# HYDROGEOLOGICAL AND ECOHYDROLOGICAL CONTROLS ON PEATLAND RESILIENCE TO WILDFIRE

# HYDROGEOLOGICAL AND ECOHYDROLOGICAL CONTROLS ON PEATLAND RESILIENCE TO WILDFIRE

By MAXWELL LUKENBACH, B.Sc.

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| AUTHOR: | Maxwell Lukenbach |
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## ABSTRACT

Peatlands represent a globally significant carbon stock and wildfire is the largest disturbance affecting these ecosystems. Climate change scenarios suggest that increases in evapotranspiration are likely to exceed increases in precipitation in northern latitudes, raising concern that peatlands will experience substantial drying. Drying may increase peat burn severity and, when coupled with expected increases in total wildfire area burned, may exceed peatland resilience to wildfire. While previous studies have examined both peatland vulnerability to wildfire and post-fire recovery, these studies have not examined the driest peatlands on the landscape that are likely to be the most susceptible to the combined effects of climate change and wildfire. For this reason, this thesis examined the hydrogeological and ecohydrological controls on burn severity and post-fire recovery in peatlands in the Boreal Plains of Alberta, where peatlands exist at the limit of their climate tolerance.

High burn severity was prevalent at the margins of a small peatland isolated from groundwater flow, where average burn depths were five-fold greater than in the middle of the peatland. Deep burning was attributable to the effect of dynamic hydrological conditions on margin peat bulk density and moisture. Following wildfire, water availability was a key determinant of post-fire moss recovery. Both high and low burn severity can decrease post-fire water availability by altering peat hydrophysical properties. Post-fire recovery was also dependent on large-scale hydrological processes that influence peatland water tables, specifically, hydrogeological setting. Small peatlands isolated from groundwater flow systems had lower peatland moss

recolonization rates at both their middles and margins due to drier conditions. This was important because the margins of these same peatlands were prone to deep burning. Therefore, deep burning is likely altering peatland margin ecohydrological function and may be facilitating a regime shift from peatland to mineral upland.

## ACKNOWLEDGEMENTS

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I am thankful for the opportunity provided by Drs. Bradford Wilcox and Clyde Munster to conduct research during my undergraduate studies at Texas A&M University. Although I doubted Dr. Wilcox at the time, his bold recommendation to pursue a PhD has been instrumental to my present achievement.

I am especially thankful to all those who assisted me in the field and the lab. I know that it was arduous at times. I am also grateful to the past and present members of the McMaster University Ecohydrology Research Group, as our discussions, collaborations, and inquiries were critical to my success.

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## DECLARATION OF ACADEMIC ACHIEVEMENT

This dissertation takes the form of a series of manuscripts in publication or to be submitted to peer-reviewed academic journals. There is a fair degree of repetition in the site description and methods. Each of the chapters can be read as a stand-alone document. The reference style in the text has been standardized for the purposes of this dissertation.

In addition to the work presented here, I contributed to two other peer-reviewed publications that are relevant to this thesis during the course of my graduate studies.

During the 2012 field campaign, I was involved in a study on post-fire water repellency in peat soils. I played a key role in the fieldwork and a minor role in the writing of the manuscript:

Kettridge N, Humphrey RE, Smith JE, Lukenbach MC, Devito KJ, Petrone RM, Waddington JM. 2014. Burned and unburned peat water repellency: Implications for peatland evaporation following wildfire. *Journal of Hydrology* **513**: 335-341. DOI: 10.1016/j.jhydrol.2014.03.019

Shortly after beginning the second year of my PhD, I had the pleasure of working with Kelly Hokanson during her MSc. While completing the analyses for my first manuscript in this thesis, it became apparent that there was a need to examine the spatial variability in burn severity at peatland margins. Kelly led the study and I significantly contributed by providing feedback on the experimental design, participating in data collection and analyses, and assisting in the writing process:

Hokanson KJ, Lukenbach MC, Devito KJ, Kettridge N, Petrone RM, Waddington JM. 2015. Groundwater connectivity controls peat burn severity in the Boreal Plains. *Ecohydrology. in press.* DOI: 10.1002/eco.1657

The work presented in this thesis is the result of collaborative research, and the contributions of the candidate are listed below.

**Chapter 2**
Title: Hydrological controls on deep burning in a northern forested peatland

Authorship: Lukenbach MC, Hokanson KJ, Moore PA, Devito KJ, Kettridge N, Thompson DK, Wotton BM, Petrone RM, Waddington JM.

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Candidate's contribution: Except for the depth of burn data collection that K.J. Hokanson assisted with, the candidate conducted all data collection and analysis. The draft manuscript was prepared by the candidate, with edits and feedback from the co-authors. All authors contributed to the experimental design, with funding secured by K.J. Devito, N. Kettridge, R.M. Petrone, and J.M. Waddington.

**Chapter 3**
Title: Burn severity alters peatland moss water availability: implications for post-fire recovery

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**Chapter 4**
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Candidate's contribution: The candidate conducted all data collection and analysis. The draft manuscript was prepared by the candidate, with edits and feedback from the co-authors. All authors contributed to the experimental design, with funding secured by K.J. Devito, N. Kettridge, R.M. Petrone, and J.M. Waddington.

**Chapter 5**
Title: Does groundwater connectivity control the ecohydrological resilience of peatland margins follow wildfire?

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Candidate's contribution: The candidate conducted all data collection and analysis. The draft manuscript was prepared by the candidate, with edits and feedback from the co-authors. All authors contributed to the experimental design, with funding secured by K.J. Devito, N. Kettridge, R.M. Petrone, and J.M. Waddington.

# CHAPTER 1: INTRODUCTION

## 1.1 Peatland wildfire disturbance

Peatlands are wetlands characterized by the accumulation of organic matter >0.40 m deep (National Wetlands Working Group, 1997). Peatlands provide valuable ecosystem services (Vitt et al., 2000), including the preservation of large carbon stocks (Frolking *et al*., 2011), maintenance of biodiversity (Rydin, 2013) and conservation of water resources (Devito *et al*., 2012). Wildfire is the largest disturbance affecting northern peatlands (Turetsky *et al.*, 2002), occurring as frequently as every 100-120 years in boreal regions (Turetsky *et al.*, 2004). While peatland wildfires typically result in complete stand mortality and the die-off of ground layer vegetation (Benscoter and Vitt, 2008; Zoltai *et al.*, 1998), northern peatlands are generally resilient to wildfire and return to a net carbon sink status within ~20 years post-fire (Wieder *et al.*, 2009). However, given that climate change scenarios suggest that increases in evapotranspiration are likely to exceed increases in precipitation in northern latitudes (Collins *et al*., 2013), there is concern that peatlands will experience substantial drying (Roulet *et al.*, 1992), thereby increasing their vulnerability to wildfire (Kettridge *et al.*, 2015; Sherwood *et al.*, 2013; Turetsky *et al.*, 2011b). Increases in organic layer burn severity, coupled with expected increases in total wildfire area burned of 25–100% (Flannigan *et al.*, 2005), may shift these ecosystems to net sources of carbon to the atmosphere (Turetsky *et al.*, 2002). Indeed, some simple models suggest that peatlands will represent a strong and persistent positive feedback to climate change (Ise *et al*., 2008). However, field evidence suggests that peatlands are generally resilient ecosystems, characterized by a set of negative feedbacks that regulate water and carbon storage (*c.f.* Morris and Waddington, 2011; Waddington *et al.*, 2015; Wieder *et al.*, 2009).

The resilience of ecosystems, such as peatlands, to disturbance is a growing research field, as ecosystem scientists have determined that a resilience framework is the best approach to develop impact mitigation and restoration/reclamation strategies (Carpenter *et al*., 2001). This thesis adopts such a resilience framework and adopts the following definition of ecosystem resilience from Carpenter *et al*. (2001), "Resilience is the magnitude of disturbance that can be tolerated before a socioecological system (SES) moves to a different region of state space controlled by a different set of processes." Of particular interest in this thesis is how wildfire alters and interacts with key hydrological processes that maintain ecosystem services (*e.g.* carbon storage) in peatlands.

### 1.2 Ecohydrological controls on burn severity in peatlands

Burn severity in peatland ecosystems, defined by the depth of peat combustion (*i.e.,* depth of burn, DOB), typically ranges from 0.05-0.10 m (Benscoter and Wieder, 2003; Shetler *et al.*, 2008). Because the flaming front rapidly (~45 seconds) passes during wildfire (Taylor *et al.,* 2004; Thompson *et al.,* 2015), smouldering, a slow moving and flameless form of combustion, is responsible for the majority of belowground organic matter combustion in peatlands (Rein *et al.*, 2008). Because smouldering is typically not limited by the amount of fuel (peat) present or oxygen availability (Benscoter *et al.,* 2011; Fransden, 1991; Miyanishi and Johnson, 2002; Rein *et al*., 2008), DOB has been shown to primarily be a function of peat hydrophysical properties (*i.e.,* moisture content and bulk density) (Benscoter *et al.,* 2011). That is, peat smouldering potential can be conceptualized as the balance between the energy source (peat) and energy sink (water) in a peat profile (Benscoter *et al*., 2011).

Because low moisture contents are associated with higher burn depths in organic soils (Miyanishi and Johnson, 2002; Rein *et al.*, 2008; Watts, 2013), peatland DOB is influenced by the water retention properties of ground layer vegetation. For example, in *Sphagnum* dominated peatlands, burn severity varies spatially as a function of hummock-hollow microtopography (Benscoter and Wieder, 2003; Benscoter *et al*., 2015; Shetler *et al*., 2008). Specifically, hummock species (*e.g., Sphagnum fuscum*) retain water more effectively than hollow species during water table (WT) declines (Hayward and Clymo, 1982; McCarter and Price, 2014) and, accordingly, they resist combustion during wildfire (Benscoter *et al*., 2011; Shetler *et al*., 2008). Similarly, site-level factors affecting moisture contents are also important factors affecting DOB. For example, afforestation was likely responsible for increased DOB in a temperate peatland in Scotland because it lowered moisture contents (Davies *et al.*, 2013). Furthermore, prolonged periods of high vapour pressure deficits or the presence of ice lowered moisture contents and were shown to influence DOB in an ombrotrophic bog in Alberta's Boreal Plains (Thompson and Waddington, 2013a).

While numerous studies have documented that decreases in moisture content coincide with increases in burn depths in organic soils (Miyanishi and Johnson, 2002; Rein *et al.*, 2008), bulk density is equally important in determining peat smouldering potential (Benscoter *et al.*, 2011). Although increases in bulk density result in higher moisture retention on a volumetric basis (Boelter, 1968; Thompson and Waddington, 2013a), the presence of dense peat (>100 kg $m^{-3}$) represents an increase in the amount of fuel available to generate energy in the smouldering process and an overall increase in its vulnerability to combustion (Benscoter *et al.*, 2011). Therefore, peat decomposition rates, which are a key control on bulk density (Belyea and Clymo, 2001; Moore and Basiliko, 2006), also influence burn severity. For example, *Sphagnum* hummock species have more recalcitrant litter than hollow species (Turetsky *et al.*, 2008) (*e.g.*, *Sphagnum angustifolium, Pleurozium schreberi, Cladonia spp.*), resulting in higher bulk densities and lower gravimetric moisture contents at depth in hollows than in hummocks in sub-humid regions (Benscoter and Wieder, 2003). This contributes to higher DOB in hollows (~0.05 to 0.15 m) than in hummocks (~0.02 m) (Benscoter *et al.*, 2011; Benscoter and Wieder, 2003).

Recent research has documented high peat burn severity (depth of burn of 0.25–0.30 m; carbon loss of 16.8 kg C $m^{-2}$) in a northern Alberta peatland where the WT position was experimentally lowered by approximately 0.25 m thirty years prior to wildfire (Turetsky *et al.*, 2011a). Drainage decreased peatland vegetation cover and moisture contents, while it increased peat bulk density (Silins and Rothwell, 1998; Sherwood *et al.*, 2013), resulting in high burn severity (Turetsky *et al.*, 2011a). Similarly, high peat burn severity (depth of burn of 0.25–0.30 m) has been observed in black spruce forests in Alaska with intermediate organic layer thickness (0.20-0.30 m), where burn depths were affected by site drainage condition, permafrost condition, and the soil thermal regime (Kasischke and Johnstone, 2005; Turetsky *et al.*, 2011b). These observations of high peat burn severity highlight how hydrological conditions play a key role in the vulnerability of peatlands to wildfire. Moreover, they indicate that transitions to drier hydrological regimes prior to fire increase DOB by altering peat hydrophysical properties (Benscoter *et al*., 2011; Thompson and Waddington, 2013a; Turetsky *et al*., 2011).

**1.3 Controls on post-fire recovery in peatlands**

Post-fire recovery in peatlands is characterized by the recolonization of peat-forming vegetation (Benscoter and Vitt, 2008). In particular, *Sphagnum* mosses have been the focus of post-fire recovery studies because of their primary role in the formation of peat soils (Benscoter and Vitt, 2008; Rydin, 2013; Thompson and Waddington, 2008). Because *Sphagnum* mosses are characteristic of drier peatland ecosystems (Rydin, 2013), understanding the post-fire recovery of this genus is critical in understanding the resilience (*i.e.*, the ability of the ecosystem to recover following a disturbance) of peatlands to the combined effects of climate change and wildfire. Moreover, these bryophytes are key ecosystem engineers that increase peatland resilience to wildfire by lowering decomposition rates (Rydin, 2013), conserving water during drought (Kettridge and Waddington, 2014), and limiting burn severity during wildfire (Shetler *et al.*, 2008).

Because *Sphagnum* mosses are characteristic of low nutrient status (Rydin, 2013), water availability plays the prevailing role in determining their recolonization after wildfire (Waddington *et al.*, 2010). As such, water stress in *Sphagnum* mosses has been correlated with depth to water table (WT) and volumetric moisture content ($\theta$) (McNeil and Waddington, 2003; Tuittila *et al.*, 2004); however, tension ($\Psi$) (i.e. pore water pressure, suction) is a more accurate indicator of stress because it links large-scale hydrological processes to cellular-scale water requirements in mosses (Thompson and Waddington, 2008). The commonly cited $\Psi$ threshold that restricts *Sphagnum* moss recovery is -100 to -400 cm (Hayward and Clymo, 1982); however, the exact threshold is disputed and is species specific (Hajeck and Beckett, 2008; Lewis, 1988; McCarter and Price, 2014). Although hummocks are more resistant to combustion than hollows (Shetler *et al*., 2008), $\Psi$ was higher in hummocks than hollows after wildfire in an ombrotrophic bog in Alberta (Thompson and Waddington, 2013b). High $\Psi$ may be attributable to low WTs (Thompson and Waddington, 2013b), the presence of ice (Thompson and Waddington, 2013b), or the presence of water repellency (Kettridge *et al*., 2014). Furthermore, the increased probability of high $\Psi$ in hummocks may explain why *Sphagnum* moss recovery was more rapid in hollows than in hummocks (Benscoter, 2006; Benscoter and Vitt, 2008). However, despite recovery occurring more slowly in hummocks than hollows, post-fire recovery usually occurs in both microforms, and is signified by a return to *Sphagnum* dominance 30-70 years post-fire (Benscoter and Vitt, 2008; Wieder *et al*., 2009).

Post-fire water availability is not only a function of small-scale ecohydrological conditions but also the water balance of a peatland (Thompson and Waddington, 2008). Notably, the combustion of the tree canopy increases the amount of available energy for evaporation following wildfire due to a decrease in shading. Despite rapid (~1 year) herbaceous shrub recovery, which reduces the amount of solar radiation reaching the peat surface (Thompson *et al*., 2015), the increase in surface evaporation can exceed the reduction in transpiration and interception resulting in a net increase in post-fire peatland evapotranspiration (Kettridge et al., 2013; Thompson *et al*., 2015). Increased surface evaporation coupled with low specific yields in burned hollows may have caused deeper WT declines after wildfire (Thompson and Waddington, 2014). However, although Thompson and Waddington (2014) observed drier post-fire conditions, the time since the previous wildfire was only 81 years, which is shorter than the average fire return interval for peatlands in the region of study (Boreal Plains in Alberta) (Turetsky *et al*., 2004; Wieder *et al*., 2009). Therefore, the burning of forested peatlands in late successional stages may raise WT positions because the loss in transpiration and interception outweighs the increase in surface evaporation (Thompson and Waddington, 2014).

Burn severity can also influence post-fire recovery of peatlands by altering peat hydrophysical properties (Benscoter and Vitt, 2008; Sherwood *et al*., 2013; Kettridge *et al*., 2015). High DOB not only alters the surface datum, typically resulting in shallower post-fire WT positions, but also exposes dense peat at the surface after fire (Thompson and Waddington, 2013a). This substrate can decrease the amount of water available to peatland mosses (Sherwood *et al*., 2013; Kettridge *et al*., 2015). Furthermore, large changes in peat hydrophysical properties likely explain why moderate drainage (~ 0.20 m lower WT position) in a drained fen followed by deep burning (~ 0.20 m) resulted in a post-fire regime shift to a non-carbon accumulating shrub-grass dominated ecosystem (Kettridge *et al*., 2015).

## 1.4 Towards a landscape-scale understanding of burn severity and post-fire recovery in peatlands

While previous research has highlighted the importance of hydrological controls on both peatland burn severity and post-fire recovery, the role of landscape-scale hydrological processes has yet to be investigated. Specifically, because hydrogeological setting defines both the mineral substrate composition (*i.e.* texture) and topographic position at a particular position on the landscape, it influences peatland WT dynamics (Winter, 1999; Winter *et al.,* 2003). As such, prior studies may only have a limited understanding of the spatial variability in burn severity and post-fire recovery in peatlands. In particular, small peatlands isolated from groundwater flow systems can undergo large changes in water storage due to high potential water demand from adjacent mineral upland ecosystems, where actual evapotranspiration is higher than in peatlands (Petrone *et al.*, 2007), resulting in the movement of water from peatlands to mineral uplands (Brown *et al.,* 2014). This is important because it not only influences WT dynamics in the middle of the peatland but also results in periodic, deep (1-2 m) WT drawdowns at peatland margins (Ferone and Devito, 2004). Therefore, peat burn severity and post-fire recovery has likely not been examined in peatlands that are most vulnerable to drying on the landscape.

The underrepresentation of peatlands situated in dry hydrological settings on the landscape is particularly important because these peatlands are likely to be the most vulnerable to climate change and wildfire. Moreover, this gap of knowledge in peatland wildfire research is especially important from a wildfire management perspective, as peat fires currently are problematic for fire management as they require increased financial and suppression resource commitments (Rein *et al.*, 2009; Davies *et al.,* 2013). Once surface and crown fires are contained by a control perimeter, smouldering peat must be extinguished to prevent subsequent flare-up and fire escape. This is very time-consuming, labour-intensive, and serves as a serious drawdown on firefighting resources for suppressing new fires.

In light of the aforementioned research gap and the growing peatland wildfire risk, the aim of this thesis is to gain a better understanding of the current and future peatland wildfire risk on the landscape by examining the hydrogeological and ecohydrological controls on burn severity and post-fire recovery in peatlands. In May 2011, a ~90,000 ha fire burned three peatlands along a hydrogeological transect that were part of a long-term hydrological monitoring network at the Utikuma Lake Research Study Area (URSA; 56.107$^{o}$N 115.561$^{o}$W) in Alberta's Boreal Plains ecozone (Devito *et al.,* 2012). This provided an opportunity to examine the spatial variability in burn severity and post-fire recovery both within and between these peatlands. Within this study the specific objectives of the study were:

1. To determine the influence of dynamic hydrological conditions on the spatial variability of peat burn severity (DOB) in peatlands.
2. To determine how post-fire ecohydrological conditions influence the spatial variability in the recovery of peat-forming vegetation in peatlands.
3. To determine how hydrogeological setting, through its influence of peatland WTs,

affects the post-fire recovery of peat-forming vegetation.

4. To determine how hydrogeological setting and burn severity interact to influence peatland ecohydrological function.

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# CHAPTER 2: HYDROLOGICAL CONTROLS ON DEEP BURNING IN A NORTHERN FORESTED PEATLAND

## 2.1 Abstract

While previous boreal peatland wildfire research has generally reported average organic soil burn depths ranging from 0.05-0.20 m, here we report on deep burning in a peatland in the Utikuma Complex forest fire (SWF-060, ~90,000 ha, May 2011) in the sub-humid climate of Alberta's Boreal Plains. Deep burning was prevalent at peatland margins, where average burn depths of 0.42 ± 0.02 m were five-fold greater than in the middle of the peatland. We examined adjacent unburned sections of the peatland to characterize the hydrological and hydrophysical conditions necessary to account for the observed burn depths. Our findings suggest that the peatland margin at this site represented a smouldering hotspot due to the effect of dynamic hydrological conditions on margin peat bulk density and moisture. Specifically, the coupling of dense peat (bulk density >100 kg $m^{-3}$) and low peat moisture (GWC < 250 %) at the peatland margin allowed for severe smouldering to propagate deep into the peat profile. We estimated carbon release from this margin 'hotspot' ranged from 10–85 kg C $m^{-2}$ (mean = 27 kg C $m^{-2}$) accounting for ~80% of the total soil carbon loss from the peatland during the wildfire. As such, we suggest that current estimations of peatland carbon loss from wildfires that exclude (and/or miss) these 'hotspots' are likely underestimating total carbon emissions from peatland wildfires. We conclude that assessments of natural and managed peatland vulnerability to wildfire should focus on identifying dense peat on the landscape that is vulnerable to drying.

## 2.2 Introduction

Peat wildfires have recently received international attention due to their challenging fire management issues and their long-range transport of pollutants such as $PM_{2.5}$ (particulate matter <2.5 um) (Betha *et al.*, 2013) and mercury (Sigler *et al.*, 2003; Turetsky *et al.*, 2006) that have detrimental and far-reaching health impacts (Rittmaster *et al.*, 2006). These impacts are exemplified by the 2010 Russian heat wave and wildfires in drained peatlands, where air pollution from these burned peatlands was responsible for over 3,000 deaths in Moscow (Shaposhnikov *et al.,* 2014). Peat wildfires also represent a large source of carbon to the atmosphere impacting over 1500 $km^2$ and releasing 4700 Gg C annually in Western Canada alone (Turetsky *et al.*, 2002). In sub-humid regions of the boreal forest (*e.g.* Western Canada), peatlands burn as frequently as mineral uplands at 100–120 year intervals (Turetsky *et al.*, 2004). The majority of carbon released from northern peatlands during wildfire is due to smouldering (Benscoter *et al.*, 2011), with typical peat burn depths ranging from 0.05–0.10 m and releasing 2–3 kg C $m^{-2}$ (Benscoter and Wieder, 2003; Shetler *et al.*, 2008). However, recent research documented high peat burn severity (depth of burn of 0.25–0.30 m; carbon loss of 16.8 kg C $m^{-2}$) in a northern Alberta peatland where the water table (WT) position was experimentally lowered by approximately 0.25 m several years prior to wildfire (Turetsky *et al.*, 2011a). Similarly, high peat burn severity (depth of burn of 0.25–0.30 m) has been observed in black spruce forests in Alaska with intermediate organic layer thickness (0.20-0.30 m), where burn depths were affected by site drainage condition, permafrost condition, and the soil thermal regime (Kasischke and Johnstone, 2005; Turetsky *et al.*, 2011b). Given that portions of natural peatlands without permafrost can undergo periodic deep WT drawdown during periods of drought (Ferone and Devito, 2004), it is likely that depth of burn 'hotspots' exist in these units of the landscape. This would suggest that a comprehensive understanding of how hydrological processes influence high burn severity is essential in identifying areas that are vulnerable to deep burning on the landscape (Kasischke and Johnstone, 2005; Benscoter *et al.,* 2011). However, we are unaware of studies that have directly linked peatland hydrological processes and peat burn severity. Consequently, the overall aim of this article was to determine the hydrological and hydrophysical controls on peat burn severity in a peatland complex. We carried out this research in Canada's Boreal Plains (BP) ecozone because they are known to be affected by wildfire (Turetsky *et al.* 2004).

Peatlands in the BP represent ~10% of the global peatland carbon stock (Vitt *et al.*, 2000) and are particularly vulnerable to wildfire because they exist at the limit of their climate tolerance, where annual precipitation typically does not exceed annual potential evapotranspiration (Devito *et al.*, 2012). Peatlands in the BP of Alberta cover ~25–75% of the landscape and forested peatlands in this region undergo large and multi-annual WT drawdown during dry periods of a decadal climate cycle (Thompson, 2012). The riparian transitional zones (~8 to 10 m wide), which Dimitrov *et al.* (2014) refer to as boreal ecotones between forested mineral uplands and peatlands (hereafter referred to as margins) are especially vulnerable to drying. Here, declines in WT position of up to 1–2 m (Ferone and Devito, 2004; Redding and Devito, 2008) occur due to high potential water demand from adjacent mineral upland ecosystems, where freshet recharge is variable and actual evapotranspiration is higher than in peatlands (Petrone *et al.*, 2007), resulting in the movement of water from peatlands to mineral uplands (Brown *et al.,* 2014). This WT decline not only reduces peat profile moisture content, but prolonged periods of low WT also increase peat bulk density prior to fire through subsidence (Price, 2003) and enhanced decomposition (Blodau *et al.*, 2004). Burn depths are deeper in organic soils with low moisture contents (Miyanishi and Johnson, 2002; Rein *et al.*, 2008), high peat bulk densities (Benscoter *et al.*, 2011) and intermediate organic layer thickness (Kasischke and Johnstone, 2005), suggesting that peatland margins may represent the prime location for peat burn severity 'hotspots' in the BP. Although peatland margins may comprise a relatively small area of a peatland, the potentially large depth of burn could represent a significant portion of carbon loss during fire. This would likely be enhanced in peatland complexes that experience greater mineral upland influence because of their smaller area to perimeter ratio (Thompson, 2012).

## 2.3 Methods

In May of 2011, a ~90,000 ha wildfire burned a portion of a small peatland complex (~5 ha) located between two esker ridges on a coarse-textured outwash plain in the Utikuma Lake Research Study Area (URSA; 56.107$^{o}$N 115.561$^{o}$W) of the BP. We measured burn depths ~2 years after fire throughout the burned portion of the peatland complex (see section 2.2). Additionally, we measured WT position, bulk density, and peat moisture retention in the middle and margins of both burned and unburned portions of the peatland complex. Our moisture and bulk density data from the unburned site was used to parameterize a simple organic soil consumption model (Van Wagner, 1972), modified for peat profiles (Benscoter *et al.*, 2011), to characterize the hydrological and hydrophysical conditions necessary to produce the observed burn severity.

*2.3.1 Study site*

The URSA is characterized by low topographic relief and deep heterogeneous glacial substrates (Vogwill, 1978). The nature of the crown fire resulted in a mosaic of burned and unburned peatlands. For this study, we examined a small peatland complex (~5 ha) located in the coarse textured glaciofluvial outwash composed of numerous small burned and unburned peatlands separated by forested uplands (Figure 2.1). The burned and unburned portions (hereafter referred to as sites) within the peatland complex (located ~300 m apart) are approximately 100 x 150 m and 90 x 150 m in size, respectively, and are both surrounded by aspen (*Populus Tremuloides*) forest with a stand age of ~135 years (cf. Devito *et al*., 2012; Brown *et al.,* 2010). Prior to the fire, both sites were characterized by feather moss lawns, *Sphagnum fuscum* hummocks, vascular vegetation cover of *Rhododendron groenlandicum* and *Rubus chamaemorus*, and a dense black spruce (*Picea mariana*) tree canopy with a stem density of approximately 7,000 and 4,000 stems per hectare at the burned and unburned sites, respectively. For more details of the peatland complex and its associated hydrology see Smerdon *et al.* (2005).

*2.3.2 Depth of burn measurements*

Depth of burn was estimated in May of 2013 by assuming the pre-fire surface between multiple reference points (adventitious roots and/or surfaces unaltered by the fire) was flat prior to fire. This difference between the burned surface and the reconstructed surface was taken to be the DOB, which has been previously used by Kasischke *et al*. (2008) and Mack *et al*. (2011). Although we assumed the adventitious roots of black spruce to be the pre-fire peat surface, black spruce trees in unburned peatlands in the URSA have no adventitious roots exposed at the surface. On average, the depth of peat and/or live moss layer above the roots is 0.055 ± 0.004 m ($n$=210), suggesting that our depth of burn measurements are conservative. This is consistent with Kasischke *et al*. (2008) that reported black spruce adventitious roots were 0.051 ± 0.002 m below the pre-fire peat surface. Because of the low mineral content of peat, there was little to no ash present at the surface (< 2 cm) and smouldering had extinguished at the time of measurement. Each site was classified into two zones (middle and margin) associated with the distance from the peatland-upland interface. The middle of the peatland was characterized by the presence of *Sphagnum* hummock microtopography. In total, 920 depth of burn measurements were collected within the two zones: middle (n = 580) and margin (n = 340) (Figure 2.2). All locations experienced burning, however, some of these locations were only singed and had no appreciable depth of burn (< 0.01 m; 35% middle and 0% margins).

*2.3.3 Hydrological measurements and peat characteristics*

Hydrological and hydrophysical data were not collected prior to wildfire at the sites; however, we monitored post-fire hydrological conditions at both the burned and unburned sites. At both sites, WT position was continuously recorded by pressure transducers (Solinst) at 20 min intervals from May 2013–September 2013 in 0.05 m diameter PVC wells at middle and margin locations (data reported from one middle and margin well per site). Hydraulic gradients were determined by pairing water level measurements with survey data of the wells from a real-time kinematic GNSS differential GPS (Trimble R8; accuracy +/- 0.015 m). Because the upper peat layers were burnt off in the middle and margins of the burned site, we characterized the hydrological and hydrophysical properties of peat at the unburned site to examine the necessary conditions for deep burning to occur. At the unburned site, tension ($\Psi$) was measured 2–3 times per week at 0.05, 0.15, and 0.30 m depths in three middle and margin locations using 0.02 m outside diameter tensiometers (Soil Measurement Systems). A total of 24 (12 middle and 12 margin) peat cores (0.45 m in length) were extracted randomly throughout the middle and margin zone from the unburned site and analyzed for bulk density at 0.05 m increments, with half of these cores (6 middle and 6 margin) also analyzed for peat moisture retention every 0.05 m. We determined the moisture retention curve of each 0.05 m peat 'puck' using saturated porous plates (Soil Moisture Equipment Corp.) with an air entry pressure of 1000 cm (1 cm = 1 mb). We conducted our measurements inside sealed acrylic chambers in which we maintained a high relative humidity to minimize evaporative losses from the samples. We subjected the peat 'pucks' to several constant tensions (10, 20, 30, 40, 50, 75, 100, 150 and 200, and 500 cm), each for 24 hours or until water outflow from the pressure plate had ceased, whichever was longer, so as to ensure pressure equilibration. Following all moisture retention measurements, we dried the peat 'pucks' at 85 °C until no change in the sample weight was observed to calculate dry bulk density. We estimated water content at saturation (Ψ=0 cm), equal to porosity, from the measured bulk density, and assuming a particle density value of 1.47 g $cm^3$ (Redding and Devito, 2006). Peat moisture retention curves were modelled using a form of the van Genuchten equation (van Genuchten, 1980):

$$\theta(\psi) = \frac{\theta_s}{[1 + (\alpha \mid \psi \mid)^n]^{1-1/n}} \qquad (1)$$

where $\theta(\Psi)$ is the water retention curve [$m^3\, m^{-3}$], $\Psi$ is the tension (cm of water), $\theta_s$ is the saturated water content ($m^3\, m^{-3}$), α is related to the inverse of air entry tension ($cm^{-1}$), and $n$ (unitless) is a measure of the pore-size distribution. A number of previous studies have successfully applied this equation to model peat moisture retention curves (*e.g.* Weiss *et al.,* 1998; Sherwood *et al.*, 2013).

*2.3.4 Energy balance model*

In order to evaluate the relative vulnerability of margin and middle peat to burning and to characterize the hydrological and hydrophysical controls on burning, we parameterized a simple energy balance model (Benscoter *et al.*, 2011) that has been applied to explain landscape patterns of burn severity in peatlands. Because the flaming front of a wildfire usually passes rapidly (~45 seconds) (Taylor *et al.,* 2004; Thompson *et al.,* 2014), landscape patterns of smouldering are almost exclusively dictated by hydrophysical properties of the peat (Benscoter *et al.*, 2011; Rein *et al.*, 2008). As such, peat smouldering potential can be approximated by determining the ratio between energy release ($H_{\text{comb}}$) and energy required for ignition ($H_{ign}$) between successive layers (0.05 m in this study) in a peat profile. The amount of energy released from a combusting peat layer is defined by:

$$H'_{comb} = \rho_{b(i)} x_{(i)} E_{(comb)} \qquad (2)$$

where $\rho_b$ is the bulk density (g m$^{-3}$), $x$ is the thickness (m) of the fuel horizon $i$ and $E_{(comb)}$ is the low heat of combustion per unit mass of peat (14.2 kJ g$^{-1}$ for milled peat from *Frandsen*, 1991). Similarly, the amount of energy required for combustion of a fuel horizon can be determined by:

$$H'_{ign(i)} = h_{(i)} \rho_{b(i)} x_{(i)} \qquad (3)$$

where $h$ is the heat of ignition for the fuel horizon. The heat of ignition is defined by:

$$h = mC_w(T_v - T_A) + L_v m + C_f(T_{comb} - T_A) + S \qquad (4)$$

where $m$ is the gravimetric moisture content (g water g$^{-1}$ fuel by dry weight), $C_{\text{w}}$ is the heat capacity of water (4.186 J g$^{-1}$ °C$^{-1}$), $C_f$ is the specific heat of the dry fuel (1.92 J g$^{-1}$ fuel °C$^{-1}$), $T_v$ is the vaporization temperature of water (100 °C), $T_{comb}$ is the combustion temperature of peat (300 °C), $T_{\text{A}}$ is the ambient temperature (°C), $L_v$ is the latent heat of vaporization of water (2250 J g$^{-1}$), and $S$ is the energy required to liberate water molecules sorbed to peat (50.4 J g$^{-1}$). Consequently, $H_{\text{comb}}/H_{ign}$ ratios <1 between two layers indicate a low smouldering potential because the energy sourced from combusting peat does not exceed the energy required for subsequent combustion of underlying peat.

However, a $H_{\mathrm{comb}}/H_{ign}$ ratio of two indicates that only 50% of the energy produced by the combusting peat layer would need to be transferred downward to the underlying layer in order for smouldering to propagate, which falls into the range of previously reported downward efficiencies (Frandsen, 1998; Schneller and Frandsen, 1998). It was not our aim to fully model the smouldering process but rather to examine relative differences in smouldering potential on the landscape to explain the burn severity measurements. For a detailed examination of smouldering in peat soils see Huang and Rein (2014).

Peat moisture content for a given Ψ varies as a function of peat properties (primarily bulk density) (Boelter, 1968; Weiss *et al.*, 1998). Therefore, estimating drying scenarios based solely on peat moisture content (measured or simulated) would not have allowed us to apply uniform drying scenarios between peat cores and would have led to inconsistencies in drying within and between middle and margin sites. Consequently, we made use of model Ψ profiles because it allowed us to account for spatial variability in peat properties and capture the influence of distinct hydrological conditions (*i.e.* drying scenarios). Model Ψ profiles conservatively estimated the relative difference in vulnerability to burning between the middle and margins of the unburned site and were informed by measured Ψ data. An exponentially declining Ψ profile with depth from the surface was applied because it provided a conservative estimate of the relative difference in vulnerability to burning between middle and margins of the unburned site (Figure 2.3, see section 3.2). We modelled this exponential declining Ψ profile using the mathematical function,

$$\psi(z) = \psi_{\max}\left(\frac{|z - ep|}{ep}\right)^2 \qquad (5)$$

where $\Psi$ is tension (cm), $\Psi_{max}$ is the difference between expected pore water tension at the surface under an equilibrium profile condition and actual pore water tension (cm), $z$ is the depth below the peat surface (m), *ep* is the 'equilibrium point' (m), or where in the peat profile pore water Ψ is under an equilibrium profile condition. At the surface, Ψ was defined as the highest Ψ observed (0.05 m depth) during the post-fire study period at either the middle or margin of the unburned site. Because Ψ measurements were not made below a depth of 0.30 m, the exponential decline in Ψ was conservatively modelled to transition to an equilibrium profile at ≥ 0.40 m deep. An equilibrium profile can be defined as a condition where Ψ is equal to the height above the WT in the unsaturated zone.

While Ψ measurements in the middle of the unburned site were of lower magnitude than the margin, we applied the margin Ψ profile to the middle cores to provide a dry ('worst-case') scenario. By using model Ψ profiles as an input parameter for measured moisture retention curves from each peat core, we estimated *m* in each 0.05 m layer. Parameterized with the observed hydrophysical properties and observed Ψ, we obtained a conservative estimate of the relative difference in smouldering potential between middle and margin sites from the energy balance model.

