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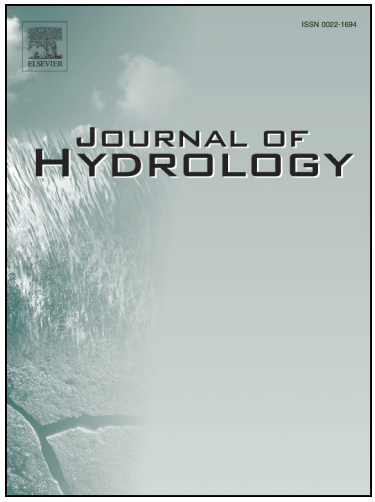

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HYDROGEOLOGICAL CONTROLS ON POST-FIRE MOSS RECOVERY IN PEATLANDS

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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,

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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.

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).

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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 studies found that the re-establishment of peatland mosses after wildfire, through repsrouting and diaspores, 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, the vertical and horizontal spatial extent of burn) affects post-fire hydrological conditions, such as soil moisture (Lukenbach *et al.*, 2015a) and depth to water table (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.*,

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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 relative to groundwater (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 the spatial extent of burn (*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 depth of burn (DOB) (>0.05 m) (Benscoter *et al.*, 2011), whereas *Sphagnum* mosses (*e.g.*, *Sphagnum fuscum*) are able to efficiently retain water during dry periods and limit DOB to <0.03 m (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.*,

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2015a; Lukenbach *et al.*, 2015b) and the presence of near-surface (top 0.10 m) 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 (see conceptual model in 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 (see Eppinga *et al.*, 2008 for a review of microtopography development), 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

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(Thompson and Waddington, 2013a). Because hummock species (*i.e. S. 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 deeper 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. In particular, 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

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hydrogeological settings isolated from groundwater sources would exhibit lower post-fire moss water availability 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 greater feather moss cover prior to fire (due to tree shading) because of lower average WT positions and that this would reduce post-fire moss water availability and limit peatland moss (*e.g. Sphagnum*) recolonization rates.

METHODOLOGY

2.1 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 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).

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We collected measurements in peatlands affected by the Utikuma complex fire (SWF-057, ~90,000 ha) that were part of the URSA's long-term hydrological monitoring network (including WT dynamics), 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 1). However, our study focused on the bog portions (based on vegetation indicators and pH, Table 1) of the three peatland complexes.

In the lake 16 catchment, an ~3 ha ephemerally perched peatland complex is positioned adjacent to a regional topographic high in a coarse-textured glaciofluvial outwash and is ephemerally perched above both local and intermediate groundwater flow systems between lakes in the region (Smerdon *et al.*, 2005). We took advantage of the presence of an ~0.5 ha burned bog (EP-Burned, *i.e. 'Ephemerally Perched Burned'*) and an ~0.5 ha unburned bog (EP-Unburned, *i.e. 'Ephemerally Perched Unburned'*) in the peatland complex. The partial burning of the peatland complex was likely due to variability in fire behaviour (*c.f.* Johnston *et al.*, 2015) and not differences in site wetness or fuel structure between the bogs (Table 1). 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 EP-Burned and EP-Unburned, 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

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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 EP. This flow through bog 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 the peatland burned during fire, providing the opportunity to instrument both the burned (FT-Burned, *i.e. 'Flow Through Burned'*) and unburned (FT-Unburned, *i.e. 'Flow Through Unburned'*) portions of the peatland. The partial burning of the peatland was likely due to variability in fire behaviour (*c.f.* Johnston *et al.*, 2015) and not differences in site wetness or fuel structure (Table 1). 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 1) represents the vast majority of the water balance at the peatland (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 burned bog is located on a lacustrine clay plain (CP-Burned, *i.e. 'Clay Plain Burned'*) 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, CP-Burned receives

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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 1). Because there were no unburned portions of CP-Burned, we instrumented the nearest, and only, unburned bog (CP-Unburned, *i.e. 'Clay Plain Unburned'*) 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 CP-Burned in lake catchment 300. CP-Unburned 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 EP-Burned/Unburned and FT-Burned/Unburned, respectively. Furthermore, Hokanson *et al.* (2015) showed that the depth to WT ranged from 0.15 – 0.60 m at CP-Burned, 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 CP-Unburned, 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 CP-Unburned has a shallower WT on average than CP-Burned.

