hydrophobicity of mineral soil (Doerr et al., 2000) and was found to be as effective as the molarity of ethanol test (Watson and Letey, 1970) for moss and peat (Kettridge et al., 2014). The WDPT test was modified for field measurements by setting the maximum measurement period at ten minutes, rather than one hour. Five small droplets of deionized water were placed on the surface using a pipette during field measurements. Based on the time taken to infiltrate into the surface each drop was categorized into one of five classes of hydrophobicity (1 – <5 s = hydrophilic, 2 – 5-59 s = slightly hydrophobic, 3 – 60-599 s = strongly hydrophobic, 4 – 600-3599 s = severely hydrophobic, and 5 – ≥3600 s = extremely hydrophobic (not used for field measurements); Dekker, 2000). Hydrophobicity was measured at 0, 2, and 5 cm below the ground surface in each plot. Layers were successively removed using scissors to expose underlying moss/peat. Average WDPT category was calculated by multiplying the number of drops within each category by its respective value, adding that together and dividing by the total number of drops used.

### 4.2.4 Water table manipulation lab experiment

To investigate links between water table depth (WTD), moss type, burn severity, and hydrophobicity, twelve samples, three of each: burned feather moss (BFM), singed feather moss (SFM), burned *Sphagnum* (BSph), and singed *Sphagnum* (SSph), were collected from the MD treatment in late May of 2018. The samples were frozen to prevent degradation until the experiment was conducted. The samples were approximately 20 x 20 x 20 cm in size, once thawed they were saturated completely for approximately 72 hours and placed into plastic containers to only allow water to escape through evaporation. The WTD was maintained at 5 cm for one week (phase 1) then 15 cm for one week (phase 2), and then the samples were completely drained from the bottom and no water was added for one week (phase 3). For the final week of the experiment the samples were rewetted to return the WTD to 5 cm to assess potential hysteretic effects (phase 4). Daily measurements of hydrophobicity using the WDPT test, and near-surface moisture content were conducted. For WDPT tests, ten drops were placed on the moss surface of each sample and the full range of WDPT categories we used (*i.e.* 1-5). VWC measurements of the top 3 cm were made for each sample using an ML3 ThetaProbe by Delta-T Devices (accurate to 1 % VWC). mV readings were recorded and then converted to VWC using empirically derived calibration curves (Lukenbach et al., 2017). Following the controlled experiment, samples were oven dried at 65°C for 72 hours and then weighed to obtain dry weight in order to calculate gravimetric water content (GWC). Mass lost due to evaporation was also measured and replaced daily to maintain a constant WTD during each phase. Air temperature and RH within the lab were measured throughout the experiment using the iButton™.

4.2.5 Drying lab experiment

We used singed *Sphagnum* and singed feather moss samples to examine the onset of hydrophobicity in relation to GWC, and compare potential differences in WDPT-GWC relations between treatment. Five replicate samples measuring roughly 15 × 15 × 5 cm were collected from both the UD and MD treatments. Equivalent samples were not collected from the HD treatment because those classifications were not present (see Table 1). Samples were air dried in a growth chamber (mean temperature and relative humidity of 20°C and 65 %, respectively) for three weeks. WDPT tests were conducted daily and were the same as in *Water table manipulation lab experiment.* Sample weight was measured daily throughout the air drying and samples were then oven-dried at 65°C for 72 hours to obtain dry weight to calculate GWC.

Results are presented as arithmetic mean and standard deviation unless otherwise stated. A one-way ANOVA with the Tukey post-hoc test was used to test for significant differences in treatment level instantaneous evaporation rate. A non-parametric t-test (Kruskal Wallis) was used to test for a significant difference in evaporation rate, and in moisture content, between plots where hydrophobicity was present and not present. Non-parametric multiple comparison tests (Dunn's method) were used to test for significant differences between evaporation rate from different ground cover types due to the relatively small sample sizes. All statistical operations were performed in Matlab © version 2017b (Matlab, 2017).

## 4.3 RESULTS

### 4.3.1 Hydrophobicity

Hydrophobicity was highest at the HD litter surface with a modal WDPT category of 4 (severely hydrophobic - 88% of drops, minimum WDPT=3), showing severe hydrophobicity, however, at depths of 2 and 5 cm hydrophobicity was non-existent (Figure 1). The HD area showed only slight hydrophobicity within bare peat (BP) having modal WDPT categories of 2 (slightly hydrophobic – 76% of drops, max WDPT=2), 1 (hydrophilic - 80% of drops, max WDPT=2), and 1 (80% of drops, max WDPT=3) for the surface, 2 cm, and 5 cm, respectively (where a WDPT category of 1 is hydrophilic). The UD and MD sites showed relatively low hydrophobicity at the surface compared to 2 and 5 cm depths, with the greatest hydrophobicity occurring at the 2 cm depth for most ground cover types (Figure 1). The greatest hydrophobicity amongst the UD and MD treatments was observed in feather moss, specifically SFM at 2 cm depth, which had modal WDPT categories of 2 (33% of drops) and 3 (strongly hydrophobic - 47% of drops), respectively. BFM within both the UD and MD treatments also showed elevated levels of hydrophobicity at 2 cm depth with modal WDPT categories of 3 (53% of drops) and 2 (33% of drops) respectively. Conversely, *Sphagnum* tended to be less hydrophobic, where within *Sphagnum* ground cover classes, BSph showed greater hydrophobicity than SSph and LSph (Figure 1).

4.3.2 Evaporation

Evaporation rates in plots with hydrophobicity present at any of the measured depths were significantly lower than the evaporation rates from plots where no hydrophobicity was present ($\chi^2_{1,46}$ = 13.25, $p < 0.01$) (Figure 2), averaging 0.8 ± 0.2 mm day$^{-1}$ and 2.9 ± 0.4 mm day$^{-1}$, respectively. Similarly, plots with hydrophobicity present tended to be drier in the near-surface (upper 6 cm) with volumetric water content (VWC) of 0.04 ± 0.02 m$^3$ m$^{-3}$ compared to 0.31 ± 0.04 m$^3$ m$^{-3}$ ($\chi^2_{1,43}$ = 9.75, $p < 0.01$). Overall, plot evaporation was significantly affected by VWC ($F_{39}$ = 10.1, $p < 0.01$), although the explained variance was low ($R^2$ = 0.21) (Supplementary Material; Figure S4.1), where VWC tended to be much higher at HD (0.49 ± 0.04 m$^3$ m$^{-3}$) compared to UD (0.16 ± 0.03 m$^3$ m$^{-3}$) and MD (0.09 ± 0.04 m$^3$ m$^{-3}$). Evaporation was lowest on average, at MD SFM plots with a mean of 0.3 ± 0.3 mm day$^{-1}$, mean VWC of 0.03 m$^3$ m$^{-3}$, and tended to be strongly hydrophobic at 2 cm depth. Evaporation was highest at MD SSph plots with a mean of 5.4 ± 0.6 mm day$^{-1}$, mean VWC of 0.12 m$^3$ m$^{-3}$, and was consistently hydrophilic at all measured depths. LSph and SSph showed similar evaporation rates in the UD (mean = 3.6 and 2.6 mm day$^{-1}$, respectively), whereas SFM and BFM were much lower (mean = 0.48 and 1.3 mm day$^{-1}$, respectively). Similarly, SSph in the MD had a much greater evaporation rate than SFM (0.28 ± 0.47 mm day-1) and BFM (0.37 ± 0.33 mm day$^{-1}$), however, BSph had low (but variable) evaporation rates (1.1 ± 1.2 mm day$^{-1}$). In the HD treatment, evaporation rates were relatively high in both Pool and FrM plots (mean = 4.2 and 3.7 mm day$^{-1}$, respectively), where VWC was also high. Overall, SFM plots had significantly lower evaporation rates than SSph, Pool and FrM, and BFM plots had significantly lower evaporation rates than SSph and Pool (Supplementary Material; Table S4.1).

Accounting for the percent cover of each ground cover type (Table 1), the mean instantaneous evaporation rate was highest within the HD treatment with an evaporation rate of 2.75 mm day$^{-1}$, followed by the UD site at 1.64 mm day$^{-1}$, and lowest with the MD site at 0.82 mm day$^{-1}$ (Figure 3a), where HD was significantly greater than UD and MD ($F_{6.4}$, $p < 0.05$). The ground cover within the HD treatment was characteristically different from the ground cover found within the UD and MD treatments, and was primarily litter (47 %) and FrM (37 %). Although litter was the dominant HD ground cover, it should be noted that the litter layer was sporadic in coverage. Given the high proportion of litter and FrM, and the relatively high evaporation rate from FrM plots, these ground cover types were also the dominant contributors to evaporation in the HD treatment, with 38 % and 50 %, respectively. Within the MD treatment, ground cover was dominated by BFM (62 %), however BFM only contributed 29 % of the total evaporation, while the dominant evaporative surface was SSph, contributing 40 % whilst only covering 7 % of the ground surface. SSph in the UD covered a similar proportion (9 %) but contributed only 19 % of evaporation. Within the UD treatment a similar trend is seen with BFM as was seen in the MD treatment – with BFM constituting a large proportion of ground cover (47 %) but contributing relatively less of the evaporation (37 %) (Figure 3b). SFM contributes very little to the evaporation in either the UD or MD treatments (Figure 3b).

4.3.3 Effect of water table depth and water content on hydrophobicity

In the water table manipulation lab experiment volumetric water content (VWC) of the top 3 cm decreased as the water table was lowered. All samples had a mean VWC of 0.3 – 0.4 $m^3$ $m^{-3}$ when the WTD was 5 cm. BFM and SSph mean VWC was similar for phase 2 and 3 (15 cm WTD and no water table, respectively). BSph VWC remained slightly higher throughout, whereas SFM had lowest mean VWC for phase 2 and 3 (Supplementary Material; Figure S4.2). Similar trends were maintained when VWC was converted to gravimetric water content (GWC) yet the effect of species became more apparent as BFM and SFM had mean GWC of < 2-3 g $g^{-1}$ for 15 cm WTD and with no water table (Supplementary Material; Figure S4.3).

Similar to field-measured hydrophobicity (Figure 1), BSph and SSph tended to be hydrophilic at all stages of the water table manipulation experiment (data not shown). BFM samples tended to stay slightly hydrophobic throughout the water table manipulation with a median WDPT category of 2 (27% of drops; 95% ≤ WPDT of 3), and with little change in average WDPT associated with changes in WTD (Figure 4). Conversely, SFM showed increased hydrophobicity with an increasing WTD. The proportion of WDPT classified as slightly to moderately hydrophobic increased, and modal WDPT categories were 1 (71% of drops), 1 (53% of drops), and 2 (42% of drops) as WTD increased between phases 1-3 (Figure 4). In phase 3 a greater proportion of SFM WDPT exhibited hydrophobicity compared to BFM, and upon rewetting (phase 4) there was a delay of 3 days before average WDPT decreased substantially. While the average WDPT category during phase 4 (1.9 ± 0.10) was higher than the average during the first two phases (5 and 15 cm WTD), the average WDPT after several days of re-wetting was commensurate with values measured throughout phase 1 (5 cm WTD) (Figure 4). Moss drying rates, under controlled conditions in the drying lab experiment, were characteristically different between SSph and SFM, with average differences between treatments being negligible (Figure 5a). In general, hydrophobicity was not observed unless GWC was less than 2-3 g $g^{-1}$ (Figure 5b). On average, SFM samples would reach a GWC of 2-3 g $g^{-1}$ after 4-6 days of drying, while SSph would take 9-11 days.

## 4.4 DISCUSSION

Ground cover types were similar in the undrained (UD) and moderately-drained (MD) treatments, and represent a typical recovery trajectory following wildfire for *Sphagnum*-feather moss co-dominated peatlands (Benscoter and Vitt, 2008) where post-fire ground cover is a result of both pre-fire ground cover and burn severity (Benscoter et al., 2005). Because feather moss species thrive in shaded conditions compared to *Sphagnum* moss species (Bisbee et al., 2001) we suggest that pre-fire feather moss dominance is likely to have been correlated with degree of drainage (*i.e.* UD < MD < HD) across treatments. Burn severity varied within and between treatments and in general increased with the intensity of drainage and stand density/basal area. Here, the HD treatment lost > 50 % of pre-fire peat carbon stock, representing extreme peat burn severity (depth of burn 36.9 ± 29.6 cm; Wilkinson et al., 2018). The differences in peat burn severity between treatments are reflected in the ground cover types. Specifically, the HD treatment differs from the UD and MD treatments; consisting of pools, bare peat, and some recovery of fire moss (*Ceratodon* *purpureus*) and liverwort, with a surface covering of wood horsetail and leaf litter. These results are similar to the lack of moss recovery found one-year post-fire from a separate wildfire in another drained and burned peatland in the region (Kettridge et al., 2015). The UD and MD treatments remain *Sphagnum*-feather moss co-dominated but the proportion of each, and moss burn severity (singed, burned), differs (Table 1).

Burned and singed feather moss peat tended to exhibit the greatest degree of hydrophobicity compared to *Sphagnum* (Figure 1) especially at a depth of 2 cm which is similar to the findings of Kettridge et al. (2014). Burned moss tends to have greater hydrophobicity than unburned (Kettridge et al., 2014; Moore et al., 2017), and further subdividing plots into burn severity classes (singed, burned) we found that singed feather moss had the highest average WDPT, including occurrences of severe hydrophobicity (Figure 1). The degree of hydrophobicity in soils is determined by the chemical changes induced by the temperature and duration of heating (DeBano, 2000), where intermediate temperatures (175 to 200°C) during soil heating have been associated with the occurrence of the most intense water repellency (March et al., 1994). Moreover, some studies have also documented an increase in burn severity causing a decrease in hydrophobicity, in peat and mineral soil (*e.g.* Elmes et al., 2019), and this has been attributed to overlying soils reaching combustion temperatures of > 300°C (Savage et al., 1974). We found that hydrophobicity, at the surface and at depth (2 and 5 cm), was generally lower in the HD treatment compared to the UD and MD, where UD and MD were similar (Figure 1) despite differences in mean depth of burn (Wilkinson et al., 2018).