To examine the sensitivity of the energy balance model to variations in bulk density, WT, and moisture conditions as well as to ensure that our results were not an artifact of the selection of specific disequilibrium profiles (*i.e.* drying scenarios), an additional theoretical modelling exercise was performed. $H_{\mathrm{comb}}/H_{ign}$ ratios were modelled under four hypothetical conditions: 1) low bulk density profile, 0.40 m WT depth, 2) high bulk density profile, 0.40 WT depth, 3) low bulk density profile, 0.80 m WT depth, 4) high bulk density profile, 0.80 m WT depth. High and low bulk density values were taken to be the average bulk density profiles for middle (low bulk density) and margin (high bulk density) sampling locations listed in Table 2.1. Because bulk density was not measured below 0.50 m, we conservatively assumed that bulk density at depths > 0.50 m remained the same as in the 0.45-0.50 m layer. In each scenario, bulk density profiles and WTs were held constant whilst $H_{\mathrm{comb}}/H_{ign}$ ratios were calculated under variable disequilibrium pore water Ψ within the unsaturated zone. Water content was estimated using the peat moisture retention dataset and parameterizations described in Moore *et al.*, 2015, which consists of *Sphagnum* peat. The use of moisture retention characteristics from Moore *et al.*, 2015 (not the middle and margin peat cores sampled in this study) allowed for an examination of the impact of dense peat and drying on $H_{\mathrm{comb}}/H_{ign}$ ratios independent of differences due to moisture retention characteristics.

### *2.3.5 Statistical analyses*

In general, statistical analyses were performed using parametric statistical tests. No transformations were performed on the data for the statistical analyses; however, where residuals were not normally distributed (*e.g.* depth of burn), non-parametric tests were used. Means and standard error of the means are reported unless otherwise stated. When reporting of the paired *t*-test results, subscripts denote the degrees of freedom.

## 2.4 Results

### *2.4.1 Peat depth of burn*

Depth of burn ranged from 0.00-1.10 m within the burned site and was highly dependent on location within the peatland complex. Mann-Whitney U tests showed that peatland margins had significantly greater burn depths (0.42 ± 0.02 m; $p < 0.001$) than middle (0.08 ± 0.01 m) locations (Figure 2.4). At the burned site, the northern margin was the most severely burned location (see Figure 2.2 for cross section) and exhibited burn depths as high as 1.10 m (Figure 2.4).

### *2.4.2 Hydrological measurements*

During the May 2013–September 2013 measurement period, the WT at the burned site was on average 0.36 m (range = 0.23–0.49 m) below the surface of the middle measurement location and 0.57 m (range = 0.35–0.73 m) above the surface at the margin measurement site due to vertical combustion of margin peat. At the unburned reference site, the WT was on average 0.85 m (range = 0.79–0.98 m) and 0.47 m (range = 0.32–0.63 m) below the surface at the margin and middle measurement locations, respectively. Hydraulic gradients indicated that the dominant direction of flow was from the middle of the peatland to its margins in both the burned and unburned sites. During the study period, hydraulic gradients averaged 0.008 ± 0.001 and 0.002 ± 0.002 between the middle and margin of the burned and unburned peatland, respectively.

In the middle of the unburned site, observed Ψ values were typically lower than the model Ψ profile that was applied (Figure 2.3). The opposite was true at the margins of the unburned site, where observed Ψ values were generally greater than model Ψ profile and this difference was most pronounced at depths of 0.15 m and 0.30 m (Figure 2.3). This resulted in the model Ψ profiles of the margin (middle) of the unburned site being wetter (drier) than were observed and decreased the relative difference in vulnerability to burning between the middle and margins of the unburned site, as indicated by modelled gravimetric water contents and $H_{\mathrm{comb}}/H_{ign}$ ratios (see section 3.4 below).

*2.4.3 Peat properties*

Bulk densities at the unburned site were significantly higher in margins (150 ± 22 kg m$^{-3}$) than middle locations (86 ± 20 kg m$^{-3}$, paired $t_{83}$ = 6.36, $p$ < 0.001). Differences in bulk density were greater in the upper 0.25 m of the peat profile (paired $t_{11}$ = >3.23, $p$ < 0.01 for each 0.05 m layer) than in the lower 0.25–0.40 m (paired $t_{11}$ = >2.50, $p$ < 0.05 for each 0.05 m layer) and were not significantly different in the 0.40–0.45 m layer (Table 2.1). Bulk density also increased significantly with depth, following a log-linear relationship with $r^2$ equal to 0.59 at middle locations ($p$ < 0.001, $n$ = 84) and 0.61 at margin locations ($p$ < 0.001, $n$ = 84). At both the middle and margin locations, a robust ordinary least squares (OLS) regression showed that bulk density explained the majority of the variation in the peat moisture retention at all Ψ ($p$ < 0.001, $n$ = 84), with $r^2$ values ranging from 0.72 to 0.82 (min and max $r^2$ at $\Psi$ = 10 cm and 200 cm, respectively).

Measured gravimetric water contents ($m$) obtained from the peat moisture retention analyses followed similar trends as bulk density, where margins exhibited lower $m$ values than middle at the unburned site (Figure 2.5). For all middle peat samples, average values were 1055 ± 189%, 692 ± 117%, 623 ± 107%, and 520 ± 93% at 10 cm, 50 cm, 100 cm (Table 2.1) and 500 cm of Ψ, respectively. Similarly, the average margin values were 467 ± 83%, 316 ± 46%, 290 ± 41%, 252 ± 35% at 10 cm, 50 cm, 100 cm and 500 cm of Ψ, respectively. Although $m$ values were typically higher in middle samples, there was substantial variation between samples at a given depth (Figure 2.5). At the 100 cm pressure step, $m$ values were significantly higher in middles for all depth increments (paired $t_5$ = >2.31, $p < 0.05$ for each 0.05 m layer) except for the 0.15-0.20 m (paired $t_5$ =1.83, $p = 0.09$) and 0.20-0.25 m ($t_5$ =1.93, $p = 0.08$) layers. In contrast to bulk density, $m$ did not show a relationship with depth for all pressure steps. The van Genuchten parameters ($n$ and $\alpha$) showed trends with location and depth (Table 2.1). The $n$ parameter was significantly higher in middle than margin samples (paired $t_{47} = 5.05$, $p < 0.001$); however, there was no significant difference in $\alpha$ between these locations. A robust OLS regression showed that depth was a significant predictor of $\alpha$, with $r^2$ equal to 0.17 and 0.48 in middle and margin samples ($p < 0.01$, $n = 84$), respectively. The same was also true for $n$, where $r^2$ was equal to 0.50 and 0.09 at these same locations, respectively ($p < 0.05$, $n = 84$).

### *2.4.4 Energy balance model*

Trends in the modelled margin and middle depth profiles of $H_{comb}/H_{ign}$ ratios closely follow those of gravimetric water content (i.e. *m*) values (see section 3.3, Figure 2.6). For all depths in the peat profile, $H_{comb}/H_{ign}$ ratios were significantly higher at margin (1.75 ± 0.10) than middle (0.90 ± 0.06) locations ($t_{84}$ = 7.06, $p < 0.001$) (Figure 2.6b). Robust OLS regression indicated there were significant, but weak, decreasing trends in $H_{\mathrm{comb}}/H_{ign}$ ratios with depth with $r^2$ equal to 0.19 and 0.13 in middle and margin locations, respectively ($p < 0.05$, $n = 84$). Figure 2.7 shows $H_{comb}/H_{ign}$ ratios for *Sphagnum* peat as a function of disequilibrium pore water Ψ under the four hypothetical scenarios (see section 2.4). $H_{comb}/H_{ign}$ ratios range from 1.0 - 1.8 and increase with higher bulk density, lower WT conditions, and increasing disequilibrium pore water Ψ.

## 2.5 Discussion

### *2.5.1 Hydrophysical and hydrological controls on deep burning 'hotspots'*

Our peatland margin burn depths were higher than previously measured boreal peat burn depths (Turetsky *et al.*, 2011b); however, middle burn depths did not differ from published literature values (Shetler *et al.*, 2008). This severe burning of peatland margins was likely caused by two primary factors: (1) the presence of high-density peat and (2) the higher susceptibility of margins to drying. While numerous studies have documented that decreases in moisture content coincide with increases in burn depths in organic soils (Miyanishi and Johnson, 2002; Rein *et al.*, 2008), bulk density is equally important in determining smouldering potential in boreal peat (Benscoter *et al.*, 2011).

Although increases in bulk density result in higher moisture retention on a volumetric basis (Boelter, 1968; Thompson and Waddington, 2013), the presence of dense peat (>100 kg $m^{-3}$) signifies an increase in the amount of peat present to generate energy in the smouldering process and an overall increase in its vulnerability to combustion (Benscoter *et al.*, 2011). This is reflected in our theoretical modelling showing that dense peat profiles corresponded to higher $H_{\text{comb}}/H_{ign}$ ratios. Furthermore, this is evident the observed differences in *m* and $H_{\text{comb}}/H_{ign}$ ratios between middles and margins under the same drying scenario, where high *m* values and $H_{\text{comb}}/H_{ign}$ ratios <1 were ubiquitous in middle profiles. Previous work has shown that boreal peat can smolder at *m* values up to 295% (Benscoter *et al.*, 2011) and, using a more conservative threshold of <250% in our 'worst-case' drying scenario, 10% and 63% of middle and margin locations fall below this value, respectively. Furthermore, the number of peatland middle samples with *m* <250% are heavily concentrated in the upper 0.10 m, with only one sample located below this depth, suggesting that middles are unlikely to undergo smouldering.

Although moisture conditions may appear to have a smaller effect on smouldering potential than the presence of a dense peat profile (Figure 2.6), actual moisture conditions at peatland margins immediately prior to fire were very likely drier than our conservative model Ψ profiles. Given the larger WT fluctuations at the margins, the disparity in moisture conditions between peatland middles and margins would be more pronounced during drought when the margin WT is lowest (Devito *et al.*, 2012). Dynamic WT fluctuations not only make margins susceptible to wildfire but they also facilitate the development of dense peat prior to fire through subsidence (Price, 2003) and decomposition (Blodau *et al.*, 2004) in the peat profile. Therefore, a positive feedback exists whereby increases in peat bulk density decrease the specific yield of the peat profile, enhancing WT fluctuation at margins and leading to further decreases (increases) in $m$ ($H_{\mathrm{comb}}/H_{ign}$ ratios) during dry periods. This highlights that dynamic hydrological conditions are essential in increasing peat smouldering potential.

*2.5.2 Implications of deep burning 'hotspots' for peatland hydrology and carbon storage*

The presence of dense, humified peat at peatland margins has been observed in the BP (Ferone and Devito, 2004) and in humid climates as well (Baird *et al.,* 2008; Lapen et al., 2005). Because saturated hydraulic conductivity ($K_{sat}$) generally decreases with increasing peat bulk density (Boelter, 1968), dense margin peat can act as a water conservation mechanism in peatlands (Baird et al., 2008; Lapen et al., 2005). By limiting lateral groundwater losses, low saturated hydraulic conductivities in margin peat may enable the peatland to grow to a greater height above the mineral surface than otherwise possible (Lapen and Baird, 2008). Therefore, if deep burning combusts margin peat, it has the potential to increase lateral groundwater losses and even lower the WT in the middle of the peatland. Given that the WT resided above the margin surface at the burned site for the duration of the study period, this highlights the potential for post-fire increases in lateral groundwater losses, which could impact post-fire vegetation recovery in the peatland. However, the significance of this water conservation mechanism must be assessed with respect to the permeability of adjacent mineral upland soils. For example, a peatland in a coarse-grained hydrogeological setting (*i.e.* higher $K_{sat}$ in adjacent mineral uplands) would likely rely much more on this low $K_{sat}$ in margin peats water conservation mechanism than a peatland situated in a fine-textured hydrogeological setting (*i.e.* low $K_{sat}$ in adjacent mineral uplands). Future research should investigate how deep burning of low $K_{sat}$ peats affects peatland hydraulics.

Deep burning 'hotspots' also have the potential to be a significant portion of the total carbon released from peatlands during wildfire. For example, by spatially weighting our depth of burn measurements and pairing them with bulk density data from the unburned site (see section 3.3) and assuming a carbon content of 51.7% (Gorham, 1991), we estimated that carbon loss from the middle and margins of the burned site ranged from 0-4 kg C $m^{-2}$ (mean = 1 kg C $m^{-2}$) and 10-85 kg C $m^{-2}$ (mean = 27 kg C $m^{-2}$), respectively. Accounting for the spatial distribution of margin and middle DOB, the site averaged carbon loss is 19.9 ± 2.0 kg C $m^{-2}$. Margin carbon release (based on mean C release for middle and margin) accounted for ~90% of the estimated carbon release despite representing only ~30% of the total peatland area. If these smouldering 'hotspots' observed at peatland margins are prevalent on the landscape, it suggests that previous studies (Turetsky *et al.,* 2011b; de Groot *et al.*, 2009) may have underestimated the amount of carbon released from peatlands in the BP. This would be especially true in landscapes where peatland water balances undergo higher mineral upland influence (Thompson, 2012). Given that the fire return interval for peatlands in this region is estimated at 120 years (Turetsky *et al.*, 2004), legacy carbon lost in these 'hotspots' may not be balanced by contemporary between-fire carbon sequestration (Wieder *et al.*, 2009). This suggests that peatland margins, under current climatic conditions, may be dynamic, laterally shifting (inwards) entities that can be impacted by high severity fires. Moreover, because climate change scenarios suggest that increases in evapotranspiration are likely to exceed increases in precipitation in northern latitudes (Collins *et al*., 2013), organic layer burn severity may increase thereby exposing denser peat at the surface. This would lead to enhanced smouldering potential at peatland margins and, coupled with expected increases in total wildfire area burned of 25–100% (Flannigan *et al.*, 2005), may exceed peatland resilience to wildfire and shift these ecosystems to net sources of carbon to the atmosphere (Turetsky *et al.*, 2002).

### *2.5.3 Identifying deep burning 'hotspots' on the landscape*

Although this study was performed at only one site, the hydrological processes controlling deep burning described herein are very likely applicable at the landscape-scale. However, we encourage future studies to examine the representativeness of our current study, given the size of the peatland and its climatic and hydrogeological setting. Nevertheless, our results suggest that the coupling of dense peat (bulk density > 100 kg $m^{-3}$) and low peat moisture ($m$ < 250 %) allow for smouldering to propagate deep into the peat profile. These controls also likely provide an explanation of the occurrence of smouldering 'hotspots' in drained and mined peatlands, both of which possess dense peat profiles susceptible to drying (Sherwood *et al.*, 2013; Turetsky *et al.*, 2011a). Therefore, it is very likely that peatland restoration, which facilitates the re-establishment of low-density moss that maintains high gravimetric water content during dry conditions

(McCarter and Price, 2013), chould be used as a strategy to reduce wildfire risk and the associated carbon losses and air pollution within managed peatlands.

In natural peatlands not affected by land-use change, peatland hydrogeological setting likely controls the distribution of peat $H_{\mathrm{comb}}/H_{ign}$ ratios on the landscape because peat bulk density and moisture are controlled by the frequency of dry conditions (low WT). For example, peatlands permanently or ephemerally disconnected from groundwater sources will likely be more vulnerable than peatlands located in continuous groundwater discharge zones because they would have both drier and denser peat. Furthermore, climate change and peatland afforestation pose substantial risks to peatland carbon stocks, as drying may exceed negative feedbacks (*e.g.* Waddington et al., 2014) that conserve water during drought (Kettridge and Waddington, 2014) and subsequently decrease peatland resilience to wildfire through a shift to hydrophysical properties that are vulnerable to smouldering. Consequently, we suggest that future research should aim to better understand landscape-scale controls on the spatial and temporal variability in peat moisture content and bulk density as a means to identify other potential deep burning 'hotspots' on the landscape.

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**Table 2.1.** Peat hydrophysical properties, unburned site[a]

| | Middle | | | | Margin | | | |
|---|---|---|---|---|---|---|---|---|
| Depth (m) | $\rho_b$ (kg m$^{-3}$) | $\alpha$ (cm$^{-1}$) | *N* | $m\%_{100cm}$ | $\rho_b$ (kg m$^{-3}$) | $\alpha$ (cm$^{-1}$) | *n* | $m\%_{100cm}$ |
| 0.00-0.05 | 36.5 (8.7) | 6.02 (1.68) | 1.39 (0.03) | 549 (107) | 80.2 (9.1) | 7.99 (1.16) | 1.27 (0.03) | 260 (12) |
| 0.05-0.10 | 40.8 (6.4) | 6.44 (1.50) | 1.34 (0.02) | 492 (87) | 91.5 (9.1) | 8.34 (1.32) | 1.23 (0.02) | 226 (18) |
| 0.10-0.15 | 44.2 (6.2) | 3.37 (1.27) | 1.35 (0.02) | 584 (56) | 100.0 (8.9) | 5.4 (1.81) | 1.24 (0.02) | 281 (38) |
| 0.15-0.20 | 52.9 (9.1) | 4.90 (1.58) | 1.32 (0.08) | 693 (183) | 118.5 (15.1) | 4.72 (1.45) | 1.19 (0.01) | 306 (50) |
| 0.20-0.25 | 69.4 (14.5) | 4.16 (1.58) | 1.25 (0.03) | 733 (202) | 150.9 (20.6) | 1.52 (0.97) | 1.22 (0.02) | 307 (48) |
| 0.25-0.30 | 100.1 (18.7) | 2.11 (1.45) | 1.36 (0.08) | 756 (90) | 170.8 (20.3) | 1.99 (1.47) | 1.22 (0.02) | 319 (58) |
| 0.30-0.35 | 114.6 (19.2) | 2.72 (1.35) | 1.21 (0.02) | 654 (74) | 191.8 (22.1) | 0.54 (0.31) | 1.21 (0.02) | 347 (76) |
| 0.35-0.40 | 136.8 (21.1) | 1.07 (0.38) | 1.22 (0.01) | 519 (65) | 225.4 (19.1) | 0.15 (0.07) | 1.21 (0.02) | 257 (37) |
| 0.40-0.45 | 182.3 (25.4) | 0.17 (0.07) | 1.19 (0.01) | 625 (100) | 233.7 (14.9) | 0.02 (0.01) | 1.26 (0.03) | 290 (28) |

[a]Mean bulk density ($\rho_b$), van Genuchten parameters $\alpha$ and $n$, and gravimetric water content ($m$) at 100 cm of $\Psi$ measured at 0.05 m increments at peatland middle and margin (standard errors in parentheses).

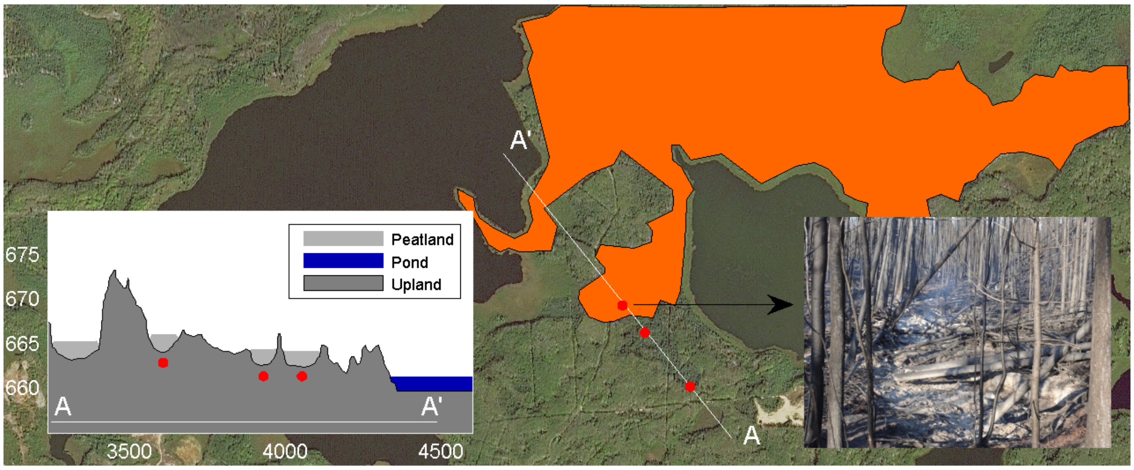


**Figure 2.1.** Aerial view of the study site with orange representing the areas of complete and partial forest canopy burning during the wildfire. The red points denote both the burned and unburned sampling locations along transect A-A' within the peatland complex. The inset cross-section highlights the similarity in size and topographical position of the sites within the coarse textured glaciofluvial outwash. The inset photo shows the burned peatland margin in Figure 2.1, where depth of burn exceeded one meter, approximately three weeks after the fire. Note smoke from smouldering.

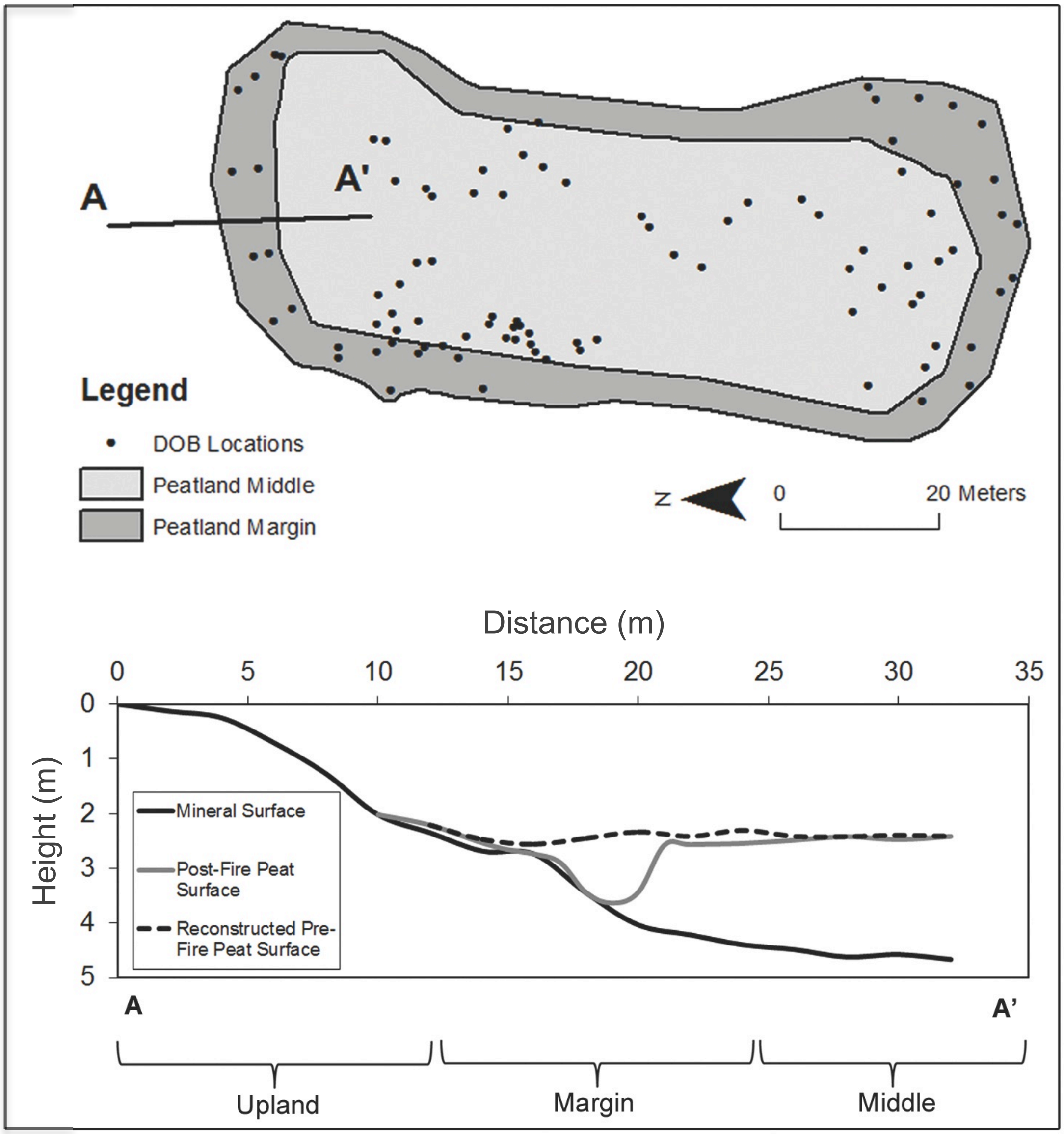


**Figure 2.2.** Burned study site with depth of burn sampling locations shown as points in middle and margin zones, where each point represents ten measurements. Transect A-A' is the most severely burned margin and is shown in cross-sectional view with the mineral, reconstructed, and post-fire peat surfaces denoted.

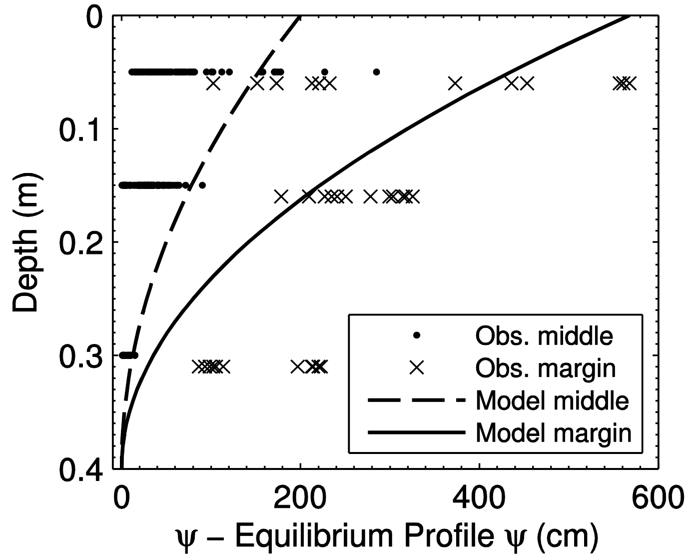


**Figure 2.3.** Measured and model Ψ profiles from the drying scenarios that were used to determine modelled gravimetric water content values. Note that the x-axis is the difference between Ψ at a given depth and an equilibrium Ψ profile (*i.e.* disequilibrium). Measured Ψ values from the middle (points) and margins (x's) are offset by 0.01 m for viewing purposes.

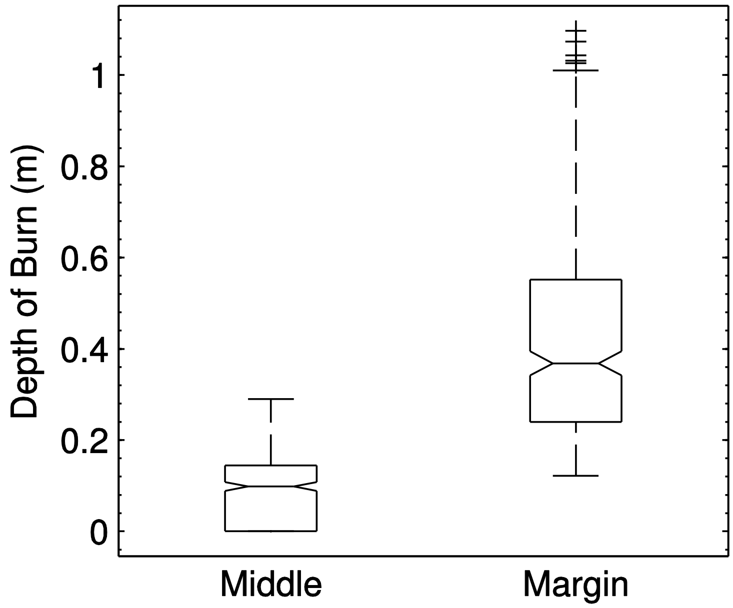

**Figure 2.4.** Depth of burn measurements classified into middle and margin locations. No overlaps between boxplot notches indicate significant differences at a 95% confidence interval.

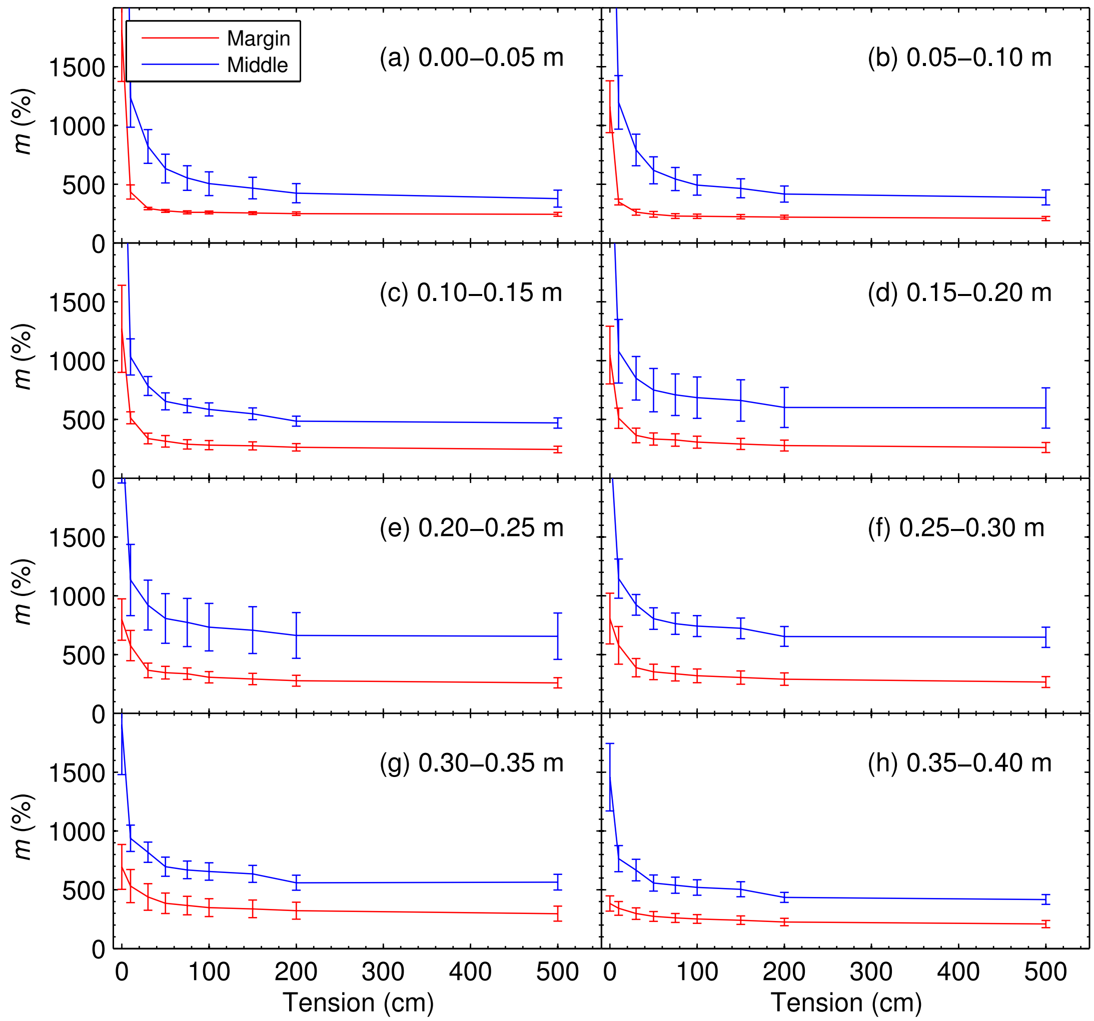


**Figure 2.5.** Mean moisture retention curves on a gravimetric scale for middle (blue) and margins (red) at depths of a) 0.00-0.05 m, b) 0.05-0.10 m, c) 0.10-0.15 m, d) 0.15-0.20 m, e) 0.20-0.25 m, f) 0.25-0.30 m, g) 0.30-0.35 m, h) 0.35-0.40 m for unburned peat cores. Error bars are standard errors of the mean.

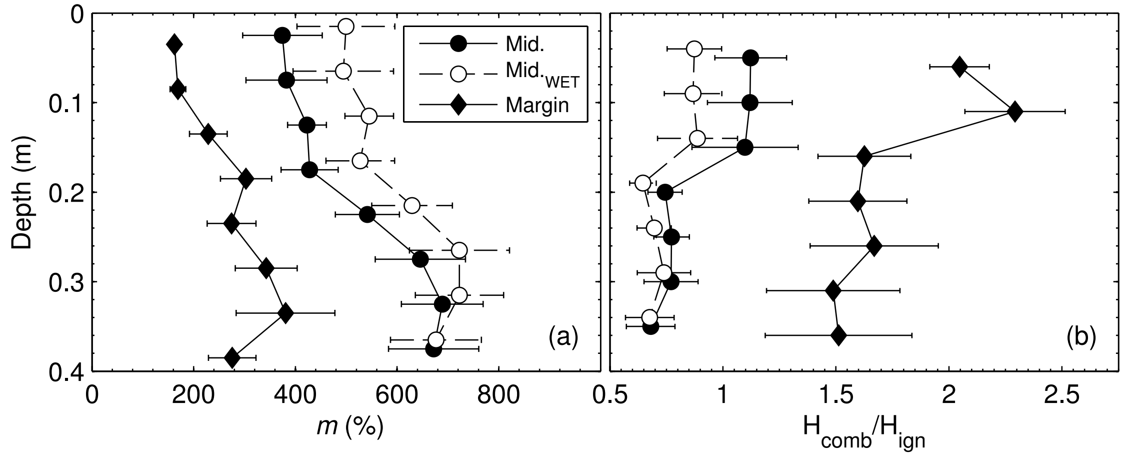


**Figure 2.6.** Depth profiles of modelled gravimetric water content (a) and $H_{\mathrm{comb}}/H_{ign}$ ratios (b) of the peatland margin and middle under two drying scenarios. Error bars are standard errors of the mean, the legend applies to both panels, and profiles are offset by 0.01 m.

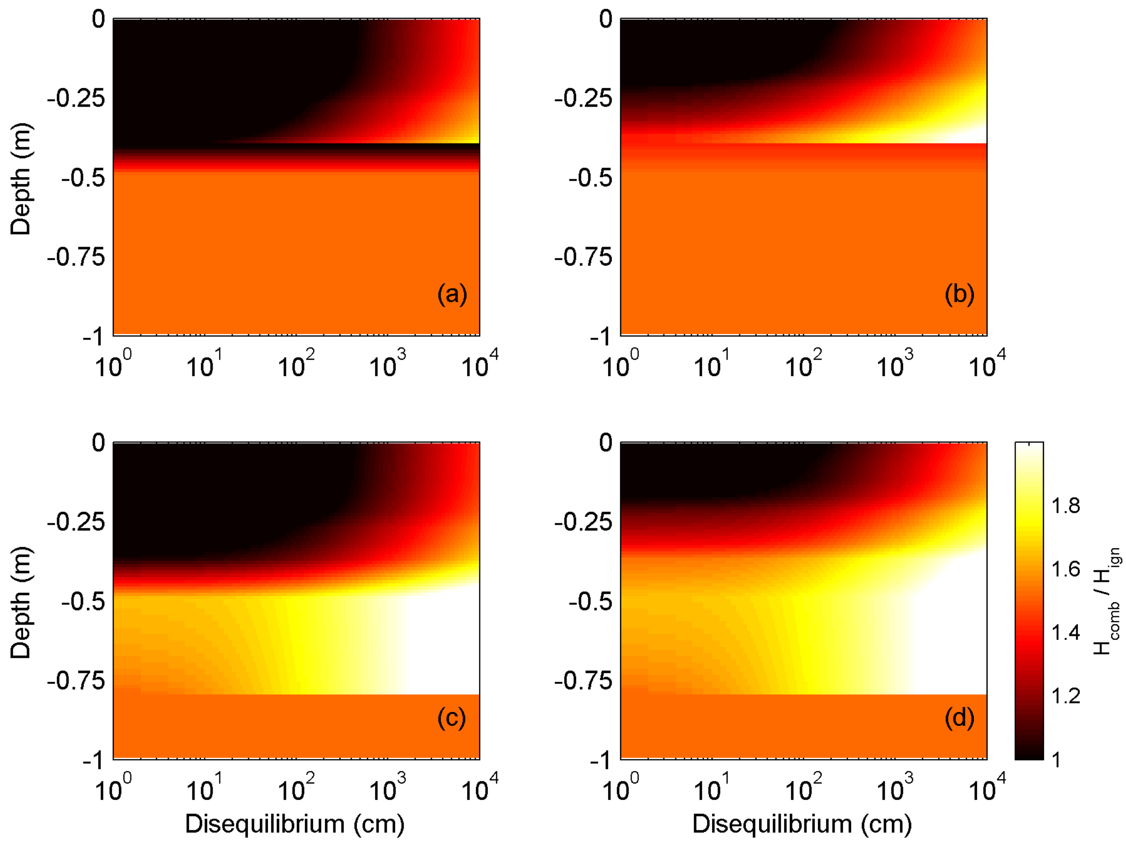


**Figure 2.7.** $H_{\text{comb}}/H_{ign}$ ratio sensitivity to disequilibrium pore water tension ($\Psi$) for four different scenarios: a) low bulk density profile, 0.40 m WT, b) high bulk density profile, 0.40 WT, c) low bulk density profile, 0.80 m WT, d) high bulk density profile, 0.80 m WT. High and low bulk density scenarios are modelled as the average bulk density profiles for middle (low bulk density) and margin (high bulk density) sampling locations listed in Table 1. Because bulk density was not measured below 0.50 m, we conservatively assumed that bulk density at depths > 0.50 m remained the same as in the 0.45-0.50 m layer. Water content is estimated according to Equation 1, assuming the peat profile consists of *Sphagnum* peat. Herein, the $H_{\text{comb}}/H_{ign}$ ratio values have been limited to values equal to or greater than one.