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Because CP-Burned did not have an unburned portion, burned and unburned sites were not analyzed using a paired approach. 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 *Pleurozium schreberi* and *Sphagnum angustifolium* were the primary surface covers in hollows. Species cover in hollows varied depending on historical site wetness (Table 1). Vascular vegetation cover was dominated by *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 sampled post-fire moss water availability using a microtopographic approach. 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

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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.

2.2 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. Soil tension (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 randomly selected hummocks and hollows at the burned and unburned sites using 0.02 m outside diameter tensiometers (*cf.* Lukenbach *et al.*, 2015a). In 2014, surface volumetric moisture content (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.

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2.3 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 EP-Burned, EP-Unburned, FT-Burned, FT-Unburned, and CP-Burned using the point-centered quarter method along 100 m long transects (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 CP-Unburned were obtained from Johnston *et al.* (2015). Stand ages were ~80, ~135, ~70, and ~80 years for FT-Burned/Unburned, EP-Burned/Unburned, CP-Burned, and CP-Unburned, 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) that captured the dominant post-fire surface covers at all the sites (Table 2). These burn severity groups were typically small patches (0.01-0.25 m^2) interspersed amongst one another throughout the entire peatland. 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 lightly burned feather moss locations (LB-F) were areas where DOB <0.03 m and residual, singed feather moss was visible. In addition to the classification of Lukenbach *et al.* (2015a), *S. angustifolium* was added as a surface cover because 1) its ubiquitous coverage at FT-Unburned and CP-Unburned (unburned) and 2) it is prone to higher DOB than *S. fuscum* (Benscoter *et al.*,

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2011). Thus, locations where DOB >0.05 m and pre-fire moss cover were not identifiable were classified as severely burned areas that had either *S. angustifolium* or feather moss present prior to fire (SB-F/Sa). DOB was measured in these locations by taking the difference in surface elevation between burned areas and surrounding unconsumed areas (*cf*. Kasischke *et al.*, 2008; Mack *et al.*, 2011). Of note, we observed no areas at our study sites where *S. angustifolium* underwent low burn severity (DOB < 0.03 m). Unburned surface covers consisted of unburned *S. fuscum* (UB-Sf), unburned *S. angustifolium* (UB-Sa), and unburned feather moss (UB-F).

Hydrological measurements along a burn severity gradient across a hydrogeological transect: Three tensiometers were installed in each burn severity group (see Table 2) in 2014 at the burned and unburned sites by instrumenting the nearest locations to randomly generated coordinate positions within each site. 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 FT-Burned and FT-Unburned (< 2% pre-fire surface cover) and UB-Sa was not common at EP-Burned and EP-Unburned (< 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 EP-Burned and EP-Unburned, this yielded three UB-Sf, three UB-F, one LB-Sf, two LB-F, two SB-Sf, and one SB-F/Sa. At FT-Burned and FT-Unburned, this yielded three UB-Sf, three UB-Sa,

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three LB-Sf, and three SB-F/Sa. Finally, at CP-Burned and 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 1st, 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 = 25 per group) at each site (n = 100 per site).

2.4 Moss and bryophyte recolonization measurements

Hydrological measurements during the IFS were paired with measurements of % bryophyte species cover and % *Sphagnum* cover (*c.f.* Rochefort, 2000), which were visually estimated in 0.10 x 0.10 m plots above each tensiometer cup and surface θ sampling plot to examine the association with bryophyte recovery. The fire resulted in the mortality or prolonged dormancy of all ground layer vegetation, thus the observations of % bryophyte species and *Sphagnum* cover are a direct measure of the recolonization following the fire. *S. fuscum* primarily recolonized by reprouting, while other *Sphagnum* species recolonized hollows via diaspores (Benscoter and Vitt, 2008). These measurements were used to obtain a site-level estimate of moss recolonization by scaling

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post-fire moss recolonization in each burn severity group to the coverage of each burn severity group at each site.

2.5 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 Table 1.