Moore et al. (2017) found that there is a threshold gravimetric water content (GWC) whereby the degree of hydrophobicity increases in both burned and unburned moss, which suggests that the hydrophobicity exhibited could be a manifestation of a difference in water content of the different species/burn severity classes. When BSph, SSph, BFM, and SFM were subjected to a constant water table depth of 5 cm and then 15 cm, the feather moss samples had lower water contents in the top 3 cm (Supplementary Material, Figure S4.2) compared to the *Sphagnum* samples. Our drying experiment found that hydrophobicity was exhibited at GWC of less than 2 – 3 g $g^{-1}$ (Figure 5b) which was reached in BFM and SFM samples when WTD was just 5 cm (Supplementary Material, Figure S4.3). Moreover, the threshold GWC of ~1.4 g $g^{-1}$ (Moore et al., 2017), was reached by BFM and SFM samples when WTD was 15 cm for a few days, a typical (if not shallow) WTD in post-fire peatlands for the Boreal Plains ecoregion (Lukenbach et al., 2015). Conversely, the SSph and BSph samples always maintained a GWC above the threshold and correspondingly did not show any hydrophobicity. Furthering the work of Moore et al. (2017) when WDPT was assessed on both BFM and SFM with different WTD, we found that the degree of hydrophobicity at the surface of BFM did not respond to changes in WTD or the concomitant decrease in water content. However, hydrophobicity exhibited by SFM increased substantially as WTD increased and water content decreased (Figure 4). The differing responses of the ground cover types to WTD likely explain the elevated levels of hydrophobicity measured at the SFM ground cover plots under field conditions in UD and MD treatments (Figure 1).

Instantaneous evaporation rates showed greater variability between ground cover types than between treatments, where ground cover was only comparable between UD and MD treatments. *Sphagnum* evaporation rates ranged from 1.1 – 5.5 mm day$^{-1}$ where SSph tends to be greater than BSph. Evaporation rates from feather moss were much lower than *Sphagnum*, especially SFM, ranging from 0.28 – 0.48 mm day$^{-1}$, comparable to Bond-Lamberty et al. (2011) and Kettridge et al. (2017). Conversely, BFM sometimes had higher evaporation rates than unburned feather moss (Bond-Lamberty et al., 2011) showing that increasing burn severity in feather moss dominated peatlands can increase evaporation rates compared to unburned ecosystems (Thompson et al., 2014). Furthering this change is the potential to burn through the total depth of feather moss peat, exposing *Sphagnum* peat and/or mineral soil below.

Areas of higher burn severity undergo greater depths of burn which effectively lowers the ground surface, bringing it closer to the water table (reducing WTD), increasing near-surface moisture content and increasing evaporation rates (Figure 3a). Depth of burn was much greater in the HD treatment and up to 60 cm of peat was removed from the surface in some places (Wilkinson et al., 2018), leading to some areas of open surface water (classified, in this study, as a Pool ground cover type). The creation of pools increases the bulk specific yield of the ground cover and may help regulate water table fluctuations (Price et al., 1996). However, depending on the efficiency of lateral water movement, pools might represent a net loss of water due to their relatively high evaporation rate. Yet pools occupied only a relatively small percent cover in the HD treatment (Figure 3b). The lack of hydrophobicity across much of the HD treatment provides surface conditions more conducive to the recolonization of pioneer moss species such as fire moss just two years post-fire, which provided the greatest contribution to HD evaporation (Figure 3b), followed closely by the expansive cover of bare peat and litter (Table 1).

Given the low burn severity (maintenance of the moss structure and capitula) and the ability of *Sphagnum* mosses to regenerate in-situ from a disturbed state, it is unsurprising that SSph is functioning similarly to LSph 2 years post-fire, and making a substantial contribution to evaporation in the UD and MD (Figure 3b). Conversely, SFM has the highest levels of hydrophobicity and significantly lower evaporation rates than pools, FrM and SSph, where its rate is only 10 % of that of SSph, and it contributes very little to evaporation in UD and MD treatments. Hence, feather moss areas experiencing only negligible depth of burn, may exhibit the lowest moisture contents and greatest hydrophobicity, reducing their favourability for recolonization by moss species, but acting to reduce evaporative losses (Figure 3a).

The reduced evaporation rates from SFM and BFM provide evidence for the persistence of the hydrophobicity-evaporation feedback (Kettridge et al., 2014) in the UD and MD treatments, resulting in significantly lower evaporation than in the HD treatment (Figure 3a). In the UD and MD, where field measures are coupled with low water content in the near-surface this feedback is strongest, supporting findings from laboratory-based studies by Moore et al. (2017). Unlike laboratory-based studies, the presence of hydrophobicity in SSph tends to have little impact on evaporation rates, potentially because of the regeneration of *Sphagnum* capitula and their comparatively high water content. Nonetheless, the hydrophobicity-evaporation feedback is effectively limiting evaporative losses (Figure 2) in the UD and MD treatments in this crucial recovery period immediately following wildfire (Figure 3a). SSph and SFM water retention characteristics were not affected by treatment (Figure 5a). Hence, the difference in evaporation rates, within ground cover types, between UD and MD treatments (MD-SFM < UD-SFM) is likely a function of WTD and near-surface water content. By manipulating the WTD in a controlled experiment, we found that SFM became substantially more hydrophobic under a WTD of 15cm and above, whereas BFM hydrophobicity showed little response to water table manipulation (Figure 4). This demonstrates the importance of water content on hydrophobicity and evaporation, and the effect of the relative proportions of ground cover classifications even within the same species, given differences in burn severity.

### 4.4.1 Implications for peatland wildfire recovery

We argue that the HD treatment surpassed a burn severity threshold whereby the post-fire ground cover exhibited little hydrophobicity and consequently evaporation rates are significantly higher than in UD and MD treatments, and rates are similar to the average values of the high burn severity plots of Kettridge et al. (2019). The breakdown of the hydrophobicity-evaporation feedback is mainly due to the loss of the hydrophobic feather moss cap, and puts the system at risk to post-fire drying whereby the water table is drawn down into the peat or underlying mineral soil (Figure 6). This has implications for the peatland carbon balance through an increase in peat decomposition rates as peat, which has been generally saturated for millenia, is oxidised and undergoes faster aerobic decomposition (Moore and Basiliko, 2006). Moreover, since dense peat is exposed at the surface after fire (Sherwood et al., 2013), soil-water tension will likely increase greatly in response to water table drawdown causing unfavourable conditions for the recolonization of the key peat-forming *Sphagnum* mosses (Price and Whitehead, 2001; Thompson and Waddington, 2013). Deciduous tree encroachment and shrubification of previously black spruce dominated peatlands are also a concern in these circumstances (Johnstone and Kasischke, 2005). The exposure of mineral soil and damaged native seed bank provide ample opportunities for colonization by deciduous trees and shrubs, as seen in the black spruce forests of interior Alaska (Johnstone and Kasischke, 2005) and immediately post-fire in Alberta (Depante et al., 2019). We expect that under current climatic conditions, the HD treatment is likely to undergo an ecosystem regime shift, similar to that described by Kettridge et al. (2015), resulting in a lack of moss recovery, smaller (or negative) carbon uptake, and a higher fire frequency (Figure 6).

In light of the predicted increase in area burned (Flannigan et al., 2005), fire weather indices (Wang et al., 2015) and fire season length in boreal regions (Flannigan et al., 2013), black spruce peatlands will likely be affected by wildfire disturbance more frequently and severely. Although, in Canada specifically, the proportion of drained peatlands is lower than other boreal regions (*e.g.* in northern and eastern Europe), an increase in the severity and duration of drought may initiate positive (self-reinforcing) feedbacks that act to enhance peatland drying leading to unprecedented peat burn severity (Wilkinson et al., 2018). Adaptive management and restoration practices (*e.g.* rewetting with *Sphagnum* transplant and/or moss transfer techniques) (see Rochefort et al. 2003) will be crucial in order to negate the initiation of positive feedbacks, and re-establish stable, shallow water table positions in disturbed black spruce peatlands, promoting ecosystem recovery and protecting remnant carbon stocks.

By assessing hydrophobicity and evaporation rates from a black spruce peatland under a range of burn severities we have evidenced the existence of a threshold whereby the hydrophobicity-evaporation feedback breaks down, lending way to a multitude of processes which may negatively affect peatland ecohydrological function in the short term, and the ability to recover these important functions in the long-term. Further work should aim to develop and evaluate adaptive peatland management and restoration techniques to reduce peat burn severity, as well as test the strength of other ecohydrological feedbacks that provide resilience to the increasing threat of wildfire disturbance in black spruce dominated peatlands.

**Acknowledgements**

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## Tables

**Table 4.1** Percent ground cover in each treatment, where ground cover types are: Burned feather moss (BFM), Singed feather moss (SFM), Burned *Sphagnum* (BSph), Singed *Sphagnum* (SSph), Live *Sphagnum* (LSph), Fire Moss/ Liverwort (FrM), Bare Peat (BrP), Litter, and Pool.

| Treatment | LSph | SSph | BSph | SFM | BFM | FrM | Litter | BrP | Pool |
|---|---|---|---|---|---|---|---|---|---|
| Undrained | 13.3 | 9.0 | 22.0 | 9.0 | 46.7 | 0 | 0 | 0 | 0 |
| Moderately drained | 2.0 | 5.0 | 14.2 | 16.4 | 62.4 | 0 | 0 | 0 | 0 |
| Heavily drained | 0 | 0 | 0 | 0 | 0 | 37.3 | 47.0 | 13.3 | 2.4 |

## List of Figures

**Figure 4.1** Summary of WDPT tests for surface type measured within the field site for 3 depths; 0 cm, 2 cm, and 5 cm. Coloured bars show percentage of water drops that infiltrated the surface in: <5 s (1 – hydrophilic), 5-59 s (2 – slightly hydrophobic), 60-599 s (3 – strongly hydrophobic), and 600+ s (4 – severely to extremely hydrophobic).

**Figure 4.2** Evaporation rate (mm day$^{-1}$) for plots where hydrophicity is present (WDPT category > 1) in any of the near-surface layers (ie. 0, 2, and 5 cm depths) and plots where hydrophobicity was not present (WDPT category = 1)

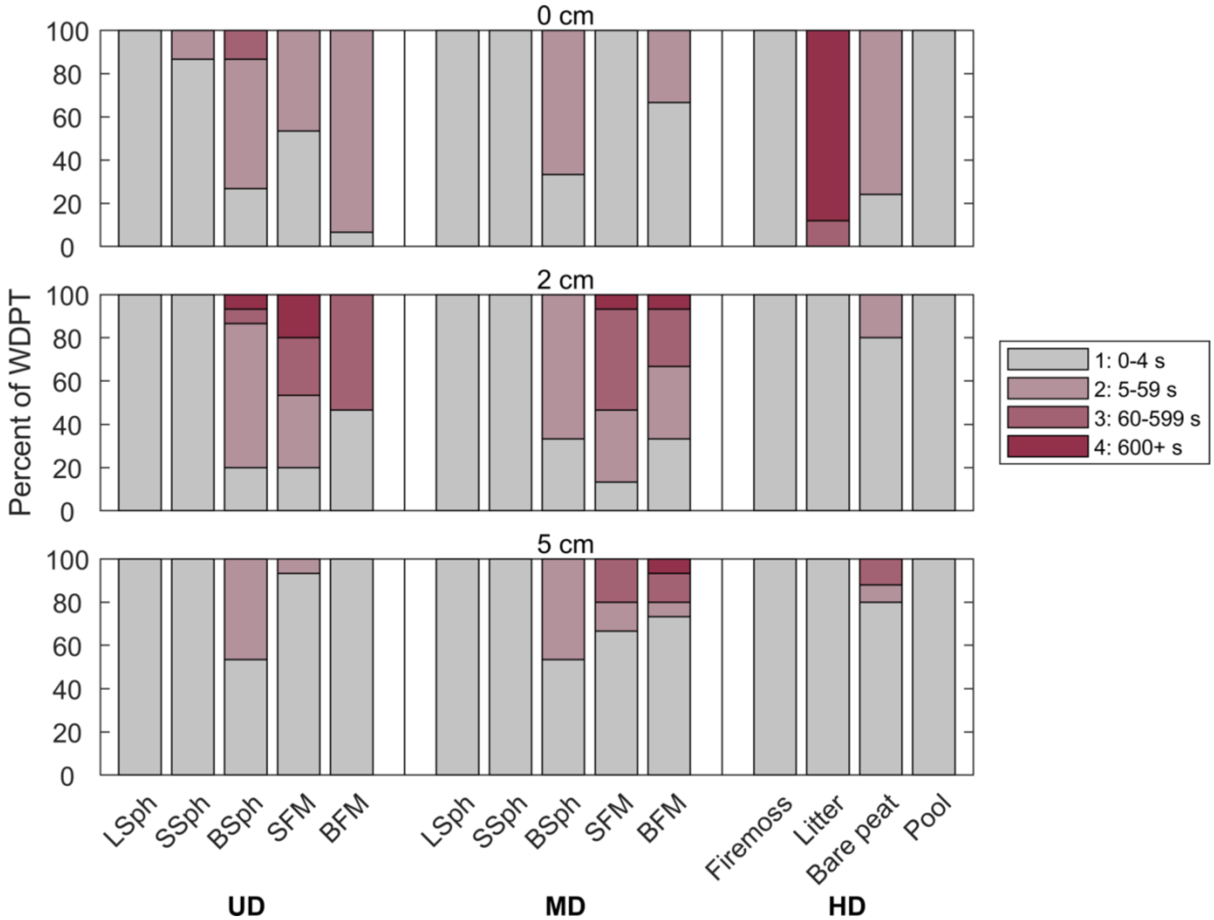
**Figure 4.3** (a) Evaporation rate for each treatment subdivided into the various components contributed by the various ground covers. Contribution calculated by multiplying average evaporation rate by percent cover for each surface type. (b) Percent ground cover and percent of total evaporation for each ground surface for each treatment.

**Figure 4.4** Summary of surface WDPT tests for BFM and SFM from the controlled lab experiment. Coloured bars show percentage of water drops that infiltrated the surface in: <5 s (hydrophilic), 5-59 s (slightly hydrophobic), 60-599 s (strongly hydrophobic), 600-3599 s (severely hydrophobic), and 3600+ s (extremely hydrophobic). BSph and SSph tended to be hydrophilic throughout the experiment and have been omitted here.

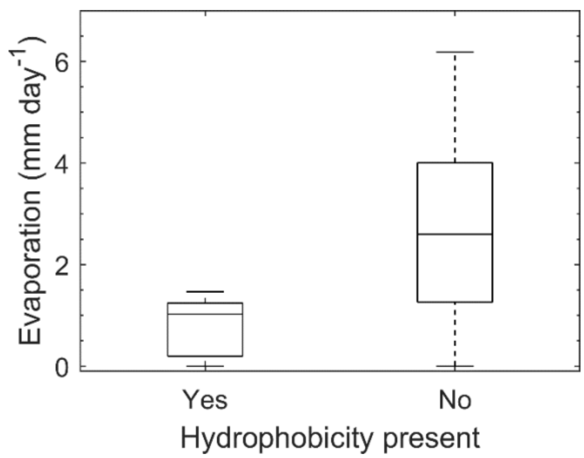
**Figure 4.5** Relation between lab measured gravimetric water content (GWC) and water repellency category for SSph and SFM at undrained (UD) and moderately-drained (MD) sites.