## CHAPTER 3: BURN SEVERITY ALTERS PEATLAND MOSS WATER AVAILABILITY: IMPLICATIONS FOR POST-FIRE RECOVERY

### 3.1 Abstract

Wildfire is the largest disturbance affecting northern peatlands, however, little is known about how burn severity (organic soil depth of burn) alters post-fire hydrological conditions that control the recovery of keystone peatland mosses (*i.e., Sphagnum*). For this reason, we assessed the impact of burn severity on moss water availability by measuring soil tension ($\Psi$) and surface volumetric moisture content ($\theta$) in burned and unburned portions of a peatland complex two years after fire. We found that both high and low burn severity decreased post-fire water availability by altering peat hydrophysical properties (moisture retention, water repellency). Locations covered by *S. fuscum* prior to fire exhibited decreasing post-fire water availability with increasing burn severity. In contrast, the lowest water availability ($\Psi$ >400 cm, $\theta$ <0.02) was observed in feather mosses that underwent low burn severity (residual branches identifiable). Deep burning (>0.20 m) in peatland margins and burn depths >0.05 m in the middle of the peatland exhibited the highest water availability ($\Psi$ <60 cm). Locations with low surface $\theta$ and high $\Psi$, notably feather mosses undergoing low burn severity, exhibited minimal moss recolonization. Such areas dominate post-fire surface cover (~40%) within late successional (mature) peatlands or peatlands located in dry hydrological settings. We argue that such environments are underrepresented in conceptual models of post-fire recovery. A new conceptual model is proposed in which: 1) deep burning is counterbalanced by rapid recolonization and 2) pre-fire species interact with burn severity to produce substantial lags in post-fire moss recovery.

### 3.2 Introduction

Wildfire is the largest disturbance affecting northern peatlands (Turetsky *et al.*, 2002), occurring as frequently as every 100-120 years within boreal regions (Turetsky *et al.*, 2004). While peatland wildfires typically result in complete stand mortality and the die-off of ground layer vegetation (Benscoter and Vitt, 2008; Zoltai *et al.*, 1998), northern peatlands are generally resilient to wildfire and return to a net carbon sink status within ~20 years post-fire (Wieder *et al.*, 2009). However, given that climate change scenarios suggest that increases in evapotranspiration are likely to exceed increases in precipitation in northern latitudes (Collins *et al*., 2013), there is concern that peatlands will experience substantial drying (Roulet *et al.*, 1992), thereby increasing their vulnerability to wildfire (Kettridge *et al.*, 2015; Sherwood *et al.*, 2013; Turetsky *et al.*, 2011b). Increases in organic layer burn severity, coupled with expected increases in total wildfire area burned of 25–100% (Flannigan *et al.*, 2005), may exceed peatland resilience to wildfire and shift these ecosystems to net sources of carbon to the atmosphere (Turetsky *et al.*, 2002).

Whilst climate plays the largest role in dictating the stability of peatland ecosystems (Gorham *et al.*, 2012), peatlands possess numerous internal ecohydrological feedbacks that enable them to self-regulate water losses (Waddington *et al*., 2015). Consequently, the potential impact of future drying and shifting wildfire regimes must be weighed against the resilience of peat-forming vegetation, specifically, *Sphagnum* mosses (Turetsky *et al.*, 2010; Waddington *et al*., 2015). These bryophytes are important ecosystem engineers that increase peatland resilience by lowering decomposition rates

(Rydin, 2013), conserving water during drought (Kettridge and Waddington, 2014), and limiting combustion during wildfire (Shetler *et al.*, 2008). Because little is known about how burn severity alters the hydrological processes controlling the recolonization of these mosses, there is an immediate need to understand how burn severity affects post-fire hydrological conditions within peatlands.

Burn severity in peatland ecosystems, defined by the depth of peat combustion (*i.e.,* depth of burn, DOB), typically ranges from 0.05-0.10 m (Benscoter and Wieder, 2003; Lukenbach *et al.*, 2015; Shetler *et al.*, 2008) and is influenced by the water retention properties of ground layer vegetation and near-surface peat (Lukenbach *et al.*, 2015; Thompson and Waddington, 2013a). Where ground layer vegetation is prone to drying (*e.g.,* feather mosses), burning is usually more severe (Benscoter *et al.*, 2011). In contrast, moss species able to efficiently retain water during dry periods, such as *Sphagnum fuscum* (Thompson and Waddington, 2013a), typically limit combustion to <0.03 m (Shetler *et al.*, 2008). Although DOB is influenced by these moss traits (Turetsky *et al.*, 2010), burn severity can exhibit substantial variability within a particular type of ground layer vegetation. For example, feather mosses may undergo little or no combustion (DOB <0.03 m) or may undergo appreciable burning (DOB >0.05 m) (Benscoter *et al.*, 2011; Kettridge *et al.*, 2014), which is likely due to small-scale variation in fuel loading or surface moisture at the time of wildfire (Miyanishi and Johnson, 2002; Thompson *et al.*, 2014). Because mosses occupy environmental niches along microtopographic gradients (Andrus *et al.*, 1983), patterns of burn severity are also apparent at the microtopographic scale (*i.e.,* hummocks and hollows) (Benscoter *et al.*, 2005; Benscoter and Wieder, 2003), creating a range of post-fire environmental conditions that influence the recolonization of peatland mosses (Benscoter *et al.*, 2005; Benscoter and Vitt, 2008).

Post-fire vegetation recovery in peatlands is often characterized by the recolonization of *Sphagnum* mosses, particularly hummock-forming species, because of their role as ecosystem engineers whereby they maintain ecosystem structure and function (*e.g.* Benscoter and Vitt, 2008; Rydin, 2013). However, the underlying controls on the post-fire recovery of *Sphagnum* mosses remain unknown. Similarly, little is known about the physical processes controlling the establishment of pioneer moss species, such as *Polytricum strictum*, which may facilitate the recovery of *Sphagnum* mosses (Benscoter, 2006). Because *Sphagnum* mosses are characteristic of low nutrient status (Rydin, 2013), water availability likely plays the prevailing role in determining their recolonization after wildfire (Waddington *et al.*, 2010). As such, post-fire hydrological processes that affect moss moisture stress are likely to account for spatial and temporal variability in recovery patterns (Thompson and Waddington, 2013b).

Water stress in *Sphagnum* mosses has been correlated with depth to water table (WT) and volumetric moisture content ($\theta$) (McNeil and Waddington, 2003; Tuittila *et al.*, 2004); however, tension ($\Psi$) (i.e. pore water pressure, suction) is a more accurate indicator of stress because it links large-scale hydrological processes to cellular-scale water requirements in mosses (Thompson and Waddington, 2008). The influence of burn severity on these hydrological variables has yet to be documented, although Thompson and Waddington (2013b) found that $\Psi$ increased in the near-surface in an ombrotrophic bog after wildfire. Moreover, advances in the understanding of post-fire hydrological processes have not been consistent with current conceptual models of how burn severity affects *Sphagnum* recovery. While high burn severity has been suggested to limit moss recovery in hollows (Benscoter and Vitt, 2008), feather moss-covered hollows that have undergone low burn severity (DOB <0.03 m, residual branches identifiable) can develop near-surface water repellency (Kettridge *et al*., 2014). This implies that low burn severity can also potentially limit water availability at the surface and impede recovery in hollows. Therefore, we suggest that the sole use of microtopography to characterize post-fire recolonization patterns may not be appropriate due to the interaction between pre-fire species cover and burn severity. Moreover, because hummock-hollow microtopography is usually only present in the middle of the peatland (Dimitrov *et al*., 2014), conceptual models of post-fire recovery employing microtopographic frameworks are likely not applicable at peatland margins. This is important given that high burn severity and deep burning (DOB >0.20 m), common to peatland margins (Hokanson *et al*., 2015; Lukenbach *et al.*, 2015), have been linked to post-fire regime shifts in peatlands (Kettridge *et al.*, 2015; Turetsky *et al.*, 2011a). Therefore, we argue there is an immediate need to investigate the impact of deep burning on post-fire moss water availability, given its role in determining peatland resilience to wildfire (Kettridge *et al.*, 2015).

This study examines how peatland burn severity alters post-fire hydrological conditions. First, the impact of deep burning on post-fire water availability is examined. Based on previous observations of post-fire moss recovery in peatlands that underwent deep burning (see Kettridge *et al.,* 2015), we hypothesized that deep burning reduces water availability in near-surface peat. Second, the control of pre-fire species cover and burn severity on post-fire water availability is examined. We hypothesized that feather mosses undergoing low burn severity (<0.03 m DOB) would limit post-fire water availability and bryophyte recolonization. Furthermore, because feather mosses can undergo a range of burn severities in hollows, we also hypothesized that microtopography (*i.e.* hummocks and hollows) is a poor indicator of post-fire water availability.

## 3.3 Methods

### *3.3.1 Study site and experimental design*

In May of 2011, a ~90,000 ha wildfire burned a portion of a small peatland complex (~5 ha) located between two esker ridges on a coarse-textured outwash plain in the Utikuma Lake Research Study Area (URSA; 56.107°N 115.561°W) of the Boreal Plains (BP) ecozone (Devito *et al*., 2012). The burned and unburned portions (hereafter referred to as sites) within the peatland complex (located ~100 m apart) are approximately 100 x 150 m and 90 x 150 m in size, respectively, and are both surrounded by aspen (*Populus Tremuloides*) forest with some jack pine (*Pinus banksiana*) with a stand age of ~135 years. During the study, the unburned portion of the peatland complex served as a reference for the burned site because no measurements of moss water availability were collected prior to fire at the either site. The sites were characterized by feather moss (>95% *Pleurozium schreberi*) hollows, *S. fuscum* hummocks, vascular vegetation cover of *Rhododendron groenlandicum* and *Rubus chamaemorus*, and a dense black spruce (*Picea mariana*) tree canopy with a stem density of approximately 7,000 and 4,000 stems $ha^{-1}$ at the burned and unburned sites, respectively. For more details of the local hydrology see Smerdon *et al.* (2005).

To examine the impact of deep burning on post-fire water availability, we examined temporal and spatial trends in Ψ, surface θ, WT position, and moss recovery between the middle (low DOB) and margins (high DOB) of a burned peatland. Given that Thompson and Waddington (2013b) observed differences in water availability between hummock and hollow microforms, we sampled five hummocks and five hollows to capture the range of water availability in the middle of the peatland. At the burned margins, deep burning was prevalent and low burn severity accounted for <5% of the surface cover, thus all sampling occurred in deeply burned areas (DOB >0.20 m).

To examine the spatial control of pre-fire species cover and burn severity on post-fire water availability in the middle of the peatland, we conducted an intensive field survey (IFS) that examined temporal and spatial trends in Ψ, surface θ, WT position, and moss recovery across a microtopographic (hummocks vs. hollows) and burn severity gradient in the middle of the peatland complex. Following the findings of Benscoter and Vitt (2008), who found that pre-fire species cover might exert a major significant control on post-fire water availability, we classified the burned site into burn severity and species groups.

*3.3.2 Hydrological measurements at the middle and margins of the peatland*

The middle and margins of the burned peatland complex were differentiated from one another because there were prominent gradients of residual peat depth, depth to WT, vegetation cover, and DOB (see Figure 3.1) between these units. The middles of the unburned and burned portions of the peatland complex were visually classified into traditional hummocks and hollows based on differences in the elevation of the peat surface. At both sites, hummock-hollow microtopography was not present at the peatland margins, although variation in DOB resulted in variability in the elevation of the peat surface at the burned margins.

Ψ was measured two to three times per week from May 2013–September 2013 at a depth of 0.05 m in five hummocks and five hollows at the burned and unburned sites using 0.02 m outside diameter tensiometers (Soil Measurement Systems, Tucson, Arizona, USA) and a UMS (Munich, Germany) Infield tensimeter, accurate to ± 2 cm. The use of tensiometers to measure Ψ in peatland mosses has been successfully employed by numerous studies (*c.f.* Ketcheson and Price, 2014; Kettridge and Waddington, 2014; McCarter and Price, 2013; Thompson and Waddington, 2013a). Because DOB was highly variable at the burned margins (see section 3.1), ten tensiometers were installed at this same depth in burned margins, while only three tensiometers were installed in unburned margins to serve as a reference for what water availability may have been like prior to deep burning. Of note, the ten tensiometers installed in the burn margins represented a drier portion of the margin because a large proportion of the burned margins were flooded. In May of 2013, an animal destroyed four tensiometers at the unburned site (two hummocks and two hollows) and one tensiometer at the burned site (one margin), resulting in the measurement of only three hummock and hollow tensiometers at the unburned site and nine burned margin tensiometers for the rest of the study period.

At both the burned and unburned site, WT position was continuously recorded by capacitance water level recorders (Odyssey Data Recording, Christchurch, New Zealand) or pressure transducers (Solinst, Georgetown, Ontario, Canada) at 20 min intervals from May 2013–September 2013 in 0.05 m diameter PVC wells at middle and margin locations. The response time of these wells to rainfall was almost instantaneous, as is common in peatlands that contain high porosity soils (Heliotis and Dewitt, 1987). Depth to WT at locations where tensiometers were installed was determined by measuring the water level in wells adjacent to tensiometers.

### *3.3.3 Pre-fire species cover and burn severity control on post-fire water availability*

*Burn severity classification*: The middles of the burned and unburned sites were sub-divided into burn severity and vegetation groups during the IFS. Hummocks were dominated by *S. fuscum* (>95% cover) and burn severity in these locations was defined by whether or not capitula were intact after wildfire (lightly burned *S. fuscum*, LB-Sf vs. severely burned *S. fuscum*, SB-Sf). Feather moss (predominantly *P. schreberi*) was the primary surface cover in hollows and burn severity in these locations was assessed by measuring DOB, where the difference in surface elevation between burned areas and surrounding unconsumed areas was measured in a manner similar to other researchers (Kasischke *et al.*, 2008; Mack *et al.*, 2011). Lightly burned feather moss locations (LB-F) were areas where DOB <0.03 m and residual, singed feather moss was visible. Hollows where DOB >0.05 m and pre-fire moss cover was not identifiable were classified as severely burned feather mosses (SB-F), as feather mosses were likely the surface cover in these locations based on species cover observations from the unburned site and because feather mosses are much more prone DOB >0.05 m than *S. fuscum* (Benscoter *et al.*, 2011). Severely burned unidentifiable margins (SB-Ma) were also characterized by the same criteria as SB-F; however, these locations had residual peat depths of <0.15 m compared to SB-F that had residual peat depths >1.00 m. The burn severity groups and their definitions are given in Table 3.1.

The coverage of the burn severity groups and microforms in the middle of the peatland were determined using the line interception method (Floyd and Anderson, 1987). Surface cover and microtopography was identified every 0.25 m ($n$=400) along two perpendicular 50 m long transects. The peatland complex was also discretized into margin, middle hummock or middle hollow units based on the classification of 2.5 cm resolution multiband 8-bit RGB aerial imagery obtained from an unmanned aerial vehicle flown over the sites at a height of 100 m. Radiometric enhancement was used to create greater contrast between hummock and hollow microforms. There was good agreement between the ground surveys and air photo interpretation in the middle of the peatland. The surface cover distributions of microtopography and burn severity groups are given in Table 3.2.

*Hydrological measurements:* Hydrological measurements were conducted on September 8$^{th}$, 2013 when water stress was anticipated to be highest in late summer, typical of the sub-humid and continental climate (Devito *et al.*, 2012). A total of 225 tensiometers were installed during the IFS in burned and unburned portions peatland complex at a depth of 0.05 m. Each tensiometer was measured five days after installation. 50 tensiometers were installed within *S. fuscum* groups ($n$=50 in each of UB-Sf, LB-Sf, and SB-Sf) to accurately capture the influence of WT position on Ψ. A total of 25 tensiometers were installed in each additional burn severity group (Table 3.1). Tensiometers were not installed in unburned margins (UB-Ma) during the IFS because feather mosses were the dominant species cover in these areas and tensiometer availability was limited.

Surface θ (both the top 0.03 m and 0.06 m) was measured at each tensiometer installation using a Thetaprobe (Delta-T Devices, Burwell, Cambridge, UK). Both 0.03 and 0.06 m measurements were calibrated following the approach of Kasischke *et al.* (2009). Surface θ at a depth of 0.03 m was measured directly above each tensiometer cup, while 0.06 m surface θ was taken directly adjacent to each tensiometer cup. An additional measurement of 0.03 m and 0.06 m surface θ was taken <0.07 m away from each tensiometer.

Although the primary goal of the IFS was to examine post-fire water availability in the middle of the burned peatland, nine tensiometers were installed in SB-Ma in order to compare post-fire water availability in these areas to the middles of the sites during the IFS. The tensiometer installation also utilized 16 hummock and hollow tensiometers previously installed to collect temporal measurements of water availability in the burn severity groups, which yielded three UB-Sf, three UB-F, one LB-Sf, four LB-F, four SB-F, and one SB-F. These were reanalyzed and compared to SB-Ma and UB-Ma locations. In addition to these manual temporal Ψ measurements, logging tensiometers were installed in one LB-Sf, three LB-F, two SB-Sf, and one SB-F to provide high frequency measurements of Ψ.

*Moss and bryophyte recolonization:* Hydrological measurements during the IFS were paired with measurements of % bryophyte species cover and % *Sphagnum* cover, which were visually estimated in 0.10 x 0.10 m plots above each tensiometer cup on September $8^{th}$, 2013, to examine the association with bryophyte recovery. The fire resulted in complete mortality of ground layer vegetation; therefore, observations of % bryophyte species and *Sphagnum* cover are a measure of recolonization since fire.

### *3.3.4 Analyses*

Because residuals were not normally distributed between the middle and margin sampling locations as well as for the burn severity groups, Ψ was natural logarithm transformed prior to statistical analyses. Welch's (unequal variance) two sample t-tests were used to test for significant differences between middle and margin sampling locations. ANOVAs were used to examine differences in water availability during the IFS. To assess the control of WT on Ψ, robust ordinary least squares regression (OLS) was employed. Where relationships were found to be statistically significant, the amount of time that an equilibrium profile (Ψ = depth to WT) existed in the unsaturated zone was calculated, which corresponded to periods when water was sourced from the WT to meet evaporative demand. We defined equilibrium profile conditions to exist when the measured difference between the depth to the WT from the tensiometer porous cup and Ψ was less than 0.10 m (*c.f.,* Thompson and Waddington, 2013b). The impact of rainfall on Ψ was only examined where disequilibrium profile conditions existed, as rainfall would have an inconsequential impact on water availability in groups where equilibrium profile conditions were present. To investigate the impact of rainfall on Ψ, measurements of Ψ during the study period were separated into two groups: i) $<$24 hours since rainfall $\geq$1 mm and ii) $>$24 hours since rainfall $\geq$1 mm. Means and standard error of the means are reported unless otherwise stated.

## 3.4 Results

*3.4.1 Trends in water availability between the middle and margins of the peatland*

DOB was significantly higher at the burned margin (0.30 m ± 0.02) compared to the middle (0.07 m ± 0.02, $t_{21}$ = 9.92, $p$ < 0.001) of the burned site (Figure 3.1). Deep burning likely resulted in a shallower WT at the burned margin compared to the unburned margin during the study period (Table 3.2). The study period was preceded by higher than average precipitation from June 2011–April 2013 and one of the largest snow melts in the past 15 years in 2013, resulting in a cumulative moisture surplus relative to the 30-year precipitation mean (Devito, unpublished data). Following snowmelt, WTs at the burned site peaked with large rain events in early and late June and then declined throughout the remainder of the study period (Figure 3.2c and 3.2d). At the unburned site, the WT in the middle of the unburned site followed this same trend, while the WT at the unburned margin did not peak until late July (Figure 3.2c and 3.2d). Throughout the study period, the WT at the unburned margin was much deeper than at the burned margin (Figure 3.2d), while WTs in the middle of the burned and unburned sites were similar (Figure 3.2c); however, depth to WT varied between microform sampling locations in the middle of both the burned and unburned sites (Table 3.2).

During the study period, Ψ was significantly higher at the unburned margin compared to the burned margin ($t_{60}$ = 9.50, $p$< 0.001, Figure 3.2, Table 3.2). At the burned site, Ψ in the burned margins was significantly lower than in the middle of the burned peatland ($t_{339}$ = 23.73, $p$< 0.001), while the opposite was true for the unburned site ($t_{49}$ = 3.06, $p$< 0.01, Table 3.2). In the middle of the study sites, Ψ was highly variable, but was significantly higher in hummocks and hollows at the burned site compared to these same microforms at the unburned site ($t_{360}$ = 14.23, $p$< 0.001, Figure

3.2a, Table 3.2). Ψ was not significantly different between hummocks and hollows at the unburned site, while at the burned site Ψ not only varied between burned hummocks and hollows but also within these microforms (Figure 3.2a & Table 3.2). In particular, one burned hollow (later classified as SB-F) and one burned hummock (later classified as LB-Sf) exhibited significantly lower Ψ than the other tensiometers within their respective microform groups (see section 3.2, Table 3.2).

*3.4.2 Spatiotemporal trends in water availability along a burn severity gradient*

DOB was <0.03 m for all sampling groups except SB-F and SB-Ma locations (Figure 3.1). Average DOB in SB-Ma was 0.30 ± 0.02, while in SB-F it was 0.10 ± 0.01 m. A two-way ANOVA of microtopography and burn severity group showed a significant effect of microtopography ($F_{1, 144} = 12.23$, $p< 0.001$) and burn severity group ($F_{2, 144} = 111.6$, $p< 0.001$) on Ψ (SB-Ma not included in ANOVA). Moreover, the interaction between these two factors was highly significant ($F_{2, 144} = 158.78$, $p< 0.001$), indicating that the effect of microtopography was dependent on the burn severity group. Figure 3.3 shows the Ψ results of Tukey's HSD post-hoc test for the burn severity groups. Ψ increased slightly in LB-Sf and dramatically in LB-F and SB-Sf compared to the unburned references, but were at or below the reference in SB-F and SB-Ma (Welch's two sample t-tests were used to compare SB-Ma to other burn severity groups). Similarly, a two-way ANOVA of microtopography and burn severity group showed a significant effect of microtopography ($F_{1, 294} = 8.24$, $p< 0.01$) and burn severity group ($F_{2, 294} = 12.24$, $p< 0.001$) on surface θ (0.03 m). Furthermore, the interaction between these two factors was highly significant ($F_{2, 294} = 283.59$, $p< 0.001$). Figure 3.4 shows the surface θ results of Tukey's HSD post-hoc test for the burn severity groups. The only notable departure between Ψ and surface θ was in UB-F, where both Ψ and surface θ were low. Temporal trends in Ψ between the burn severity groups were similar to those observed during the IFS (Figure 3.5). A one-way ANOVA ($F_{7, 552}$ = 174.2, $p < 0.001$) showed that Ψ differed across burn severity groups and a Tukey's HSD post-hoc test showed significant differences between burn severity groups during the study period. These data are shown in Table 3.2.

*3.4.3 WT control on water availability*

During the IFS, there were significant WT-Ψ relationships in UB-Sf, LB-Sf, and SB-F ($p < 0.001$; $r^2$ = 0.80, 0.30, and 0.53 in UB-Sf, LB-Sf, and SB-F, respectively) (Figure 3.6). In *Sphagnum* groups (UB-Sf, LB-Sf, SB-Sf), increasing burn severity was linked to a decline in the correlation between WT and Ψ. In contrast, high burn severity in feather moss groups resulted in a strong relationship between WT and Ψ (*i.e.* SB-F). Data from SB-Ma were not statistically evaluated from the IFS due to low sample size; however, these data are shown alongside SB-F on Figure 3.6f. For locations where WT position explained some of the variation in Ψ, equilibrium profile conditions were present in 60%, 10%, and 96% of UB-Sf, LB-Sf, and SB-F, respectively (Figure 3.6). Comparatively, equilibrium profile conditions existed 90%, 60%, and 100% of the time during the rest of the study period in these same groups, respectively, indicating how much drier it was during the IFS. Although the SB-Ma WT-Ψ relationship was not evaluated during the IFS, these locations had strong WT-Ψ relationships during the rest of the study period ($p < 0.001$, $r^2$ = 0.61) and equilibrium profile conditions were present 100% of the time.

*3.4.4 Hydrological response to rainfall*

Rainfall had a moderating effect on differences in Ψ between burn severity groups. For groups that exhibited disequilibrium conditions during the study period (see section 3.3), Rainfall had a significant effect on Ψ in UB-Sf ($t_{46}$ = 6.67, $p$< 0.001), LB-Sf ($t_7$ = 4.92, $p$< 0.01), SB-Sf ($t_{36}$ = 5.38, $p$< 0.001), and UB-Ma ($t_{25}$ = 2.65, $p$< 0.05). During periods >24 hours since ≥1 mm, average Ψ was 57 ± 8 cm, 75 ± 14 cm, 189 ± 58 cm, 176 ± 31 cm in UB-Sf, LB-Sf, SB-Sf, and UB-Ma, respectively. While during periods <24 hours since rainfall ≥1 mm, average Ψ was 35 ± 3 cm, 51 ± 8 cm, 89 ± 39 cm, and 78 ± 27 in these same groups, respectively. Two rainfall events (≥1 mm) of 4 mm and 7 mm were captured by the logging tensiometers on August 20$^{th}$ and August 25$^{th}$, respectively (Figure 3.7). While Ψ declined in LB-Sf and SB-Sf, rainfall did not appear to have an effect on Ψ in LB-F, which conforms to the results observed in non-logging tensiometers placed in UB-F and LB-F (note that no logging tensiometers were installed in UB-Ma). Diurnal fluctuations in logging tensiometers were likely attributable to temperature fluctuations, as documented by previous studies (Redding and Devito, 2010; Warrick *et al.*, 1998). However, it is possible that Ψ at the 0.05 m depth lagged drying at the surface by several hours. As such, there is uncertainty regarding whether or not temperature fluctuations were the cause of diurnal fluctuations in Ψ. For this reason, temperature corrections were not conducted.

*3.4.5 Spatial trends in moss and bryophyte recolonization along a burn severity gradient*

In the unburned site, bryophyte cover was >95% and there was no *S. fuscum* present within the feather moss plots and vice versa. High post-fire water availability corresponded to high post-fire % bryophyte cover (Table 3.3). In LB-F, which displayed the lowest water availability, no bryophyte recolonization was observed. Within the other groups, bryophyte recolonization was observed and *Sphagnum* spp. was only observed in LB-Sf (Table 3.3).

## 3.5 Discussion

### *3.5.1 Post-fire ecohydrological controls on moss water availability*

Contrary to our first hypothesis, deep burning did not result in low post-fire water availability; in fact, burned margins (*i.e.* SB-Ma) exhibited the highest moss water availability throughout the study period and during the IFS. In the middle of the burned site, increasing burn severity was not consistently associated with low water availability; rather, water availability was also dependent on pre-fire species cover. Moreover, variability in $\Psi$ within microforms was as high as between microforms, highlighting the importance of pre-fire species cover at a given microtopographic position and the interaction between pre-fire species cover and burn severity.

Low burn severity in *S. fuscum* mildly decreased water availability when compared to its unburned state, while high burn severity led to much drier post-fire surface soil conditions. Because *S. fuscum* relies on high water retention and upward capillary flow from the WT to satisfy its water requirements (Hayward and Clymo, 1982), severe burning either: (1) lowered its ability to retain water by altering its pore structure, and/or (2) induced water repellency. Wildfire has been shown to lower moisture retention in *S. fuscum* (Thompson and Waddington, 2013a); however, the extent to which water repellency concurrently reduces moisture retention is unknown. Kettridge *et al.* (2014) showed that *S. fuscum* became slightly water repellent after wildfire and

Valat *et al.* (1991) demonstrated that *Sphagnum* mosses become highly water repellent after severe drying that may be analogous to drying occurring during wildfire. To what extent decreased moisture retention and increased water repellency interact to limit water availability in *S. fuscum* after wildfire remains unknown.

While low water availability was associated with high burn severity in *S. fuscum* locations, the opposite was true for feather mosses. Low burn severity resulted in small changes in surface $\theta$ in feather mosses, while $\Psi$ increased substantially when compared to its unburned state. Lightly burned feather mosses did not exhibit a disparity between surface $\theta$ and $\Psi$; rather, both measurements indicated that it was dry, suggesting that water may not reenter this water repellent matrix during rainfall (Kettridge *et al*., 2014) or that higher post-fire evaporative demand (*i.e.,* no shading) was able to increase $\Psi$ (Kettridge and Waddington, 2014). DOB >0.05 m in middle and margin locations likely occurred in areas covered by low moisture feather mosses (Benscoter *et al*., 2011); therefore, high burn severity in feather mosses is likely associated with increased post-fire water availability. Burn depths >0.05 m not only reduced depth to WT, especially in burned margins, but likely increased capillary flow by removing water repellent feather mosses (Kettridge *et al.,* 2014), and together likely explains the occurrence of high water availability in SB-F and SB-Ma. However, Thompson and Waddington (2013a) and Sherwood *et al.* (2013) suggest that the exposure of dense peat after DOB >0.05 m can limit water availability during steep WT declines, although this was not observed in our study even when the depth to WT was 0.50 m. Therefore, post-fire water availability in locations dominated by pre-fire feather moss cover is likely characterized by an inverse relationship to increasing burn severity, whereby severe burning increases water availability. The shape of this burn severity (DOB)-water availability relationship (i.e. threshold or linear) is unknown, although the outlier (DOB = 0.08 m) in SB-F (see Figure 3.6f) and the nature of feather moss peat would suggest a threshold-based response that is dependent upon burning being deep enough to remove the water repellent and low moisture feather moss peat matrix.

Although some unburned feather moss locations exhibited low surface $\theta$ and high $\Psi$, some $\Psi$ measurements departed from this anticipated scenario. The concurrent presence of low surface $\theta$ and low $\Psi$ in some unburned feather moss locations may be attributable to the tortuous and horizontally layered pore spaces within feather mosses, which have low moisture retention (Voortman *et al.*, 2013), coupled with one or several of the following: (1) low evaporative demand where unburned feather moss is present (*i.e.,* beneath trees) (Brown *et al.*, 2010; Kettridge *et al.*, 2013; Kettridge and Waddington, 2014); (2) very low unsaturated hydraulic conductivities due to the water repellent nature of feather mosses (Kettridge *et al.,* 2014), resulting in the slow drainage of small amounts of water downward and the stable presence of vertical hydraulic gradients from the near-surface to WT; (3) a dry surface moss layer resulting in an "evaporative cap", restricting water losses to only slow vapour fluxes and causing water to condense (saturation humidity) within the moss matrix (Goetz and Price, 2015). Although $\Psi$ was wide ranging in unburned feather moss, low surface $\theta$ indicates that water availability was low in these locations.

### *3.5.2 Patterns of post-fire moss recolonization*

Locations with high surface $\theta$ and low $\Psi$ exhibited substantial moss and other bryophyte recolonization, indicating that water availability is a primary control on recovery patterns observed in previous studies (Benscoter, 2006; Benscoter and Vitt, 2008). The establishment of early peatland succession colonizers (*e.g., P. strictum*) not only in SB-F locations but also in SB-Ma locations suggests that these locations possess favourable hydrological conditions. However, a large proportion of the post-fire peat surface exhibits low water availability. In particular, SB-Sf and LB-F locations had little or no post-fire bryophyte recolonization. Given that these locations accounted for ~50% of the total peatland area, recovery likely initiates in (not surprisingly) areas with high water availability. Furthermore, if post-fire hydrological processes limiting water availability are long lasting, we suggest that the recovery of locations with low water availability may originate through the lateral spread of mosses from areas of high water availability. This ecosystem response to severe burning, which is characterized by rapid bryophyte recolonization in areas with the greatest DOB (WT closest or above surface), may be vital in stabilizing peatland hydrology (Waddington *et al.,* 2015), and critical in enhancing resilience to wildfire. Furthermore, because high $\Psi$ likely corresponds to low evaporation (Kettridge and Waddington, 2014), water losses are limited in some areas of the peatland allowing for the allocation of water resources to areas with severe burning, which may be critical for jump-starting recovery. While this peatland-scale water redistribution mechanism is beneficial in middle locations, its role in the recovery of peatland margins is likely more complex. Nevertheless, because 27% of the total peatland complex area was SB-Ma, high nutrient availability and greater competition from mineral upland species may limit recovery and the long-term success of peatland mosses and other bryophytes even though post-fire water availability is high. Future studies should examine how post-fire nutrient availability and competition affect recovery along these peatland-upland interfaces after wildfire.

In areas with hydrological conditions that supported bryophyte recovery, such as LB-Sf, the frequency of low WTs likely plays a large role in how effectively these areas recolonize. The study period was preceded by wetter than normal conditions (see section 3.1), which resulted in shallow WTs in May 2013 and highlights that conditions were favourable for recolonization in the first two years post-fire. All locations that contained high water availability after wildfire had significant WT-$\Psi$ relationships, indicating that capillary flow from the WT was a key water source in areas conducive to recolonization. Because disequilibrium conditions and high $\Psi$ existed during extended periods without rainfall and when depth to WT was >0.6 m, the occurrence of drought following wildfire may be a particularly severe scenario because it would highly stress areas conducive to recolonization.

### *3.5.3 Implications for the trajectory of post-fire recovery*

Our study aligns with the findings of Benscoter and Vitt (2008) where high burn severity in *S. fuscum* hummocks was characterized by little or no bryophyte recolonization, and low burn severity resulted in the regeneration of *S. fuscum* from remaining capitula in the first three years following fire. However, although Benscoter and Vitt (2008) observed that feather mosses can comprise as much as 50% of pre-fire surface cover in peatlands, they did not provide a description of the trajectory of recovery in these locations. Rather, they describe the trajectory of recovery in locations dominated by *Sphagnum angustifolium* prior to fire, characterized by the rapid recolonization by pioneer mosses irrespective of burn severity. Our study site had minimal (<5% cover) amounts of *S. angustifolium,* and feather mosses were the primary surface cover due to the dry hydrological setting and late successional stage (~135 years since fire) of this peatland complex. Although feather mosses tend to be more vulnerable to combustion (Benscoter *et al.,* 2011), peatlands can contain large amounts of lightly burned feather mosses even after stand-replacing wildfires (Kettridge *et al.,* 2014). Furthermore, because *S. angustifolium* is indicative of shallow WTs and high moisture, this suggests that peatlands located in drier hydrological settings may be underrepresented in current conceptual models of post-fire recovery. The underrepresentation of peatlands situated in dry hydrological settings on the landscape is particularly important because these peatlands are likely to be the most vulnerable to climate change and wildfire. Therefore, we suggest a new conceptual model of post-fire bryophyte recovery based on our results and the findings of Benscoter and Vitt (2008), where water availability controls post-fire recolonization patterns (Figure 3.8). We suggest a substantial lag in recolonization is observed in areas of high burn severity in *Sphagnum* hummock species and in areas of low burn severity in feather mosses. However, if feather mosses undergo appreciable burning, there is the potential for rapid recolonization. Low burn severity in *S. fuscum* fosters capitula regeneration, while the degree of burn severity in *S. angustifolium* only slightly affects the rapid post-fire recolonization observed in these locations.

On the landscape, peatland WTs are influenced spatially by hydrogeological setting (*i.e.*, groundwater flow systems) and temporally by climate (Devito *et al.,* 2012), highlighting that peatland resilience to wildfire is also dependent on broader hydrological scales. This indicates that there may be substantial spatiotemporal variability in post-fire recovery between peatlands. Therefore, we suggest that future studies should assess peatland resilience to wildfire across hydrological scales, where local post-fire hydrological processes can limit capillary flow to the peat surface (this study) and groundwater flow systems affect pre and post-fire WT position.