2.6 Analyses

Due to the challenge of instrumenting a large number of sites and the need to take advantage of sites where long-term hydrological monitoring had occurred, 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 pseudoreplicated (*c.f.* Hurlbert, 1984). In particular, we examined statistical differences between sites, instead of between hydrogeological settings, and then interpreted site differences in the context of their hydrogeological setting. In both the microtopographic sampling approach (experiment 1) and burn severity group sampling approach (experiment 2), Ψ was natural

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logarithm transformed and surface θ 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, which included interactions between variables not nested within one another, was used to account for multiple measurements in the same Ψ/surface θ plot (modelled as a random effect) 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 hydrogeological setting). 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 the hydraulic head difference between the near-surface peat (0.05 m depth) and the WT did not exceed 10 cm was calculated (*c.f.*, Thompson and Waddington, 2013b). This condition implies that evaporative water losses from the moss

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surface are rapidly replenished by water fluxes from the WT 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 does not appreciably decline when the hydraulic head difference between the near-surface peat and the WT does not exceed 10 cm. 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.

RESULTS

3.1 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 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 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 2c).

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In 2012, WTs at all sites peaked in June following large rain events (Figure 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 3b). In 2014, WTs at all sites peaked following snowmelt and declined throughout the rest of the summer due to dry conditions (Figure 3c). Depth to WT did not vary between EP-Burned and EP-Unburned. Similarly, depth to WT did not vary between FT-Burned and FT-Unburned. In contrast, the WT at CP-Unburned was shallower than at CP-Burned. With the exception of CP-Unburned in 2012, FT-Burned and FT-Unburned had the shallowest WTs 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 FT-Burned and FT-Unburned. In contrast, EP-Burned, EP-Unburned, and CP-Burned 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 EP-Burned, EP-Unburned, and CP-Burned near the end of the study period in 2014. At all sites, hummocks had ~0.2-0.4 m higher surface elevations than adjacent hollows, resulting in deeper WTs beneath hummocks.

Ψ values were generally higher at the burned sites and sites isolated from groundwater flow systems, while they were highly variable both within and between microforms (Figure 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 × burn status ($F_{2,\ 1806}$ = 15.1, p < 0.001) on Ψ (post-

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hoc test results in Figure 4). Year ($F_{24, 1806} = 0.92$, $p = 0.58$) and microtopography × burn status ($F_{3, 1806} = 0.99$, $p = 0.40$) were not significant 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 × burn status ($F_{2, 357} = 6.07$, $p < 0.01$) on surface θ (post-hoc test results in Figure 5a). Microtopography × burn status ($F_{3, 357} = 0.66$, $p = 0.59$) did not have a significant effect on surface θ and, 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 3). Furthermore, hydraulic head differences not exceeding 10 cm m between near-surface peat and the WT were more common at sites with shallower WTs (Table 3).

3.2 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 EP-Burned followed by CP-Burned and FT-Burned (Table 1). Fuel loading followed this same trend, as basal diameters were larger at EP-Burned and CP-Burned (Table 1). Stand densities at the unburned reference sites did not follow this same hydrogeological gradient, as CP-Unburned had the highest stand density compared to unburned EP-Unburned and FT-Unburned. However, trends in fuel loads did follow the same hydrogeological gradient as the burned sites as indicated by larger basal diameters at EP-Unburned than CP-Unburned (Table 1).

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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 6c). Ψ was higher at EP-Burned and CP-Burned than FT-Burned, particularly for the LB-Sf and SB-F/Sa. 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 6). With the exception of UB-F at CP-Unburned and EP-Unburned, trends in surface θ were similar to those observed for Ψ (Figure 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 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 *S. fuscum* locations (*i.e.* SB-Sf) (Table 3). Where significant WT-Ψ relationships were observed (UB-Sa, UB-Sf, LB-Sf and SB-F/Sa at all sites), hydraulic head differences not exceeding 10 cm between near-surface peat and the WT were frequent, albeit variable, across sites (Table 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 ($F_{9, 478}$ =

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322, $p < 0.001$). Post-hoc test results for Ψ and surface θ are shown in Figure 8a and 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 not exceeding 10 cm between near-surface peat and the WT were present 75% and 100% of LB-Sf and SB-F/Sa at FT-Burned, respectively, while hydraulic head differences not exceeding 10 cm between near-surface peat and the WT were present at EP-Burned in 25% and 87.5% of LB-Sf and SB-F/Sa, respectively. At CP-Burned, hydraulic head differences not exceeding 10 cm between near-surface peat and the WT were present in 25% and 62.5% of LB-Sf and SB-F/Sa, respectively.