**Figure 4.6** Conceptual figure showing the breakdown of the hydrophobicity-evaporation feedback (HEF) based on depth of burn (DOB). ET stands for evapotranspiration.

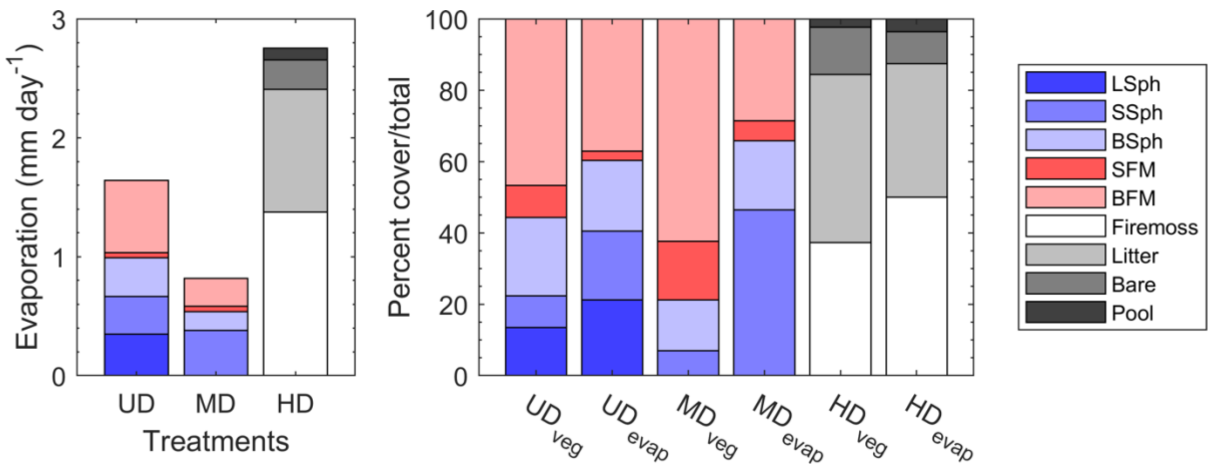
## Figures




[Figure 4.1]

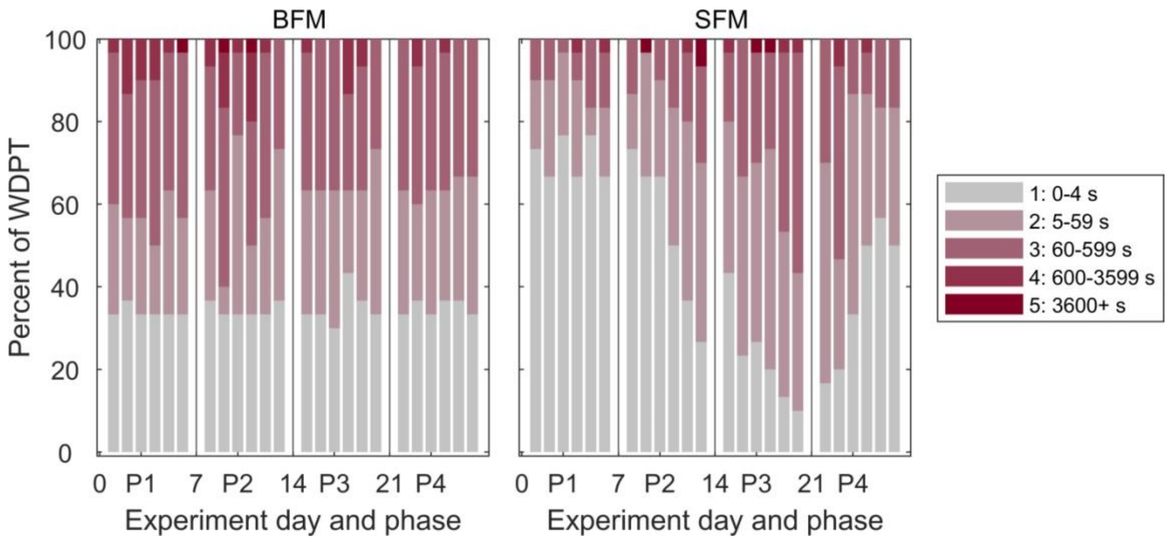



[Figure 4.2]

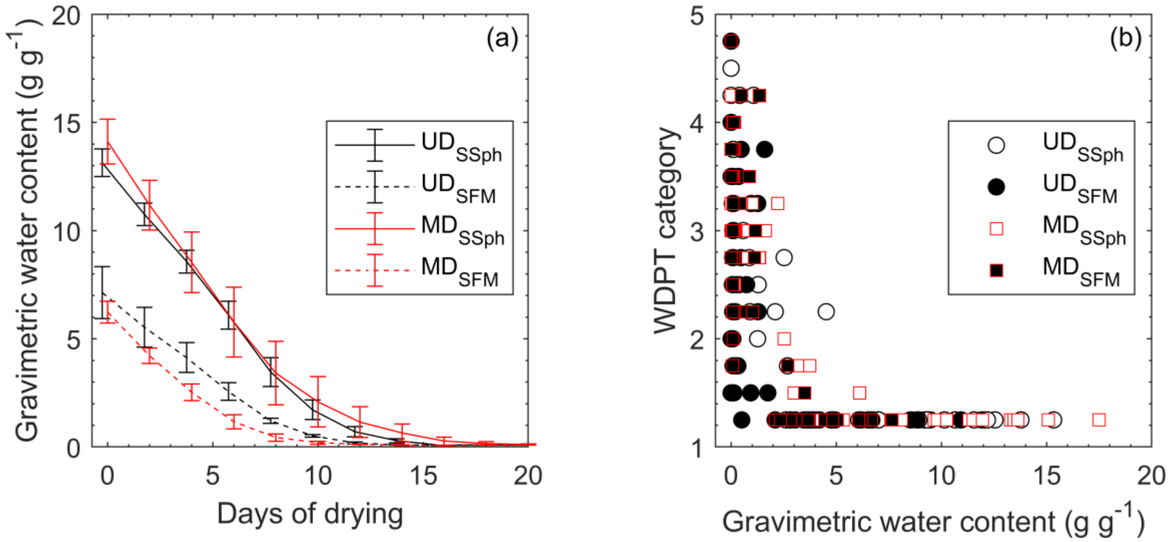


Evaporation (mm day$^{-1}$)
3
2
1
0
UD
MD
HD
Treatments
Percent cover/total
100
80
60
40
20
0
UD veg
UD evap
MD veg
MD evap
HD veg
HD evap
LSph
SSph
BSph
SFM
BFM
Firemoss
Litter
Bare
Pool


[Figure 4.3]

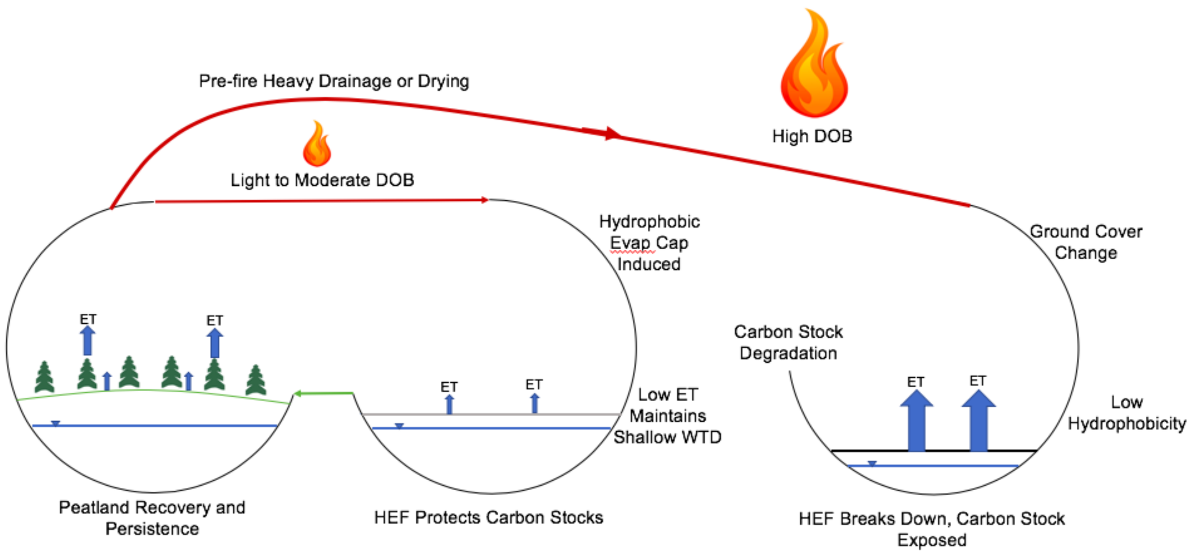

BFM
SFM
Percent of WDPT
100
80
60
40
20
0
0 P1 7 P2 14 P3 21 P4
Experiment day and phase
0 P1 7 P2 14 P3 21 P4
Experiment day and phase
1: 0-4 s
2: 5-59 s
3: 60-599 s
4: 600-3599 s
5: 3600+ s

[Figure 4.4]



(a)
Gravimetric water content (g g-1)
Days of drying
UDSSph
UDSFM
MDSSph
MDSFM
(b)
WDPT category
Gravimetric water content (g g-1)


[Figure 4.5]


Pre-fire Heavy Drainage or Drying
High DOB
Light to Moderate DOB
Hydrophobic
Evap Cap
Induced
Ground Cover
Change
Carbon Stock
Degradation
ET
ET
ET
ET
ET
ET
Low ET
Maintains
Shallow WTD
Low
Hydrophobicity
Peatland Recovery and
Persistence
HEF Protects Carbon Stocks
HEF Breaks Down, Carbon Stock
Exposed

[Figure 4.6]

# CHAPTER 5

## THE EFFECTS OF BLACK SPRUCE FUEL MANAGEMENT ON SURFACE FUEL CONDITION AND PEAT BURN SEVERITY IN AN EXPERIMENTAL FIRE


## ABSTRACT

In the boreal plains ecozone, black spruce (*Picea mariana*) peatlands can represent large parts of the expanding wildland-urban interface (WUI) and wildland-industry interface (WII). The boreal plains wildfire regime is predicted to increase in areal extent and intensity, amplifying the need for wildfire management to protect the WUI and WII. Forested peatland ecosystems can burn at high intensity and present challenges for wildfire managers such as severe smouldering combustion and large carbon loss. Fuel management techniques such as mulching treatments (converting surface and canopy fuel to a masticated fuelbed) can be applied to black spruce peatlands, yet the impact on fuel load, condition and peat burn severity is unclear. Using observations from an experimental fire we found that a mulch-thinning fuel-treatment could reduce peat depth of burn. However, where peat bulk density was increased by compaction, this led to an increased peat combustion carbon loss relative to the control. Furthermore, near-total combustion of the mulch layer resulted in significantly more surface fuel carbon emission from thinned and stripped fuel-treated areas compared to the control. We argue that while fuel-treatment may benefit smouldering combustion suppression efforts, surface fuel carbon loss should be considered before treatments are implemented on a large scale.

## 5.1 INTRODUCTION

The areal extent, frequency and intensity of wildfires across Canada's boreal region are predicted to increase under future climate scenarios (Flannigan *et al.*, 2005, 2013), while urban and industrial land-use area continues to expand (Robinne *et al.*, 2016). Consequently, the need for effective wildfire management in the wildland-urban interface (WUI) and wildland-industry interface (WII) is increasing (Flannigan *et al.*, 2009; Seto *et al.*, 2011). The boreal plains ecozone (BP) spans continental western Canada where wildfire is the dominant disturbance (Turetsky *et al.*, 2002) and 21 % of the land area is covered by peatlands (Vitt *et al.*, 2000). This is especially important because while peatlands can act as fire-breaks on the landscape (*e.g.* Shetler *et al.*, 2008), recent research has shown they can not only sustain high intensity fires (Johnston *et al.*, 2015) but may aid in the development of large fire events (Flannigan *et al*., 2009). The densely treed, black spruce (*Picea mariana*) dominated peatlands present the greatest peatland fire management challenge in the BP because their total fuel accumulation is comparable to upland forest stands (Thompson *et al.*, 2017) and they are the dominant peatland type by area (ABMI, 2016). Peatlands in the BP can sustain dense black spruce stands (>6000 stems/ha) on peat over 3 m deep and can accumulate canopy fuels capable of carrying a crown fire after ~80 years since fire (ysf) (Johnston *et al.*, 2015), in a landscape where the average fire return interval is 120 years (Turetsky *et al.*, 2004).

The ground surface of mature (>80 ysf) black spruce peatlands is often dominated by feathermosses that have poor water retention and can become extremely dry and hydrophobic (Moore *et al.*, 2017), causing them to be readily flammable. Depending on the peat fuel load and condition (*i.e.* peat bulk density and moisture content), the heat energy input from fire can lead to the propagation of smouldering combustion (Van Wagner, 1972; Benscoter *et al.*, 2011). Where peat is dense and moisture content is low, peatlands can sustain smouldering combustion >1 m deep into the peat profile (Lukenbach *et al.*, 2015; Wilkinson *et al.*, 2018). Smouldering combustion occurs at a lower temperature (~300 °C) than flaming combustion (>800 °C), however, slow movement, long residence times and vertical penetration into the peat profile can lead to challenging suppression scenarios and high resource demand for suppression (*e.g.* Davies *et al.*, 2013). Peat smouldering also releases particulate matter, carbon dioxide and trace metals into the air causing both visibility and air quality issues (Shaposhnikov *et al.*, 2014) that exacerbate fire suppression challenges. Overwintering and re-ignition of surface fires have also been highlighted as key issues associated with peat smouldering fires, making them challenging and costly issues for fire managers.

Fuel treatment is one component of FireSmart Canada's approach to community wildfire risk mitigation. The primary aim of fuel treatment is to reduce forest fuel ignition potential, reduce fire intensity and aid in the efficacy of fire suppression efforts, whilst minimizing the socioeconomic impacts of wildfires (Hirsch *et al.*, 2001). Due to the constraints of administering landscape-scale prescribed fires in the WUI and WII, management goals are usually achieved by the implementation of fuel modification treatments, such as thinning and mulching treatments (Hirsch *et al.*, 2001; Agee and Skinner, 2005). Mulching uses equipment such as rotary drums to reduce the amount and connectivity of forest fuels by mechanically masticating surface fuels and standing trees. Crown bulk density is reduced and easily available surface and ladder fuels are converted to a compacted masticated fuelbed on the ground surface (Kane *et al.*,2009). In Canada, the testing and implementation of the two common mulching techniques (mulch-thinning and strip-mulching) is usually limited to upland forest stands (Agee and Skinner, 2005; Hudak *et al.,* 2011; Stephens *et al.,* 2012). Consequently, the effect of mulching treatments on black spruce peatland fuel load and condition is a critical research gap and is currently unknown.