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**Table 3.1.** Classification of microform subdivisions into burn severity groups, their definition, location, and (*n*) number of tensiometers deployed during the intensive field survey on September 8th, 2013 in the burned and unburned portions of the peatland complex.

| Acronym | Group | Definition and Description | Location | Microform | *n* |
|---|---|---|---|---|---|
| UB-Sf | Unburned *S. fuscum* | *S. fuscum* | Middle, Unburned peatland | Hummock | 50 |
| UB-F | Unburned feather moss | Feather moss | Middle, Unburned peatland | Hollow | 25 |
| UB-Ma | Unburned Margin | Transitional zone, feather moss and bare peat | Margin, Unburned peatland | N/A | N/A |
| LB-Sf | Lightly burned *S. fuscum* | Capitula intact | Middle, Burned peatland | Hummock | 50 |
| LB-F | Lightly burned feathermoss | DOB <0.03 m, residual feather moss visible | Middle, Burned peatland | Hollow | 25 |
| SB-Sf | Severely burned *S. fuscum* | Capitula not intact | Middle, Burned peatland | Hummock | 50 |
| SB-F | Severely burned feather moss | DOB >0.05 m, pre-fire moss cover un-identifiable | Middle, Burned peatland | Hollow | 25 |
| SB-Ma | Severely burned margin | DOB >0.05 m, pre-fire moss cover un-identifiable | Margin, Burned peatland | N/A | 9 |

**Table 3.2.** Summary of temporal measurements for the middle and margin of the burned and unburned portions of the peatland complex and these same measurements classified into burn severity groups (see Table 3.1). Average depth to WT (cm) for the study period, mean ± one standard deviation of soil tension (cm) for the study period, the percent of the burned (includes margin) and unburned (includes margin) portions of the peatland complex covered by each category and burn severity group, and the number of tensiometers deployed during the study period (May – September). Soil tensions with the same lowercase letter are not statistically different (least significant difference α=0.05)

| Category | Location | Depth to WT | Soil Tension | % Cover | *n* tensiometers |
|---|---|---|---|---|---|
| Unburned Hummock | Middle, Unburned peatland | 47 | $53 \pm 22^{a}$ | 31 | 3 |
| Unburned Hollow | Middle, Unburned peatland | 10 | $50 \pm 49^{a}$ | 45 | 3 |
| Unburned Margin | Margin, Unburned peatland | 81 | $143 \pm 171^{bf}$ | 24 | 3 |
| Burned Hummock | Middle, Burned peatland | 54 | $152 \pm 118^{bf}$ | 26 | 5 |
| Burned Hollow | Middle, Burned peatland | 33 | $346 \pm 246^{c}$ | 44 | 5 |
| Burned Margins | Margin, Burned peatland | 5 | $12 \pm 13^{d}$ | 30 | 9 |
| **Burn Severity Group** | | | | | |
| UB-Sf | Middle, Unburned peatland | 47 | $53 \pm 22^{a}$ | 26 | 3 |
| UB-F | Middle, Unburned peatland | 10 | $50 \pm 49^{a}$ | 42 | 3 |
| UB-Ma | Margin, Unburned peatland | 81 | $143 \pm 171^{bf}$ | 24 | 3 |
| LB-Sf | Middle, Burned peatland | 62 | $73 \pm 21^{ab}$ | 4 | 1 |
| LB-F | Middle, Burned peatland | 34 | $446 \pm 194^{e}$ | 40 | 4 |
| SB-Sf | Middle, Burned peatland | 52 | $172 \pm 124^{f}$ | 10 | 4 |
| SB-F | Middle, Unburned peatland | 25 | $28 \pm 12^{g}$ | 13 | 1 |
| SB-Ma | Margin Unburned peatland | 5 | $12 \pm 13^{d}$ | 27 | 9 |

**Table 3.3.** Average % species and % bryophyte cover (standard error of the mean in parentheses) within 0.1 x 0.1 m plots along a burn severity gradient collected at tensiometer installations in the peatland complex during the intensive field survey on September 8th, 2013 (*n=25* per group).

| | **Percent Cover** | | | | | |
|---|---|---|---|---|---|---|
| **Group** | *Ceratodon purpureus* | *Polytricum strictum* | *Marchantia polymorpha* | *Sphagnum fuscum* | *Pleurozium schreberi* | Total |
| UB-Sf | 0 | 0 | 0 | 98 (1) | 0 | 98 (1) |
| UB-F | 0 | 0 | 0 | 0 | 97 (2) | 97 (2) |
| LB-Sf | 8 (3) | 4 (2) | 0 (0) | 12 (3) | 0 | 24 (4) |
| LB-F | 0 | 0 | 0 | 0 | 0 | 0 |
| SB-Sf | 1 (1) | 1 (1) | 0 | 0 | 0 | 1 (1) |
| SB-F | 43 (5) | 2 (1) | 1 (1) | 0 | 0 | 46 (5) |
| SB-Ma | 33 (8) | 2 (1) | 27 (10) | 0 | 0 | 61 (10) |

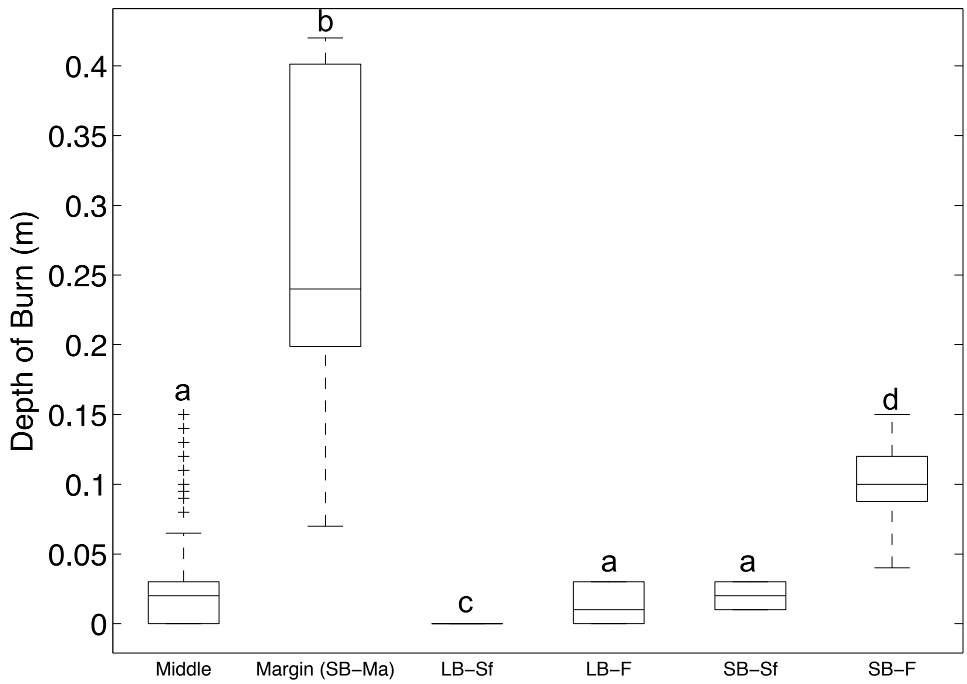


**Figure 3.1.** DOB measured in the middle ($n$=100) and margin ($n$=25) of the burned portion of the peatland complex and classified into the five burn severity groups where tensiometers were installed in the burned portion of the peatland complex during the intensive field survey: SB-Ma ($n$=25), LB-Sf ($n$=50), LB-F ($n$=25), SB-Sf ($n$=50), and SB-F ($n$=25). The horizontal line in the middle of each boxplot is the median DOB value and the next horizontal lines denote the 25 and 75 percentiles. Groups with the same lowercase letter do not have statistically different DOB (least significant difference $\alpha$=0.05). Note that all margin DOB locations were classified as SB-Ma and 16 additional measurements of DOB were collected for SB-Ma (only 9 tensiometers installed) where surface moisture $\theta$ was measured.

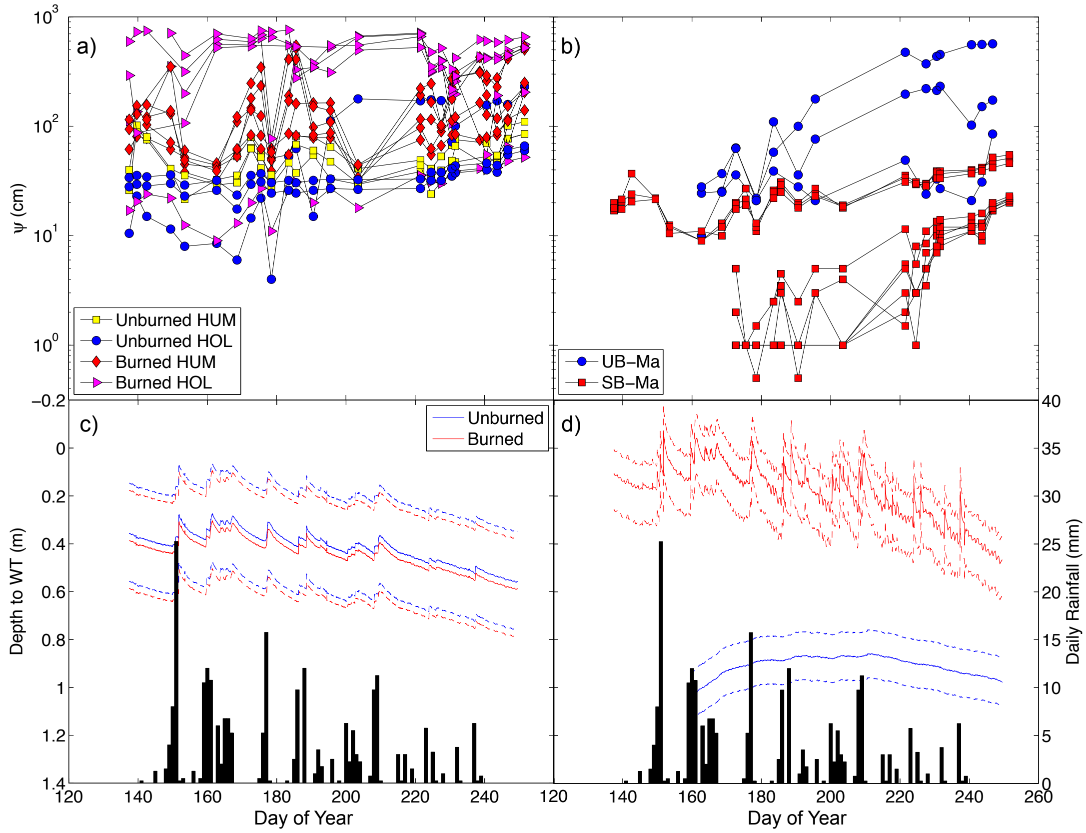


**Figure 3.2.** a) Soil tension from May–September 2013 in burned hummocks ($n$ =5), burned hollows ($n$ =5), unburned hummocks ($n$ =3), and unburned hollows ($n$ =3) in the peatland complex (y-axes are on $\log_{10}$ scales for viewing purposes). b) Soil tension from May–September 2013 in burned margins ($n$ = 9) and unburned margins ($n$ = 3). Continuous average depth to WT beneath tensiometers in the c) middles and d) margins of the burned and unburned portions of the peatland complex (dashed lines represent maxima and minima) with daily rainfall during the study period.

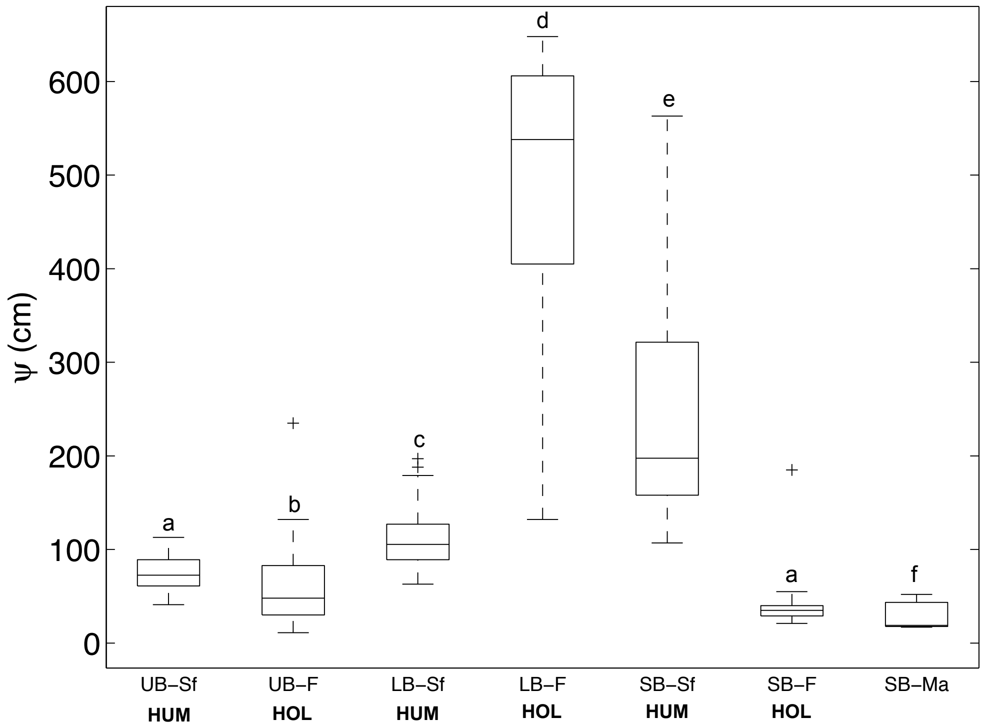


**Figure 3.3** Soil tension in the seven burn severity groups (see Table 3.1) and microform subdivisions (HUM & HOL) collected during the intensive field survey on September 8th, 2013 in the peatland complex: UB-Sf (*n*=50), UB-F (*n*=25), LB-Sf (*n*=50), LB-F (*n*=25), SB-Sf (*n*=50), SB-F (*n*=25), and SB-Ma (*n*=9). Soil tensions with the same lowercase letter are not statistically different (least significant difference α=0.05).

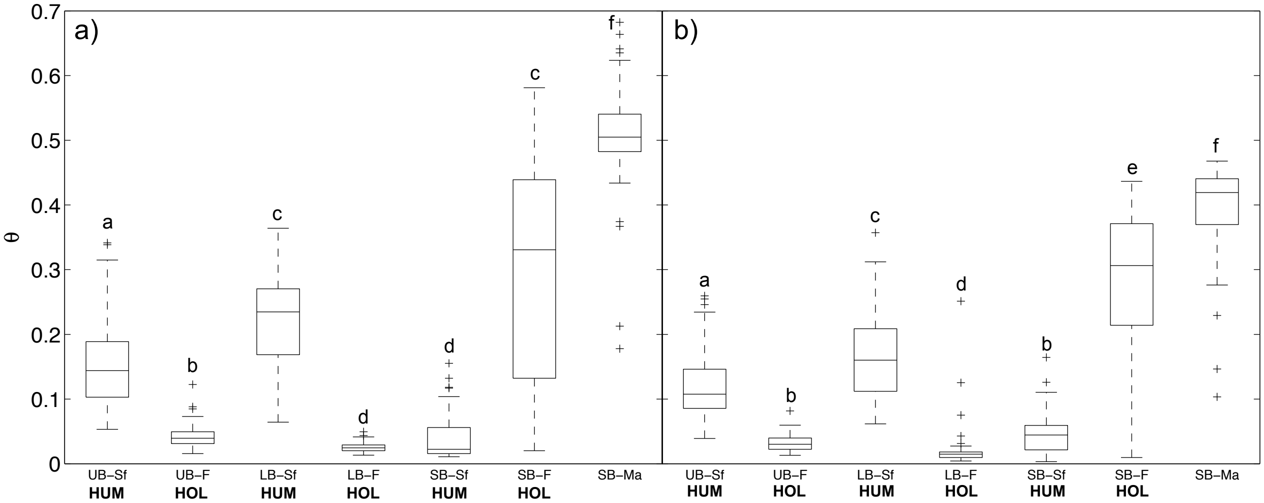


**Figure 3.4.** Surface soil moisture measured in a) top 0.03 and b) 0.06 m of peat in the seven burn severity groups (see Table 3.1) and microform subdivisions (HUM & HOL) collected during the intensive field survey on September 8th, 2013 in the peatland complex: UB-Sf (*n*=100), UB-F (*n*=50), LB-Sf (*n*=100), LB-F (*n*=50), SB-Sf (*n*=100), SB-F (*n*=50), and SB-Ma (*n*=25). Burned margin sampling locations did not include flooded areas.

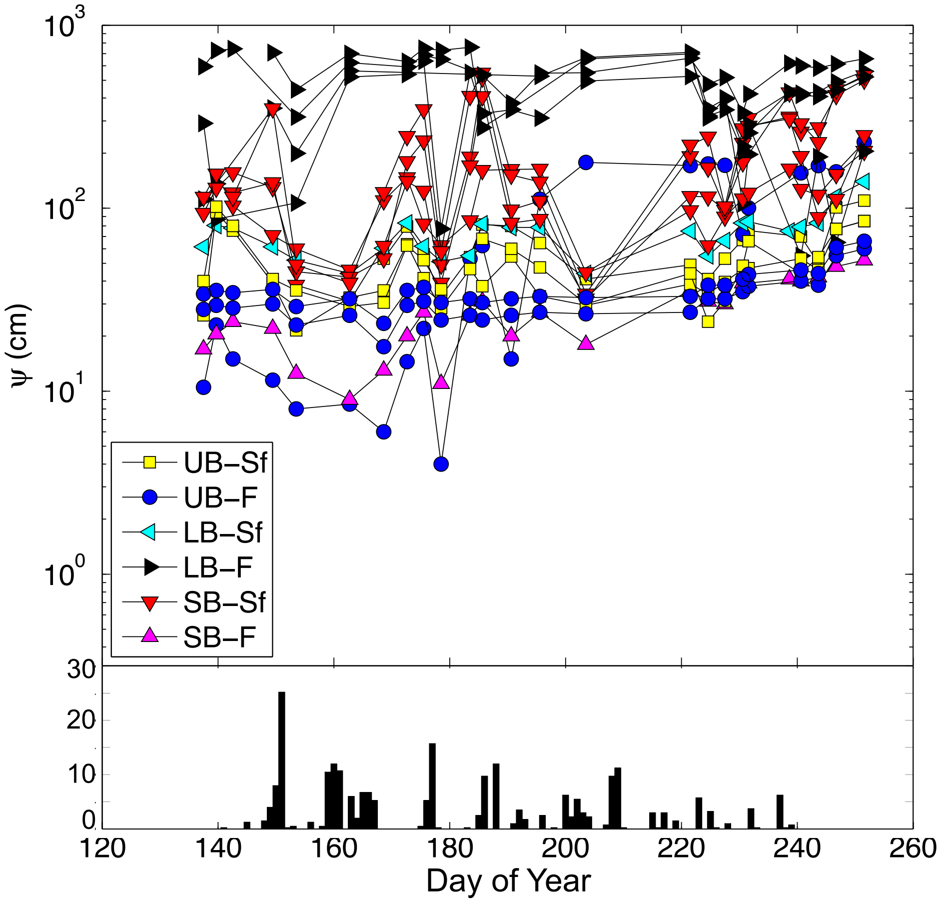


**Figure 3.5.** Soil tension classified into burn severity groups (see Table 3.1) from May–September 2013 in UB-Sf (*n* =3), UB-F (*n* =3), LB-Sf (*n* =1), LB-F (*n* =4), SB-Sf (*n* =4), and SB-F (*n* =1) in the peatland complex (y-axes are on $\log_{10}$ scales for viewing purposes). Note that SB-Ma has been previously shown in Figure 3.2.

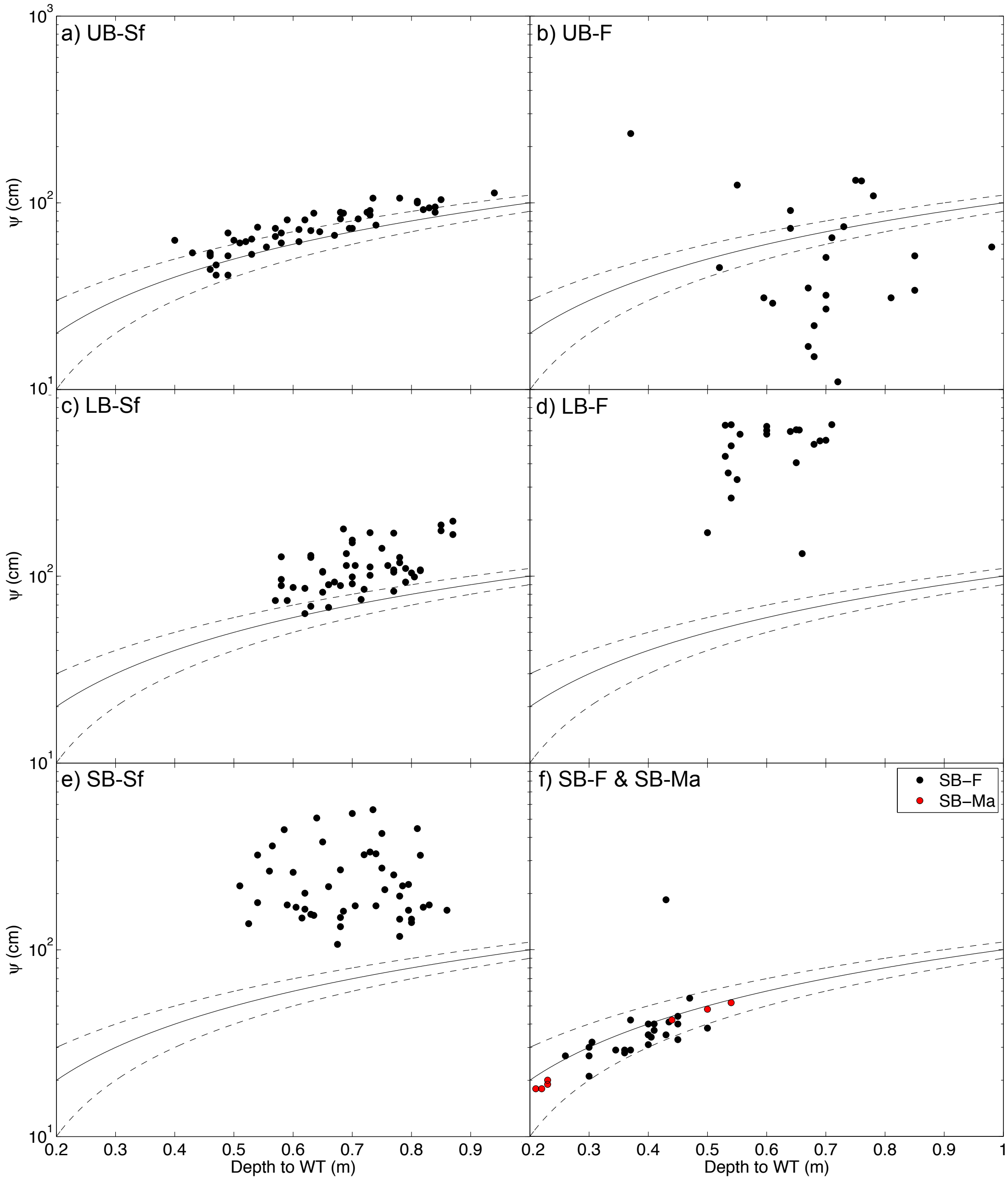


**Figure 3.6.** WT vs. soil tension relationship (y-axes are on $\log_{10}$ scales for viewing purposes) during the intensive field survey on September 8th, 2013 in the peatland complex for a) UB-Sf ($n$=50), b) UB-F ($n$=25), c) LB-Sf ($n$=50), d) LB-F ($n$=25), e) SB-Sf ($n$=50), and f) SB-F ($n$=25) & SB-Ma ($n$=9). The solid line surrounded by the two dashed lines represent the region where an equilibrium profile was calculated (*i.e.* evaporative demand does not outstrip supply from the WT).

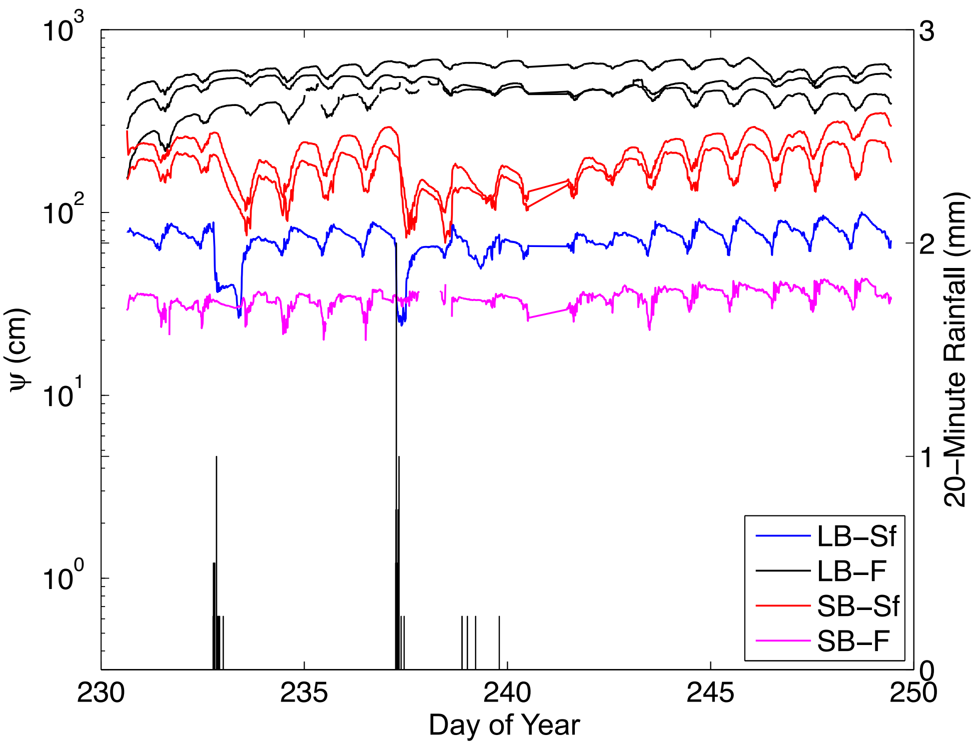


**Figure 3.7.** Soil tension from logging tensiometers measured every 10 minutes and rainfall binned into 20-minute increments within the burned portion of the peatland complex. The blue line is LB-Sf (*n*=1), the black lines are LB-F (*n*=3), the red lines are SB-Sf (*n*=2), and the magenta line is SB-F (*n*=1).

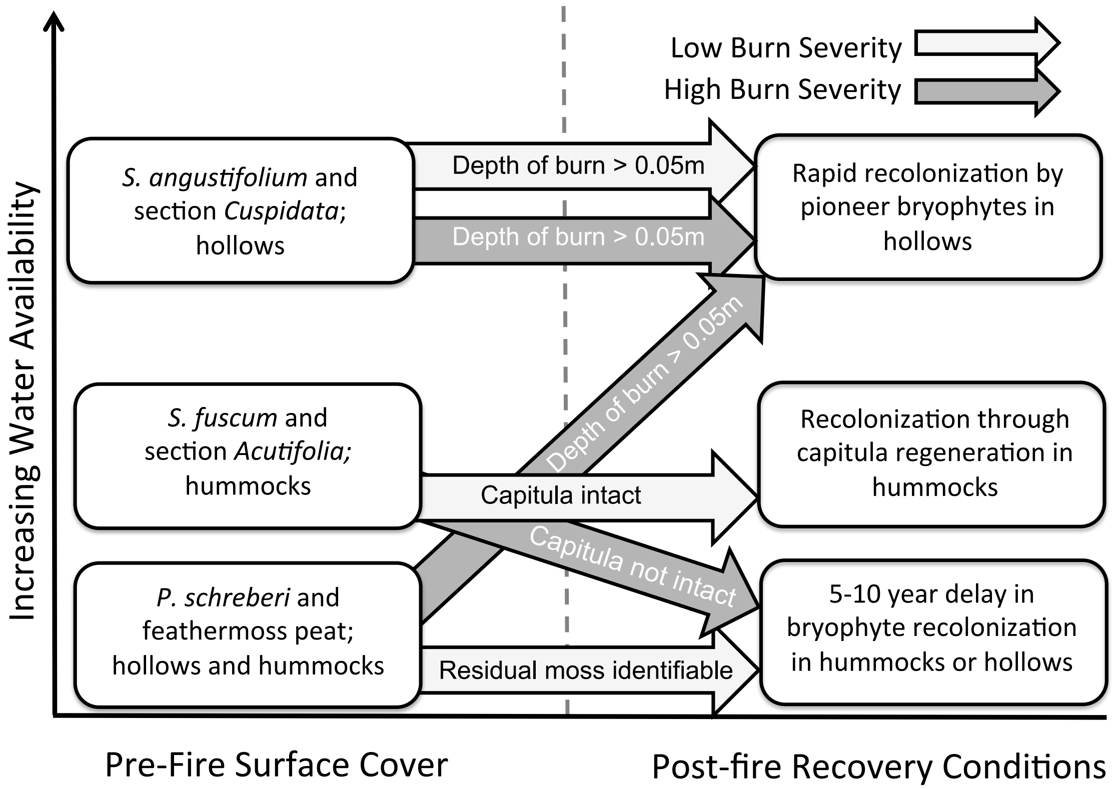


**Figure 3.8.** Conceptual model of how burn severity and pre-fire species interact to alter post-fire water availability and bryophyte recovery. The definitions of high and low burn severity for each pre-fire species are described within the arrows. The trajectory of recovery for *S. angustifolium* is based on Benscoter and Vitt (2008).

# CHAPTER 4: HYDROGEOLOGICAL CONTROLS ON POST-FIRE MOSS RECOVERY IN PEATLANDS

## 4.1 Abstract

Wildfire is the largest disturbance affecting boreal peatlands, however, little is known about the controls on post-fire peatland vegetation recovery. While small-scale variation in burn severity can reduce post-fire moss water availability, high water table (WT) positions following wildfire are also critical to enable the re-establishment of keystone peatland mosses (*i.e. Sphagnum*). Thus, post-fire moss water availability is also likely a function of landscape-scale controls on peatland WT dynamics, specifically, connectivity to groundwater flow systems (*i.e.* hydrogeological setting). For this reason, we assessed the interacting controls of hydrogeological setting and burn severity on post-fire moss water availability in three burned, *Sphagnum*-dominated peatlands in Alberta's Boreal Plains. At all sites, variation in burn severity resulted in a dichotomy between post-fire surface covers that: 1) exhibited low water availability, regardless of WT position, and had minimal (<5%) moss re-establishment (*i.e.* lightly burned feather mosses and severely burned *Sphagnum fuscum*) or 2) exhibited high water availability, depending on WT position, and had substantial (>50%) moss re-establishment (*i.e.* lightly burned *Sphagnum fuscum* and where depth of burn was >0.05 m). Notably, hydrogeological setting influenced the spatial coverage of these post-fire surface covers by influencing pre-fire WTs and stand characteristics (*e.g.* shading). Because feather moss cover is controlled by tree shading, lightly burned feather mosses were ubiquitous (>25%) in drier peatlands (deeper pre-fire WTs) that were densely treed and had little connection to large groundwater flow systems. Moreover, hydrogeological setting also controlled post-fire

WT positions, thereby affecting moss re-establishment in post-fire surface covers that were dependent on WT position (*e.g.* lightly burned *Sphagnum fuscum*). Accordingly, higher recolonization rates were observed in a peatland located in a groundwater flow through system that had a shallow post-fire WT. Therefore, we argue that hydrogeological setting influences post-fire recovery in two ways: 1) by influencing vegetation structure prior to wildfire, thereby controlling the coverage of post-fire surface covers and 2) by influencing post-fire WT positions. These results suggest that post-fire moss recovery in peatlands isolated from groundwater flow systems may be particularly susceptible to droughts and future climate change.

### 4.2 Introduction

The boreal forest accounts for ~29% of the Earth's forest cover (FAO, 2006) and stores ~367 to 1716 Pg C (Bradshaw and Warkentin, 2015), primarily in peatlands (Bradshaw and Warkentin, 2015; NWWG, 1997). Wildfire is the largest disturbance affecting peatlands in this zone, areally accounting for >97% of all disturbances in these ecosystems (Turetsky *et al.*, 2002). Although peatland wildfires typically result in complete stand mortality and the die-off of ground layer vegetation (Benscoter and Vitt, 2008; Zoltai *et al.*, 1998), peatlands are generally resilient to wildfire in that they return to a net carbon sink status within ~20 years post-fire (Wieder *et al.*, 2009). However, given that climate change scenarios suggest that increases in evapotranspiration are likely to exceed increases in precipitation in northern latitudes (Collins *et al.*, 2013), there is concern that peatlands will experience substantial drying (Roulet *et al.*, 1992), thereby increasing their vulnerability to wildfire (Thompson and Waddington, 2013a; Turetsky *et al.*, 2011b), and shift to net sources of carbon to the atmosphere (Turetsky *et al.*, 2004).

The potential impact of future drying and shifting wildfire regimes must be weighed against the recovery of peat-forming vegetation. In particular, peatland mosses (*e.g. Sphagnum*) are critical in maintaining ecosystem resilience because of their role as ecosystem engineers (van Breeman, 1995) whereby they lower decomposition rates (Rydin *et al.*, 2013), conserve water during drought (Kettridge and Waddington, 2014; Waddington *et al.*, 2015), and limit combustion during wildfire (Shetler *et al.*, 2008). Previous studies have provided an understanding of the spatiotemporal patterns of post-fire moss recovery in peatlands (Benscoter, 2006; Benscoter and Vitt, 2008; Wieder *et al.*, 2009); however, the underlying physical processes controlling this recovery have only recently been investigated (Sherwood *et al.*, 2013; Thompson and Waddington, 2013b; Lukenbach *et al.*, 2015a). These inquiries found that the re-establishment of peatland mosses after wildfire was primarily dependent on hydrological factors (*i.e.* depth to water table, soil moisture) (Lukenbach *et al.*, 2015a; Kettridge *et al.*, 2015). Specifically, burn severity (*i.e.* depth of burn) affects post-fire hydrological conditions, such as soil moisture (Lukenbach *et al.*, 2015a) and depth to WT (Kettridge *et al.*, 2015; Sherwood *et al.*, 2013), thereby altering water available to peatland mosses. However, these studies were limited in that they were site-specific, limited in temporal scope, or focused on plot-scale hydrological processes (Lukenbach *et al.*, 2015a; Thompson *et al.*, 2013b). Therefore, there is an immediate need to examine these hydrological controls on post-fire recovery in peatlands at the landscape-scale. Given that hydrogeological setting is a first order landscape-scale control on peatland WTs (Aldous *et al.*, 2015; Demers *et* *al.*, 2013; Devito *et al.*, 2012; Duval *et al.*, 2011; Godwin *et al.*, 2002; Winter, 1999) and has been shown to influence patterns of burn severity in peatlands (Hokanson *et al.*, 2015), we hypothesized that hydrogeological setting would also impart a strong control on the post-fire recovery of keystone peatland mosses (*i.e. Sphagnum*).

Hydrogeological setting defines both the mineral substrate composition (*i.e.* texture) and topographic position at a particular location on the landscape (Winter, 1999). Consequently, hydrogeological setting controls peatland connectivity to groundwater flow systems, the magnitude and composition of hydrological fluxes (Winter, 1999), and thus the frequency of low WT positions in peatlands (Duval *et al.*, 2011; Winter *et al.*, 2003). For example, on regional topographic lows in coarse-textured glaciofluvial outwashes, peatlands are commonly located in groundwater discharge zones, resulting in less dynamic and shallow WT positions (Winter, 2000). Because peatlands located in different hydrogeological settings exhibit differences in WT dynamics, hydrogeological setting is linked to peatland vegetation cover (Godwin *et al.*, 2002). This is important because moss species exert a strong control on peat burn severity (*i.e.* depth of burn) (Hokanson *et al.*, 2015; Benscoter *et al.*, 2011; Shetler *et al.*, 2008). In peatlands, feather mosses are prone to drying and typically undergo higher burn severity (Benscoter *et al.*, 2011), whereas *Sphagnum* mosses (e.g. *Sphagnum fuscum*) are able to efficiently retain water during dry periods and limit depth of burn (DOB) (Shetler *et al.*, 2008; Thompson and Waddington, 2013a). Such variability in burn severity alters the trajectory of post-fire moss recovery by influencing hydrological conditions, such as depth to WT (Hokanson *et al.*, 2015; Lukenbach *et al.*, 2015a; Lukenbach *et al.*, 2015b) and the presence of water repellency (Kettridge *et al.*, 2014). Indeed, peatlands with higher feather moss cover exhibit lags in post-fire recolonization due to reduced water availability after fire because feather mosses facilitate the development of near-surface water repellency (Kettridge *et al.*, 2014; Lukenbach *et al.*, 2015a). Moreover, because feather moss cover is controlled by canopy closure (*i.e.* shading) prior to fire (Bisbee *et al.*, 2001; Kettridge *et al.*, 2013), peatlands located in hydrogeological settings with lower average WT positions likely contain higher feather moss cover due to enhanced tree growth (Leiffers and MacDonald, 1990) and recruitment (Lieffers and Rothwell, 1986). Therefore, peatlands located in hydrogeological settings that are prone to drying, such as those that have water balances dominated by precipitation inputs (Winter, 2000), may be less resilient to wildfire because higher feather moss cover can both increase burn severity (Benscoter *et al.*, 2011) and limit post-fire recolonization rates (Lukenbach *et al.*, 2015a).

While hydrogeological setting likely influences post-fire recovery through its control on peatland WT positions, peatlands also possess internal ecohydrological feedbacks that are critical for post-fire moss re-establishment (Waddington *et al.*, 2015). The presence of peatland microtopography, a form of self-organized spatial patterning (Eppinga *et al.*, 2008), is responsible for some of these ecohydrological feedbacks. In peatlands, species occupy hydrological niches along microtopographic gradients (Rydin *et al.*, 2013) and, as a result, microforms (*i.e.* hummocks and hollows) exhibit differences in their water retention properties (Thompson and Waddington, 2013a). Because hummock species (*i.e. Sphagnum fuscum*) retain water more efficiently during drought, DOB is lower in hummocks (<0.03 m) than hollows (Shetler *et al.*, 2008). This variability in depth of burn maintains peatland microtopography over long-time scales (Benscoter *et al.*, 2015), which can increase ecosystem stability by increasing habitat heterogeneity (Tilman *et al.*, 2006). This ecosystem stability is further enhanced by ecohydrological feedbacks that affect peatland water balances. For example, while the amount of energy available for evaporation increases after wildfire due to the combustion of the tree canopy (Thompson *et al.*, 2015), evaporation and WT drawdowns are limited through the water table depth—moss surface resistance feedback (Waddington *et al.*, 2015) where moss surface resistance increases due to a reduction in capillary flow from the WT (Kettridge *et al.*, 2014). While this and other ecohydrological feedbacks are critical in peatlands, the importance of certain feedbacks is dependent on a peatland's hydrogeological setting. For example, peatlands located in groundwater discharge zones have water balances dominated by groundwater fluxes, thus a post-fire reduction in evaporation is unlikely to affect post-fire WT positions. As such, there is a need to integrate the current understanding of post-fire ecohydrological processes with landscape-scale properties (*i.e.,* hydrogeological setting) to better understand post-fire recovery in peatlands.

Here, we present the first inter-annual, multi-site measurements of post-fire hydrological conditions in peatlands that are linked to landscape-scale properties (*i.e.* hydrogeological setting) in Alberta's Boreal Plain. First, we examined how hydrogeological setting affected post-fire moss water availability. We hypothesized that peatlands located in hydrogeological settings isolated from groundwater sources would exhibit lower post-fire moss water availability (WT, $\Psi$, surface $\theta$) due to their deeper and more dynamic WTs. Second, given the strong control that vegetation imparts on post-fire moss water availability and recovery in peatlands (Benscoter and Vitt, 2008; Lukenbach *et al.*, 2015a), we examined how hydrogeological setting affected pre-fire vegetation structure and how this interacted with post-fire hydrological processes (*e.g.,* WT dynamics). We hypothesized that peatlands located in hydrogeological settings that were isolated from groundwater sources would have higher stand densities and greater feather moss cover prior to fire due to lower average WT positions and that this would reduce post-fire moss water availability and limit moss recolonization rates.