3.3 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 1, Figure 9). Across all sites, high post-fire water availability corresponded to high post-fire % bryophyte recolonization in the burn severity groups (Table 1, Figure 9). *Sphagnum* spp. were observed in LB-Sf at all the burned sites, while *Sphagnum* spp. were also common in SB-F/Sa at FT-Burned (Figure 9). No bryophyte recolonization was observed at all sites in LB-F. 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 between

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sites, ranging from 49% at EP-Burned to 95% at FT-Burned (Table 1). *S. fuscum* was the dominant species recolonizing LB-Sf at all sites, accounting for 98%, 63%, and 97% of the LB-Sf cover at FT-Burned, EP-Burned, and CP-Burned, respectively. *P. strictum* and *C. purpureus* were also present in LB-Sf, but only at EP-Burned 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 was less variable across the sites than in LB-Sf, ranging from 54% at CP-Burned to 69% at FT-Burned (Table 1). However, SB-F/Sa exhibited the greatest variability in the species recolonizing these areas both within and between sites. At FT-Burned, post-fire surface cover in SB-F/Sa was 36% *S. angustifolium*, 12% *Sphagnum magellanicum*, 9% *Polytricum strictum*, 7% *Ceratodon purpureus*, 4% *Aulacomnium palustre*, and 1% *Dicranum spp.*, respectively. In contrast, post-fire surface cover in SB-F/Sa at EP-Burned was 53% *C. purpureus*, 8% *P. strictum*, 3% *Marchantia polymorpha*, 1% *A. palustre*, respectively, and at CP-Burned was 32% *C. purpureus*, 18% *P. strictum*, 1% *M. polymorpha*, 1% *S. angustifolium*, 1% *A. palustre*, respectively.

DISCUSSION

4.1 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

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severity in hollows (*i.e.* SB-F/Sa), regardless of whether pre-fire surface cover was feather mosses (UB-F) or *S. angustifolium* (UB-Sa), increased post-fire water availability. Given that this pattern of water availability in the burn severity groups was consistent across all sites (Table 3, Figures 5b, 8) and between years (Figure 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.

4.2 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 surface covers 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, both of which can limit upward capillary flow, and 2) high water availability LB-Sf and SB-F/Sa surface covers 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 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.

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Densely forested sites (*i.e.* EP-Burned) had higher canopy closure, 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 between sites in different hydrogeological settings (Table 1). The ~80 year old stand at FT-Burned and FT-Unburned 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 following wildfire (*i.e.* 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 FT-Burned (Hokanson, 2014), where the adjacent peatland's position in the groundwater flow system resulted in an ~0.30 m deeper WT than FT-Burned (Hokanson, 2014). Likewise, the substantially lower canopy fuel load at CP-Burned compared to EP-Burned 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 1). This not only resulted in low post-fire water availability but also rendered >25% of surface at EP-Burned and CP-Burned unable to recolonize for at least the first three years following wildfire (Table 1 and Figure 9).

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Deeper WTs at CP-Burned and EP-Burned likely increased the frequency of SB-Sf surface cover at EP-Burned and CP-Burned compared to FT-Burned. 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 between sites in different hydrogeological settings. This is particularly important because SB-Sf accounted for 5%, 14%, and 22% of the surface cover at FT-Burned, EP-Burned, and CP-Burned, respectively, and little (< 5%) moss recolonization was observed in this burn severity group. Combining the spatial coverage of SB-Sf with LB-F at EP-Burned and CP-Burned highlights that >40% of the surface 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.