Fuel management operations using heavy machinery will likely compress the peatland surface, increasing near-surface peat bulk density (Brais and Camiré, 1998). If the increased fuel density is not offset by increased moisture content smouldering combustion can propagate downwards and peat burn severity *i.e.* depth of burn, will increase (Van Wagner, 1972; Benscoter *et al.,* 2011). Alternatively, the "mulching effect" may reduce depth of burn by reducing evaporation from the moss surface and maintaining high peat moisture content (Price *et al.,* 1998; Kreye *et al.* 2012), moisture retention may also be increased through compression (Golubev and Whittington, 2018). Since the relative strength of each feedback is currently unknown, so too is the effect of fuel management on peat burn severity and carbon loss. Therefore, the aims of this study are: 1) to assess the impact of two common fuel treatments on peat fuel load and condition (bulk density and moisture content) 2) to evaluate peat burn severity and surface fuel carbon loss from fuel-treated stands and a Control stand, and 3) to suggest pathways for future research and the development of best practice fuel treatments in black spruce peatlands.

## 5.2 METHODS

### 5.2.1 Study Area

This research was conducted at the Red Earth Creek Research site (56.547°N, 115.107°W), 5.5 km east of the hamlet of Red Earth Creek, Alberta, Canada. The area is located in the central mixed wood region of the BP, comprising of deciduous uplands and black spruce lowlands although relief is usually limited (Natural Subregions Committee, 2006). The region has a sub-humid climate where potential evapotranspiration exceeds precipitation by an average of ~30 mm annually (Environment Canada, 2013). More than 50 % of annual precipitation occurs in May–September, however, average daytime temperatures of >21°C, low relative humidity, and frequent drought periods can cause high fire danger ratings (Environment Canada, 2013; Amiro *et al.,* 2005).

The overstory species within the study site was 100 % black spruce [*Picea mariana* (Mill.) B.S.P.], with a *Rhododendron groenlandicum* dominated understory. Surface vascular vegetation also included *Rubus chamaemorus, Vaccinium oxycoccos* and *Equisetum* species. Moss cover was primarily feathermoss; *Pleurozium schreberi*, with some *Ptilium crista-castrensis* and *Hylocomium splendens*. *Sphagnum fuscum* was present on some hummocks and other *Sphagnum* species in intermediate and lower (hollow) areas. The ~120-year-old black spruce stand grows on 2-3 m of peat which is then underlain by clay. This vegetation community and peat depth is representative of mature black spruce peatlands often found in proximity to communities in the area (Housman, 2017).

5.2.2 Fuel treatments and sampling

In addition to a *Control* treatment, the site was divided into two forest stand treatments that are frequently applied in the Lesser Slave Wildlife Management Area: (1) mulch-thinning; and (2) strip-mulching (Figure 1), hereafter referred to as "*Thinned*" and "*Stripped*", respectively. Treatments took place in the winter of 2013-2014 when the ground was frozen to reduce disturbance and aid in the use of heavy machinery. For the *Thinned* treatment (0.71 ha), stand density was reduced by tree felling and mulching in place. The *Stripped* treatment (0.65 ha) consisted of 5 m wide clear cut and mulched strips, alternating with 5 m wide natural forest stands. The orientation of the strips was roughly east-west, perpendicular to the direction of wind for the experimental fire. Pre- and post-treatment stand characteristics and fuel loads were sampled according to the Alberta Wildland Fuels Inventory Program (AWFIP) (Alberta Agriculture and Forestry, 2015) (Table 1). Two 30 m transects were established in each fuel-treatment (including *Control*) and plots were set up every metre. Transects were orientated roughly north-south in all treatments, specifically to capture both treated and un-treated areas of the *Stripped* treatment. All pre-fire measurements were taken midday roughly 24 hrs prior to the experimental fire, while post-fire measurements took place within 48 hrs of the fire.

5.2.3 Fire weather and behaviour

The fire was initiated by a heli-torch, along the ignition line in the *Control* treatment on May 14th 2015 (Figure 1). The fire weather danger rating on the day of ignition was "high" for the region. Using historical weather data (1981 to present) for the Red Earth lookout tower weather station (15 km to the north), Fire Weather Index (Van Wagner, 1987) codes were determined as follows: hourly Initial Spread Index value was 16 (at the 98th percentile) and the Build Up Index value was 51 (at the 83rd percentile). Fine Fuel Moisture Code values were high (94.2), and the Duff Moisture Code and Drought Code were moderate-high (33 and 282, respectively). Average wind speeds were between 14 and 19 km $h^{-1}$, with fluctuations of 3–6 km $h^{-1}$ due to gusts during the fire passage through the treatments.

5.2.4 Surface fuel load and condition

Surface temperature was measured for each plot between 10 am and 2 pm EST one day before the fire using infra-red imagery from a FLIR One camera attachment with a 0.1°C thermal resolution. Mean and maximum plot temperature were recorded and transect measurement order was pseudo-randomized to minimize effect of time on temperature measurements. Mulch depth was recorded in all plots, and gravimetric water content (GWC) was assessed for 15 plots (0.3×0.3 m) per treatment. GWC of the mulch was assessed by weighing samples in the field, and then drying the samples at 65°C until constant mass. Mulch samples were separated into upper (top 2 cm) and lower (everything below 2 cm) sections before weighing in order to calculate mulch bulk density and changes in GWC with depth. Mulch bulk density was assessed alongside mulch GWC. Peat volumetric water content (VWC) was measured separately in the top 0–3 cm and 0–6 cm using a ML3 theta probe in each plot. Apparent dielectric permittivity for the 0–3 cm measurements were corrected for the dielectric permittivity of air, with VWC calculated using a third order polynomial for poorly decomposed peat (Kellner and Lundin, 2001). Peat cores were taken to a depth of 30 cm from all treatments, they were separated in to 5 cm sections and then dried at 65°C for 48 hours to assess bulk density with depth.

### 5.2.5 Burn severity and surface fuel carbon loss

Burn pins were installed every metre along each transect to measure depth of burn (DOB). Depth of burn was calculated as the difference in height above the ground surface measured pre- and post-fire. In locations where the ground cover was mulch, the depth of mulch was recorded in order to partition DOB between mulch and peat. Carbon loss estimates of surface fuel (mulch and peat) were calculated using DOB and treatment average fuel bulk density. For peat fuel, the depth dependence of bulk density was also included. Where DOB was greater than pre-fire mulch depth it was assumed there was total mulch combustion. Typical loss on ignition (LOI) for moss, near-surface peat, and woody vegetation was evaluated by heating a small amount of sample in a muffler furnace at 550 °C for 4 hours. Using a carbon content of 52 % for peat organic matter (Bhatti and Bauer, 2002) and 49 % for black spruce (*i.e.* mulch) (Matthews, 1993) combustion carbon loss was estimated using the product of LOI and carbon content.

### 5.2.6 Vegetation surveys

The dominant moss type was identified to family level within a 0.1 m radius of each burn pin. In cases where moss cover was not homogeneous, co-dominant moss type was also recorded. For cases where mulch completely obscured the underlying moss, moss type was identified in locations where near-surface VWC was measured (see above). Average distance to tree was calculated based on distance to nearest tree in four quadrants surrounding each burn pin. Tree diameter at breast height (DBH) was also measured for the closest tree in each quadrant using a tree calliper, with DBH measured to the nearest 1 mm. Canopy openness was measured along the transect using fish-eye photography, taken at breast height above the sub-canopy shrub layer. Relative differences in canopy openness were assessed by classifying images into tree and sky using colour and intensity threshold-based criteria in Gap Light Analyzer software (Fraser *et al.,* 1999).

5.2.7 Statistical analyses

In general, we report mean and standard deviation, but report median values when data is heavily skewed. A principal component analysis (PCA) (MATLAB *pca*) was used to identify candidate explanatory variables for predicting DOB. Variables were transformed to z-scores prior to running the PCA. Variables with similar vector loadings were considered redundant with one another, while variables with small loading were considered to have low explanatory power. A general linear model was used to predict peat DOB. DOB data was square-root transformed to satisfy assumption of normality and confirmed using the Shapiro-Wilk test (w=0.98, p=0.09). Our initial null model was constructed using only VWC as a continuous variable. An ANOVA was used to test whether other factors (i.e. DBH, average tree distance, canopy gap fraction, surface temperature, mulch depth, ice depth, moss species) had a significant effect on DOB. In cases where a significant effect was found a post-hoc Tukey test was then applied.

## 5.3 RESULTS

### 5.3.1 Above-ground fuel condition

Treatment had a significant effect on canopy openness ($F_{2,182} = 114.8$; $p < 0.001$) and average distance to tree ($F_{2,183} = 298.0$; $p < 0.001$), both increased from *Control* < *Stripped* < *Thinned* (Table 2). While the *Stripped* treatment greatly reduced canopy fuel bulk density (Table 1), the effect on treatment average canopy openness was comparatively small (Table 2). While not expressly a component of the *Thinned* treatment, it appears that the *Thinned* treatment was biased towards leaving larger trees, where Tukey's HSD post-hoc test showed that DBH at the *Thinned* treatment was significantly greater than the *Stripped* and *Control* treatments ($F_{2,183} = 179.6$, $p < 0.001$).

### 5.3.2 Surface fuel condition

Where present, mulch depth tended to be deeper in the *Thinned* treatment (Table 2), although the difference was not significant ($t_{63} = 0.91$, $p = 0.38$). Overall, relative mulch cover was lower in the *Stripped* treatment (Table 2), but when adjusted to account for un-mulched strips, the percent cover in treated strips itself was estimated to be 77 % compared to 66 % in the *Thinned* treatment. Gravimetric water content (GWC) of mulch was substantially lower in the top 2 cm (0.17 $g_{water}\ g^{-1}_{mulch,dry}$) compared to mulch sampled below 2 cm (0.46 $g_{water}\ g^{-1}_{mulch,dry}$), while average differences in GWC between fuel treatments was small for both mulch samples at 0-2 cm (0.01 $g_{water}\ g^{-1}_{mulch,dry}$) and 2+ cm (0.06 $g_{water}\ g^{-1}_{mulch,dry}$), respectively. Mulch bulk density averaged 74.3 ± 34 kg $m^{-3}$ across both fuel treatments.

Overall, median near-surface VWC of moss/peat tended to be relatively low (0.09 $cm^3$ $cm^{-3}$), where both treatment ($F_{2,368}$ = 43.3; $p$ <0.001) and measurement depth ($F_{1,368}$ = 9.1; $p$ = 0.003) had a significant effect on water content. In the fuel-treated areas, mulch presence had a significant effect on near-surface water content ($F_{1,244}$ = 15.32; $p$ <0.01), where effect size (0.06 $cm^3$ $cm^{-3}$) was similar between treatments (mulch presence × treatment; $F_{2,368}$ = 0.02; $p$ = 0.88). Treatment mean GWC for the top 6 cm of peat was 4.9, 3.0 and 5.2 $g_{water}$ $g^{-1}_{peat,dry}$ , in the *Control, Stripped* and *Thinned* treatments, respectively. In general, the transects were dominated by feathermoss (Table 2). Because of the small number of transect points in the *Thinned* treatment with *Sphagnum*, moss type was not included in the above ANOVA. However, aggregated to site level, moss type had a significant effect on near-surface VWC ($t$ = 7.0; $p$ <0.001), where median VWC was 0.07 and 0.17 $cm^3$ $cm^{-3}$ for feathermoss and *Sphagnum*, respectively.

Mean surface temperature was significantly different between the fuel-treated and *Control* treatments ($p$ <0.001). Treatment level mean temperatures tended to be ~10°C higher in the *Thinned* and *Stripped* treatments compared to the *Control* treatment (Table 2). Moreover, the range of maximum measured temperature in plots with mulch were high and relatively narrow (~50–70 °C) compared to plots without mulch (~20–60 °C).

### 5.3.3 Surface fuel load

Peat showed no significant difference in 0-30 cm depth-integrated bulk density between treatments ($p$ >0.05). However, the 5-10 cm depth increment in the *Stripped* treatment was significantly denser than in the *Control* treatment ($F_{2,27}$, $p$ <0.05) (Figure 2). Moreover, when the *Stripped* treatment cores were separated into cores taken in the treated strips and in the un-treated strips, the bulk density in the treated strips was significantly greater than the un-treated strips and the *Control*. Additionally, the *Thinned* treatment was significantly denser than un-treated strips ($F_{4,227}$ $p$ <0.001).

5.3.4 Depth of burn and carbon loss

Peat depth of burn (DOB) varied between fuel-treatments, where the *Thinned* treatment had significantly lower DOB than the *Stripped* treatment, and neither *Thinned or Stripped* treatment was significantly different to the *Control* ($F_{2,185}$ $p$ <0.001) (Figure 3). Treatment median peat DOB were 55 ± 51 mm, 50 ± 54 mm and 31 ± 33 mm for *Control*, *Stripped* and *Thinned* treatments, respectively. In developing a simple empirical model for DOB, a PCA was used to help select predictor variables (Figure 4). The first two principal components explained ~60 % of variance in the data set. $VWC_{0\text{-}3\,cm}$ and DBH were eliminated as potential predictor variables due to the close correspondence in loading vectors with $VWC_{0\text{-}6\,cm}$ and average tree distance, respectively. General linear model analysis found that moisture content had the greatest effect on DOB, where moss type and mulch presence provided statistically significant additional explanatory power (Table S5.1). In particular, plots with *Sphagnum* or mulch present had lower DOB (Figure 5).

Although peat DOB followed the trend *Thinned* < *Stripped* < *Control*, due to the higher bulk density of the near-surface peat in the *Stripped* and *Thinned* treatments, peat carbon loss followed the trend *Control* < *Thinned* < *Stripped* (Figure 6a). Treatment average peat carbon loss was 0.56 ± 0.61 kg C $m^{-2}$, 1.03 ± 0.89 kg C $m^{-2}$, 0.56 ± 0.54 kg C $m^{-2}$ for

*Control*, *Stripped* and *Thinned*, respectively. When carbon loss from mulch combustion is accounted for, the treatment level surface fuel carbon losses are 0.56 ± 0.61 kg C $m^{-2}$, 1.5 ± 1.3 kg C $m^{-2}$ and 1.5 ± 1.2 kg C $m^{-2}$, where both fuel-treatments are significantly different to the *Control*, but not to each other ($F_{2,185}$ $p$ <0.001) (Figure 6b).