## 4.3 Methods

### *4.3.1 Study sites and experimental design*

In May 2011, a ~90,000 ha fire burned a large portion of the Utikuma Lake Research Study Area (URSA; 56.107$^{o}$N 115.561$^{o}$W) in Alberta's Boreal Plains ecozone (Devito *et al.*, 2012, Figure 4.1). The URSA is part of a long-term hydrogeological study that has examined the local and regional hydrology of a number of pond-peatland-upland complexes since 1999. The URSA region is characterized by low topographic relief and deep heterogeneous glacial substrates, such as lacustrine clay plains, fine-textured disintegration moraines, and coarse-textured glaciofluvial outwashes overlying marine shale (Vogwill, 1978; Devito *et al.*, 2012). The climate is sub-humid, with annual potential evapotranspiration (PET) often exceeding annual precipitation (Devito *et al.*, 2012).

Long-term hydrological data (including WT dynamics) was available for several peatlands that were affected by the Utikuma complex fire (SWF-057, ~90,000 ha), providing an opportunity to examine wildfire impacts in peatlands located in different hydrogeological settings. Our study was carried out in the same peatland complexes as Hokanson *et al.* (2015), specifically, in lake catchments 16, 208, and 171 along a hydrogeological transect at the URSA (Figure 4.1). However, our study focused on the bog portions (based on vegetation indicators and pH, Table 4.1) of the three peatland complexes.

In the lake 16 catchment, an ~5 ha ephemerally perched peatland complex (16-OEP, *i.e. 'Outwash, Ephemerally Perched'*) is positioned adjacent to a regional topographic high in a coarse-textured glaciofluvial outwash and is ephemerally connected to both local and intermediate groundwater flow systems between lakes in the region (Smerdon *et al.*, 2005). We took advantage of the presence of a burned bog (~0.5 ha) and an unburned bog (~0.5 ha) in the peatland complex. These bogs do not have surface flows and are isolated lobes of the larger peatland complex (Hokanson *et al.*, 2015). Vertical hydraulic gradients were 0.02 ± 0.01 and 0.02 ± 0.01 at the burned and unburned bog, respectively, indicating recharge to the underlying groundwater flow system. Therefore, even though groundwater is an important component of the water balance in these bogs, bog-like vegetation is present at the peatland surfaces because of its isolation from solute rich surface flows and groundwater.

In the lake 208 catchment, an ~1 ha kettle hole bog is located on a regional topographic low in the same coarse-textured glaciofluvial outwash as 16-OEP. This outwash bog (208-OFT, *i.e. 'Outwash, Flow Through'*) intersects a large-scale groundwater flow system comprised of several large lakes (~450 – 900 ha) (Hokanson *et al.*, 2015). These larger scale groundwater flows that develop in the coarse material moderate the WT position and minimize WT fluctuations in the bog during drought (Redding, 2009). Approximately half of 208-OFT burned during fire, providing the opportunity to instrument both the burned and unburned portions of the peatland. Vertical hydraulic gradients (0.05 ± .01) in the bog indicated recharge to large groundwater flow system. Furthermore, because oligotrophic groundwater (see electrical conductivity in Table 4.1) represents the vast majority of the water balance at 208-OFT (Redding 2009; Devito *et al.*, 2012) and surface flows do not occur in the bog, species endemic to bogs characterized the vegetation cover.

In lake catchment 171, an ~4 ha bog is located on a lacustrine clay plain (171-CP, *i.e. 'Clay Plain'*) and is the isolated portion of larger peatland complex that connects to lake 171 (Ferone and Devito, 2004; Hokanson *et al.* 2015). Because of the fine-textured substrate composition of the lacustrine clay plain, this bog receives minimal groundwater fluxes (<5 mm $yr^{-1}$) and no surface flows (Ferone and Devito, 2004). Thus, its water influxes were almost entirely comprised of precipitation. This precipitation-dominated water balance coupled with a vertical hydraulic gradient (0.02 ± 0.01) to the underlying mineral substrate, resulted in nutrient poor conditions and the presence of species endemic to bogs (Table 4.1). Because there were no unburned portions of 171-CP, we instrumented the nearest, and only, unburned bog that had a long-term hydrological record at URSA in the same lacustrine clay plain (Thompson *et al.*, 2014). This ~50 ha unburned bog is located ~11 km south of 171-CP in lake catchment 300. The bog (300-CP, '*Clay Plain'*) is isolated from surface flows and groundwater fluxes from adjacent mineral uplands are minimal, thus the site is characterized by species endemic to bogs (Thompson *et al.*, 2014).

Based on long-term hydrological monitoring at URSA presented in Hokanson *et al.* (2015), depth to WT between 2000-2012 was 0.25 – 0.75 m and 0.00 – 0.40 m beneath hollow microforms at 16-OEP and 208-OFT, respectively. Furthermore, Hokanson *et al.* (2015) showed that the depth to WT ranged from 0.15 – 0.60 m at 171-CP, but this was in area that had a high contributing area of within peatland groundwater flows (*i.e.* non-isolated). Thus, the isolated bog portion likely had a 0.10 - 0.20 m deeper WT on average (Lukenbach, unpublished data). At 300-CP, long-term monitoring began in 2008 and depth to WT between 2008-2012 beneath hollow microforms was 0.00 – 0.40 m (Thompson *et al.*, 2014), indicating that 300-CP has a shallower WT on average than 171-CP.

Because 171-CP did not have an unburned portion, burned and unburned sites were not analyzed using a paired approach. Moreover, long-term hydrological monitoring indicates that the large size of 300-CP resulted in the presence of within peatland groundwater flow that moderated WT drawdowns compared to 171-CP. Therefore, the unburned sites primarily provided data on unburned moss water availability under a large range of WT positions. This was important because no measurements of moss water availability were collected prior to fire at the burned sites.

At all the burned and unburned sites, hummock-hollow microtopography was present and *S. fuscum* dominated the surface cover on hummocks (>95% cover), while *P. schreberi* and *Sphagnum angustifolium* were the primary surface covers in hollows. Species cover in hollows varied depending on historical site wetness (Table 4.1). Vascular vegetation cover consisted of *Rhododendron groenlandicum*, *Chamaedaphne calyculata,* and *Rubus chamaemorus*, while the canopies were comprised of black spruce (*Picea mariana*).

To examine how hydrogeological setting affected post-fire water availability, we conducted two experiments. First, given that peatland microtopography has been shown to affect post-fire moss recovery (Benscoter *et al.*, 2005; Benscoter and Vitt, 2008) and that Thompson and Waddington (2013b) observed large differences in post-fire moss water availability between microforms, we randomly sampled post-fire moss water availability (WT, Ψ, surface θ) in an equal number of hummocks and hollows in the middle of each peatland to capture the range of water availability. We then used these observations to examine how moss water availability varied with hydrogeological setting, burn status, and peatland microtopography.

Second, given that microtopography may be limited in its ability to explain patterns of post-fire water availability because burn severity and pre-fire species exert a major control on post-fire water availability (Benscoter and Vitt, 2008; Lukenbach *et al.*, 2015a), we examined how stand characteristics and moss species cover varied prior to fire with hydrogeological setting. We based our analysis on historical WT data (see above) that was indicative of site wetness along the hydrogeological transect. We then linked this pre-fire vegetation control to post-fire recovery using the approach of Lukenbach *et al.* (2015a), which consisted of classifying the bogs into burn severity and species groups and sampling moss water availability in these units of each peatland. We then examined how these observations varied with hydrogeological setting.

*4.3.2 Experiment 1: Microform hydrological measurements across a hydrogeological transect*

The unburned and burned portions of the peatlands were visually classified into hummocks and hollows based on differences in the elevation of the peat surface. Ψ was measured one to three times per week from May – September in 2012, 2013, and 2014 at a depth of 0.05 m in three hummocks and hollows at the burned and unburned sites using

0.02 m outside diameter tensiometers (*cf.* Lukenbach *et al.*, 2015a). In 2014, surface θ (top 0.03 m) was concurrently measured with Ψ directly above each tensiometer cup using a Thetaprobe. Measurements were calibrated following the approach of Kasischke *et al.* (2009). At both the burned and unburned study sites, depth to WT was recorded by capacitance water level recorders at 20 minute intervals in 0.05 m diameter PolyVinyl Chloride (PVC) wells. Depth to WT at locations where tensiometers were installed was determined by measuring the water level in wells adjacent to tensiometers.

*4.3.3 Experiment 2: Measurements of pre-fire vegetation structure, burn severity, and moss water availability along a hydrogeological transect*

*Tree stand characteristics*: Stand density and basal diameters were measured at both the burned and unburned portions of 208-OFT, 16-OEP, and 171-CP using the point-centered quarter method (Mitchell, 2007). At all sites, *Picea mariana* accounted for >95% of the stand cover. Using stand density and basal diameter measurements, canopy fuel load was estimated using an allometric fuel-loading model for forested peatlands in Alberta's Boreal Plains (Johnston *et al.*, 2015). Stand densities and canopy fuel loads at 300-CP were obtained from Johnston *et al.* (2015). Stand ages were ~80, ~135, ~70, and ~80 years for 208-OFT, 16-OEP, 171-CP, and 300-CP, respectively.

*Burn severity classification*: The burned and unburned sites were sub-divided into burn severity and vegetation groups using a slightly modified version of the classification scheme of Lukenbach *et al.* (2015a). Burn severity in *S. fuscum* was defined by whether or not capitula were intact after wildfire (lightly burned *S. fuscum*, LB-Sf vs. severely burned *S. fuscum*, SB-Sf), while burn severity in feather moss locations was assessed by measuring DOB, where the difference in surface elevation between burned areas and surrounding unconsumed areas was measured (*cf*. Kasischke *et al.*, 2008; Mack *et al.*, 2011). Lightly burned feather moss locations (LB-F) were areas where DOB <0.03 m and residual, singed feather moss was visible. Locations where DOB >0.05 m and pre-fire moss cover were not identifiable were classified as severely burned feather mosses (SB-F). In addition to the classification of Lukenbach *et al.* (2015a), *S. angustifolium* was added as a surface cover because 1) its ubiquitous coverage at 208-OFT (unburned) and 300-CP (unburned) and 2) it is prone to higher DOB than *S. fuscum* (Benscoter *et al.*, 2011). Pre-fire moss cover not identifiable was also included to classify areas where either *S. angustifolium* and feather moss may have been present prior to fire (SB-F/Sa). Of note, we observed no areas at our study sites where *S. angustifolium* underwent low burn severity (DOB < 0.03 m).

*Hydrological measurements along a burn severity gradient across a hydrogeological transect*: Three tensiometers were installed in each burn severity group in 2014 at the burned and unburned sites. Ψ was measured one to three times per week from May – September 2014 at a depth of 0.05 m. Because UB-F was not common at 208-OFT (< 2% pre-fire surface cover) and UB-Sa was not common at 16-OEP (< 2% pre-fire surface cover), tensiometers were not installed in these locations. Measurements of Ψ were paired with measurements of surface θ (top 0.03). Although temporal Ψ measurements in the burn severity groups were only collected during 2014, the previously installed tensiometers in the microforms from experiment 1 were classified into burn severity groups and utilized to examine temporal trends in moss water availability in 2012 and 2013. At 16-OEP (burned and unburned), this yielded three UB-Sf, three UB-F, one LB-Sf, two LB-F, two SB-F, and one SB-F/Sa. At 208-OFT (burned and unburned), this yielded three UB-Sf, three UB-Sa, three LB-Sf, and three SB-F/Sa. Finally, at 171-CP (burned) and 300-CP (unburned), this yielded three UB-Sf, three UB-Sa, two LB-Sf, one LB-F, one SB-Sf, and one SB-F/Sa.

These temporal measurements in the burn severity groups were supplemented by an intensive field survey (IFS) at the burned sites. During the IFS, eight tensiometers were installed in each burn severity group at the burned sites (*i.e.* LB-Sf, LB-F, SB-Sf, SB-F/Sa) at a depth of 0.05 m, yielding a total of 32 tensiometers at each site (unburned sites not instrumented). These tensiometers were measured on September 1$^{st}$, 2014, five days after installation, and all data were collected within a 4-hour period. Surface θ (top 0.03 m) was paired with Ψ measurements and 32 additional measurements of surface θ were taken in each burn severity group ($n$ = 40 per group) at each site ($n$ = 100 per site).

*4.3.4 Moss and bryophyte recolonization measurements*

Hydrological measurements during the IFS were paired with measurements of % bryophyte species cover and % *Sphagnum* cover, which were visually estimated in 0.10 x 0.10 m plots above each tensiometer cup and surface θ sampling location to examine the association with bryophyte recovery. The fire resulted in complete mortality of ground layer vegetation, thus the observations of % bryophyte species and *Sphagnum* cover are a direct measure of the recolonization following the fire. The burn severity groups accounted for the majority (>80%) of surface cover at both the burned and unburned sites, and these measurements were used to obtain a site-level estimate of moss recolonization by scaling post-fire moss recolonization in each burn severity group to the coverage of each burn severity group at each site.

*4.3.5 Spatial coverage measurements*

The coverage of microtopography and burn severity groups was determined using the line interception method (Floyd and Anderson, 1987). Surface cover was identified every 0.25 m ($n$ = 400) along two perpendicular 50 m long transects at each site. The study sites were also discretized into microtopography and burn severity groups based on the classification of 2.5 cm resolution multiband 8-bit RGB aerial imagery obtained from an unmanned aerial vehicle flown over the sites at a height of 100 m. Radiometric enhancement was used to create greater contrast between burn severity groups. There was good agreement between the ground surveys and air photo interpretation in the peatlands. The distributions of microtopography and burn severity groups are presented in Tables 4.1 and 4.2, respectively.

*4.3.6 Analyses*

Due to the challenge of instrumenting a large number of sites, only one site was instrumented in each hydrogeological setting. As such, we cautiously interpreted the results of our statistical analyses because the research design was pseudoreplicted (c.f. Hurlbert, 1984). In experiments 1 and 2, $\Psi$ was natural logarithm transformed and surface $\theta$ was converted to a % and square root transformed prior to statistical analyses because residuals were not normally distributed and there were unequal variances in the data collected. In experiment 1, repeated measures ANOVA was used to account for multiple measurements in the same $\Psi$/surface $\theta$ plot throughout the study period ($n$ ranged from 10 – 37 in each plot $yr^{-1}$) and tested the effect of site (3 levels), burn status (2 levels), microtopography (2 levels, nested within site), and year (3 levels, nested within the aforementioned effects) on Ψ and surface θ. Similarly, a repeated measures ANOVA ($n$ = 10 in each plot for the 2014 study period) was used in experiment 2 that included two factors, site (3 levels) and burn severity group (7 levels, nested within site). Because Ψ data in 2012 and 2013 for the burn severity groups were obtained by classifying tensiometers from experiment 1 and sample sizes were small, analyses were only conducted on data from 2014. Data from the IFS in experiment 2 were analyzed using a 2-way ANOVA of site and burn severity group (nested within site). For all aforementioned ANOVAs, post-hoc tests consisted of applying the Holm-Bonferonni correction to Welch (unequal variance) two sample *t*-tests.

For both experiments, robust ordinary least squares regression (OLS) was employed to assess the control of WT on Ψ. Where relationships were found to be statistically significant, the amount of time a hydraulic head difference of <0.10 m existed between near-surface peat (0.05 m depth) and the water table was calculated (*c.f.,* Thompson and Waddington, 2013b). This condition implies that evaporative water losses from the moss surface are rapidly replenished by water fluxes from the water table because the unsaturated hydraulic conductivity of shallow peat is high under low soil tensions (Price *et al.*, 2008). As such, this condition is indicative of moss water stress because the moisture content in near-surface moss layers is does not appreciably decline when the hydraulic head difference is <0.10 m. Because only eight tensiometers were installed in each burn severity group during the IFS, data was pooled across sites, yielding 24 measurements per burn severity group for the analysis of WT-Ψ relationships.

## 4.4 Results

*4.4.1 Experiment 1: Trends in microform water availability along a hydrogeological transect*

The study was carried out during a wet period of Alberta's Boreal Plains climate cycle and was preceded by higher than average precipitation from June 2011 to April 2012 (Environment Canada, 2000). The sampling period from May to September 2012 was characterized by average precipitation (Figure 4.2a) followed by an unusually wet fall and one of the largest snow melts in the past 15 years in spring 2013. Precipitation during the sampling period from May to September 2013 and during the fall of 2013 was near historical averages (Figure 4.2b), while this was followed by a higher than average snow melt in spring of 2014. The sampling period from May – September 2014 was characterized by 50% less precipitation than the historical average, resulting in continuous drying until the end of the study period (Figure 4.2c).

In 2012, WTs at all sites peaked in June following large rain events (Figure 4.3a). A similar pattern was observed in 2013, although the timing of the peaks differed between the sites due to the variability in the size of rain events at each site (Figure 4.3b). In 2014, WTs at all sites peaked following snowmelt and declined throughout the rest of the summer due to dry conditions (Figure 4.3c). Depth to WT did not vary between burned and unburned sites at 16-OEP and 208-OFT, while the WT at 300-CP (unburned) was shallower than at 171-CP (burned). With the exception of 300-CP in 2012, 208-OFT had the shallowest WT on the hydrogeological transect throughout the study period and exhibited a lagged response to the climate cycle, as evidenced by sequential decreases in depth to WT in 2013 and 2014. This resulted in 2012 being the driest year and 2014 being the wettest year during the study period at 208-OFT. In contrast, 16-OEP and 171-CP did not exhibit a lagged response to the climate cycle during the study period. As a result, steep WT declines (> 1.0 m in some hummocks) were observed at 16-OEP and 171-CP near the end of the study period in 2014. At both the burned and unburned sites, depth to WT varied between microforms and was deeper beneath hummocks.

Ψ values were generally higher at the burned sites and at 16-OEP, while they were highly variable both within and between microforms (Figure 4.4). There was a significant effect of site ($F_{2, 1806} = 44.8$, $p < 0.001$), microtopography ($F_{3, 1806} = 16.8$, $p < 0.001$), burn status ($F_{1, 1806} = 19.5$, $p < 0.001$), plot ($F_{72, 1806} = 30.1$, $p < 0.001$), and site X burn status ($F_{2, 1806} = 15.1$, $p < 0.001$) on Ψ (post-hoc test results in Figure 4.4). Year ($F_{24, 1806} = 0.92$, $p = 0.58$) was not a significant factor when nested within the main effects. There was a significant effect of site ($F_{2, 357} = 14.63$, $p < 0.001$), plot ($F_{24, 357} = 13.4$, $p < 0.001$), and site X burn status ($F_{2, 357} = 6.07$, $p < 0.01$) on surface θ (post-hoc test results in Figure 4.5a). In contrast to Ψ, burn status ($F_{1, 357} = 2.25$, $p = 0.15$) and microtopography ($F_{3, 357} = 1.50$, $p = 0.24$) were not significant effects on surface θ. WT was a strong predictor of Ψ in unburned hummocks at all sites, while WT-Ψ relationships in other microforms varied between sites (Table 4.3). Furthermore, WT-Ψ relationships and hydraulic head differences < 0.10 m between near-surface peat and the water table were more common at sites with shallower WTs (Table 4.3).

### *4.4.2 Experiment 2: Trends in pre-fire vegetation structure, burn severity, and moss water availability along a hydrogeological transect*

*Stand characteristics:* At the burned sites, stand densities were highest at 16-OEP followed by 171-CP and 208-OFT (Table 4.1). Fuel loading followed these same trends at the burned sites, as basal diameters were larger at 16-OEP and 171-CP (Table 4.1). Stand densities at the unburned reference sites did not follow this same hydrogeological gradient, as 300-CP had the highest stand density compared to unburned 16-OEP and 208-OFT. However, trends in fuel loads did follow the same hydrogeological gradient as the burned sites as indicated by larger basal diameters at 16-OEP than 300-CP (Table 4.1).

*Trends in water availability along a burn severity gradient across a hydrogeological transect:* There was a significant effect of site ($F_{2, 621} = 6.00$, $p < 0.01$), burn severity group ($F_{18, 621} = 25.3$, $p < 0.001$), and plot ($F_{40, 621} = 2.85$, $p < 0.001$) on Ψ (post-hoc test results in Figure 4.6c). Ψ was higher at 16-OEP and 171-CP than 208-OFT, particularly for the LB-Sf and SB-F/Sa groups. Across all sites, LB-F and SB-Sf exhibited the highest Ψ values, while UB-Sa and SB-F/Sa had the lowest Ψ values (Figure 4.6). With the exception of UB-F at the sites, trends in surface θ were similar to those observed for Ψ (Figure 4.5b). There was a significant effect of burn severity group ($F_{18, 587} = 28.6$, $p < 0.001$) and plot ($F_{40, 587} = 4.01$, $p < 0.001$) on surface θ (post-hoc test results in Figure 4.5b). In contrast to Ψ, site was not a significant main effect ($F_{2, 587} = 2.19$, $p = 0.13$). At all sites, WT-Ψ relationships indicated that the severe burning of feather mosses increased WT connectivity (*i.e.* SB-F/Sa), but decreased WT connectivity in *Sphagnum fuscum* locations (*i.e.* SB-Sf) (Table 4.3). Where significant WT-Ψ relationships were observed (UB-Sa, UB-Sf, LB-Sf and SB-F/Sa at all sites), hydraulic head differences < 0.10 m between near-surface peat and the water table were frequent, albeit variable, across sites (Table 4.3).

*Intensive field survey:* The IFS exhibited similar trends as the continuous measurements from experiment 2; however, because sampling was carried out during an extended dry period, Ψ values were generally higher and surface θ values were lower than during the rest of 2014. There was a significant effect of site ($F_{2, 95} = 4.09$, $p = 0.02$) and burn severity group ($F_{9, 95} = 49.9$, $p < 0.001$) on Ψ and these same effects were also significant for surface θ as follows: site ($F_{2, 478} = 22.8$, $p < 0.001$) and burn severity group ($F_{9, 478} = 322$, $p < 0.001$). Post-hoc test results for Ψ and surface θ are shown in Figure 4.8a and 4.8b, respectively.

During the IFS, WT-Ψ relationships were only significant in LB-Sf ($r^2 = 0.44$) and SB-F/Sa ($r^2 = 0.32$). Hydraulic head differences < 0.10 m between near-surface peat and the water table were present 75% and 100% of LB-Sf and SB-F/Sa locations at 208-OFT, respectively, while hydraulic head differences < 0.10 m between near-surface peat and the water table were present at 16-OEP in 25% and 87.5% of LB-Sf and SB-F/Sa locations, respectively. At 171-CP, hydraulic head differences < 0.10 m between near-surface peat and the water table were present in 25% and 62.5% of LB-Sf and SB-F/Sa locations, respectively.

### *4.4.3 Trends in moss and bryophyte recolonization along a burn severity gradient across a hydrogeological transect*

The dominant surface covers at the unburned sites were *S. fuscum*, *P. schreberi*, and, *S. angustifolium* (Table 4.1, Figure 4.9). Across all sites, high post-fire water availability corresponded to high post-fire % bryophyte recolonization (Table 4.1, Figure 4.9). *Sphagnum* spp. were observed in LB-Sf at all sites, while *Sphagnum* spp. were also common in SB-F/Sa locations at 208-OFT (Figure 4.9). No bryophyte recolonization was observed at all sites in LB-F locations. Within the other burn severity groups, bryophyte recolonization was observed. Little recolonization occurred in SB-Sf at all sites and was characterized by the occurrence of only *C. purpureus*, ranging from 1-5% cover when present. The amount of recolonization was most variable in LB-Sf locations between sites, ranging from 49% at 16-OEP to 95% at 208-OFT (Table 4.1). *S. fuscum* was the dominant species recolonizing LB-Sf locations at all sites, accounting for 98%, 63%, and 97% of the LB-Sf cover at 208-OFT, 16-OEP, and 171-CP. *P. strictum* and *C. purpureus* were also present in LB-Sf locations, but only at 16-OEP did *C. purpureus* account for > 1% of the surface cover (15%). The amount of recolonization that had occurred three years post-fire in SB-F/Sa locations was less variable across the sites than in LB-Sf locations, ranging from 54% at 171-CP to 69% at 208-OFT (Table 4.1). However, SB-F/Sa locations exhibited the greatest variability in the species recolonizing these areas both within and between sites. At 208-OFT, post-fire surface cover in SB-F/Sa locations was 36% *S. angustifolium*, 12% *S. magellanicum*, 9% *P. strictum*, 7% *C. purpureus*, 4% *A. palustre*, and 1% *Dicranum spp.*, respectively. In contrast, post-fire surface cover in SB-F/Sa locations at 16-OEP was 53% *C. purpureus*, 8% *P. strictum*, 3% *M. polymorpha*, 1% *A. palustre*, respectively, and at 171-CP was 32% *C. purpureus*, 18% *P. strictum*, 1% *M. polymorpha*, 1% *S. angustifolium*, 1% *A. palustre*, respectively.

## 4.5 Discussion

### *4.5.1 Microtopography is a poor indicator of post-fire water availability in peatlands*

While we hypothesized that drier sites (deeper WTs) would have lower post-fire water availability in experiment 1, 16-OEP had significantly lower water availability than

171-CP (Figures 4.4 & 4.5a), even though these sites had similar post-fire WT positions. This is likely because a range of burn severities and moss species can occur in one microform type (*e.g.* LB-F or SB-F/Sa in hollows; Lukenbach *et al.*, 2015a), resulting in the variability in post-fire water availability within microforms being greater than between microforms, sites, and burn statuses. Therefore, while sampling post-fire water availability along a microtopographic gradient is likely adequate for identifying the general post-fire distribution of water availability at a specific peatland, we argue that it cannot be used to explain why high/low post-fire water availability is present. As such, post-fire classification schemes based on microtopography alone cannot be used as indicators of post-fire water availability in *Sphagnum*-dominated peatlands.

*4.5.2 Burn severity and vegetation cover are useful indicators of post-fire recovery at the landscape-scale*

Low burn severity in *S. fuscum* (*i.e.* LB-Sf) mildly decreased water availability when compared to its unburned state, while high burn severity (*i.e.* SB-Sf) led to much drier post-fire surface soil conditions. The opposite was true for feather mosses, where low burn severity (*i.e.* LB-F) resulted in low post-fire water availability. Moreover, high burn severity in hollows (*i.e.* SB-F/Sa), regardless of whether pre-fire surface cover was feather mosses or *S. angustifolium,* increased post-fire water availability. Given that this pattern of water availability in the burn severity groups was consistent across all sites (Table 4.3, Figures 4.5b, 4.8) and between years (Figure 4.6), it highlights that the classification adapted from Lukenbach *et al.* (2015a), which comprised >80% of the surface cover at each site, can be employed as an easily measurable and effective indicator of post-fire water availability and moss recovery in *Sphagnum*-dominated peatlands.

*4.5.3 Hydrogeological controls on pre-fire peatland vegetation structure that influence post-fire moss recovery*

The peatland surface following wildfire can be conceptualized as a dichotomy between: 1) low water availability LB-F and SB-Sf locations that cannot be rapidly recolonized regardless of WT position due to their water repellent nature (Kettridge *et al.*, 2014) and/or their inability to retain moisture, and 2) high water availability LB-Sf and SB-F/Sa locations that can recolonize quickly and rely on upward capillary flow from the WT, as evidenced by significant WT-Ψ relationships in these burn severity groups (Figure 4.7a). Hydrogeological setting influenced the spatial distribution of these burn severity groups by influencing pre-fire stand characteristics and surface vegetation cover, which determined the areal extent of the burn severity groups at the sites after fire. Densely forested sites (*i.e.* 16-OEP) had higher canopy closure (*i.e.* shading), thereby resulting in high spatial coverage of feather mosses (UB-F) prior to fire and, consequently, increasing the spatial coverage of LB-F locations after fire. Although LB-F exhibited low water availabilities at all of the burned sites, the spatial coverage of this burn severity group varied extensively with hydrogeological setting (Table 4.1).

The ~80 year old stand at 208-OFT exhibited both a low canopy fuel load and stand density, resulting in low spatial coverage of feather mosses prior to wildfire (*i.e.* UB-F and LB-F). Given that canopy fuel loads and *Picea mariana* stand densities increase with time since fire in peatlands (Johnston *et al.*, 2015), the shallow WT at the site appears to have limited the growth and recruitment of *Picea mariana* prior to wildfire. This is further substantiated by the higher stand density (2200 stems ha$^{-1}$) and canopy fuel load (4800 kg ha$^{-1}$) at an adjacent peatland (<100 m away) to 208-OFT (Hokanson, 2014), where the adjacent peatland's position in the groundwater flow system resulted in an ~0.30 m deeper WT than 208-OFT (Hokanson, 2014). Likewise, the substantially lower canopy fuel load at 171-CP compared to 16-OEP is likely attributable to differences in stand ages (~70 vs. ~135 years) and not site wetness, since these sites exhibited similar depth to WTs both before and after wildfire. Nevertheless, the common presence of feather mosses at these sites prior to fire led to the frequent occurrence of LB-F locations after wildfire (Table 4.1). This not only resulted in low post-fire water availability but also rendered >25% of surface at 16-OEP and 171-CP unable to recolonize for at least the first three years following wildfire (Table 4.1 and Figure 4.9).

Deeper WTs at 171-CP and 16-OEP also likely increased the frequency of SB-Sf surface cover at 16-CP and 171-CP compared to 208-OFT after wildfire. Specifically, low WTs at the time of fire probably lowered the moisture content of *S. fuscum* capitula, making them more prone to combustion. As such, the spatial coverage of SB-Sf (low water availability) versus LB-Sf (high water availability) varied with hydrogeological setting. This is particularly important because SB-Sf accounted for 5%, 14%, and 22% of the surface cover at 208-OFT, 16-OEP, and 171-CP, respectively, and little (< 5%) moss recolonization was observed in this burn severity group. Combining the spatial coverage of SB-Sf with LB-F at the sites highlights that >40% of the surface at 16-OEP and 171-CP was unable to recolonize for the first three years after fire. Given that precipitation only temporarily (<1 day) increases moisture in SB-Sf locations and does not affect moisture in LB-F locations (Lukenbach *et al.*, 2015a), we suggest that future research should examine how the recovery of SB-Sf and LB-F progresses over time and whether: 1) these areas are laterally recolonized by adjacent recovering areas, or 2) peat that is currently water repellent (or unable to retain water) breaks down over time and transitions to a more recolonizable medium.

*4.5.4 Post-fire hydrogeological controls on peatland moss recovery*

Through its control of post-fire WT position, hydrogeological setting appeared to determine how recovery progressed in areas that were rapidly recolonizable (*i.e.* LB-Sf and SB-F/Sa). This is exemplified by differences in $\Psi$ in LB-Sf and SB-F/Sa locations at 208-OFT compared to these same burn severity groups at 16-OEP and 171-CP during both the study period and the IFS (Figures 4.6 and 4.8). Because $\Psi$ is a more precise indicator of moss stress than surface $\theta$ (Thompson and Waddington, 2008), it provided a useful means of comparing across sites with different WT positions. Although evaporative demand frequently exceeded supply from the WT (*i.e.* hydraulic head differences > 0.10 m between near-surface peat and the water table) at all the burned sites (Figure 4.7a), deeper WTs at 16-OEP and 171-CP resulted in higher soil tension $\Psi$ values (WTs >0.75 m in Figure 4.7a). Moreover, by examining the exceedance probability (Figure 4.7b) of 100 cm of soil tension $\Psi$, which has been frequently used as an important threshold in prior studies (*e.g.* Price, 1997; Thompson and Waddington 2008), the effect of hydrogeological setting on post-fire water availability is readily observable. While LB-Sf locations at 208-OFT exceed this threshold <2% of the time, LB-Sf locations at 16-OEP and 171-CP exceed this threshold ~20% and ~10% of the time, respectively, during the 2012-2014 study period. This may explain differences in recolonization between the burned sites three years after wildfire, where 95% of LB-Sf at

208-OFT was recolonized, primarily by *S. fuscum,* while only 49% of LB-Sf at 16-OEP was recolonized, of which, *C. purpureus* comprised 15% of the post-fire surface cover (Table 4.1, Figure 4.9). While SB-F/Sa locations did not exhibit large differences in the absolute amount of recolonization between sites (Table 4.1), the species recolonizing these locations were variable across hydrogeological settings. In particular, *S. angustifolium* and *S. magellanicum* had already recolonized some SB-F/Sa locations at 208-OFT, while this did not occur at 16-OEP and was rare at 171-CP (Figure 4.9). This may be attributable to the presence of a near-surface WT at 208-OFT (frequently <0.10 m), which may have facilitated the rapid recolonization of *Sphagnum spp.*

Since this study was carried out during a wet period of the climate cycle, our observations of post-fire moss water availability and moss recolonization may be higher than during a dry post-fire period. Furthermore, peatland WT response to the climate cycle varied between hydrogeological settings, which likely affected recolonization patterns observed herein. As such, it is a possible that a post-fire drought at 171-CP and 16-OEP may resemble the end of the 2014 study period, although these sites may be dissimilar during other periods of the climate cycle because of 16-OEP's ephemeral connection to groundwater sources. Likewise, the lagged response of 208-OFT to the climate cycle would likely result in lower WT positions during a post-fire drought, potentially delaying the recolonization of *Sphagnum* spp. at the site compared to the spatial coverage observed herein.

Variability in the recolonization of *Sphagnum* spp. is likely to have large implications for post-fire carbon balances and the resilience of *Sphagnum*-dominated peatlands to wildfire. The rapid recolonization of *S. angustifolium* and *S. magellanicum* at

208-OFT and marginal moss recolonization at 16-OEP represent extremes in the post-fire recolonization distribution when compared to previous studies (*c.f.* Benscoter 2006; Benscoter and Vitt, 2008). Given that Benscoter and Vitt (2008) observed that post-fire *Sphagnum* cover in peatlands ranges from 5-30% ~5 years after fire, the results herein provide a mechanistic understanding of this variability, asserting that site-level differences in *Sphagnum* recolonization are due to factors controlling site wetness (*i.e.* hydrogeological setting).

While this study highlights how drier peatlands with limited groundwater connectivity experience lower moss recolonization rates, such peatlands may have vegetation structures optimized to suit their hydrogeological setting. For example, a high areal extent of low post-fire water availability surface covers (*e.g.* lightly burned feather mosses) may provide a peatland-scale water conservation mechanism by limiting post-fire evaporation (Kettridge *et al.*, 2014; Lukenbach *et al.*, 2015a), thereby accelerating post-fire moss recolonization and carbon accumulation in areas that experienced higher DOB (*i.e.* SB-F/Sa). This self-regulation of water losses may be an important mechanism that makes peatlands isolated from or ephemerally connected to groundwater flow systems (*i.e.* drier) resilient to wildfire.

**4.6 Conclusion**

Hydrogeological setting influences post-fire recovery and water availability in two ways: 1) by influencing vegetation structure prior to wildfire, thereby controlling the coverage of burn severity groups after wildfire, and 2) by influencing post-fire WT positions. The vegetation-based classification scheme presented herein can be linked to landscape-scale properties and thus provides a rapid and effective means of understanding post-fire recovery in *Sphagnum*-dominated peatlands. Given that previous studies were typically conducted in large peatland complexes and were not linked to landscape-scale properties (Benscoter and Vitt, 2008), peatlands with lower average WT positions may not be accurately represented in current conceptual models of post-fire recovery in peatlands. Indeed, peatlands in late successional stages and those situated in hydrogeological settings that are not well connected to groundwater flow systems are likely to be the most vulnerable to the combined effects of wildfire and climate change. Furthermore, ecohydrological feedbacks examined in previous studies (Waddington *et al.*, 2015; Kettridge *et al.*, 2014) must be evaluated in the context of landscape-scale properties. In particular, negative feedbacks within peatlands that reduce WT drawdowns during dry periods (Waddington *et al.*, 2015) may only be important in peatlands with limited groundwater connectivity. Future studies should continue to link local ecohydrological processes with landscape-scale processes in order to better understand the resilience of peatlands to wildfire.