4.3 Post-fire hydrogeological controls on peatland moss recovery

Through its control of post-fire WT position at the sites, 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 Ψ in LB-Sf and SB-F/Sa locations at FT-Burned compared to these same burn severity groups at EP-Burned and CP-Burned during both the study period and the IFS (Figures 6 and 8). Because Ψ is a more precise

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indicator of moss stress than surface θ (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 WT) at all the burned sites (Figure 7a), deeper WTs at EP-Burned and CP-Burned resulted in higher soil tension (Ψ) values (WTs >0.75 m in Figure 7a). Moreover, by examining the exceedance probability (Figure 7b) of 100 cm of soil tension (Ψ), 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 FT-Burned exceed this threshold <2% of the time, LB-Sf locations at EP-Burned and CP-Burned 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 FT-Burned was recolonized, primarily by *S. fuscum*, while only 49% of LB-Sf at EP-Burned was recolonized, of which, *C. purpureus* comprised 15% of the post-fire surface cover (Table 1, Figure 9). While SB-F/Sa locations did not exhibit large differences in the absolute amount of recolonization between sites (Table 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 FT-Burned, while this did not occur at EP-Burned and was rare at CP-Burned (Figure 9). This may be attributable to the presence of a near-surface WT at FT-Burned (frequently <0.10 m), which may have facilitated the rapid recolonization of *Sphagnum spp.*

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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 CP-Burned and EP-Burned may resemble the end of the 2014 study period, although these sites may be dissimilar during other periods of the climate cycle because of EP-Burned's ephemeral connection to groundwater sources. Likewise, the lagged response of FT-Burned 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.

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 FT-Burned and marginal moss recolonization at EP-Burned 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).

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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.

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

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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 1: Summary of pre-fire and post-fire 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 (FT = flow-through, EP = ephemerally perched, and CP = clay plain). See Table 2 for the nomenclature of the burn severity groups. From 2012-2014, mineral electrical conductivity was monitored in wells in adjacent uplands at each site, while peat electrical conductivity and pH (mean) was measured in wells in the middle of each peatland. Hydraulic conductivity (geometric means shown) 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					
	FT-B	**FT-UB**	**EP-B**	**EP-UB**	**CP-B**	**CP-UB**
Peat 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

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Table 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

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Table 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 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 did not exceed 0.10 m between near-surface peat and the WT was calculated (% SWT – *i.e.* supplied by water table).

	Experiment 1 - Temporal Analyses					
	FT-Burned & FT-Unburned		**EP-Burned & EP-Unburned**		**CP-Burned & CP-Unburned**	
Microform	WT-Ψ r^2	% SWT	WT-Ψ r^2	% SWT	WT-Ψ r^2	% SWT
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					
	FT-Burned & FT-Unburned		**EP-Burned & EP-Unburned**		**CP-Burned & CP-Unburned**	
Group	WT-Ψ r^2	% SWT	WT-Ψ r^2	% SWT	WT-Ψ r^2	% SWT
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

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Figure 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. B and UB stand for burned and unburned at each site along the hydrogeological transect (FT = flow-through, EP = ephemerally perched, and CP = clay plain). For comparisons with other URSA research, the FT, EP, and CP sites are located in lake catchments 16, 208, and 171, respectively.