## 5.4 DISCUSSION

### 5.4.1 Fuel load

Canopy fuel load in the *Control* was greater than standard values for the boreal spruce, C-2, fuel type (0.8 kg/$m^2$) used in the Canadian Fire Behaviour Prediction System (Forestry Canada, 1992). Canopy fuel load and bulk density was effectively reduced in both the *Stripped* and *Thinned* treatments although the *Thinned* treatment tended to leave larger trees in place. Given that relatively high intensity fire was experienced in all treatments (Hvenegaard *et al.,* 2016) with significant torching and consequent consumption of available aerial fuels, the above-ground fuel consumption reflected the overall reduction in fuel load from each fuel modification treatment. Surface fuel load was altered by the addition of mulch (masticated fuelbed) to the ground surface. Mulch fuel loads were similar to others reported in the literature (*e.g.* Kane *et al.*, 2009), with mean mulch depth being 4.1 ± 3.0 and 3.3 ± 3.6 cm for the *Thinned* and *Stripped* treatments, respectively, and bulk density averaging 74 ± 34 kg $m^{-3}$. The distributions of fuel particle size and diameter are decreased by mastication which increases the amount of fine fuels compared to un-treated stands (Kane *et al.*, 2009). This likely influenced fuel condition and fire behaviour (Shicks *et al.*, 2015). Near-surface peat bulk density was greater in fuel-treated areas compared to the *Control* area, and compaction appears to be concentrated in the 5-10 cm depth interval (Figure 2). Moreover, peat underlying the treated strips in the *Stripped* treatment had significantly greater bulk density than in the un-treated strips. The use of heavy machinery to masticate the forest fuels likely compressed the underlying peat leading to the increase in bulk density. Although previous studies have not found this effect on upland soils (Stephens *et al.,* 2012) we suggest this effect is due to the high porosity and compressibility of peat (Golubev and Whittington, 2018), owing to relatively low strength even when frozen.

5.4.2 Fuel condition

The masticated surface fuels were exposed to more incoming solar radiation due to higher canopy openness in fuel-treated stands, and likely greater wind speeds due to less sheltering compared to the *Control* treatment, as a result of the removal of understory and canopy vegetation. This has been found to lead to increased evaporation and heating in fuel-treated areas (Schiks and Wotton, 2015). Average surface temperatures in the *Stripped* and *Thinned* treatments were significantly higher than in the *Control* treatment 24 hours prior to the burn, and temperatures were similar to those reported in other studies of mulch condition (*e.g.* Schiks *et al.,* 2015). These higher temperatures could reflect the lack of evaporative cooling from the surface of the mulch due to its dry condition, which is likely in equilibrium with the atmosphere in fire-prone periods (Amiro *et al.,* 2001), making sustained ignition more probable (Schiks and Wotton, 2014).

All treatments were dominated by feathermoss, where the *Thinned* area had the lowest proportion of *Sphagnum* moss, and the *Control* had the greatest proportion. *Sphagnum* moss had significantly greater moisture content than feathermoss in the top 6 cm; a trend which is often seen in peatlands (Swanson and Flanagan, 2001; Shetler *et al.*, 2008). Moss/peat becomes susceptible to combustion when the moisture content drops below a threshold, such that the energy required to drive off remaining water and raise the fuel to combustion temperature (>300 °C), is exceeded by the energy output of the adjacent fire (Van Wagner, 1972; Benscoter *et al.*, 2011). Correspondingly, lower DOB was associated with the presence of *Sphagnum* moss (Figure 5, Figure S1).

Peat moisture content was significantly greater under the presence of a mulch layer and evidence of this "mulching effect" was also recorded within the mulch layer itself, where GWC below 2 cm was greater than GWC in the uppermost 2 cm, similar to Shicks et al. (2015). The layer of mulch likely acted as an evaporative cap leading to increased moisture content of the near-surface peat (Price *et al.*, 1998). This was more prominent in the *Thinned* treatment, where peat VWC was significantly higher than in the *Stripped* and *Control* areas. Where mulch was present, the *Thinned* treatment had higher average VWC compared to the *Stripped* treatment. This supports the observed weak positive relation between mulch depth and peat water content (Figure 4). Although increased peat bulk density increases smouldering propagation potential, it is also associated with a decrease in pore-size distribution and increased moisture retention (Boelter, 1969; Sillins and Rothwell, 1998) which likely further increased the moisture content of near-surface peat in the *Thinned* treatment and mulched strips in the *Stripped* treatment. Recent research by

Golubev and Whittington (2018) found that compression of *Sphagnum* moss can increase unsaturated hydraulic conductivity by an order of magnitude, enabling better conductance of water to the surface of the moss. The combination of physical and environmental changes on the near-surface peat in fuel-treated areas likely facilitated higher moisture contents under high fire danger weather conditions.

5.4.3 Peat burn severity and surface fuel carbon loss

The flammability of the mulch is evidenced by the consumption of the full mulch depth in >99% of plots that had a mulch layer (n=120). Peat DOB followed the trend *Thinned* < *Stripped* < *Control* (Figure 3). The DOB in the *Control* treatment (55 ± 51 mm) and *Stripped* treatment (50 ± 54 mm) are similar to those measured in natural wildfires in untreated black spruce peatlands across the region (*e.g.* Benscoter and Wieder, 2003), whereas in the *Thinned* treatment DOB was significantly lower than the stripped treatment (31 ± 33 mm) (Figure 5.3). DOB decreased as moisture content increased and therefore tended to be lower in *Sphagnum* moss than feathermoss (Figure 5), highlighting the importance of maintaining *Sphagnum* ground cover to reduce peat burn severity (Shetler *et al.*, 2008). Overall, peat moisture content was the strongest predictor of DOB in all treatments (Table S1). High canopy openness values and a lack of mulch cover in fuel-treated areas were associated with higher DOB, likely due to increased evaporative losses and the subsequent effect on peat moisture content.

DOB in the *Stripped* treatment was similar to the *Control* but compaction lead to increased peat bulk density and therefore higher peat combustion carbon loss (Figure 6a). The variability of fire behaviour in masticated fuel beds (Kreye *et al.*, 2014), and the preservation of a high canopy fuel load in un-treated strips may have contributed to a more intense fire in the *Stripped* treatment, while high canopy openness in treated strips lead to surface drying, leading to instances of high DOB. High peat burn severity can expose denser peat, leading to high near-surface pore-water tensions, that are detrimental to the recovery of the keystone *Sphagnum* mosses (Thompson and Waddington, 2008). Moreover, moderate burn severity or singeing of feathermosses can create an extremely hydrophobic post-burn surface that is difficult for secondary recolonization (Moore *et al.*, 2017), and can persist for years following wildfire (Mackinnon, 2017).

Due to the higher bulk density of peat in the *Thinned* treatment, peat carbon loss was similar to the *Control* despite DOB being lower. Peat carbon loss estimates (<1 kg C $m^{-2}$) are at the lower end of the observed range from wildfires in natural black spruce peatlands (Benscoter and Wieder, 2003). However, when the carbon loss from mulch combustion is accounted for this estimate increases three-fold for the *Stripped* and *Thinned* treatments (Figure 6b), averaging 1.5 kg C $m^{-2}$ in both the fuel-treated areas compared to 0.56 kg C $m^{-2}$ in the *Control* treatment. This is important to consider if fuel-treatments are to be implemented at a large-scale, as will need to be, to protect the growing WUI and WII. It is currently unclear whether fuel-treated black spruce peatlands will follow a post-fire recovery trajectory similar to that of their natural counterparts (*e.g.* Benscoter and Vitt, 2008) and this will determine the long-term carbon dynamics of these ecosystems. Sources of uncertainty include the effect of residual mulch/charcoal on moss recovery and the effect of mulch combustion on the degree of hydrophobicity present at the peatland surface. We argue that longer-term studies of burned and unburned fuel-treated black spruce peatlands are needed to better understand the effects on long-term vegetation communities and the recovery of important ecological services.

## 5.5 CONCLUSIONS

Canopy fuel load was decreased by both *Thinned* and *Stripped* fuel treatments. Surface fuel load was increased in the *Thinned* and *Stripped* treatment areas by the conversion of stand and surface vegetation to a masticated fuelbed, and by the compaction of underlying peat. The combined effects of the mulch layer and enhanced moisture retention due to compression lead to the greater near-surface peat moisture content in the *Thinned* treatment, compared to the *Control*. Consequently, DOB followed the trend *Thinned* < *Stripped* < *Control*, with treatment average DOB within typical observed ranges for natural peatland wildfires. Surface fuel carbon loss (mulch and peat) was not significantly different between *Stripped* and *Thinned* fuel treatments but was significantly greater than losses from the *Control* treatment. Therefore, through the implementation of fuel treatments such as mulch-thinning, the challenges associated with high peat burn severity may be avoided in black spruce peatlands, however, the carbon losses incurred must be included in cost-benefit assessments before large-scale implementation is employed.

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## Tables

**Table 5.1** Pre- and post-treatment stand characteristics, and fuel loads assessed by field sampling by Alberta Wildland Fuels Inventory Program crews in 2013 and 2015, respectively. Average (± standard deviation).

| | **Stem Density (Stems/Ha)** | | **Mean Height (m)** | **Live Crown Base Height (m)** | **Understory Fuel Load** | **Canopy Fuel Load (kg m-2)** | **Canopy Bulk Density (kg m-3)** |
|---|---|---|---|---|---|---|---|
| | **Overstory** | **Understory** | | | **(kg m-2)** | | |
| **Pre-Treatment/ Control** | 1260 (550) | 4000 (2200) | 10.3 (1.6) | 4.1 (1.1) | 0.04 (0.03) | 1.2 (0.4) | 0.2 (0.1) |
| **Thinned** | 400 | 0 | 10.3 (1.6) | 4.1 (1.1) | 0 | 0.3 (0.1) | 0.05 (0.02) |
| **Stripped** | 650 | 0 | 10.3 (1.6) | 4.1 (1.1) | 0 | 0.5 (0.1)* | 0.02 (0.02)* |

* Average across treated and non-treated strips

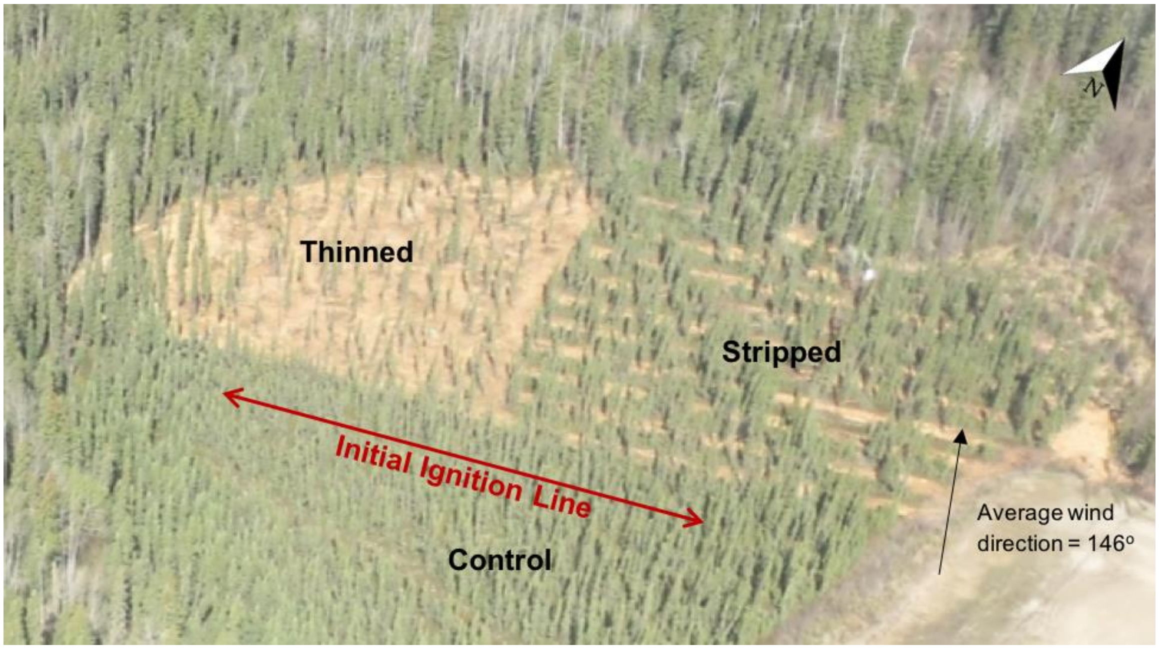
**Table 5.2** Average (± standard deviation) surface fuel characteristics aggregated to the treatment level. Values for mulch depth are for locations where mulch was present. Percent mulch represents the proportion of transect points where mulch was present at the burn pin. Surface temperature is based on FLIR measurement at mid-day, ~24 h prior to prescribed burn.

| Treatment | Mulch Depth (cm) | Percent Mulch (%) | Feathermoss / Sphagnum (%)* | DBH (cm) | Tree Distance (m) | Gap Fraction (%) | Surface Temp (°C) |
|---|---|---|---|---|---|---|---|
| **Control** | N/A | N/A | 77 / 21 | 4.4 (1.4) | 1.2 (0.5) | 63 (13) | 26.9 (7.0) |
| **Thinned** | 4.1 (3.0) | 66 | 95 / 5 | 11.0 (1.7) | 4.4 (0.8) | 91 (5) | 40.1 (4.4) |
| **Stripped** | 3.3 (3.6) | 39 | 71 / 29 | 5.7 (2.8) | 2.5 (0.9) | 75 (11) | 37.4 (6.6) |

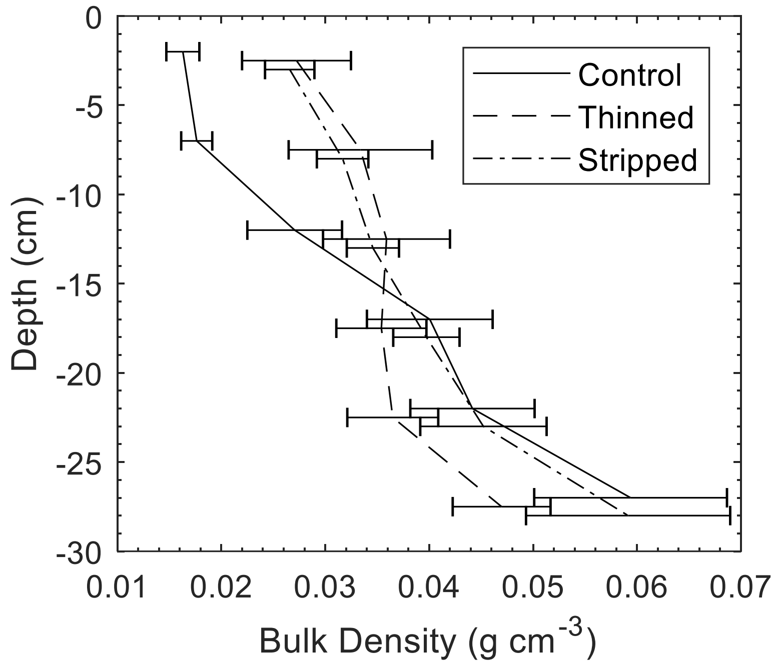
*Data excludes additional moss species therefore may not equal 100%