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**Table 4.1.** Summary of site characteristics, surface cover of microforms/burn severity groups, and recolonization in the burn severity groups at each site. B and UB stand for burned and unburned at each site along the hydrogeological transect (OFT = outwash flow-through, OEP = outwash ephemerally perched, and CP = clay plain). See Table 4.2 for the nomenclature of the burn severity groups. From 2012-2014, mineral electrical conductivity was monitored in wells in adjacent uplands in each hydrogeological setting, while peat electrical conductivity was measured in wells in the middle of each peatland. Hydraulic conductivity was measured in piezometers adjacent to these wells. All other variables were measured within the peatland at each site. The first % listed with the burn severity groups refers to their % coverage at each site, while the % following the forward slash was how much recolonization had occurred in these same groups by September 1st, 2014, three years post-fire.

| | **Site** | | | | | |
|---|---|---|---|---|---|---|
| | **208 (B)** | **208 (UB)** | **16 (B)** | **16 (UB)** | **171 (B)** | **300 (UB)** |
| pH | 4.5 | 4.5 | 4.0 | 4.0 | 4.0 | 4.0 |
| Peat depth (m) | 1.5 | 1.5 | 2.9 | 1.6 | 3.9 | 1.7 |
| Peat electrical conductivity (uS $cm^{-1}$) | 40-60 | 40-60 | 40-70 | 100-150 | 40-60 | 40-100 |
| Mineral electrical conductivity (uS $cm^{-1}$) | 20-40 | 20-40 | 300-1000 | 300-1000 | 1000-3000 | 1000-3000 |
| Mineral hydraulic conductivity (m $s^{-1}$) | $10^{-3}$ | $10^{-3}$ | $10^{-8}$ | $10^{-8}$ | $10^{-8}$ | $10^{-8}$ |
| Stand Density >1.3 m (stems $ha^{-1}$) | 1300 | 1300 | 7100 | 4875 | 2350 | 8000 |
| Canopy fuel load > 1.3 m (kg $ha^{-1}$) | 1760 | 1760 | 10600 | 6000 | 5500 | 5300 |
| % Hummock | 64 | 64 | 41 | 56 | 53 | 70 |
| % Hollow | 36 | 36 | 59 | 44 | 47 | 30 |
| % UB-Sf | 0 | 62 | 0 | 32 | 0 | 49 |
| % UB-F | 0 | 3 | 0 | 55 | 0 | 17 |
| % UB-Sa | 0 | 23 | 0 | 1 | 0 | 23 |
| % LB-Sf / % recolonized | 57/95 | 0 | 5/49 | 0 | 30/72 | 0 |
| % LB-F / % recolonized | 2/0 | 0 | 55/0 | 0 | 26/0 | 0 |
| % SB-Sf / % recolonized | 5/1 | 0 | 14/1 | 0 | 22/1 | 0 |
| % SB-F/Sa / % recolonized | 24/69 | 0 | 18/65 | 0 | 15/54 | 0 |

**Table 4.2.** Classification of peatland surface cover into burn severity groups, their definition, location, and which microform they are typically present in at the sites.

| Acronym | Category | Definition | Location | Microform |
|---|---|---|---|---|
| UB-Sf | Unburned *S. fuscum* | *S. fuscum* | Unburned Peatland | Hummock |
| UB-Sa | Unburned *S. angustifolium* | *S. angustifolium* | Unburned Peatland | Hollow |
| UB-F | Unburned *P. schreberi* | *P. schreberi* | Unburned Peatland | Hollow |
| LB-Sf | Lightly burned *S. fuscum* | Capitula intact | Burned Peatland | Hummock |
| LB-F | Lightly burned *P. schreberi* | DOB <0.03 m, residual *P. schreberi* visible | Burned Peatland | Hollow |
| SB-Sf | Severely burned *S. fuscum* | Capitula not intact | Burned Peatland | Hummock |
| SB-F/Sa | Severely burned feather moss/*angustifolium* | DOB >0.05 m and pre-fire moss cover not identifiable | Burned Peatland | Hollow |

**Table 4.3.** Robust ordinary least squares regression relationships between water table (WT) and soil tension (Ψ) for microforms (experiment 1) and burn severity groups (experiment 2, see Table 4.2 for nomenclature). UHUM, UHOL, BHUM, and BHOL represent unburned hollows, unburned hummocks, burned hummocks, and burned hollows, respectively. Where WT-Ψ relationships were significant ($p$ < 0.05, denoted by asterisk), the percent of time hydraulic head differences < 0.10 m between near-surface peat and the water table (*i.e.* evaporative demand does not outstrip supply from the WT) were present was calculated.

| | **Experiment 1 - Temporal Analyses** | | | | | |
|---|---|---|---|---|---|---|
| | **208-OFT** | | **16-OEP** | | **171-CP & 300-CP** | |
| **Microform** | WT-Ψ $r^2$ | % Eq. | WT-Ψ $r^2$ | % Eq. | WT-Ψ $r^2$ | % Eq. |
| UHUM | 0.64* | 95 | 0.33* | 64 | 0.68* | 98 |
| UHOL | 0.41* | 83 | 0.14 | - | 0.38* | 94 |
| BHUM | 0.58* | 85 | 0.16 | - | 0.05 | - |
| BHOL | 0.49* | 87 | 0.27 | - | 0.30* | 54 |
| | **Experiment 2 - Temporal Analyses** | | | | | |
| | **208-OFT** | | **16-OEP** | | **171-CP & 300-CP** | |
| **Group** | WT-Ψ $r^2$ | % Eq. | WT-Ψ $r^2$ | % Eq. | WT-Ψ $r^2$ | % Eq. |
| UB-Sf | 0.63* | 93 | 0.33* | 78 | 0.68* | 96 |
| UB-Sa | 0.45* | 50 | - | - | 0.38* | 90 |
| UB-F | - | - | 0.14 | - | 0.12 | - |
| LB-Sf | 0.60* | 70 | 0.48* | 60 | 0.45* | 75 |
| LB-F | 0.19 | - | 0.17 | - | 0.09 | - |
| SB-Sf | 0.20 | - | 0.17 | - | 0.22 | - |
| SB-F/Sa | 0.63* | 55 | 0.36* | 72 | 0.66* | 49 |

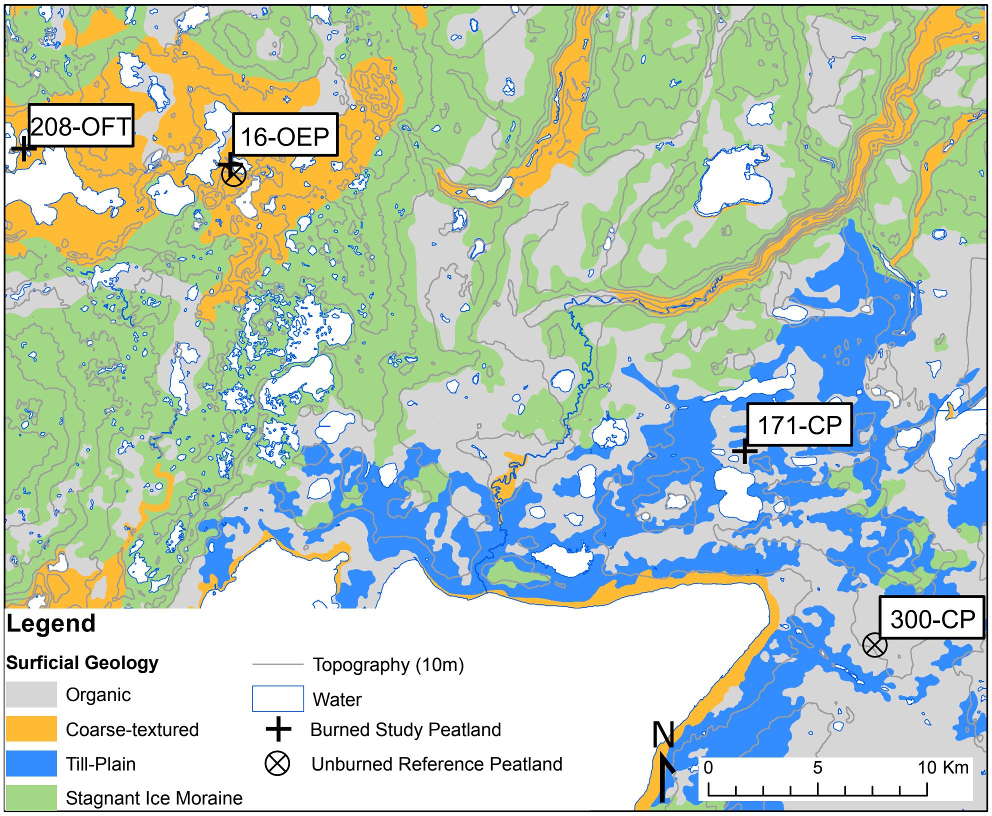


**Figure 4.1.** Map of Utikuma Region Study Area (URSA), adapted from Fenton *et al.* (2004) and Hokanson *et al.* (2015), showing the location of the burned and unburned sites along a hydrogeological transect.

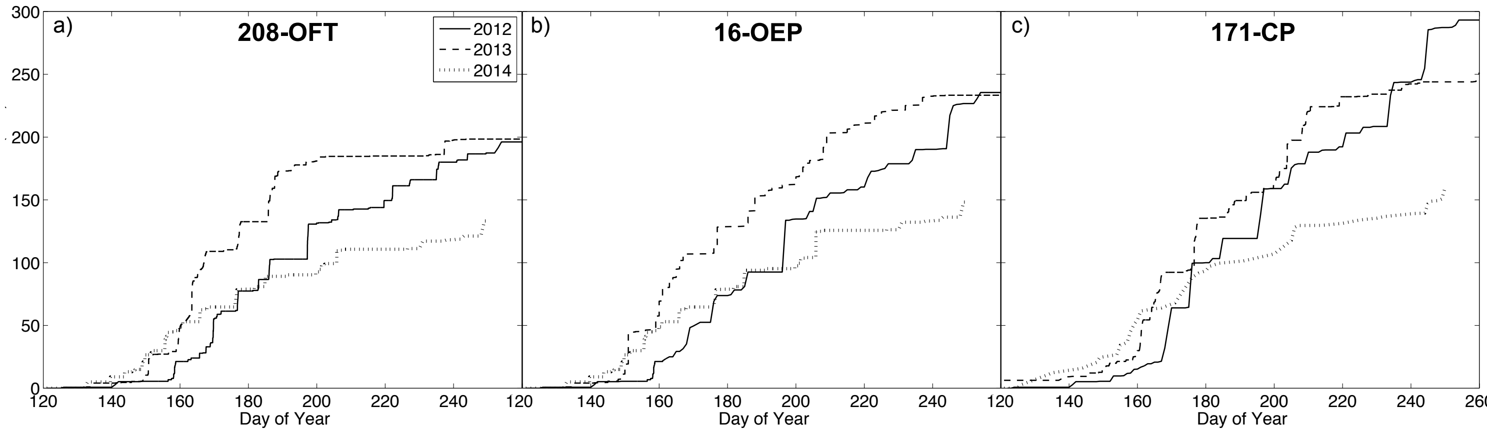


**Figure 4.2.** Cumulative rainfall from May – September in 2012, 2013, 2014 at a) 208-OFT, b) 16-OEP, and c) 171-CP.

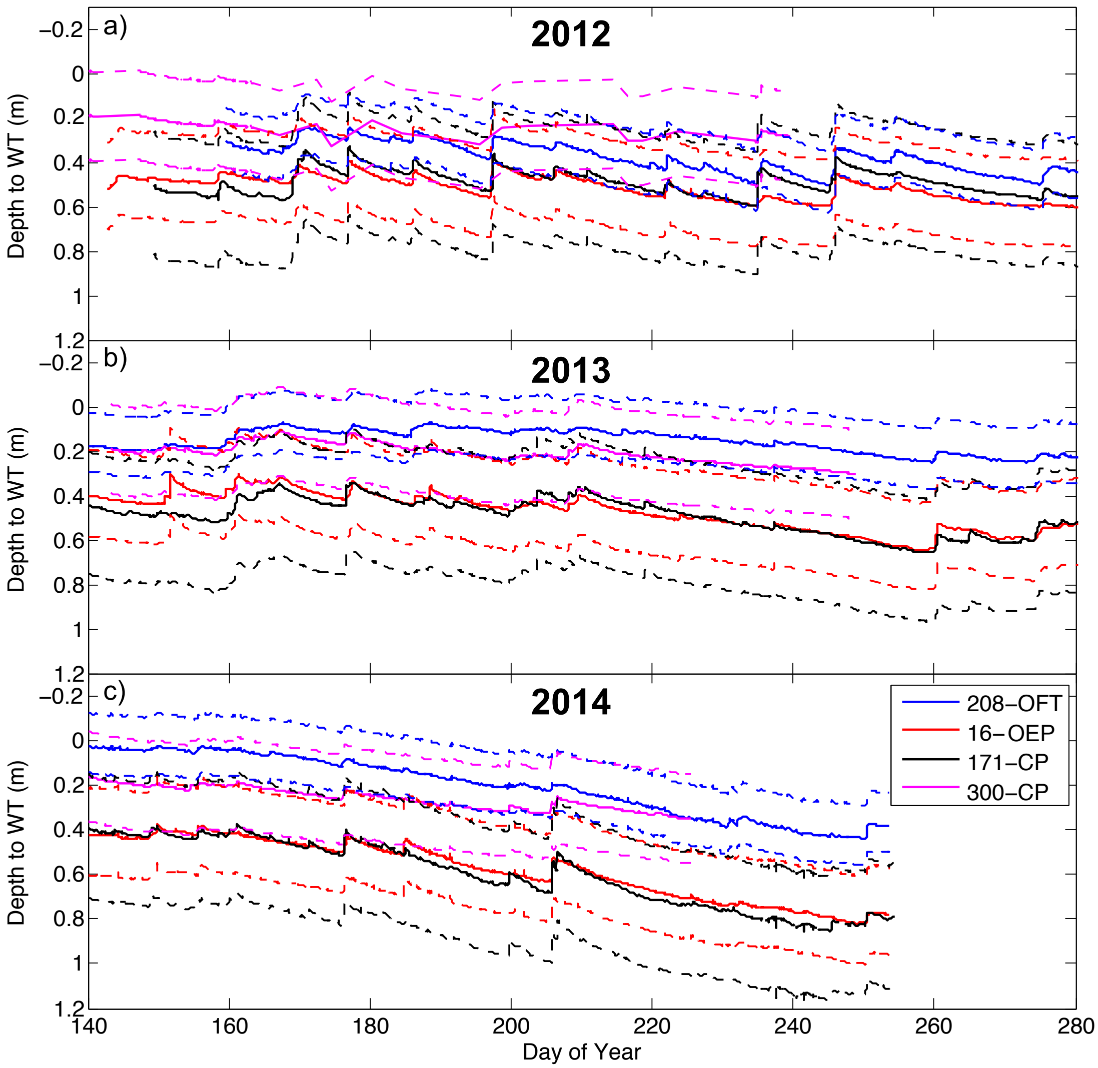


**Figure 4.3.** Average depth to WT (solid lines) beneath tensiometers installed in microforms (3 hummocks and 3 hollows per site) at the burned and unburned sites for a) 2012, b) 2013, and c) 2014. Dashed lines represent the minima and maxima depth to WT beneath microforms for each site. Differences in averages, minima, and maxima between the burned and unburned sites at 208-OFT and 16-OEP were less than 0.05 m, thus the unburned traces are not shown for visual purposes. Rapid rises in depth to WT are caused by rainfall events.

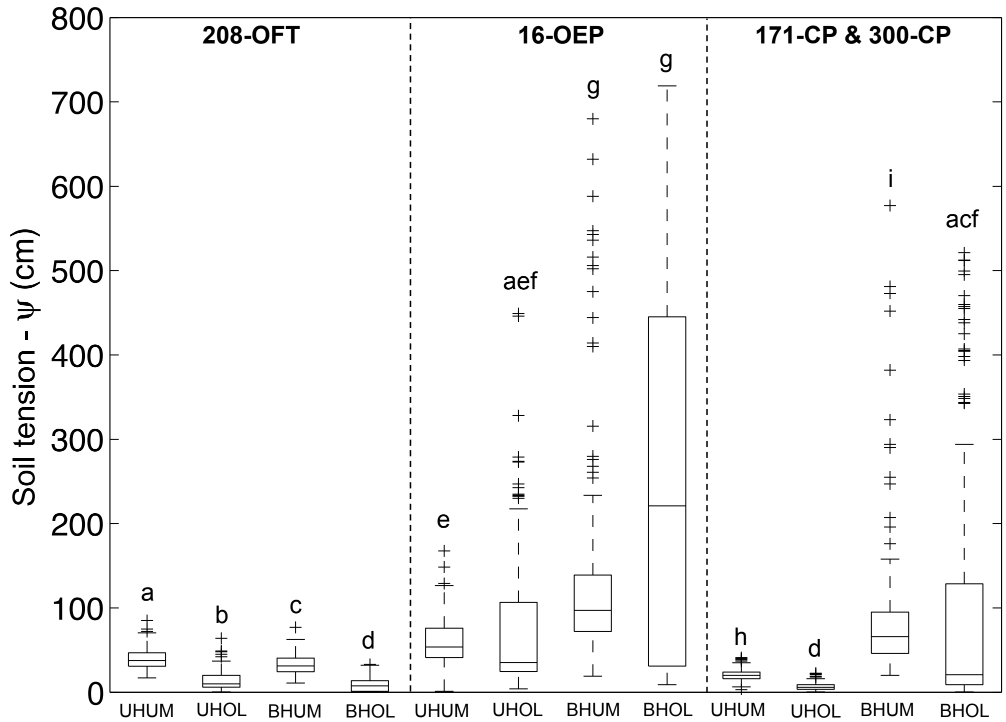


**Figure 4.4.** Experiment 1: Soil tension (Ψ) measured at least once per week from May – September in 2012, 2013, and 2014 in three hummocks and hollows at both the burned and unburned sites. Because year was not a significant factor affecting Ψ, data are grouped together for all years. UHUM, UHOL, BHUM, and BHOL represent unburned hollows, unburned hummocks, burned hummocks, and burned hollows, respectively. Ψ with the same lowercase letter are not statistically different (least significant difference α=0.05).

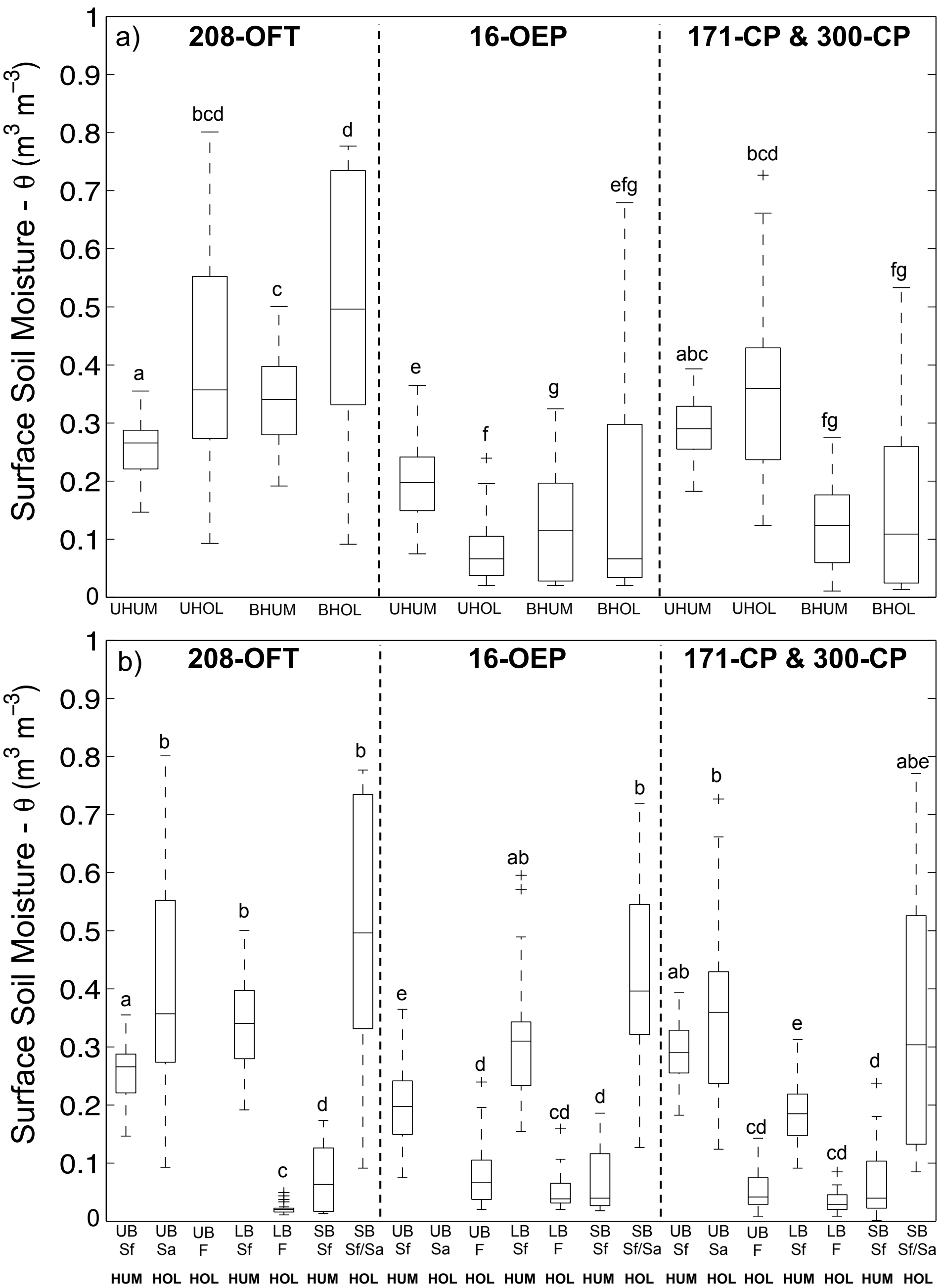


**Figure 4.5.** a) Experiment 1: Surface volumetric moisture content (θ, top 0.03 m) in 2014 in three hummocks and hollows at both the burned and unburned sites. UHUM, UHOL, BHUM, and BHOL represent unburned hollows, unburned hummocks, burned hummocks, and burned hollows, respectively. b) Experiment 2: Surface θ in 2014 in each burn severity group ($n$ = 3 per group) at each site (microform also noted) at both the burned and unburned sites. The nomenclature for the burn severity groups is listed in Table 4.2. Surface θ with the same lowercase letter are not statistically different (least significant difference α=0.05).

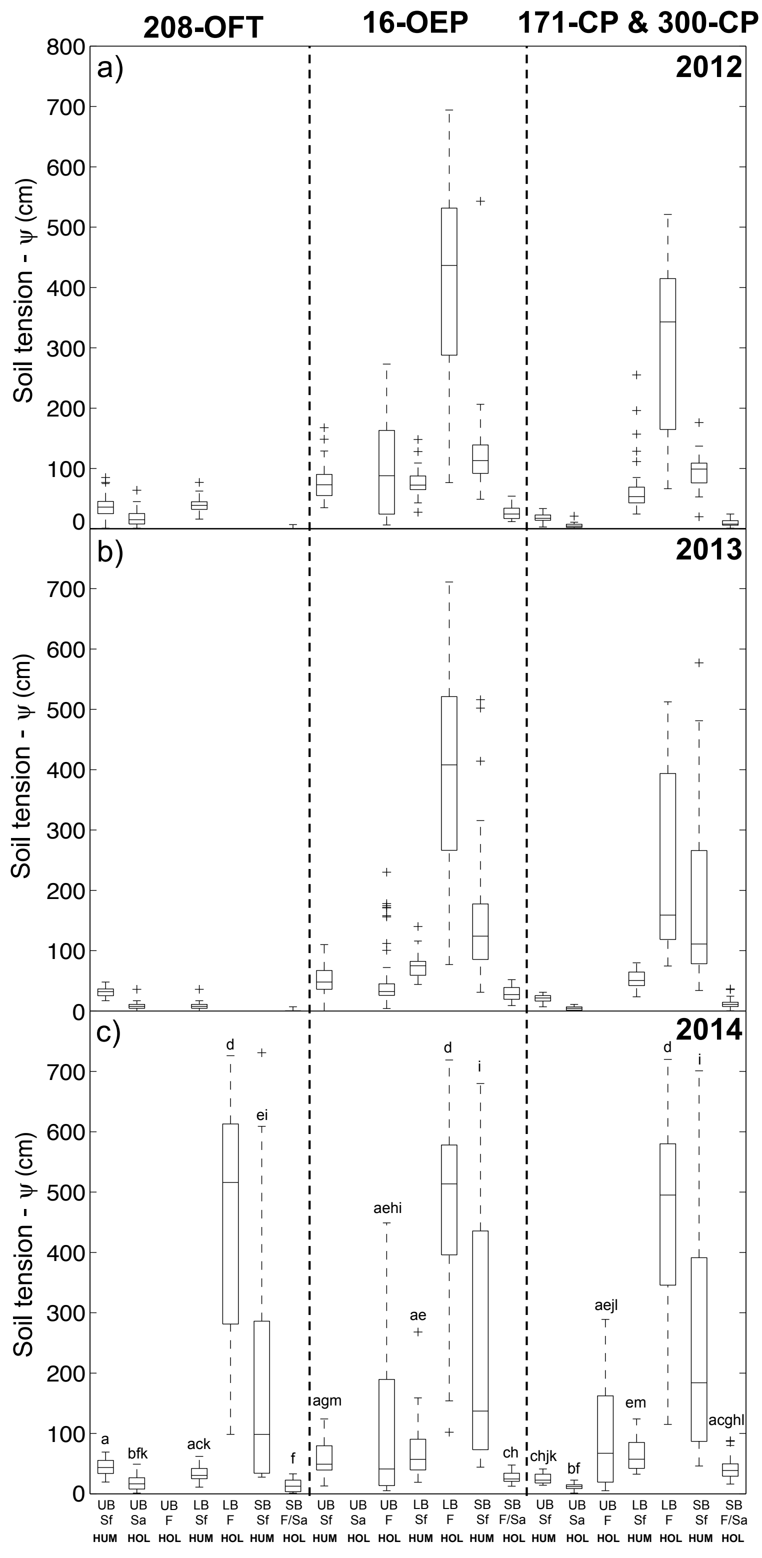


**Figure 4.6.** Experiment 2: Soil tension (Ψ) in burn severity groups (microform also noted) at both the burned and unburned sites. The nomenclature for the burn severity groups is listed in Table 4.2. Ψ data from 2012 and 2013 was obtained by classifying the

tensiometers placed in microforms from experiment 1 into burn severity groups. Because of the low sample sizes in 2012 and 2013, statistical analyses were not conducted. For 2014 ($n = 3$ per burn severity group), $\Psi$ with the same lowercase letter are not statistically different (least significant difference $\alpha$=0.05).

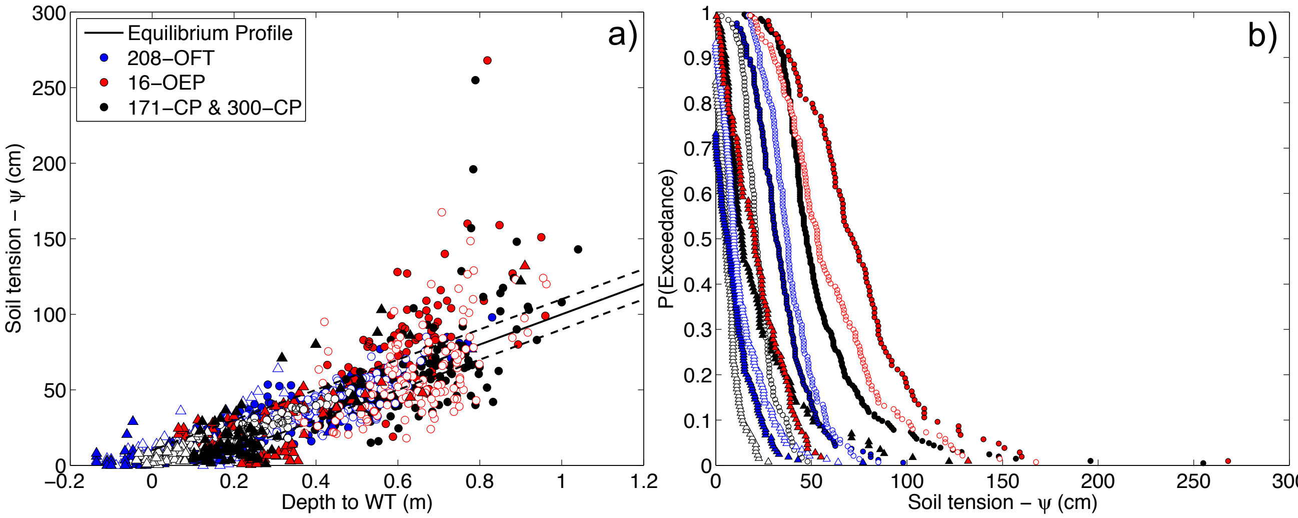


**Figure 4.7.** a) Depth to WT versus soil tension (Ψ) at all sites for the 2012-2014 study period for burn severity groups that had significant WT-Ψ relationships (see Table 4.3). Colors denote hydrogeological setting, open/closed symbols denote burned/unburned, circles denote *Sphagnum fuscum* groups, and triangles denote *Sphagnum angustifolium*/depth of burn >0.05 m groups. The solid line surrounded by the two dashed lines represent the region where hydraulic head differences were < 0.10 m between near-surface peat and the water table. b) The same Ψ data as in panel a, but plotted as exceedance probability distributions.

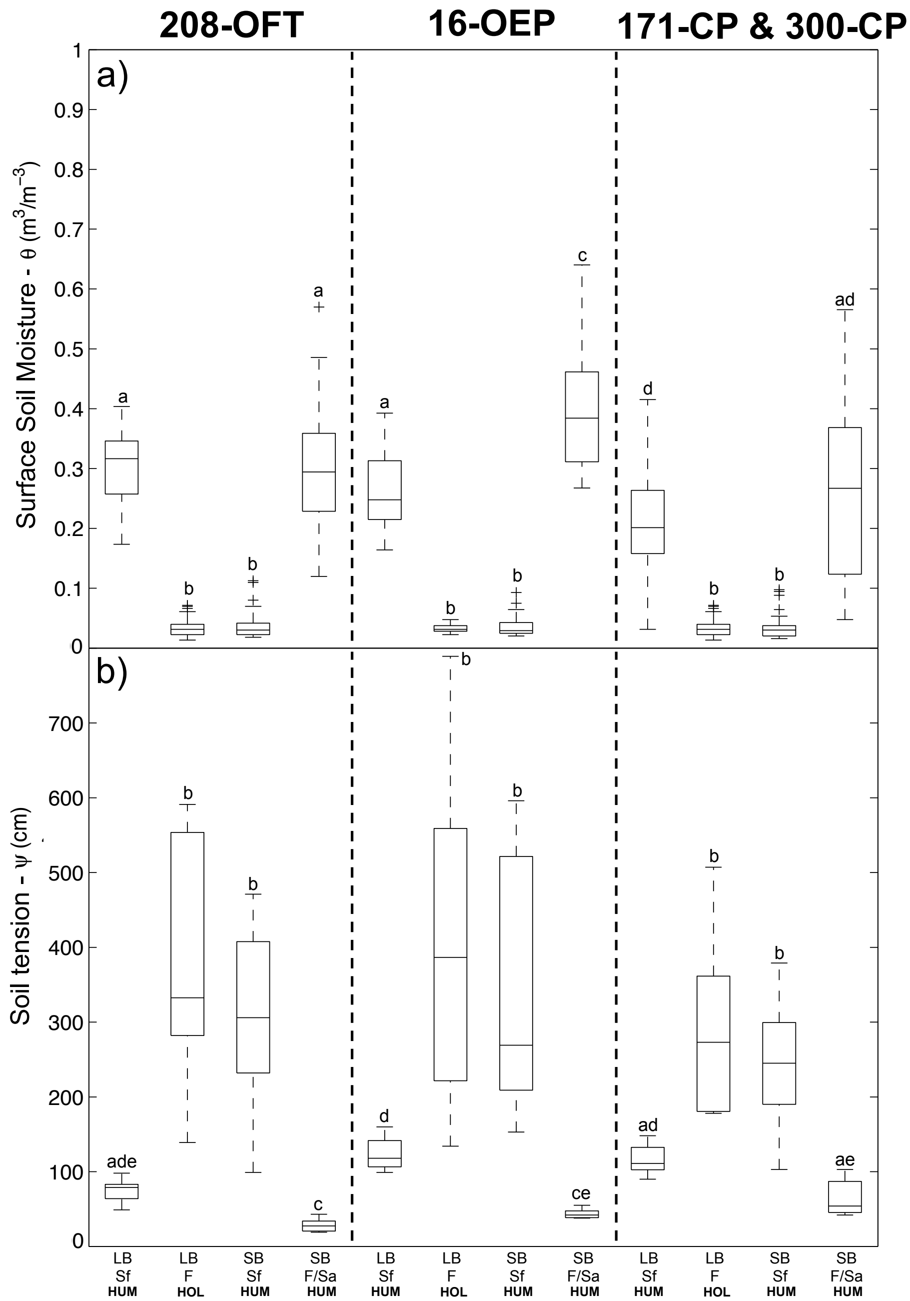


**Figure 4.8.** a) Surface volumetric moisture content (θ, top 0.03 m) and b) soil tension (Ψ) in the burn severity groups (microform also noted) at only the burned sites (see Table 4.2 for nomenclature) collected during the intensive field survey (IFS) on September 1st, 2014. For Ψ, $n$ = 8 per burn severity group, while $n$ = 40 per burn severity group for surface θ. All measurements across the hydrogeological transect were collected within the same four hour period. Ψ and surface θ with the same lowercase letter are not statistically different (least significant difference α=0.05).

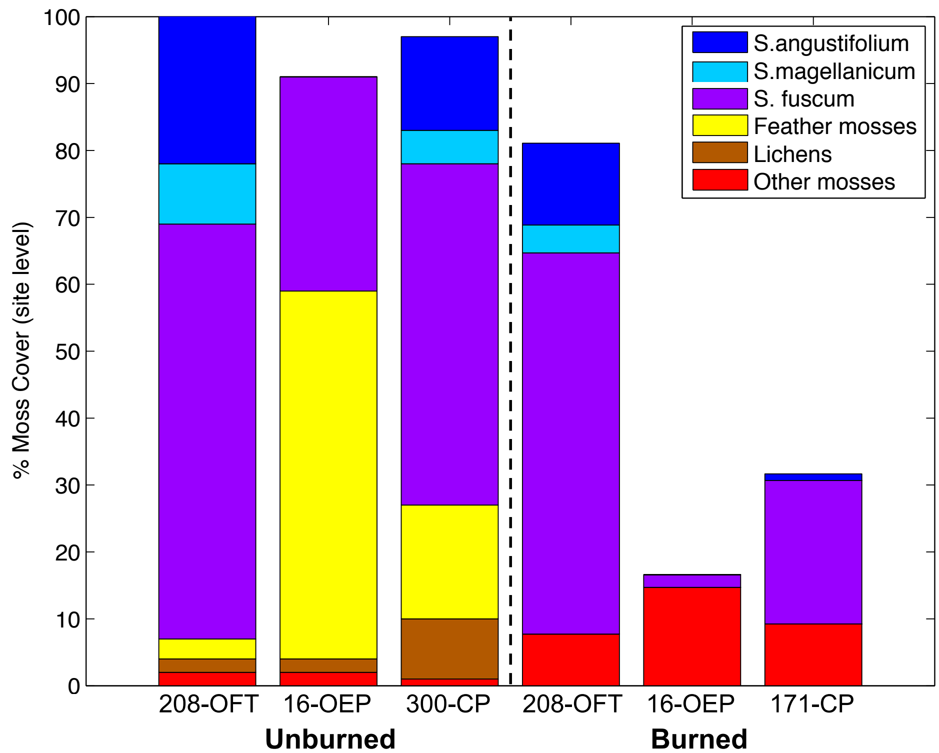


**Figure 4.9.** Percent moss species cover at the burned and unburned sites along the hydrogeological transect scaled to the site level by accounting for the coverage of each burn severity group (see Table 4.1).

# CHAPTER 5: DOES GROUNDWATER CONNECTIVITY CONTROL THE ECOHYDROLOGICAL RESILIENCE OF PEATLAND MARGINS FOLLOWING WILDFIRE?

## 5.1 Abstract

Peatland margins in sub-humid regions of the boreal forest are highly susceptible to deep burning and large carbon losses; however, little is known about how deep burning alters post-fire hydrological conditions that control the recovery of keystone peatland mosses (*i.e., Sphagnum*). With peatland margin burn severity having been shown to be higher in locations isolated from groundwater flow systems due to low water table (WT) positions, we examined if post-fire hydrological conditions (WT, soil tension, and surface moisture content) at peatland margins were suitable for peatland moss recolonization in these same hydrogeological settings. For the majority of our multi year study period, wet conditions resulted in flooding or shallow WTs at peatland margins in all hydrogeological settings, resulting in high post-fire moss water availability. However, an extended dry period induced steep (>1.0 m) WT declines at peatland margins isolated from groundwater flow systems, which limited post-fire moss water availability. We also determined that deep burning exposed dense peat and mineral soil at peatlands margins that, when coupled with dynamic hydrological conditions, created unfavorable conditions for peatland moss re-establishment. Thus, while bryophyte recolonization rates were high at all sites, peatland moss re-establishment was particularly low in peatlands isolated from groundwater flow systems. Moreover, given that this study was carried out during a wet portion of the climate cycle, peatland moss recolonization rates are likely higher than would typically occur. Therefore, we argue that deep burning at the margins of small (< 5

ha) peatlands isolated from groundwater flow systems results in low peatland moss recolonization rates and shifts in ecohydrological function. As such, it is likely that carbon losses at peatland margins are not balanced by contemporary between-fire carbon sequestration.

## 5.2 Introduction

Peatlands, which are wetlands characterized by thick organic deposits greater than 0.40 m in depth (NWWG, 1997), have the highest carbon density per unit area of any ecosystem in the boreal forest (Bradshaw and Warkentin, 2015). Wildfire accounts for greater than 97% of all disturbances in peatlands and burn severity is defined by the depth of peat combustion in these ecosystems (*i.e.,* depth of burn, DOB). While peatland DOB typically ranges from 0.05 to 0.10 m (Benscoter and Wieder, 2003; Lukenbach *et al.*, 2015b; Shetler *et al.*, 2008), deep burning (> 0.20 m) has recently been observed at peatland margins in Alberta's Boreal Plains (BP) (Hokanson *et al.*, 2015; Lukenbach et al., 2015b). Carbon losses at peatland margins ranged from 2 to 85 kg C $m^{-2}$ and can be as high as 50-90% of the total peatland carbon loss, even in large peatlands (> 50 ha) (Hokanson *et al*., 2015). Given that climate change scenarios suggest that increases in evapotranspiration are likely to exceed increases in precipitation in northern latitudes (Collins *et al*., 2013), there is concern that carbon losses at peatland margin 'hot spots' may not be balanced by contemporary between-fire carbon sequestration (Hokanson *et al*., 2015; Kettridge *et al.,* 2015 Lukenbach *et al*., 2015b). Moreover, organic soil losses at peatland margins may have long-lasting impacts on ecohydrological function, whereby shifts in water and solute transfers between peatlands and adjacent mineral upland ecosystems alter ecosystem structure and function (Hokanson *et al.*, 2015).