**Figure List**

**Figures**

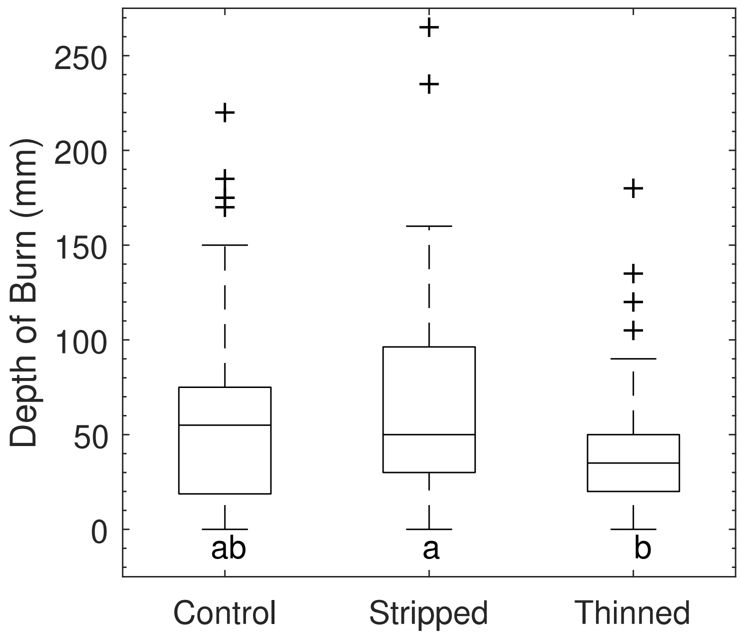



[Figure 5.1]

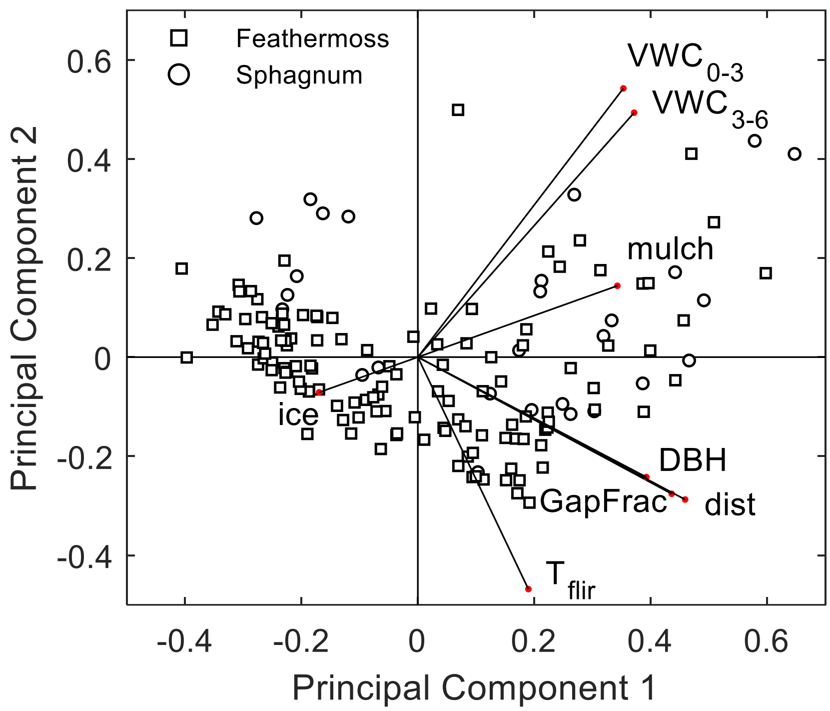


Control
Thinned
Stripped
Depth (cm)
0
-5
-10
-15
-20
-25
-30
0.01
0.02
0.03
0.04
0.05
0.06
0.07
Bulk Density (g cm-3)

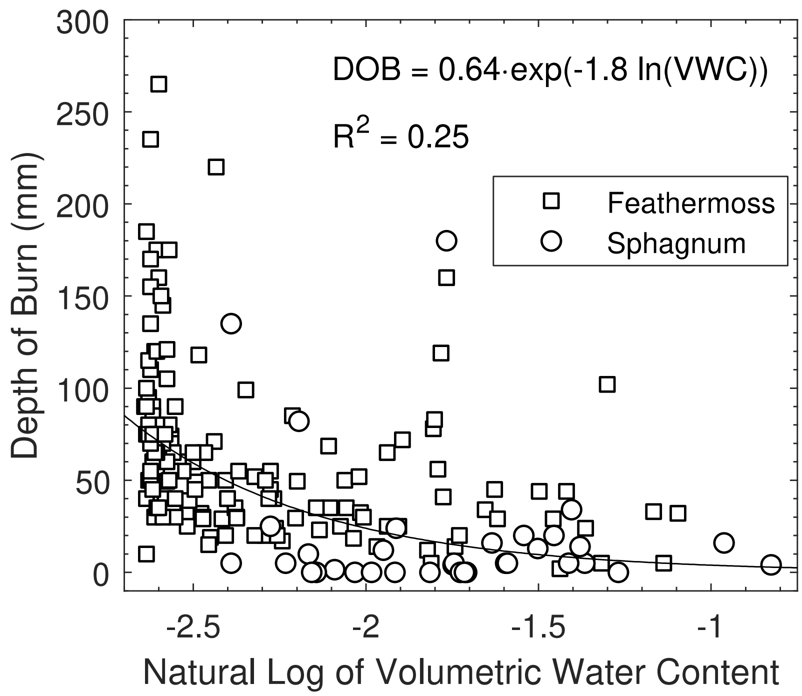

[Figure 5.2]


Depth of Burn (mm)
250
200
150
100
50
0
ab
a
b
Control
Stripped
Thinned

[Figure 5.3]

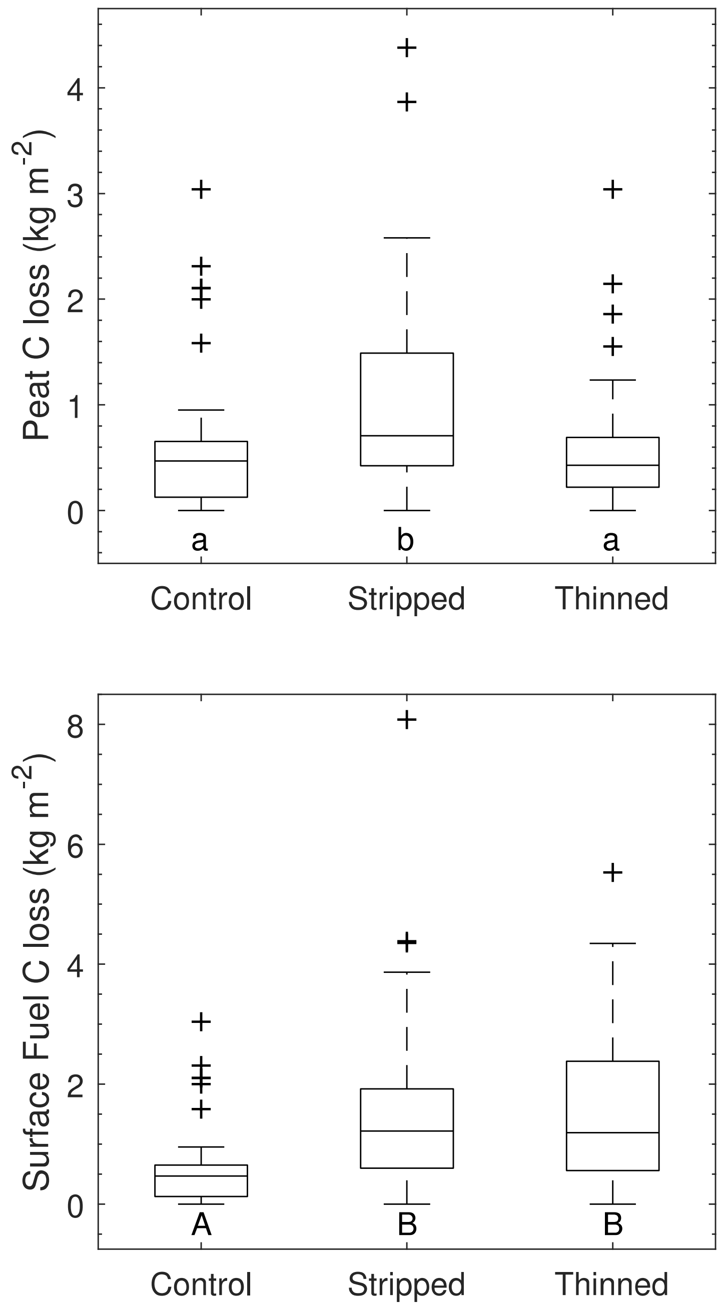

Feathermoss
Sphagnum
VWC 0-3
VWC 3-6
mulch
ice
DBH
GapFrac
dist
T flir
0.6
0.4
0.2
0
-0.2
-0.4
-0.4
-0.2
0
0.2
0.4
0.6
Principal Component 1
Principal Component 2

[Figure 5.4]



DOB = 0.64·exp(-1.8 ln(VWC))
R2 = 0.25
Feathermoss
Sphagnum
Depth of Burn (mm)
Natural Log of Volumetric Water Content
300
250
200
150
100
50
0
-2.5
-2
-1.5
-1


[Figure 5.5]


Peat C loss (kg m-2)
0
1
2
3
4
a
b
a
Control
Stripped
Thinned
Surface Fuel C loss (kg m-2)
0
2
4
6
8
A
B
B
Control
Stripped
Thinned

[Figure 5.6]

# CHAPTER 6

# CONCLUSIONS

## 6.1 Summary and conclusions

A landscape-scale assessment found substantial variability in peat hydrophysical properties within and between black spruce dominated peatlands in the BP, where peat properties can act to buffer or enhance peat smouldering combustion vulnerability. Peatland middles (hummocks and hollows) exhibit resistance to smouldering combustion across the range of time-since-fire (stand ages) across the landscape. However, the presence of high density peat in peatland margins was evidenced across all stand ages and hydrogeological settings in the BP, which highlights their vulnerability to high peat burn severity, especially when peatlands are disconnected from regional groundwater systems (Hokanson et al., 2016). Based on average peat properties and the response of the water table in margins, under moderate to severe water deficits common in the BP, margins are vulnerable to combustion throughout their entire depth. Moreover, the high peatland margin: total area in glaciofluvial (coarse) and heterogeneous (moraine) hydrogeological settings (Mayner et al., 2018) results in a greater proportion of the total peat carbon stock being vulnerable to combustion when peatlands are subjected to a water deficit. In these hydrogeological settings carbon accumulation rates in margins do not recover carbon lost from the margins within the fire return interval (Ingram et al., 2019), hence high peat burn severity in margins could alter the long-term carbon balance of some peatlands.

Although the assessment of peat properties found that black spruce dominated peatland middles are less vulnerable to high severity peat burn than margins, the disturbance of peatlands by anthropogenic disturbance (*e.g.* drainage or climate change) can cause increased peat burn severity throughout black spruce dominated peatlands. The substantial canopy fuel load in black spruce dominated peatlands (Johnston et al., 2015; Thompson et al., 2017) transfers energy to the peat surface during the passage of a wildfire front (Thompson et al., 2015). However, the stand characteristics also affect pre-fire ground cover (*i.e.* the dominance of feather moss over *Sphagnum* moss; Bisbee et al., 2001), and pre-fire peat properties/condition (*i.e.* density and moisture content). Assessment of a partially-drained burned peatland found a strong linear correlation between black spruce basal diameter and depth of burn (DOB). The threshold to high peat burn severity was bounded by average basal diameters of 3.2 and 7.9 cm, suggesting that stand characteristics can be used to identify thresholds to high peat burn severity in black spruce dominated peatlands. Tree ring analysis suggests that a strong positive feedback, the water table depth-afforestation feedback (Waddington et al., 2015) was initiated by water table drawdown (drainage) leading to enhanced tree productivity and transpiration, and sustained low water table positions in the heavily-drained area. In combination with high-extreme fire weather indices, this resulted in mean DOB of 36.9 cm, where 51 % of the pre-fire peat carbon was lost through combustion, and mineral soil was exposed in some areas. It is hypothesised that the area of high peat burn severity will likely undergo an ecosystem regime shift, similar to Kettridge et al. (2015), where ecosystem resilience is overcome and pre-fire ecosystem functions (*e.g.* pre-fire carbon accumulation rates) are not recovered.

This hypothesis is strengthened by data that evidences the exceedance of a threshold peat burn severity that resulted in the breakdown of a key evaporation-limiting feedback. Typically, the hydrophobicity-evaporation feedback limits evaporative losses in post-fire peatlands (Kettridge et al., 2017), however, its relative strength has been shown to be dependent on peat burn severity (Moore et al., 2017; Kettridge et al., 2019). The high peat burn severity (DOB) experienced in the heavily-drained area of the partially-drained peatland resulted in a change in ground cover, a concomitant decrease in surface hydrophobicity, and increase in evaporation rates. Whilst areas of low-moderate peat burn severity (DOB) demonstrated ecosystem resilience through the maintenance of *Sphagnum* and feather moss ground cover and the functioning of the hydrophobicity-evaporation feedback, the area of high peat burn severity was covered by bare peat and pools and was quickly colonised by pioneer moss species *e.g.* fire moss, that dominated evaporative losses. Overall evaporation rates were significantly greater in the area of high peat burn severity compared to the areas of moderate and low peat burn severity. Increased evaporation rates in post-fire peatlands, in a sub-humid climate such as the BP, will result in peat drying, subsequently increasing the amount of aerobic decomposition compared to slow anaerobic decomposition. This will likely release more carbon to the atmosphere and further degrade the remnant carbon stock (Kettridge et al., 2019).

High peat burn severity causes extensive carbon loss through smouldering combustion (12–17 kg C $m^{-2}$ in this study), and exposure of the remnant carbon stock (Kettridge et al., 2019). However, particulate matter and smoke are also emitted during smouldering combustion which can result in hazardous air quality. Since smouldering combustion can be sustained in low-oxygen conditions, peat fires can burn for weeks to months (including over winter) and relatively deep underground (Rein et al., 2008). Hence, they are not only a priority for wildfire managers but also a substantial resource draw. Due to the potential for extensive carbon loss, air pollution and wildfire management challenges, it is argued that there is a need for fuel modification treatments to be developed for black spruce dominated peatlands in order to protect the wildland-urban interface (WUI) and wildland-industry interface (WII) in the BP.

Testing two common fuel modification treatments using an experimental fire in a black spruce dominated peatland found that although canopy fuel load was reduced by both treatments (thinned-mulched and strip-mulched), surface fuel load was increased due to the conversion of above-ground fuels to surface fuels via mulching, and the compaction of near-surface peat. However, coverage of the surface with mulch (reducing evaporation), and the increased peat moisture retention due to compression, lead to higher peat moisture content in the thinned-mulched treatment compared to the control. All else being equal this would reduce the efficiency of the peat smouldering reaction and lead to lower peat burn severity. However, due to the increased surface fuel load, carbon loss was greater in the treated areas than the control area. Therefore, through the implementation of fuel modification treatments some of the challenges of peat smouldering combustion may be avoided or reduced, however, increased carbon losses may be incurred. This experimental fire was conducted in mid-May, early in the BP fire season when ground ice was often < 20 cm from the surface, which may have limited carbon losses in the control treatment as later season burns have been correlated with increased peat burn severity in black spruce dominated peatlands in Alaska (Turetsky et al., 2011). Moreover, fuel modification was completed just one year prior to the experimental fire and it is likely that if peatland wildfire management was implemented on a large scale there would be a number of years post-treatment before wildfire disturbance. Consequently, there is still much research needed in this field to develop best management practices in peatland wildfire management.

## 6.2 Future work

To further the work completed in this thesis, and to better predict the stability of northern peatland ecosystems in the future and protect the WUI and WII from the effects of high severity peat burn, future work is required. The timescales associated with peatland wildfire research in the BP cause limitations to field studies *e.g.* decadal climate cycles (Devito et al., 2012), ~120-year fire return interval (Turetsky et al., 2004), and vegetation succession over similar time periods (Benscoter and Vitt, 2008). Moreover, there are inherent logistical challenges and risks of using experimental fires, hence, future research would benefit from using a modelling approach.

Since hydrogeological setting, time-since-fire, and black spruce stand characteristics were found to affect peat burn severity they should firstly be incorporated into a peatland ecohydrological model and then used to better understand and predict peat burn severity. Work has begun to incorporate these drivers of cross-scale variability and their interactions into a 2D hydrological model focussing only on saturated flow (modified from DigiBog; Baird et al., 2012). However, research should also consider the unsaturated (vadose) zone in peatlands and how unsaturated peat properties (*e.g.* unsaturated hydraulic conductivity) affects the transport of water to the surface, peat water content, and ultimately the vulnerability of surface moss/peat to smouldering combustion. In addition to the hydrophobicity-evaporation feedback, other hydrological feedbacks that reduce peat burn severity or contribute to ecosystem resilience should be tested under a range of peat burn severities and future hydrologic regimes and climate scenarios. Exploratory modelling would allow for the inclusion of hydrological feedbacks, and the large variability of peat properties, stand characteristics and hydrogeological settings assessed in this thesis.

To assess the vegetation and fuel load changes of black spruce dominated peatlands following fuel modification, field studies and modelling should be used in parallel to identify vegetation/fuel-load trajectories through time to better inform wildfire management decisions such as the need for re-treatment. To most effectively reduce peat burn severity fuel modification treatments should address above- and below-ground fuels. Since *Sphagnum* mosses have been found to be resistant to burning and therefore protect the underlying carbon stock, research efforts should be focussed on the propagation of *Sphagnum* mosses on the landscape. Some work has already begun by conducting *Sphagnum* transplant experiments in post-treatment and/or post-fire black spruce dominated peatlands in the BP.

By better understanding the interaction of peat smouldering combustion vulnerability with the peatland hydrological regime, hydrological feedbacks, and climate through modelling, and by developing trajectories of black spruce peatlands following fuel modification treatment, more progress can be made towards reducing peat burn severity and the associated carbon loss, smoke and pollution, whilst sustaining the globally significant carbon stores of northern peatlands.

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# APPENDICES

## APPENDIX A

### Supplementary Material Chapter 2

**Table S2.1:** Summary of significant differences (Kruskal Wallis; $p < 0.05$) of depth-integrated peat properties for within-peatland locations; hummock, hollow, margin.

| Variable | Location | Letters denoting significant differences | | | Mean |
|---|---|---|---|---|---|
| **Bulk density (kg m$^{-3}$)** | Hummock | A | | | 55 |
| | Hollow | | B | | 103 |
| | Margin | | | C | 137 |
| **Specific Yield** | Hummock | A | | | 0.51 |
| | Hollow | | B | | 0.31 |
| | Margin | | B | | 0.35 |
| **VWC$_{-200}$ (m$^{3}$ m$^{-3}$)** | Hummock | A | | | 0.28 |
| | Hollow | | B | | 0.46 |
| | Margin | | | C | 0.40 |

**Table S2.2:** Steel-Dwass test; bulk density for within-peatland locations. * denote significant differences at p-Values ≤ 0.05.

| **ChiSquare** | **DF** | **Prob>ChiSq** |
|---|---|---|
| 195.7873 | 2 | <.0001* |

| **Level** | **- Level** | **Z** | **p-Value** |
|---|---|---|---|
| Margin | Hummock | 11.6660 | <.0001* |
| Margin | Hollow | 4.7957 | <.0001* |
| Hummock | Hollow | -11.6357 | <.0001* |

**Table S2.3:** Steel-Dwass test; specific yield for within-peatland locations.

| **ChiSquare** | **DF** | **Prob>ChiSq** |
|---|---|---|
| 67.0456 | 2 | <.0001* |

| **Level** | **- Level** | **Z** | **p-Value** |
|---|---|---|---|
| Hummock | Hollow | 7.95484 | <.0001* |
| Margin | Hollow | 1.37339 | 0.3549 |
| Margin | Hummock | -5.36872 | <.0001* |

**Table S2.4:** Steel-Dwass test; volumetric water content at -200 hPa for within-peatland locations.

| **ChiSquare** | **DF** | **Prob>ChiSq** |
|---|---|---|
| 86.0510 | 2 | <.0001* |

| **Level** | **- Level** | **Z** | **p-Value** |
|---|---|---|---|
| Margin | Hummock | 5.50284 | <.0001* |
| Margin | Hollow | -2.91987 | 0.0098* |
| Hummock | Hollow | -9.12474 | <.0001* |

**Table S2.5:** Steel-Dwass test; Margins bulk density for hydrogeological settings. Moraine (Hetergeneous), Fine (Glaciolacustrine) and Coarse (Glaciofluvial).

| **ChiSquare** | **DF** | **Prob>ChiSq** |
|---|---|---|
| 11.1826 | 2 | 0.0037* |

| **Level** | **- Level** | **Z** | **p-Value** |
|---|---|---|---|
| Moraine | Fine | 3.29421 | 0.0028* |
| Moraine | Coarse | 0.71013 | 0.7575 |
| Fine | Coarse | -2.07502 | 0.0950 |

**Table S2.6:** Steel-Dwass test; Margins specific yield for hydrogeological settings. Moraine (Hetergeneous), Fine (Glaciolacustrine) and Coarse (Glaciofluvial).

| ChiSquare | DF | Prob>ChiSq |
|---|---|---|
| 5.9852 | 2 | 0.0502 |

| Level | - Level | Z | p-Value |
|---|---|---|---|
| Fine | Coarse | 0.41028 | 0.9114 |
| Moraine | Coarse | -1.49103 | 0.2952 |
| Moraine | Fine | -2.34499 | 0.0498* |

**Table S2.7:** Steel-Dwass test; Hollows bulk density for time-since-fire categories, 0-20 years, 21-80 years and 81-120+ years.

| ChiSquare | DF | Prob>ChiSq |
|---|---|---|
| 6.2138 | 2 | 0.0447* |

| Level | - Level | Z | p-Value |
|---|---|---|---|
| 81-120 | 21-80 | 1.32721 | 0.3800 |
| 81-120 | 0-20 | -0.46707 | 0.8867 |
| 21-80 | 0-20 | -2.49063 | 0.0341* |

**Table S2.9:** Steel-Dwass test; Margins bulk density for time-since-fire categories, 0-20 years, 21-80 years and 81-120+ years.

| **ChiSquare** | **DF** | **Prob>ChiSq** |
|---|---|---|
| 6.8184 | 2 | 0.0331* |

| **Level** | **- Level** | **Z** | **p-Value** |
|---|---|---|---|
| 81-120 | 21-80 | 1.05658 | 0.5412 |
| 81-120 | 0-20 | -1.02637 | 0.5602 |
| 21-80 | 0-20 | -2.66206 | 0.0212* |

**Table S2.10:** Steel-Dwass test; Hollows specific yield for time-since-fire categories, 0-20 years, 21-80 years and 81-120+ years.

| **ChiSquare** | **DF** | **Prob>ChiSq** |
|---|---|---|
| 6.2962 | 2 | 0.0429* |

| **Level** | **- Level** | **Z** | **p-Value** |
|---|---|---|---|
| 21-80 | 0-20 | 2.53507 | 0.0302* |
| 81-120 | 0-20 | 0.93693 | 0.6169 |
| 81-120 | 21-80 | -1.06458 | 0.5361 |

**Table S2.11:** Steel-Dwass test; Hummocks specific yield for time-since-fire categories, 0-20 years, 21-80 years and 81-120+ years.

| **ChiSquare** | **DF** | **Prob>ChiSq** |
|---|---|---|
| 12.4229 | 2 | 0.0020* |

| **Level** | **- Level** | **Z** | **p-Value** |
|---|---|---|---|
| 81-120 | 0-20 | 3.336314 | 0.0024* |
| 21-80 | 0-20 | 2.393558 | 0.0440* |
| 81-120 | 21-80 | 1.428546 | 0.3262 |

**Table: S2.12** Water table depth (m) where -200 hPa soil-water pressure reached.

| Location | Margin | | |
|---|---|---|---|
| Time-since-fire (years) | 0-20 | 21-80 | 81-120 |
| Water table depth (m) | 1.12 | 1.13 | 1.15 |

Specific Yield logarithmic equations: @(p,x)exp(polyval(p,x))

Time-since-fire categories 0-20, 21-80, 81-120 years correspond to i, ii and iii, respectively.

Middle and margin peat profiles correspond to rows 1 and 2 respectively.

i) =
-0.0381 0.0653

-0.0544 0.7429

ii) =
-0.0342 0.1935

-0.0537 1.3033

iii) =
-0.0412 0.4819

-0.0865 2.4339

**Table S2.13:** Water table depth – soil-water pressure linear relationship used.

| Source | Equation | $R^2$ |
|---|---|---|
| Lindholm and Markkola, 1984 – drained hollow | -1.6219x – 0.165 | 0.77 |

## APPENDIX B

### Supplementary Material Chapter 3

**Table S3.1** Percent cover of post-burn surface cover according to the scheme presented in Lukenbach et al. (2015). UD, MD and HD are undrained, moderately-drained and heavily-drained areas respectively.

| Site | Plot | % Live Sphagnum | % Singed Sphagnum | % Burnt Sphagnum | % Singed Feathermoss | % Burnt Feathermoss | % Exposed Root | % Ash |
|---|---|---|---|---|---|---|---|---|
| UD | 1 | 40 | 30 | 5 | 0 | 25 | 0 | 0 |
| UD | 2 | 0 | 15 | 5 | 70 | 10 | 0 | 0 |
| UD | 3 | 0 | 0 | 0 | 0 | 100 | 0 | 0 |
| UD | 4 | 88 | 0 | 0 | 0 | 12 | 0 | 0 |
| UD | 5 | 8 | 5 | 7 | 0 | 80 | 0 | 0 |
| UD | 6 | 8 | 15 | 20 | 0 | 57 | 0 | 0 |
| UD | 7 | 0 | 0 | 0 | 0 | 100 | 0 | 0 |
| UD | 8 | 0 | 10 | 10 | 15 | 65 | 0 | 0 |
| UD | 9 | 0 | 0 | 0 | 30 | 70 | 0 | 0 |
| UD | 10 | 3 | 85 | 0 | 0 | 12 | 0 | 0 |
| UD | 11 | 0 | 0 | 20 | 0 | 80 | 0 | 0 |
| UD | 12 | 35 | 65 | 0 | 0 | 0 | 0 | 0 |
| UD | 13 | 0 | 60 | 0 | 0 | 40 | 0 | 0 |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| UD | 14 | 0 | 0 | 0 | 0 | 100 | 0 | 0 |
| UD | 15 | 95 | 5 | 0 | 0 | 0 | 0 | 0 |
| UD Avg. | | 18 | 19 | 4 | 8 | 50 | 0 | 0 |
| MD | 1 | 0 | 0 | 0 | 0 | 100 | 0 | 0 |
| MD | 2 | 0 | 8 | 90 | 0 | 2 | 0 | 0 |
| MD | 3 | 0 | 0 | 0 | 90 | 10 | 0 | 0 |
| MD | 4 | 0 | 0 | 82 | 3 | 15 | 0 | 0 |
| MD | 5 | 0 | 0 | 0 | 30 | 70 | 0 | 0 |
| MD | 6 | 15 | 20 | 0 | 20 | 45 | 0 | 0 |
| MD | 7 | 0 | 0 | 0 | 10 | 90 | 0 | 0 |
| MD | 8 | 7 | 15 | 3 | 10 | 65 | 0 | 0 |
| MD | 9 | 0 | 0 | 0 | 5 | 95 | 0 | 0 |
| MD | 10 | 0 | 0 | 0 | 0 | 100 | 0 | 0 |
| MD | 11 | 0 | 50 | 0 | 0 | 50 | 0 | 0 |
| MD | 12 | 0 | 0 | 15 | 0 | 85 | 0 | 0 |
| MD | 13 | 0 | 10 | 20 | 0 | 70 | 0 | 0 |
| MD | 14 | 0 | 0 | 0 | 0 | 100 | 0 | 0 |
| MD | 15 | 0 | 0 | 0 | 75 | 25 | 0 | 0 |
| MD Avg. | | 1 | 7 | 14 | 16 | 61 | 0 | 0 |

| | | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| HD | 1 | 0 | 0 | 0 | 0 | 93 | 7 | 0 |
| HD | 2 | 0 | 0 | 0 | 0 | 92 | 8 | 0 |
| HD | 3 | 0 | 0 | 0 | 0 | 70 | 5 | 25 |
| HD | 4 | 0 | 0 | 0 | 0 | 98 | 2 | 0 |
| HD | 5 | 0 | 0 | 40 | 0 | 55 | 5 | 0 |
| HD | 6 | 0 | 0 | 0 | 0 | 85 | 15 | 0 |
| HD | 7 | 0 | 5 | 20 | 0 | 72 | 3 | 0 |
| HD | 8 | 0 | 0 | 0 | 15 | 80 | 0 | 0 |
| HD | 9 | 0 | 0 | 0 | 0 | 95 | 5 | 0 |
| HD | 10 | 0 | 0 | 0 | 0 | 97 | 3 | 0 |
| HD | 11 | 0 | 0 | 5 | 0 | 80 | 15 | 0 |
| HD | 12 | 0 | 0 | 0 | 0 | 0 | 0 | 100 |
| HD | 13 | 0 | 0 | 0 | 0 | 50 | 35 | 15 |
| HD | 14 | 0 | 0 | 0 | 0 | 60 | 0 | 40 |
| HD | 15 | 0 | 0 | 0 | 0 | 10 | 10 | 80 |
| HD Avg. | | 0 | 0 | 4 | 1 | 69 | 8 | 17 |