Deep burning at peatlands margins in the BP is attributable to large WT drawdowns during drought that expose dense peat to low moisture contents (Lukenbach *et al.*, 2015b). Because the frequency of low WTs at peatland margins is influenced by their connectivity to groundwater flow systems (Winter *et al.*, 2003), hydrogeological setting affected where peatland margins experienced deep burning at the landscape-scale (Hokanson *et al.*, 2015). Given that hydrogeological setting also influences post-fire peatland WTs (Chapter 4), it is likely that the same peatland margins that underwent deep burning are also prone to low post-fire WT positions and, accordingly, lags in the post-fire recolonization of peatland mosses. This lower ecohydrological resilience to wildfire could initiate a positive feedback, whereby the lack of peat accumulation alters peatland-upland water and solute transfers (Devito *et al.*, 2012; Winter, 2001) and enables species endemic to uplands to colonize burned peatland margins (Johnstone and Chapin, 2006). As such, there is an immediate need to examine post-fire hydrological conditions at peatland margins in different hydrogeological settings in order to better understand the overall resilience of peatlands to wildfire in the BP.

Following wildfire, high water table (WT) positions are critical to enable the re-establishment of keystone peatland mosses (*i.e., Sphagnum*) (Kettridge *et al.*, 2015). Because hydrogeological setting defines both the mineral substrate composition (*i.e.* texture) and topographic position at a particular position on the landscape, it influences peatland WT dynamics (Winter, 1999) as well as post-fire recovery in peatland ecosystems (Chapter 4). In much of the BP, deep and heterogeneous surficial glacial deposits form a mosaic of fine and coarse-textured mineral substrates on the landscape

(Devito *et al*., 2012), resulting in highly variable groundwater contributions to peatland water balances. In fine-textured hydrogeological settings, groundwater fluxes are small due to the low permeability of mineral substrates (Ferone and Devito, 2004). Here, declines in WT position at peatland margins of up to 1 to 2 m (Ferone and Devito, 2004; Redding and Devito, 2008) occur due to high potential water demand from adjacent mineral upland ecosystems, where snowmelt recharge is variable and actual evapotranspiration is higher than in peatlands (Petrone *et al*., 2007; Brown *et al.,* 2014), resulting in the movement of water from peatlands to mineral uplands (Brown *et al.,* 2014). This strong vegetation control also occurs at peatland margins that are perched or mounded on low permeability sediments above groundwater flow systems near topographic highs in coarse-textured hydrogeological settings (Hokanson *et al*., 2015; Riddell, 2008). In contrast, in low-lying, coarse-textured hydrogeological settings peatland margins exhibit strong groundwater connectivity and less dynamic WT fluctuations that are similar in magnitude to adjacent mineral upland ecosystems (Redding, 2009). Such landscape-scale variability in WT dynamics explained why DOB at peatland margins was lower in coarse-textured, low-lying topographic positions (~0.10 m) when compared fine-grained hydrogeological settings (~0.25 m) and at topographic highs in coarse-textured hydrogeological settings (~0.40 m) (Hokanson *et al*., 2015). Therefore, it is likely that hydrogeological setting also controls post-fire water availability and peatland moss recolonization at peatland margins.

Although hydrogeological setting influences hydrodynamics at peatland margins, the combustion of 0.20 to 1.20 m of peat during deep burning also affects post-fire hydrological conditions (Sherwood *et al*., 2013; Hokanson *et al*., 2015; Lukenbach *et al*., 2015a). Deep burning not only alters the surface datum, typically resulting in shallower post-fire WT positions (Lukenbach *et al*., 2015a), but also exposes dense peat and/or mineral soil at the surface after fire (Lukenbach *et al*., 2015b). This new substrate tends to have a lower specific yield (Sy) (Boelter 1968; Sherwood *et al*., 2013), causing flashy WT responses and dynamic soil moisture conditions that are not conducive to peatland moss recovery (Sherwood *et al.*, 2013; Kettridge *et al*., 2015). Shifts in hydrophysical properties (*i.e.* Sy, bulk density) may explain why moderate drainage (~ 0.20 m lower WT position) in a drained fen followed by deep burning (~ 0.20 m) resulted in a post-fire regime shift to a non-carbon accumulating shrub-grass dominated ecosystem (Kettridge et al., 2015; Sherwood *et al*., 2013). While deep burning at peatland margins may result in similar post-fire hydrological conditions as drained peatlands, it is important to note that peatland margins also alternate between wet and dry states (Devito *et al*., 2012). Periodic wetting likely maintains peatland vegetation communities by limiting the encroachment of species endemic to uplands (Bhatti *et al.*, 2006; Dimitrov *et al.*, 2014). However, high burn severity at peatland margins also has the potential to increase water losses (Lukenbach *et al.*, 2015b) and facilitate the establishment of upland species (Johnstone and Chapin, 2006), thereby drying peatland margins following wildfire. This is important because shifts in hydrological conditions at a peatland's margins will influence overall ecosystem stability by affecting the water balance of a peatland (Devito *et al.*, 2012; Hokanson *et al.*, 2015; Lukenbach *et al.*, 2015b). Thus, there is an immediate need to examine how these wildfire-induced changes at peatland margins interact with hydrogeological setting to affect post-fire hydrological conditions.

In order to better understand the resilience of peatland ecosystems to wildfire, the aim of this study was to better understand how post-fire hydrological conditions varied at peatland margins across the landscape. Here, we present the first inter-annual, multi-site measurements of post-fire hydrological conditions in burned peatland margins that are linked to landscape-scale properties (*i.e.* hydrogeological setting) in Alberta's Boreal Plains. We hypothesized that peatland margins located in hydrogeological settings isolated from groundwater sources would exhibit lower post-fire moss water availability (WT, soil tension ($\Psi$), surface volumetric moisture content ($\theta$)) due to their deeper and more dynamic WTs. Furthermore, we hypothesized that deep burning would expose dense peat at peatlands margins that would interact with hydrogeological setting to further decrease post-fire moss water availability (WT, $\Psi$, surface $\theta$) and limit moss recolonization rates.

## 5.3 Methods

### *5.3.1 Study sites and experimental design*

In May 2011, a ~90,000 ha fire burned a large portion of the Utikuma Research Study Area (URSA; 56.107$^{o}$N 115.561$^{o}$W) in Alberta's Boreal Plains ecozone (Devito *et al.*, 2012, Figure 5.1). The URSA is part of a long-term hydrogeological study that has examined the local and regional hydrology of a number of pond-peatland-upland complexes since 1999. The URSA region is characterized by low topographic relief and deep heterogeneous glacial substrates, such as lacustrine clay plains, fine-textured disintegration moraines, and coarse-textured glaciofluvial outwashes overlying marine shale (Vogwill, 1978; Devito *et al.*, 2012). The climate is sub-humid, with annual potential evapotranspiration (PET) often exceeding annual precipitation (Devito *et al.*, 2012).

Our study was carried out in the same three burned peatland complexes as Hokanson *et al.* (2015), specifically, in lake catchments 16, 208, and 171 along a hydrogeological transect at the URSA (Figure 5.1). However, our study focused on burned bog margins (based on vegetation indicators and pH) because they typically undergo the highest burn severity in their respective hydrogeological settings (Hokanson *et al.*, 2015). Furthermore, we only examined post-fire hydrological conditions at one representative peatland margin in each bog.

In the lake 16 catchment, an ~3 ha ephemerally perched peatland complex (16-OEP, *i.e. 'Outwash, Ephemerally Perched'*) is positioned adjacent to a regional topographic high in a coarse-textured glaciofluvial outwash and is ephemerally connected to both local and intermediate groundwater flow systems between lakes in the region (Smerdon *et al.*, 2005). A small burned bog (~0.5 ha), which is an isolated lobe of the peatland complex, does not receive surface flows (Hokanson *et al.*, 2015). Therefore, even though groundwater is an important component of the water balance, bog-like vegetation is present at the peatland surface because of its isolation from solute rich surface flows and groundwater (Chapter 4).

In the lake 208 catchment, an ~0.5 ha burned kettle hole bog is located on a regional topographic low in the same coarse-textured glaciofluvial outwash as 16-OEP. This outwash bog (208-OFT, *i.e. 'Outwash, Flow Through'*) intersects a large-scale groundwater flow system comprised of several large lakes (~450 – 900 ha) (Hokanson *et al.*, 2015). These larger scale groundwater flows that develop in the coarse material moderate the WT position and minimize WT fluctuations in the bog during drought (Redding, 2009). Furthermore, because oligotrophic groundwater (see electrical conductivity in Table 5.1) represents the vast majority of the water balance at 208-OFT (Redding 2009; Devito *et al.*, 2012) and surface flows do not occur in the bog, species endemic to bogs characterize the vegetation cover (Chapter 4).

In lake catchment 171, an ~4 ha burned bog is located on a lacustrine clay plain (171-CP, *i.e. 'Clay Plain'*) and is the isolated portion of a larger peatland complex that connects to lake 171 (Ferone and Devito, 2004; Hokanson *et al.* 2015). Because of the fine-textured substrate composition of the lacustrine clay plain, this bog receives minimal groundwater fluxes (<5 mm $yr^{-1}$) and no surface flows (Ferone and Devito, 2004). Thus, its water influxes were almost entirely comprised of precipitation. This precipitation-dominated water balance resulted in oligotrophic conditions and the presence of species endemic to bogs (Chapter 4).

The peatland margin was defined as the transitional riparian zone bordering the forested upland (often 8 to 10 m wide). Prior to fire, this zone is characterized by a limited LFH (organic horizon of litter + fibric + hemic material) layer (*i.e.*, little to no transition between the surface litter layer and the underlying humus), a lack of the *Sphagnum* hummock microtopography typically found in the peatland middle, and shallower peat depths than the peatland middle. Dimitrov *et al*. (2014) refers to this zone as a boreal ecotone between forested uplands and peatlands and offers a brief overview of the historic categorization of this zone.

### *5.3.2 Groundwater measurements*

To understand peatland margin hydrological interactions in the context of hydrogeological setting, we took advantage of a long-term monitoring network of wells and piezometers that defined the groundwater flow systems at each site (Ferone and Devito, 2004; Smerdon *et al*., 2005; Redding 2009). Groundwater wells and piezometers along an upland-peatland transect were also installed at each site to characterize peatland margin hydrological conditions. Except for 208-OFT, which did not have a mineral upland installation, these installations were positioned at 1) the peatland, 2) the peatland margin, 3) the mineral margin, and 4) the mineral upland. The peatland groundwater well was positioned at the closest *Sphagnum* hummock-hollow microform to the peatland margin, while the peatland margin was located where residual peat depths were >0.10 m and DOB was >0.05 m. The mineral margin well was located just beyond the edge of the peatland and was characterized by <0.03 m residual organic matter, while the mineral upland well was upslope from the mineral margin and characterized by these same substrate conditions. The distances between the peatland well location and the mineral margin varied little between the sites, highlighting the similar width of the peatland margins at each site (Figure 5.1).

Water levels and electrical conductivities were measured weekly to monthly from May to September in 2013 and 2014 using a temperature-level-conductivity (TLC) meter (Solinst, Georgetown, Ontario, Canada) in 0.05 m diameter PolyVinyl Chloride (PVC) wells and 0.025 m PolyVinyl Chloride (PVC) piezometers. In select wells, capacitance water level recorders (Odyssey Data Recording, Christchurch, New Zealand) or pressure transducers (Solinst, Georgetown, Canada) continuously recorded depth to WT at one-hour intervals in 0.05 m diameter PolyVinyl Chloride (PVC) wells. Hydraulic heads along each transect were determined by pairing water level measurements with survey data of the wells and piezometers from a real-time kinematic GNSS differential GPS (Trimble R8; accuracy +/- 0.015 m). This survey data was also used to calculate the proportion of flooding between the peatland and the mineral margin well installations. Saturated hydraulic conductivities were measured in piezometers using the Hvorslev (1951) method.

*5.3.3 Post-fire moss water availability measurements*

At each site, Ψ was measured one to three times per week from May – September in 2013 and 2014 at a depth of 0.05 m in three peatland margin (residual peat depths > 0.10 m) locations using 0.02 m outside diameter tensiometers (Soil Measurement Systems, Tucson, Arizona, USA) and a UMS (Munich, Germany) Infield tensimeter, accurate to ± 2 cm. In 2014, supplemental measurements of surface θ (top 0.03 m) were concurrently measured with Ψ using a Thetaprobe (Delta-T Devices, Burwell, Cambridge, UK) directly above each tensiometer cup. Measurements were calibrated following the approach of Kasischke *et al.* (2009). Depth to WT at locations where tensiometers were installed was determined by measuring the water level in wells adjacent to tensiometers.

*5.3.4 Intensive field survey*

*Spatial measurements of post-fire moss water* availability: Temporal hydrological measurements were supplemented by an intensive field survey (IFS) to further assess the spatial variability of post-fire moss water availability across hydrogeological settings as well as examine the impact of an extended dry period on post-fire moss water availability. Water stress was anticipated to be highest in early September due to late summer drying that typically occurs in this sub-humid and continental climate (Lukenbach *et al*., 2015a). During the IFS, a total of eight tensiometers were installed in peatland margins at each site at a depth of 0.05 m. Of note, the same locations where tensiometers were temporally measured during 2013 and 2014 were included in this sampling. At all the sites, tensiometers were measured on September 1$^{st}$, 2014, five days after installation, and all data were collected within a 4-hour period. Surface θ (top 0.03 m) was paired with Ψ measurements and 32 additional measurements of surface θ were taken in peatland margins at each site (*n* = 40 per site). This sampling was paired with DOB measurements, which were approximated by measuring the difference in surface elevation between burned areas and surrounding unconsumed areas in a manner similar to previous studies (Kasischke *et al.*, 2008; Lukenbach *et al*., 2015b; Mack *et al.*, 2011). Notably, DOB was >0.05 m at >95% of the margin at both 16-OEP and 171-CP, while ~26% of 208-OFT's margin was covered by lightly burned feather mosses (DOB < 0.03 m). Because a number of previous studies have shown that lightly burned feather mosses limit post-fire moss recolonization (*c.f.* Lukenbach *et al.*, 2015a; Chapter 4) due to their water repellent nature (Kettridge *et al.*, 2014), they were not sampled in this study.

*Moss and bryophyte recolonization measurements*: Hydrological measurements during the IFS were paired with measurements of % bryophyte species cover and % *Sphagnum* cover, which were visually estimated in 0.10 x 0.10 m plots above each tensiometer cup and surface θ sampling location to examine the association with bryophyte recovery. Because the fire resulted in the complete mortality of ground layer vegetation, the observations of % bryophyte species and *Sphagnum* cover are a direct measure of the recolonization following the fire.

*5.3.5 Peat hydrophysical properties*

At each site, five surface peat cores (0.05 m in length) were extracted randomly at the margins where residual peat depths were >0.05 m. We determined the moisture retention curve of each 0.05 m peat 'puck' using saturated porous plates (Soil Moisture Equipment Corp.) with an air entry pressure of 1000 cm (1 cm = 1 mb). We conducted our measurements inside sealed acrylic chambers in which we maintained a high relative humidity to minimize evaporative losses from the samples. We subjected the peat 'pucks' to several constant tensions (10, 20, 30, 40, 50, 75, 100, 150 and 200, and 500 cm), each for 24 hours or until water outflow from the pressure plate had ceased, whichever was longer, so as to ensure pressure equilibration. Following all moisture retention measurements, we dried the peat 'pucks' at 85 °C until no change in the sample weight was observed to calculate dry bulk density. We estimated water content at saturation ($\Psi$=0 cm), equal to porosity, from the measured bulk density, and assuming a particle density value of 1.47 g $cm^3$ (Redding and Devito, 2006).

*5.3.6 Analyses*

Due to the challenge of instrumenting a large number of sites, only one site was instrumented in each hydrogeological setting. As such, we cautiously interpreted the results of our statistical analyses because the research design was pseudoreplicted (c.f. Hurlbert, 1984). Because residuals were not normally distributed and there were unequal variances in the data collected, $\Psi$ and was natural logarithm transformed and surface $\theta$

was converted to a % and square root transformed prior to statistical analyses. For the temporal measurements, repeated measures ANOVA was used to account for multiple measurements in the same Ψ/surface θ plot throughout the study period (*n* ranged from 10 – 15 in each plot $yr^{-1}$) and tested the effect of site (3 levels) and year (2 levels, nested within site) on Ψ and surface θ. Data from the IFS were analyzed using a one-way ANOVA of site and Tukey's HSD post-hoc test was used to examine differences between sites.

Robust ordinary least squares regression (OLS) was employed to assess the control of WT on Ψ. Where relationships were found to be statistically significant, the amount of time a hydraulic head difference of <0.10 m existed between near-surface peat (0.05 m depth) and the water table was calculated (*c.f.,* Thompson and Waddington, 2013b). This condition implies that evaporative water losses from the moss surface are rapidly replenished by water fluxes from the water table because the unsaturated hydraulic conductivity of shallow peat is high under low soil tensions (Price *et al.*, 2008). As such, this condition is indicative of moss water stress because the moisture content in near-surface moss layers is does not appreciably decline when the hydraulic head difference is <0.10 m. Because only eight tensiometers were installed during the IFS, data was pooled across sites, yielding 24 measurements for the analysis of WT-Ψ relationships in peatland margins.

Bulk density and gravimetric water contents (GWC) obtained from the moisture retention analyses were natural logarithm transformed prior to statistical analyses. A one-way ANOVA of site and Tukey's HSD post-hoc test were used to examine differences in bulk density and GWC between sites. Robust OLS was used to examine the relationship between bulk density and θ at different pressure steps from the peat moisture retention measurements.

## 5.4 Results

### *5.4.1 Post-fire peatland margin hydrology*

The study was carried out during a wet period of Alberta's BP climate cycle (Environment Canada, 2000) and was preceded by above average precipitation in 2011 and 2012 and one of the largest snow melts in the past 15 years in spring 2013. The sampling period from May to September 2013 and during the fall of 2013 was near historical averages, while this was followed by a higher than average snow melt in spring of 2014. The sampling period from May to September 2014 was characterized by 50% less precipitation than the historical average, resulting in continuous drying until the end of the study period.

Flow direction along upland-peatland transects varied between hydrogeological settings and was dependent on site wetness (Figures 5.1 and 5.2). At 208-OFT, the mineral substrate was thick, coarse-textured sand exhibiting high hydraulic conductivities (Table 5.1, Figure 5.1a). Hydraulic heads did not typically vary by more than 0.05 m along the upland-peatland transect and there was a flat WT between the middle of the peatland and its margin (Figures 5.1a, 5.2a, 5.2b). Furthermore, hydraulic heads along the upland-peatland transect were similar to the water level in the regional groundwater flow system (Figure 5.2a, 5.2b), highlighting the peatland's intersection with a large groundwater flow system. Although lateral hydraulic gradients were not detectable, a vertical head gradient was present in the peatland middle (0.05 ± 0.01) and mineral margin (0.01 ± 0.003), indicating recharge to the underlying aquifer.

In contrast to 208-OFT, horizontal hydraulic head differences (Figures 5.2c, 5.2d) were greater than vertical hydraulic head differences (0.02 ± 0.01) in the peatland at 16-OEP. The mineral substrate primarily consisted of low hydraulic conductivity silts beneath the peatland (Table 5.1), while mineral uplands exhibited fine sands overlying a siltier substrate (Figure 5.1b). Overall, the mineral substrates surrounding the peatland were finer textured than the rest of the groundwater flow system at catchment 16 (Smerdon *et al*., 2005). Consequently, the peatland and its margin were mounded on a thick silt lens above the coarse-textured regional groundwater flow system west of the lake (Figure 5.1b, see Smerdon *et al*., 2005 for details). Along the upland-peatland transect, water movement was consistently from the peatland to the mineral upland throughout the study period, although very brief flow reversals occurred during precipitation events (Figures 5.2c, 5.2d).

Due to the fine-textured nature of the substrate at lake catchment 171, regional groundwater interactions were limited by low hydraulic conductivities (Table 5.1) and a vertical recharge gradient (0.02 ± 0.01) to the underlying mineral substrate was present in the peatland. Thus, local groundwater exchanges defined post-fire hydrological interactions along the upland-peatland transect at 171-CP (Figure 5.1c). Throughout the entire study period, the hydraulic head in the peatland was higher than the peatland margin and mineral margin (Figures 5.2e, 5.2f). During wet periods, a groundwater mound developed in the mineral upland, indicating that groundwater fluxes were focused to the peatland margin from the mineral upland and peatland (Figure 5.1c). As drying occurred, this groundwater mound receded and water moved from the peatland margin to the mineral upland (Figures 5.1c, 5.2f). Further drying at the end of the study period in 2014 resulted in a water level decline at the mineral margin well below that of the peatland margin, coinciding with substantial margin WT declines (Figure 5.2f).

In general, electrical conductivities along the upland-peatland transects decreased moving from the peatland to the mineral upland. Electrical conductivities at 16-OEP and 171-CP were lower at the peatland margins than the mineral margins, as solute poor water from the peatland migrated to this region (Table 5.1). In contrast, electrical conductivities were lower in the mineral upland than the peatland at 208-OFT, although these absolute differences were small (Table 5.1).

In 2013, WTs at all sites peaked in June during large rain events (Figure 5.3a). In 2014, WTs at all sites peaked following snowmelt and declined throughout the rest of the summer (Figure 5.3b). Peat margin WTs were similar between hydrogeological settings throughout 2013; however, the WT at 208-OFT was substantially shallower than the other sites during 2014. Steep WT declines (> 1.0 m) were observed at 16-OEP and 171-CP near the end of the study period in 2014 (Figure 5.3b). Flooding frequently occurred at the margins of 16-OEP and 171-CP in 2013 and at the beginning of 2014 due to wet conditions coupled with deeper burning at these sites (Figure 5.4). However, the proportion of the margin that was flooded rapidly decreased at these sites during an extended dry period in 2014 (Figure 5.4b). In contrast, flooding occurred less frequently at 208-OFT due to shallower burn depths and regional groundwater control of the WT along the upland-peatland transect.

*5.4.2 Temporal measurements of post-fire moss water availability*

Because peat margin WTs were similar between hydrogeological settings for the majority of 2013 and 2014, Ψ did not vary across sites ($F_{2, 240}$ = 1.5, $p$ = 0.26) or between years ($F_{3, 240}$ = 1.82, $p$ = 0.20) (Figure 5.3c, 5.3d). However, plot had a significant effect on Ψ ($F_{12, 240}$ = 12.93, $p$ < 0.001) (Figure 5.3c, 5.3d). Similarly, in 2014, surface θ was significantly different between plots ($F_{6, 87}$ = 5.74, $p$ < 0.001), but not across sites ($F_{2, 87}$ = 3.19, $p$ = 0.11). At all the sites, WT was a strong predictor of Ψ in 2013 (Figure 5.5a, $p$< 0.001, $r^2$ = 0.89, 0.94, 0.54 at 208-OFT, 16-OEP, and 171-CP, respectively and at 208-OFT ($p$< 0.001, $r^2$ = 0.88) and 16-OEP ($p$< 0.001, $r^2$ = 0.84) in 2014 (Figure 5.5b). Furthermore, in 2013 a hydraulic head difference of <0.10 m between near-surface peat and the water table was present 100%, 98%, and 98% of the time at the peat margins of 208-OFT, 16-OEP, and 171-CP, respectively (Figure 5.5a), while in 2014 they were present 90% and 94%, of the time at the peat margins of 208-OFT and 16-OEP, respectively (Figure 5.5b).

### *5.4.3 Intensive field survey*

*Spatial measurements of post-fire moss water availability*: In locations where water availability was sampled, DOB averaged 0.07 ± 0.01 m, 0.26 ± 0.02 m, and 0.16 ± 0.01 m at 208-OFT, 16-OEP, and 171-CP. The IFS exhibited similar trends as the continuous measurements of post-fire moss water availability; however, because sampling was carried out during an extended dry period, Ψ values were generally higher and surface θ values were lower than during 2013 and the rest of 2014 (Figure 5.6). Because the site response to drying at the end of 2014 varied between the sites (see section 3.1), there was a significant effect of site on Ψ ($F_{2, 21}$ = 25.1, $p$ < 0.001) and surface θ ($F_{2, 21}$ = 4.38, $p$ < 0.001). Post-hoc test results for Ψ and surface θ are shown in Figure 5.6a and 5.6b, respectively.

Where tensiometers were installed, WTs averaged 0.42 ± 0.02 m, 0.72 ± 0.08 m, and 0.94 ± 0.05 m at 208-OFT, 16-OEP, and 171-CP, respectively, during the IFS (Figure 5.5b). After pooling WT and Ψ measurements across the sites, robust OLS indicated that WT was a significant predictor of Ψ (Figure 5.5b, $p$ < 0.001, $r^2$ = 0.67). A hydraulic head difference of <0.10 m between near-surface peat and the water table was present 100%, 100%, and 50% at the peat margins of 208-OFT, 16-OEP, and 171-CP, respectively. Thus, the primary reason for increased Ψ during the IFS was the occurrence of deeper WTs.

*Moss and bryophyte recolonization*: Across all sites, high post-fire water availability corresponded to high post-fire % bryophyte recolonization (Table 5.2). Across hydrogeological settings, peatland margins exhibited similar total recolonization rates; however, species compositions varied between sites. Notably, *Sphagnum* spp. were only observed at the peat margins of 208-OFT (Table 5.2).

### *5.4.4 Peat hydrophysical properties*

Surface (top 0.05 m) bulk densities at peat margins varied between sites ($F_{2, 12}$ = 15.6, $p$ < 0.001). Tukey's HSD post-hoc test indicated that peatland margins at 208-OFT (44.5 ± 7.71 kg m$^{-3}$, $p$ < 0.05) were significantly less dense than peatland margins at 171-CP (93.9 ± 5.29 kg m$^{-3}$) and 16-OEP (90.3 ± 8.47 kg m$^{-3}$). By pooling bulk density data across sites and applying a robust ordinary least squares (OLS) regression, bulk density explained a large portion of the variation in VWC moisture retention at all Ψ steps except 10 cm, with $r^2$ values ranging from 0.47 to 0.74 (p < 0.001, $n$ = 15, min and max $r^2$ at *Ψ*

= 20 cm and 200 cm, respectively). Measured volumetric water contents (VWC) and gravimetric water contents (GWC) obtained from the peat moisture retention analyses followed similar trends as bulk density between sites. For VWC, these differences were only statistically significant at the 150, 200, and 500 cm pressure steps ($F_{2, 12}$ = >3.94, $p <$ 0.05) and only between peatland margins at 208-OFT and 171-CP ($p < 0.05$, Figure 5.7a). In contrast, gravimetric water contents (GWC) were significantly higher at 208-OFT than 171-CP and 16-OEP for all pressure steps ($F_{2, 12}$ = >16.2, $p < 0.001$, Figure 5.7b).

## 5.5 Discussion

### *5.5.1 Hydrogeological controls on post-fire peatland margin hydrology and water availability*

Although peatland margin WTs were similar between the sites for the majority of the study period, drying at the end of the study period in 2014 effectively highlighted the differences in site-level controls on WT dynamics. At 208-OFT, the peatland and its margins intersected a large groundwater flow system, resulting in less dynamic WTs and minimal WT drawdowns. In contrast, 16-OEP and 171-CP experienced substantial WT declines at their margins at the end of 2014 because they only received local groundwater contributions (*i.e.* within peatland). Furthermore, due to the fine-textured mineral substrates and rapid recovery of mineral upland vegetation at these same sites (Depante, unpublished data), evapotranspiration in mineral uplands likely resulted in larger soil water storage changes and, subsequently, greater WT declines.

Through its influence on peatland margin WTs, hydrogeological setting influenced post-fire moss water availability, as evidenced by the strong relationship between depth to WT and Ψ. With the exception of the IFS, peatland margin post-fire moss water availability was high and similar across hydrogeological settings due to shallow WT positions. Nevertheless, it is likely that post-fire moss water availability and depth to WT at peatland margins were higher than average because this study was carried out during a wet portion of the climate cycle.

The significant control of peatland margin WTs on post-fire water availability (*i.e.* Ψ) was likely stronger than it was prior to fire (Lukenbach *et al*., 2015a) because DOB >0.05m typically increases WT-Ψ relationships by removing water repellent and low moisture feather moss peat (Lukenbach *et al*., 2015a; Chapter 4). High DOB at 171-CP and 16-OEP not only increased connectivity to the WT but also lowered the surface datum at peatland margins, thereby decreasing depth to WT and increasing the frequency of flooding (Figures 5.1, 5.4). While differences in DOB between peatland margins across hydrogeological settings minimized differences in Ψ and surface θ between sites, deep burning at 171-CP and 16-OEP exposed mineral soil and dense surface peat at peatland margins (Figure 5.7). This likely accounted for the similarity of surface θ between sites, whereby denser peat at 16-OEP and 171-CP retained more water on a volumetric basis than 208-OFT under high Ψ (Boelter, 1968). Furthermore, high Ψ at 171-CP during the IFS was likely attributable to steep WT declines coupled with deep burning that altered surface soil physical properties, as both dense peats (Sherwood *et al.*, 2013) and fine-textured mineral soils (Hillel, 1998) have the potential to lower water availability, especially during dry periods.

*5.5.2 Controls on post-fire peatland margin moss recolonization and its implications for peatland carbon balances*

High water availability corresponded to high bryophyte recolonization rates and, notably, total bryophyte recolonization rates were higher in the first three years following wildfire compared to previous observations in peatlands (Benscoter and Vitt, 2008). However, the trajectory of recovery at peatland margins was distinctly different from that occurring in peatland middles (Chapter 4) and varied across hydrogeological settings (Table 5.2). In particular, the most common species (*i.e. Marchantia* polymorpha and *Ceratadon purpureus*) recolonizing peatland margins at 171-CP and 16-OEP are characteristic of post-fire recovery in mineral upland areas (Bradbury, 2006), while the primary species recolonizing peatland margins at 208-OFT (*i.e. Polytricum strictum*) were similar to those observed in peatland middles (Benscoter and Vitt, 2008). Moreover, *Sphagnum* mosses had rapidly recolonized some peatland margin locations at 208-OFT, while no *Sphagnum* mosses were observed at 16-OEP and 171-CP.

Variability in peatland moss recolonization between hydrogeological settings was likely influenced by variability in WT dynamics between peatland margins across hydrogeological settings. While the peatland margin WT at 208-OFT was moderated by groundwater flow, dynamic hydrological conditions were present at the peatland margins of 16-OEP and 171-OEP, characterized by flooding during wet conditions and rapid WT drawdowns during dry periods. Moreover, flooding may not only have created conditions that were too wet for peatland moss recolonization but may have also resulted in an influx of solute rich water that has been shown to inhibit *Sphagnum* moss success (Granath *et al*., 2010). This may further explain why *Sphagnum* moss recolonization was higher at 208-OFT than 16-OEP and 171-CP. Specifically, although 208-OFT was located in a groundwater flow system, electrical conductivities at the site suggest that the groundwater had low solute concentrations. In contrast, fine-textured mineral uplands at 16-OEP and 171-CP likely had solute rich groundwater that migrated towards the peatland during wet periods. This coupled with steep WT declines during dry periods may explain why 16-OEP and 171-CP exhibited no *Sphagnum* moss and little peatland moss recolonization despite having high WT positions for the majority of the study period. Future studies should examine how post-fire solute/nutrient availability and competition affect post-fire recovery at peatland margins.

Because this study was carried out during a wet portion of the climate cycle, peatland moss recolonization rates were likely higher than would typically occur in the first three years following wildfire. Furthermore, steep peatland margin WT (> 1.0 m) declines that frequently occur during dry periods in small peatlands isolated from groundwater flow systems (*i.e.* 16-OEP and 171-CP) indicate that *Sphagnum* mosses will have difficulty recolonizing in the long-term (Devito *et al.*, 2012). Given that the fire return interval for peatlands in this region is estimated at 120 years (Turetsky *et al.*, 2004) and that small peatlands isolated from groundwater flow systems also experienced the highest DOB (Hokanson *et al*., 2015), legacy carbon lost during deep burning is unlikely to be balanced by contemporary between-fire carbon sequestration (Wieder *et al.*, 2009).

### *5.5.3 Post-fire ecohydrological conditions at peatland margins: implications for peatland resilience to wildfire*

Although deep burning was prevalent at peatland margins at 171-CP and 16-OEP, organic soil losses do not appear to have altered the fundamental hydrological interaction between peatlands and adjacent uplands in each hydrogeological setting in the BP (Devito *et al.*, 2012; Ferone and Devito, 2004; Redding, 2009). Specifically, peatland groundwater mounds were present both before and after wildfire at 171-CP and 16-OEP and water moved from peatland into adjacent mineral uplands during dry periods (Ferone and Devito, 2004). Moreover, a hydraulic gradient between the peatland and mineral upland at 208-OFT was difficult to detect, highlighting that the WT was primarily controlled by the regional groundwater flow system (Redding, 2009). Therefore, it is likely that climate and hydrogeological setting remained dominant controls on peatland margin hydrology after wildfire, especially in peatlands that intersect groundwater flow systems.

While landscape-scale controls were the primary factors affecting upland-peatland water exchange, deep burning may have altered peatland margin hydrology in sites isolated from groundwater flow systems (*i.e.,* 16-OEP and 171-CP). In particular, organic soil losses of >0.20 m not only have the potential to increase lateral groundwater losses by combusting low hydraulic conductivity margin peat (Lukenbach *et al*., 2015b) but may also increase water losses by exposing dense peat and mineral soil that have low Sy. Specifically, low Sy substrates are associated with increased WT fluctuation (Price and Schlotzhauer, 1999; Thompson and Waddington, 2013a) that, when coupled with the potential for higher evaporation at peatland margins due to shallower WT positions (Kettridge and Waddington, 2014), may increase hydraulic gradients from the middle to the margin of the peatland. Therefore, increased water losses due to deep burning have the potential to shift the peatland to a drier hydrological regime. This is especially true for small peatlands (<5 ha) isolated from groundwater flow systems, where total peatland water storage is more easily influenced by changes in peatland margin hydrology due to their lower perimeter to area ratios (Riddell, 2008).

Although flooding may have limited the encroachment of upland species into peatland margins, it was not associated with increases in peatland moss recolonization at 171-CP and 16-OEP (section 4.2). Furthermore, flooding was followed by rapid WT drawdowns during dry periods. Therefore, upland species may successfully encroach into peatland margins during dry, or even normal, periods of the climate cycle. Given that aspen (*Populus tremuloides*) accounts for the majority of transpiration in fine-textured mineral uplands in the BP (Brown *et al.*, 2014) and can rapidly recolonize severely burned organic and mineral substrates (Johnstone and Chapin, 2006), this may further enhance peatland margin WT drawdowns. Coupled with the aforementioned changes in soil physical properties, small peatlands isolated from groundwater flow systems may undergo a positive feedback loop whereby post-fire WT drawdowns following deep burning increase drying at the peatland and its margins. Furthermore, given that climate change is likely to increase total wildfire area burned (Flannigan *et al.*, 2005) and organic layer burn severity (Kettridge *et al.*, 2015; Turetsky *et al.,* 2011) in the boreal forest, deep burning at peatland margins may alter the ecohydrological function of peatland margins and facilitate a regime shift from peatland to mineral upland, resulting in the lateral shrinkage peatlands in the BP. Future research should continue to evaluate the impact of deep burning at peatland margins on the resilience of peatlands to wildfire, especially in small peatlands isolated from groundwater flow systems.