**Canopy closure estimate equation (Housman, 2017)**

$y = 0.0005x + 13.44$ $R^2=0.729$

Where y = canopy closure (%), and x = tree/tall shrub biomass (kg/ha)

**Specific allometric equations used to calculate above-ground biomass**

Johnston et al. (2015)- basal diameter (BD) based

| | |
|---|---|
| Black Spruce Branch Fuel Load (g): | EXP(2.915+2.089*(LN(BD))) |
| Black Spruce Foliage Fuel Load (g): | EXP(3.831+1.579*(LN(BD))) |
| Black Spruce Lichen Fuel Load (g): | EXP(0.454+2.415*(LN(BD))) |
| Black Spruce Total Crown Fuel Load (g): | SUM(Branch+Foliage+Lichen) |

Bond-Lamberty et al. (2002)- basal diameter (BD) based, without age condition

| | |
|---|---|
| Black Spruce Total Biomass (g): | 10^(1.743+(2.401*(LOG10(BD)))) |
| Willow Total Biomass (g): | 10^(1.534+(2.733*(LOG10(BD)))) |
| Paper Birch Total Biomass (g): | 10^(1.546+(2.41*(LOG10(BD)))) |

**Residual peat depth distribution**

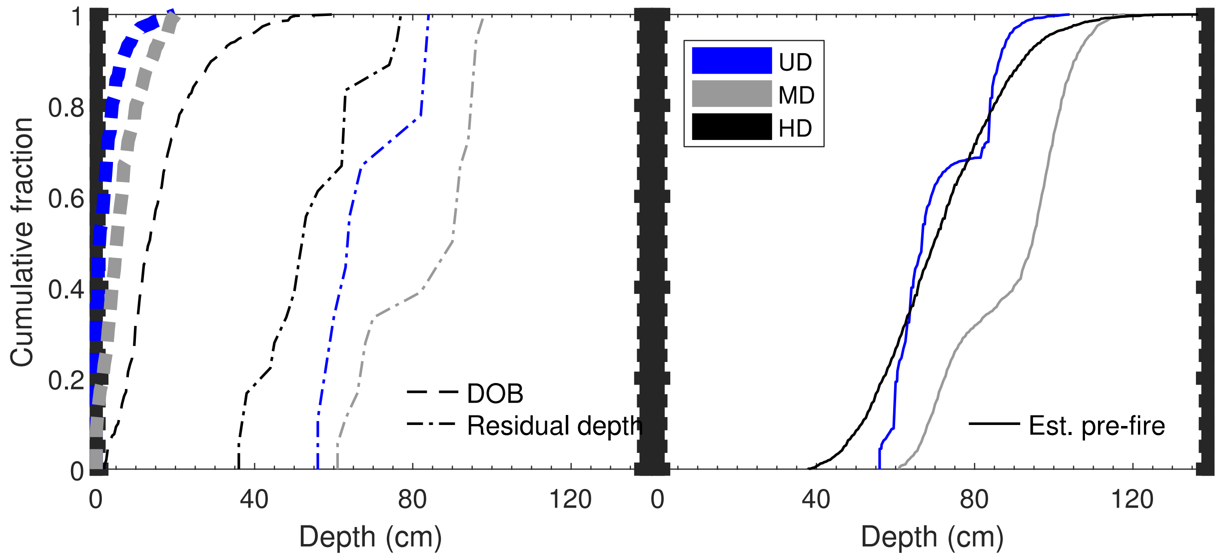



**Figure S3.1** Distribution of measured depth of burn (DOB; dashed lines), measured residual (post-fire) peat depth (dash-dot lines), and estimated pre-fire DOB (solid lines) for the undrained (UD), moderately drained (MD), and heavily drained (HD) treatments. Pre-fire estimates are derived using random resampling (10,000 times) from the DOB and residual peat depth distributions. DOB and residual peat depth presented for the HD treatment does not include the burned-to-mineral portion of the treatment.

## APPENDIX C

### Supplementary Material Chapter 4

**Table S4.1** Results of the nonparametric multiple comparisons tests conducted on plot evaporation rates per ground cover type: Burned feather moss (BFM), Singed feather moss (SFM), Burned *Sphagnum* (BSph), Singed *Sphagnum* (SS), Live *Sphagnum* (LSph), Fire Moss/ Liverwort (FrM) and Bare Peat (BrP). Dunn Method for joint ranking accounting for multiple comparisons. (*, **, *** corresponds to p≤ 0.1, p≤ 0.05, p≤ 0.01, respectively).

| **Level** | **- Level** | **Score Mean Difference** | **Std Err Dif** | **Z** | **p-Value** |
|---|---|---|---|---|---|
| SS | SFM | 32.2500 | 8.080710 | 3.99099 | 0.0024*** |
| SS | BFM | 28.0833 | 8.080710 | 3.47535 | 0.0184** |
| SFM | FrM | -27.2333 | 8.475120 | -3.21333 | 0.0472** |
| SFM | Pool | -30.6333 | 8.475120 | -3.61450 | 0.0108** |
| Pool | BFM | 26.4667 | 8.475120 | 3.12287 | 0.0645* |
| SS | BSph | 23.7500 | 8.080710 | 2.93910 | 0.1185 |
| SFM | LS | -25.4583 | 9.034508 | -2.81790 | 0.1740 |
| FrM | BFM | 23.0667 | 8.475120 | 2.72169 | 0.2338 |
| Pool | BSph | 22.1333 | 8.475120 | 2.61157 | 0.3245 |
| LS | BFM | 21.2917 | 9.034508 | 2.35670 | 0.6638 |

| | | | | | |
|---|---|---|---|---|---|
| SS | BP | 20.1167 | 8.475120 | 2.37361 | 0.6341 |
| FrM | BSph | 18.7333 | 8.475120 | 2.21039 | 0.9748 |
| Pool | BP | 18.5000 | 8.851974 | 2.08993 | 1.0000 |
| LS | BSph | 16.9583 | 9.034508 | 1.87706 | 1.0000 |
| FrM | BP | 15.1000 | 8.851974 | 1.70583 | 1.0000 |
| SS | Litter | 14.8167 | 8.475120 | 1.74825 | 1.0000 |
| LS | BP | 13.3250 | 9.388936 | 1.41922 | 1.0000 |
| Pool | Litter | 13.2000 | 8.851974 | 1.49119 | 1.0000 |
| Litter | BFM | 13.0667 | 8.475120 | 1.54177 | 1.0000 |
| Litter | BSph | 8.7333 | 8.475120 | 1.03047 | 1.0000 |
| LS | Litter | 8.0250 | 9.388936 | 0.85473 | 1.0000 |
| BP | BFM | 7.7667 | 8.475120 | 0.91641 | 1.0000 |
| SS | LS | 6.5417 | 9.034508 | 0.72408 | 1.0000 |
| Litter | BP | 5.1000 | 8.851974 | 0.57614 | 1.0000 |
| Pool | LS | 4.9250 | 9.388936 | 0.52455 | 1.0000 |
| SS | FrM | 4.8167 | 8.475120 | 0.56833 | 1.0000 |
| BSph | BFM | 4.1667 | 8.080710 | 0.51563 | 1.0000 |
| Pool | FrM | 3.2000 | 8.851974 | 0.36150 | 1.0000 |
| SS | Pool | 1.4167 | 8.475120 | 0.16716 | 1.0000 |
| LS | FrM | -1.5250 | 9.388936 | -0.16243 | 1.0000 |
| BS | BP | -3.4333 | 8.475120 | -0.40511 | 1.0000 |

| | | | | | |
|---|---|---|---|---|---|
| SFM | BFM | -4.0000 | 8.080710 | -0.49501 | 1.0000 |
| SFM | BS | -8.3333 | 8.080710 | -1.03126 | 1.0000 |
| Litter | FrM | -9.8000 | 8.851974 | -1.10710 | 1.0000 |
| SFM | BP | -11.9333 | 8.475120 | -1.40804 | 1.0000 |
| SFM | Litter | -17.2333 | 8.475120 | -2.03340 | 1.0000 |

**Figure S4.1** Relation between field-measured chamber evaporation rate and average volumetric water content (VWC) in the top 6 cm.

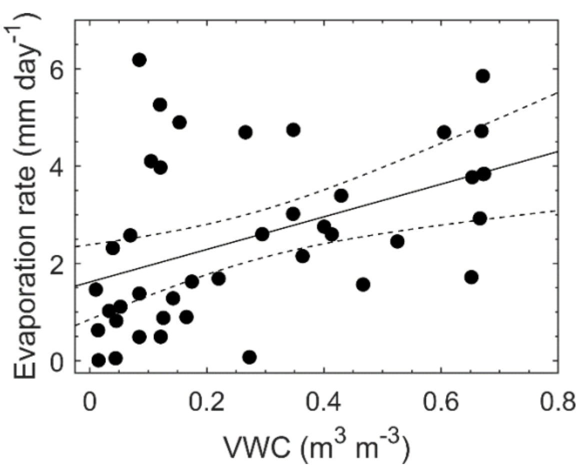


**Figure S4.2** Volumetric water content (VWC) of the top 3 cm of peat sample throughout the water table depth (WTD) manipulation experiment. Phase 1 = 5 cm WTD, phase 2 = 15 cm WTD, phase 3 = no WT, phase 4 = 5 cm (rewetted) WTD.

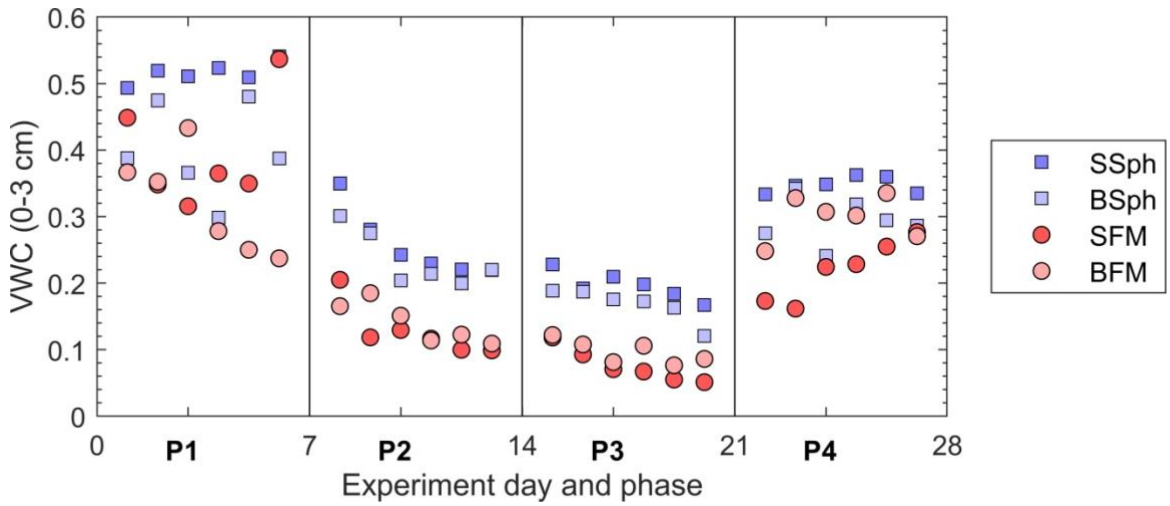
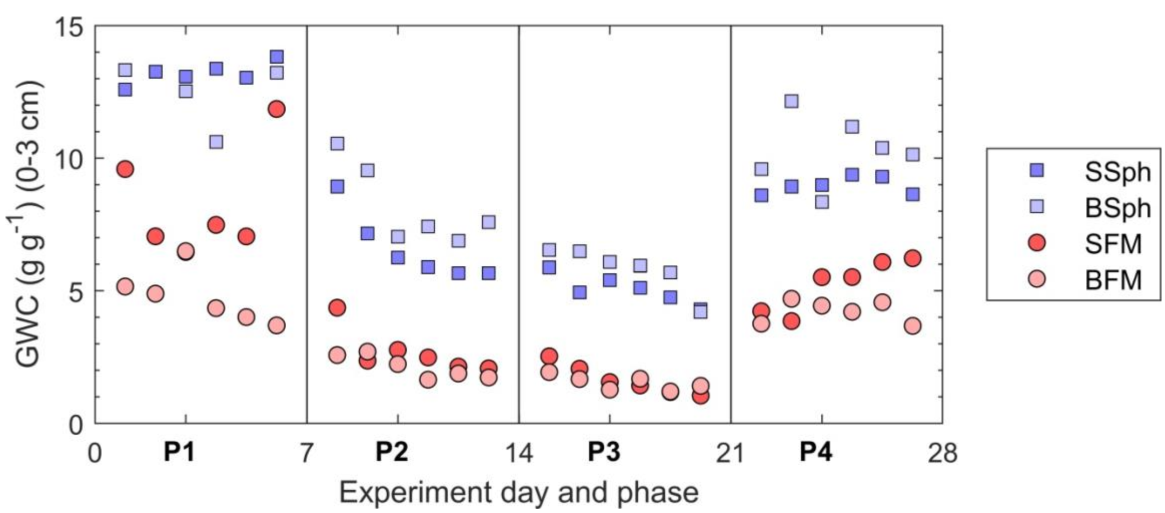



**Figure S4.3** Gravimetric water content (GWC) of the top 3 cm of peat samples throughout the water table depth (WTD) manipulation experiment. Phase 1 = 5 cm WTD, phase 2 = 15 cm WTD, phase 3 = No WT, phase 4 = 5 cm (rewetted) WTD.



**APPENDIX D**

**Supplementary Material Chapter 5**

**Table S5.1** General linear model of square-root transformed depth of burn (DOB) using log-transformed volumetric water content (VWC) as the null model predictor. ANOVA test results of alternate models against the null are shown, in addition to adjusted $R^2$ of linear model. Model specification uses Wilkinson notation.

| **Model** | **$R^2$ adj** | **dob~vwc** $\chi^2$ | **d.f.** | ***p*** |
|---|---|---|---|---|
| dob~vwc | 0.205 | – | – | – |
| dob~vwc + moss type | 0.290 | 14.7 | 1 | 0.0001* |
| dob~vwc + tree distance | 0.199 | 0.03 | 1 | 0.864 |
| dob~vwc + surf temperature | 0.204 | 0.89 | 1 | 0.346 |
| dob~vwc + mulch presence | 0.240 | 7.77 | 1 | 0.005* |
| dob~vwc + ice depth | 0.217 | 3.33 | 1 | 0.068 |

* - denotes significant difference using a 0.05 significance level.