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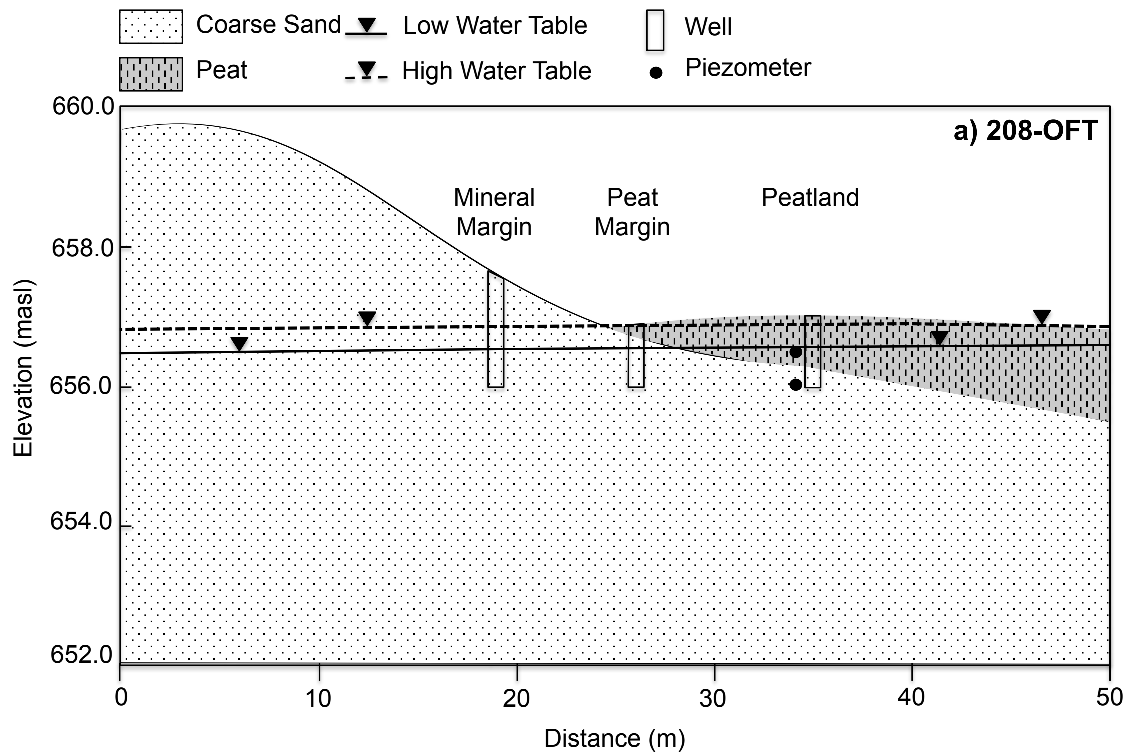
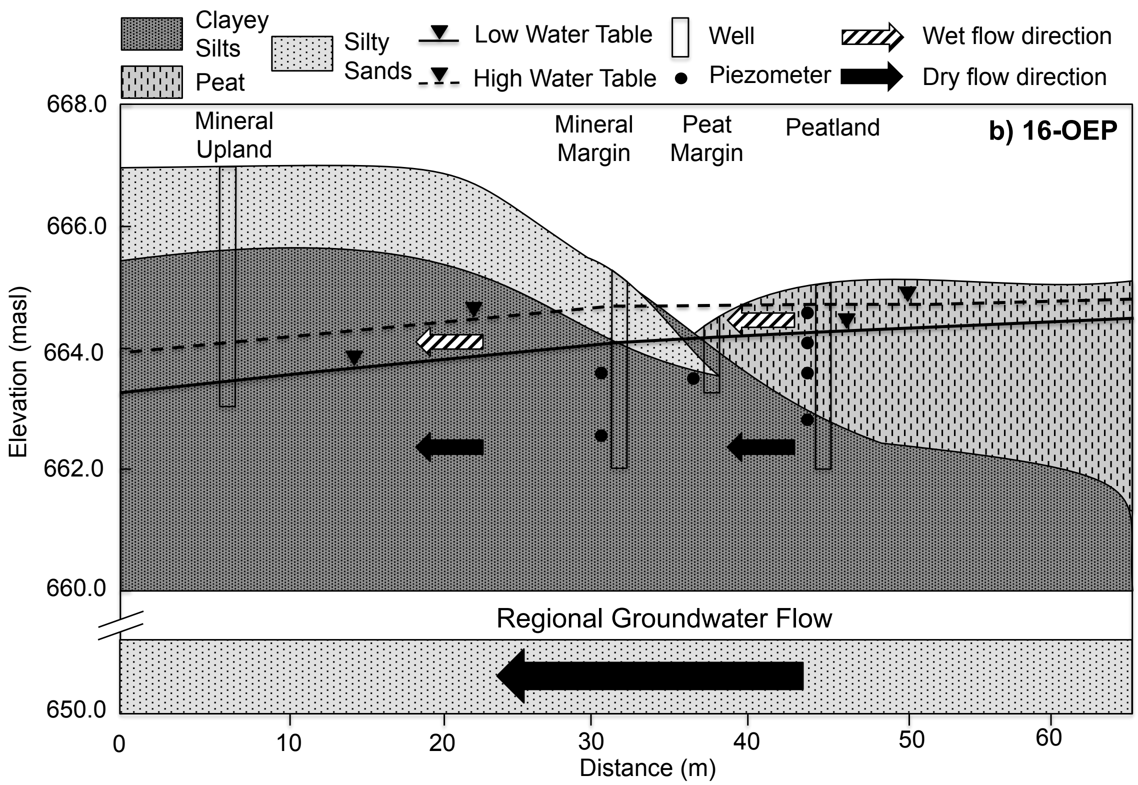
**Table 5.1.** Hydrological variables along the upland-peatland transect at each site. Abbreviations for locations are as follows: P (peatland), PM (peatland margin), MM (mineral margin), and MU (mineral upland). Abbreviations for variables are as follows: EC (electrical conductivity), Ksat (hydraulic conductivity), and DOB (depth of burn). Means and standard errors (in parentheses) are reported. Electrical conductivity was measured in shallow wells.

| | **Site** | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | **208-OFT** | | | | **16-OEP** | | | | **171-CP** | | | |
| | **P** | **PM** | **MM** | **MU** | **P** | **PM** | **MM** | **MU** | **P** | **PM** | **MM** | **MU** |
| Shallow Well EC (μS $cm^{-1}$) | 54 (3) | 40 (3) | 22 (1) | - | 77 (11) | 94 (8) | 113 (7) | 986 (80) | 91 (5) | 104 (9) | 959 (56) | 3100 (249) |
| Shallow Peat Ksat (< 1.0 m) | $10^{-4}$ | $10^{-4}$ | - | - | $10^{-5}$ | $10^{-6}$ | - | - | $10^{-5}$ | $10^{-7}$ | - | - |
| Deep Peat Ksat (> 1.0 m) | $10^{-6}$ | - | - | - | $10^{-5}$ | - | - | - | $10^{-6}$ | - | - | - |
| Mineral Ksat (m $s^{-1}$) | $10^{-3}$ | $10^{-3}$ | $10^{-3}$ | - | $10^{-8}$ | $10^{-8}$ | $10^{-8}$ | - | $10^{-8}$ | $10^{-8}$ | $10^{-8}$ | - |
| DOB (cm) | 4 (1) | 7 (1) | - | - | 6 (1) | 26 (2) | - | - | 4 (1) | 16 (1) | - | - |
| Peat Depth (m) | 0.65 | 0.15 | - | - | 2.4 | 0.35 | - | - | 0.9 | 0.6 | - | - |

**Table 5.2.** Depth of burn (DOB) at the margin of each peatland and percent recolonization of each species three years after fire (*n*=25) where tensiometers and surface moisture was measured during the IFS. Means are reported and standard errors are shown in parentheses.

| | | **Percent Cover** | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Site** | **DOB (cm)** | *Ceratodon purpureus* | *Polytricum strictum* | *Polytricum juniperium* | *Aulocomnium palustre* | *Funaria hygrometrica* | *Pohlia nutans* | *Marchantia polymorpha* | *Sphagnum angustifolium* | *Sphagnum magellanicum* | *Sphagnum fuscum* |
| 208-OFT | 7 (1) | 8 (3) | 36 (7) | 0 | 0 | 0 | 1 (1) | 0 | 7 (3) | 5 (4) | 1 (1) |
| 16-OEP | 26 (2) | 13 (5) | 1 (1) | 10 (4) | 1 (1) | 3 (2) | 1 (1) | 59 (7) | 0 | 0 | 0 |
| 171-CP | 16 (1) | 34 (7) | 15 (6) | 2 (2) | 12 (5) | 3 (3) | 1 (1) | 2 (1) | 0 | 0 | 0 |


Coarse Sand
Low Water Table
Well
Peat
High Water Table
Piezometer
a) 208-OFT
Mineral Margin
Peat Margin
Peatland
660.0
658.0
656.0
654.0
652.0
Elevation (masl)
0
10
20
30
40
50
Distance (m)


Clayey Silts
Peat
Silty Sands
Low Water Table
High Water Table
Well
Piezometer
Wet flow direction
Dry flow direction
Mineral Upland
Mineral Margin
Peat Margin
Peatland
b) 16-OEP
Elevation (masl)
668.0
666.0
664.0
662.0
660.0
650.0
Regional Groundwater Flow
0
10
20
30
40
50
60
Distance (m)

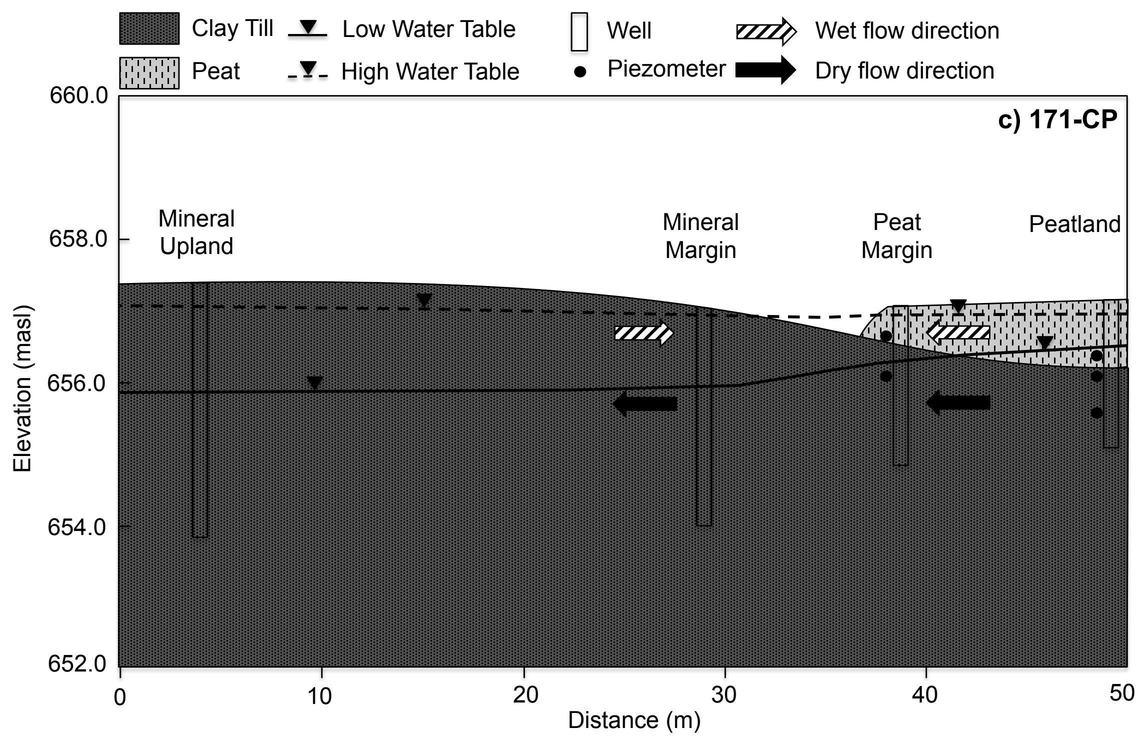



**Figure 5.1.** a) Cross section of 208-OFT showing that the site intersected a regional groundwater flow system that has a flow direction perpendicular to the cross section shown, b) cross section of 16-OEP showing that the site was mounded above the regional groundwater flow system, and c) cross section of 171-CP showing that the site was positioned on clay till, thus there was little interaction with regional groundwater flow. Within all the peatlands, vertical hydraulic gradients indicated recharge to the underlying groundwater flow systems.

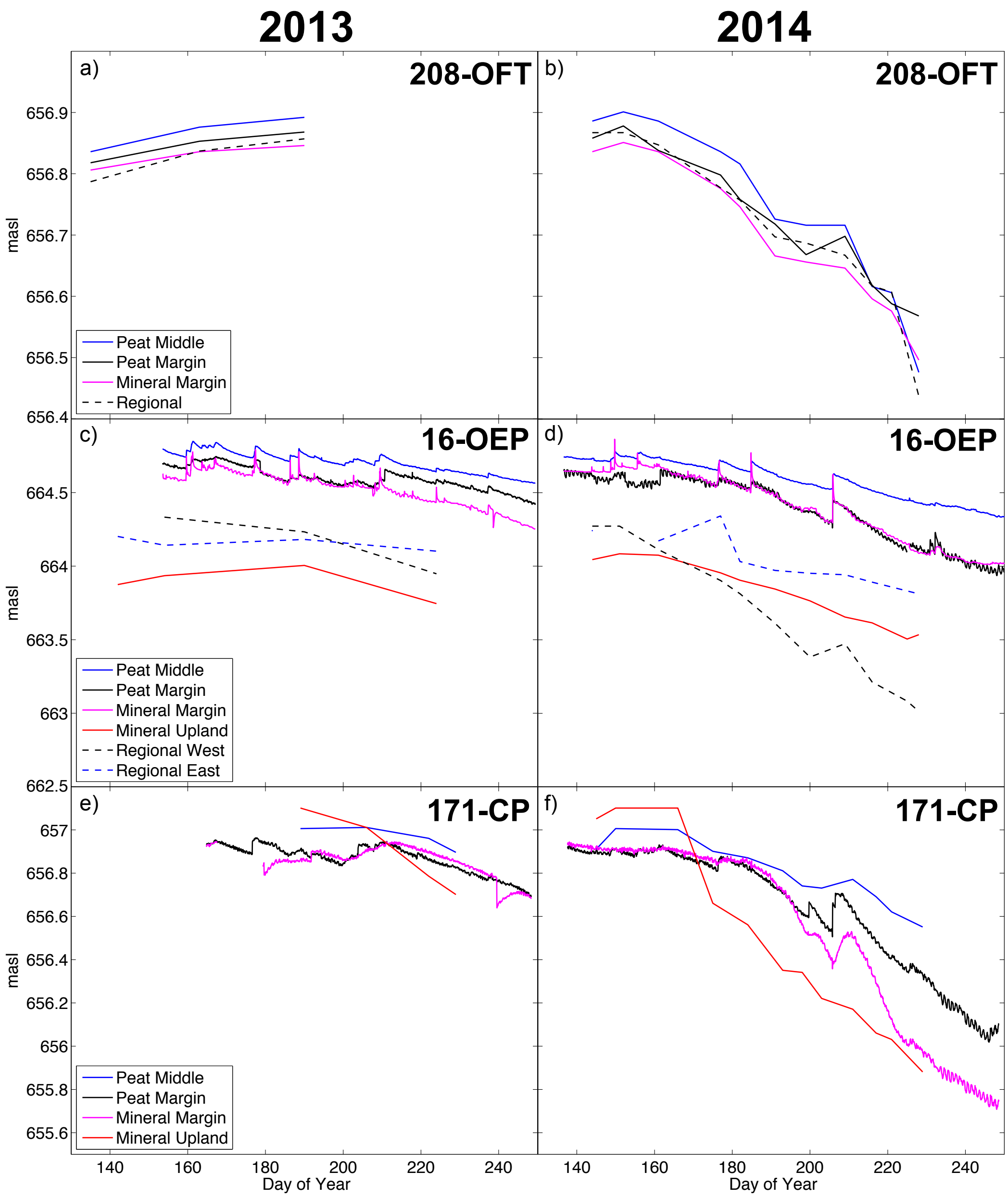



**Figure 5.2.** Water levels at 208-OFT (a &b), 16-OEP (c & d), and 171-CP (e & f) in the middle of the peatland, peatland margin, mineral margin, and mineral upland in 2013 and 2014. For 208-OFT and 16-OEP, regional groundwater water levels are shown to highlight the connectivity of these peatlands to groundwater flow systems. Note different y-axis scales between sites.

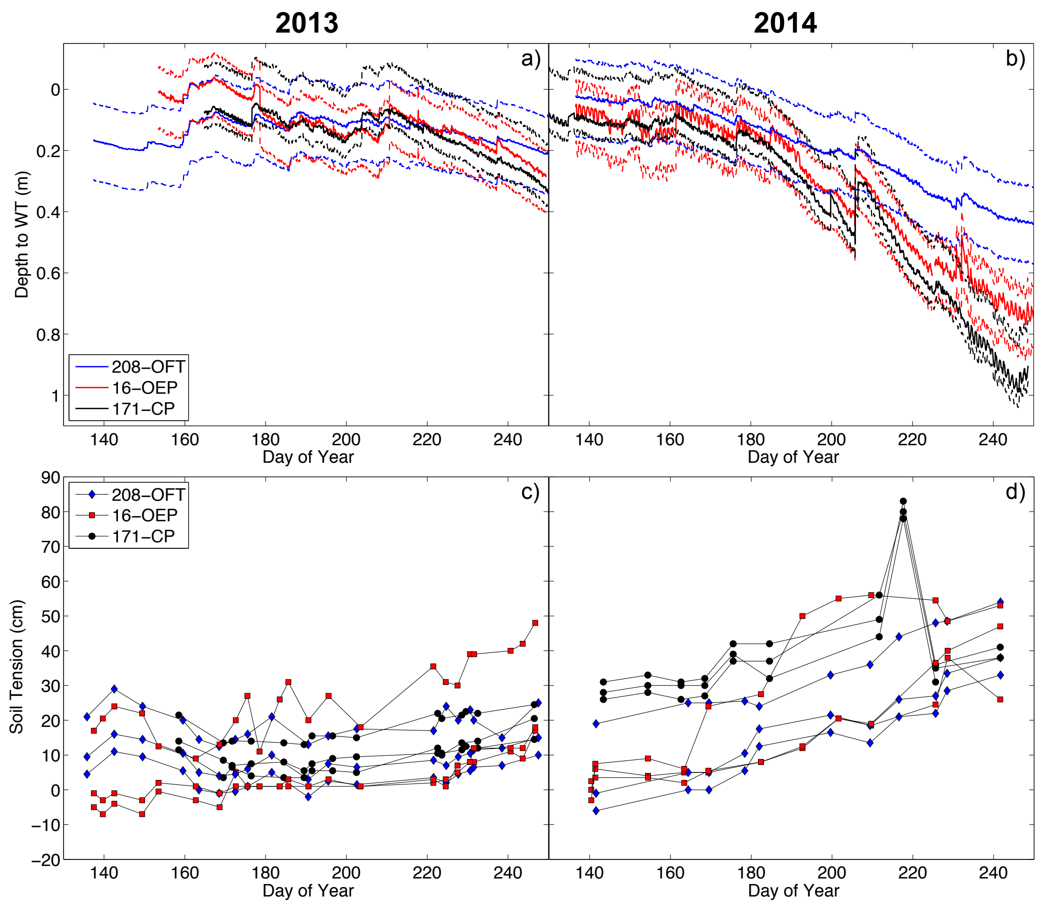



**Figure 5.3.** Depth to water table (WT) beneath tensiometers in 2013 (a) and 2014 (b) at the peatland margins of each site. Soil tension ($n$ = 3 per site) in 2013 (c) and 2014 (d) at the peatland margins of each site.

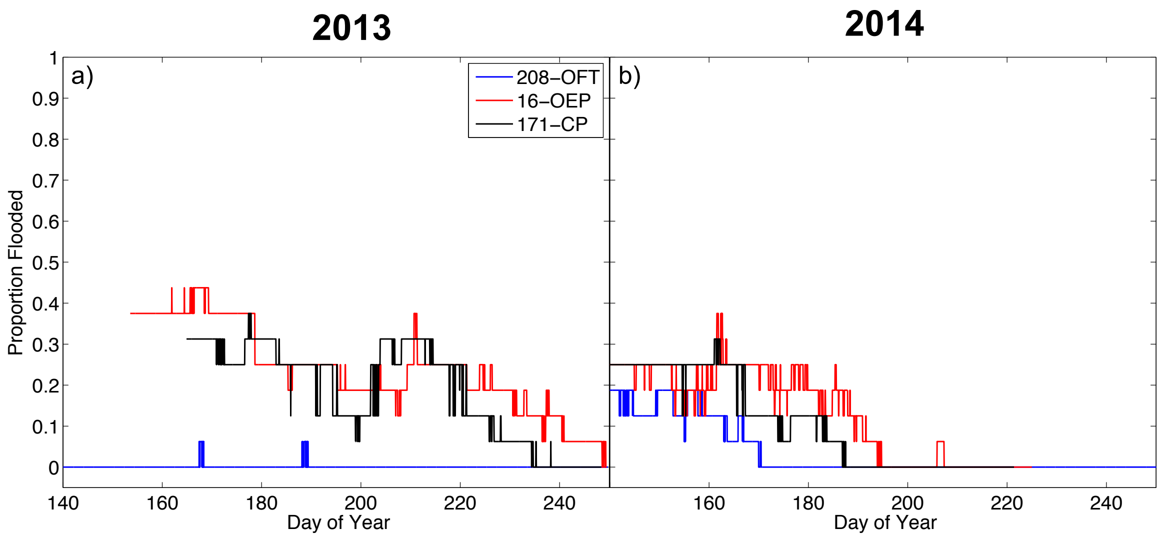


**Figure 5.4.** Proportion of distance on transect between the peatland margin and mineral margin flooded at each site in a) 2013 and b) 2014.

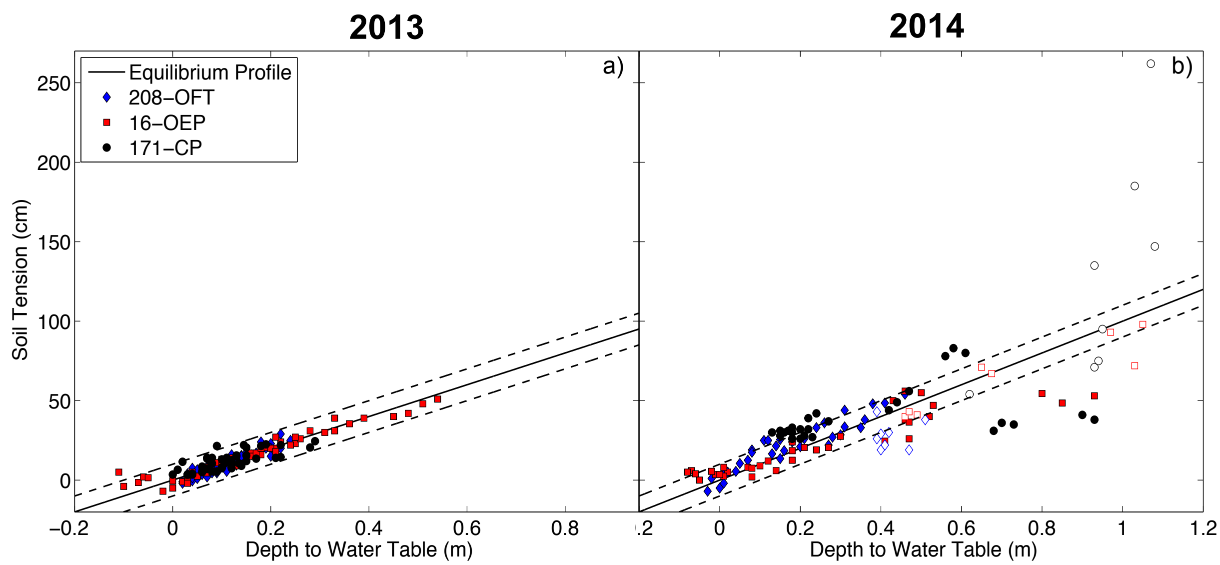



**Figure 5.5.** Water table versus soil tension (0.05 m depth) in 2013 (a) and 2014 (b) at the peatland margins of each site. The solid line surrounded by the two dashed lines represent the region where hydraulic head differences were < 0.10 m between near-surface peat and the water table. Data collected during the IFS are denoted by open symbols in 2014.

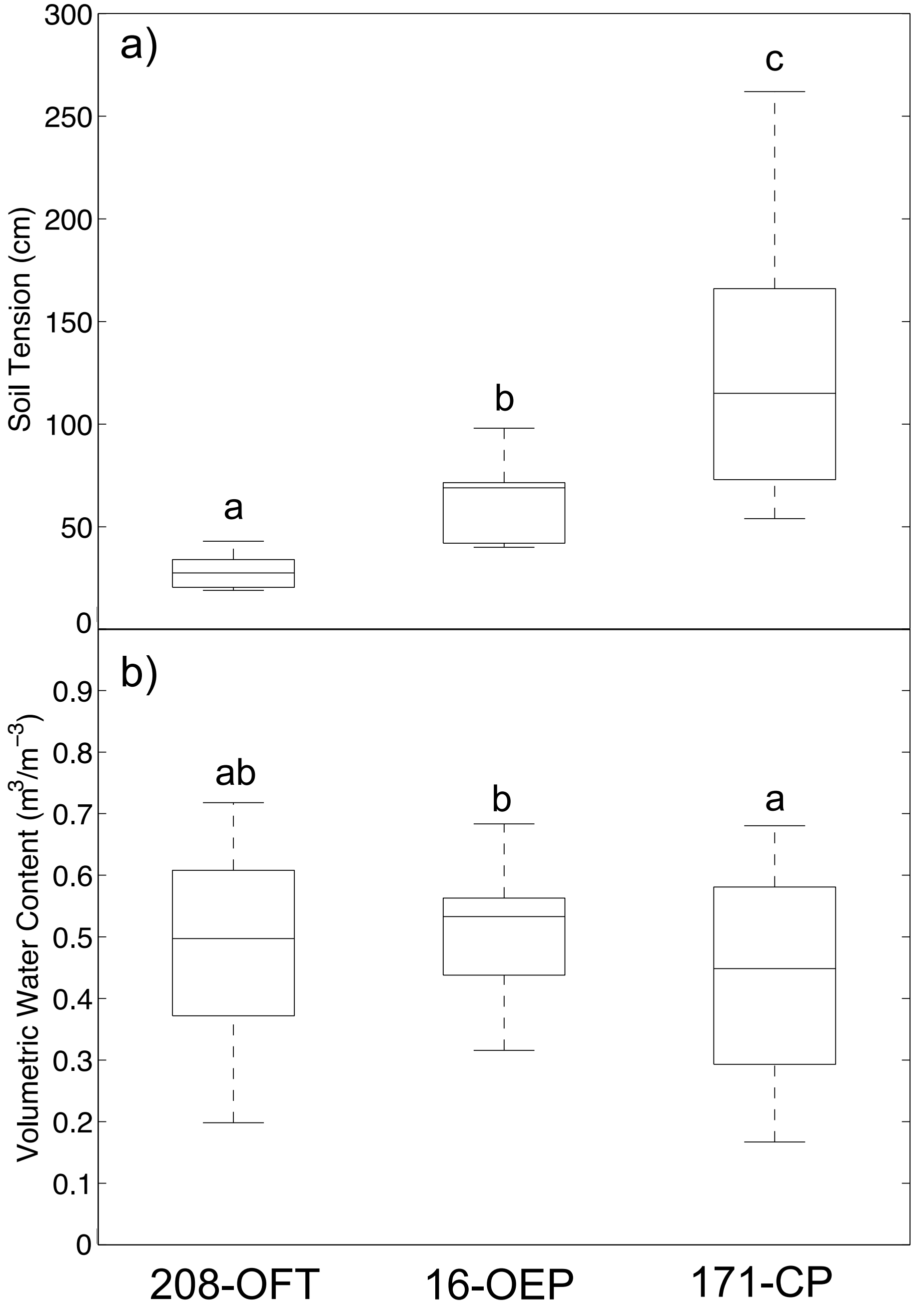



**Figure 5.6.** a) Soil tension (*n*=8) and b) surface volumetric moisture content (*n*=40) measured during the IFS. Letters above boxplots denote post-hoc test results (least significant difference α=0.05).

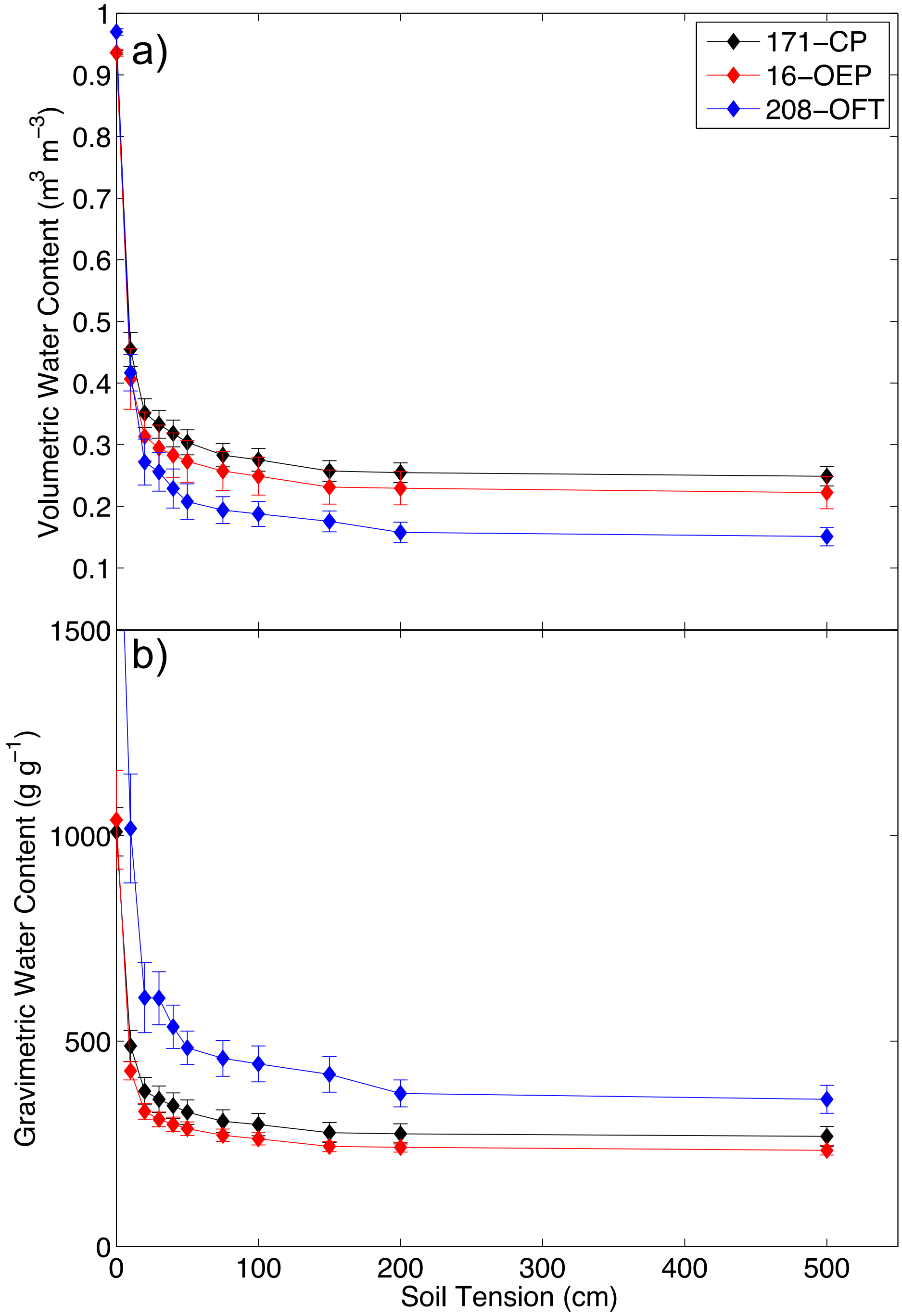



**Figure 5.7.** Peat moisture retention curves shown on a volumetric (a) and gravimetric (b) basis for the surface (0-0.05 m) layer for the peatland margins (*n*=5 per site) at each site.

# CHAPTER 6: SUMMARY AND CONCLUSIONS

## 6.1 Summary

Cross-scale hydrological processes affect wildfire impacts in peatland ecosystems. While previous studies primarily examined peat burn severity in the middles of large peatland complexes (*e.g.* Benscoter and Vitt, 2003), data herein suggests that small peatlands isolated from large groundwater flow systems in sub-humid climates are particularly vulnerable to deep burning at their margins. Specifically, high potential water demand from adjacent mineral upland ecosystems (Petrone *et al.*, 2007; Brown *et al.,* 2014), coupled with a sub-humid and highly variable decadal climate cycle (Devito *et al.*, 2012), results in dynamic hydrological conditions at peatland margins. Thus, when fire danger is high, large WT declines cause large deposits of dense peat to have low peat moisture contents, thereby facilitating deep burning.

Carbon losses at peatland margin 'hot spots' ranged from 10–85 kg C $m^{-2}$ (mean = 27 kg C $m^{-2}$) and accounted for ~80% of the total soil carbon loss from a small peatland. Similarly, Hokanson *et al.* (2015) demonstrated that burn severity is higher at peatland margins than peatland middles at the landscape-scale. Furthermore, connectivity to groundwater flow systems determined which peatlands were most at risk to deep burning (Hokanson *et al*., 2015). For this reason, assessments of the vulnerability of peatlands to wildfire, especially those examining the impact of future climate change, should conduct their analyses in the context of landscape-scale factors (*e.g.* hydrogeological setting).

Water availability, as indicated by soil tension and surface volumetric moisture content, is a key determinant of post-fire moss recovery in *Sphagnum*-dominated peatlands. In the middle of the peatland, both high and low burn severity can decrease post-fire water availability by altering peat hydrophysical properties (moisture retention, water repellency). Locations covered by *Sphagnum fuscum* prior to fire exhibited decreasing post-fire water availability with increasing burn severity. In contrast, the lowest water availability was observed in feather mosses that underwent low burn severity (residual branches identifiable), while burn depths >0.05 m in the middle of the peatland exhibited the highest water availability. These results are the first to provide an understanding of the underlying physical processes that control post-fire moss recolonization in peatlands. For this reason, a conceptual model was proposed in which: 1) severe burning (depth of burn > 0.05 m) is counterbalanced by rapid moss recolonization and 2) pre-fire species interact with burn severity to produce substantial lags in post-fire moss recovery. Data herein also suggests that microtopography alone is a poor indicator of post-fire moss recovery.

Post-fire recovery is also dependent on large-scale hydrology much more than previously anticipated. Landscape-scale factors controlling peatland WT dynamics appear to influence the trajectory of post-fire recovery in peatland middles. It was shown that hydrogeological setting influences post-fire recovery in peatland middles in two ways: 1) by influencing vegetation structure prior to wildfire, thereby controlling the coverage of post-fire surface covers and 2) by influencing post-fire WT positions. Peatlands that were located in drier hydrological settings, such as those isolated from groundwater flow systems, and those in late successional stages (*i.e.* older stands) had higher feather moss cover prior to fire. This was important because wildfire in these same peatlands resulted in the ubiquitous presence of lightly burned feather mosses. Indeed, when lightly burned feather moss was the post-fire surface cover, no moss recolonization was observed in any of the peatlands studied herein. Moreover, lower water table positions after wildfire in peatlands isolated from groundwater flow systems also stressed the recolonization of peatland mosses in other surface covers (*e.g.* lightly burned *Sphagnum fuscum*).

Given that the incidence of post-fire surface cover (*e.g.* lightly burned feather mosses) in peatland middles can be linked to landscape-scale properties, a classification scheme is given that provides a rapid and effective means of understanding post-fire recovery in *Sphagnum*-dominated peatlands. These results also indicate that post-fire recovery, and by extension peatland resilience to wildfire, is contingent upon large-scale hydrological processes affecting site wetness (*e.g.* hydrogeological setting). For this reason, peatlands in late successional stages and those situated in hydrogeological settings that are not well connected to groundwater flow systems are likely to be the most vulnerable to the combined effects of wildfire and climate change. Furthermore, ecohydrological feedbacks examined in previous studies (Waddington *et al.*, 2015; Kettridge *et al.*, 2014) must be evaluated in the context of landscape-scale properties. In particular, negative feedbacks within peatlands that reduce WT drawdowns during dry periods (Waddington *et al.*, 2015) may only be important in peatlands with limited groundwater connectivity.

While the range of moss recolonization observed in peatlands middles in different hydrogeological settings represents extremes in the post-fire recolonization distribution when compared to previous studies (*c.f.* Benscoter and Vitt, 2008), this study provides the first data on post-fire moss recolonization at peatland margins. Because peatland margins are prone to deep burning, the recolonization of peatland mosses are essential to ensure that peat accumulation maintains peatland-upland water and solute transfers that characterize peatland margin ecohydrological function. Although deep burning at the margins of peatlands altered peat hydrophysical properties after wildfire, post-fire water availability remained high, in part because this study was carried out during a wet period of the climate cycle. We anticipated that this high water availability would facilitate the recovery of keystone peatland mosses; however, moss species endemic to uplands frequently recolonized deeply burned peatland margins. The extent to which upland species recolonized peatland margins was dependent on a peatland's hydrogeological setting. It appears that dynamic hydrological conditions, characterized by flooding during wet conditions and rapid WT drawdowns during dry periods, in peatlands isolated from groundwater flow systems severely limited peatland moss recolonization. In contrast, stable hydrological conditions at a peatland situated in an oligotrophic groundwater flow system resulted in high peatland moss recolonization. Therefore, these results suggest that solute concentrations likely played a key role in determining whether or not peatland mosses are able to re-establish and/or outcompete species endemic to uplands, even though this effect was not explicitly quantified herein.

Because this study was carried out during a wet portion of the climate cycle, upland species may even more successfully colonize the margins of peatlands isolated from groundwater flow systems during dry, or even normal, periods of the climate cycle. In these instances, aspen (*Populus tremuloides*) may rapidly recolonize severely burned organic and mineral substrates (Johnstone and Chapin, 2006). This may further enhance peatland margin WT drawdowns because aspen accounts for the majority of transpiration in fine-textured mineral uplands in the BP (Brown *et al.*, 2014). Coupled with changes in soil physical properties, small peatlands isolated from groundwater flow systems may undergo a positive feedback loop whereby post-fire WT drawdowns following deep burning increase drying at the peatland and its margins. In summary, deep burning at peatland margins is likely altering the ecohydrological function of peatland margins and facilitating a regime shift from peatland to mineral upland, resulting in the lateral shrinkage of some peatlands in the BP. This transition may accelerate due to increases in total wildfire area burned (Flannigan *et al.*, 2005) and organic layer burn severity (Kettridge *et al.*, 2015; Turetsky *et al.,* 2011) associated with climate change in the boreal forest.

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