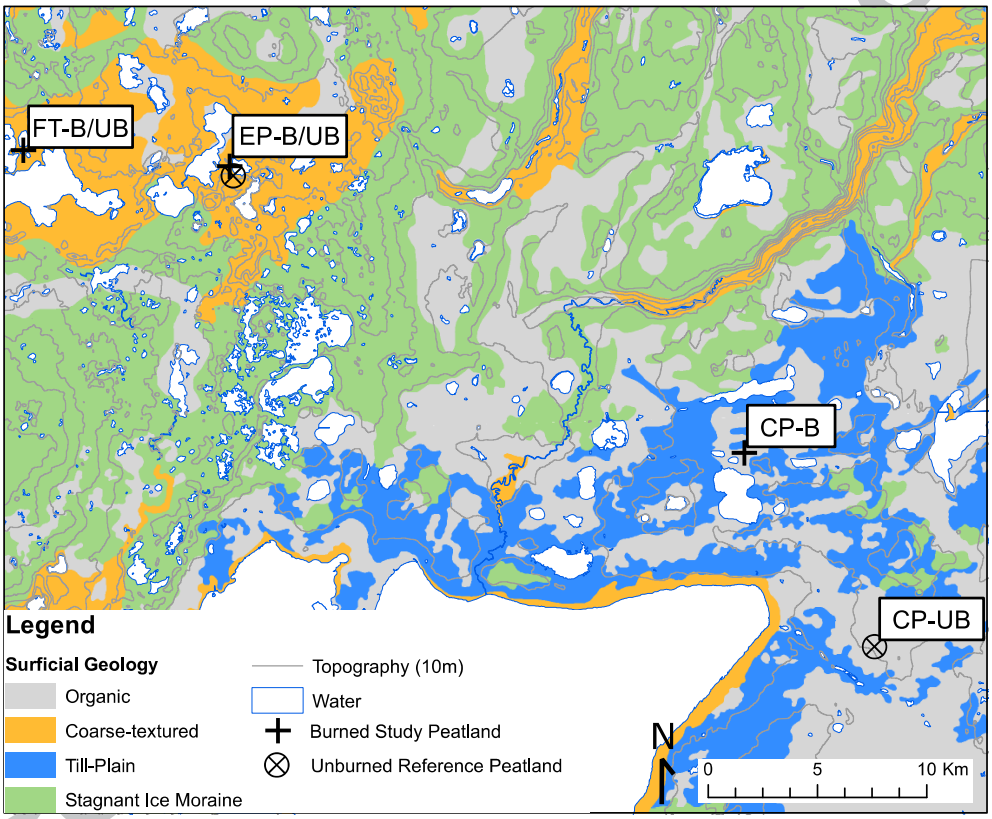

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Figure 2: Cumulative rainfall from May – September in 2012, 2013, 2014 at a) FT-Burned, b) EP-Burned, and c) CP-Burned.

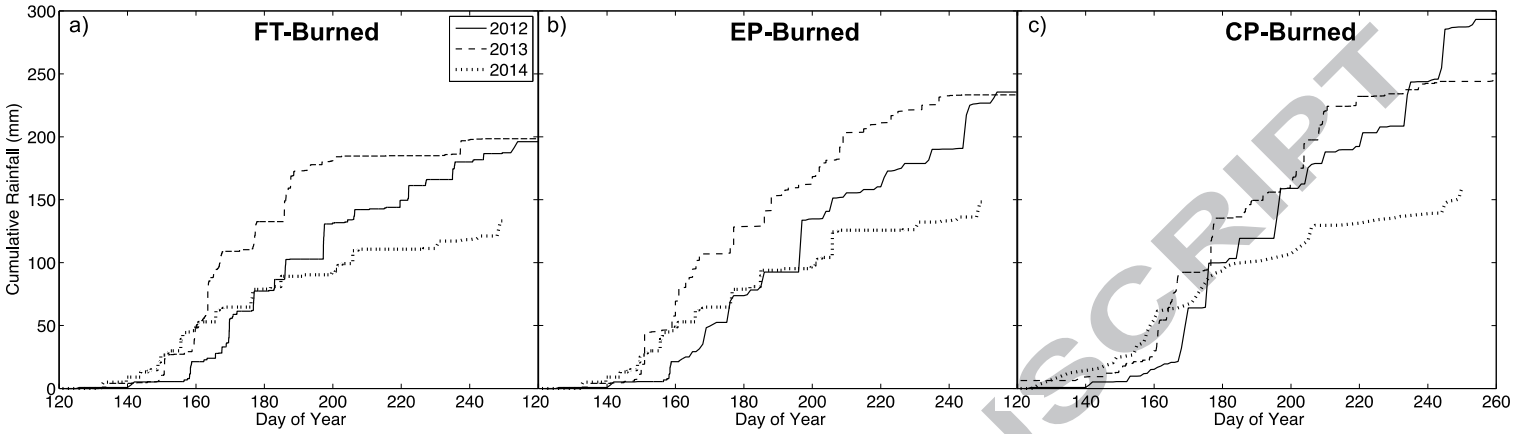

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Figure 3: Average depth to WT (solid lines) beneath tensiometers installed in microforms (3 hummocks and 3 hollows per site) at the burned and unburned sites for a) 2012, b) 2013, and c) 2014. Dashed lines represent the minima and maxima depth to WT for each site. Differences in averages, minima, and maxima between FT-Burned and FT-Unburned were less than 0.05 m. This was also true when comparing EP-Burned to EP-Unburned, thus the unburned traces of FT-Unburned and EP-Unburned are not shown for visual purposes. Rapid rises in depth to WT are caused by rainfall events.

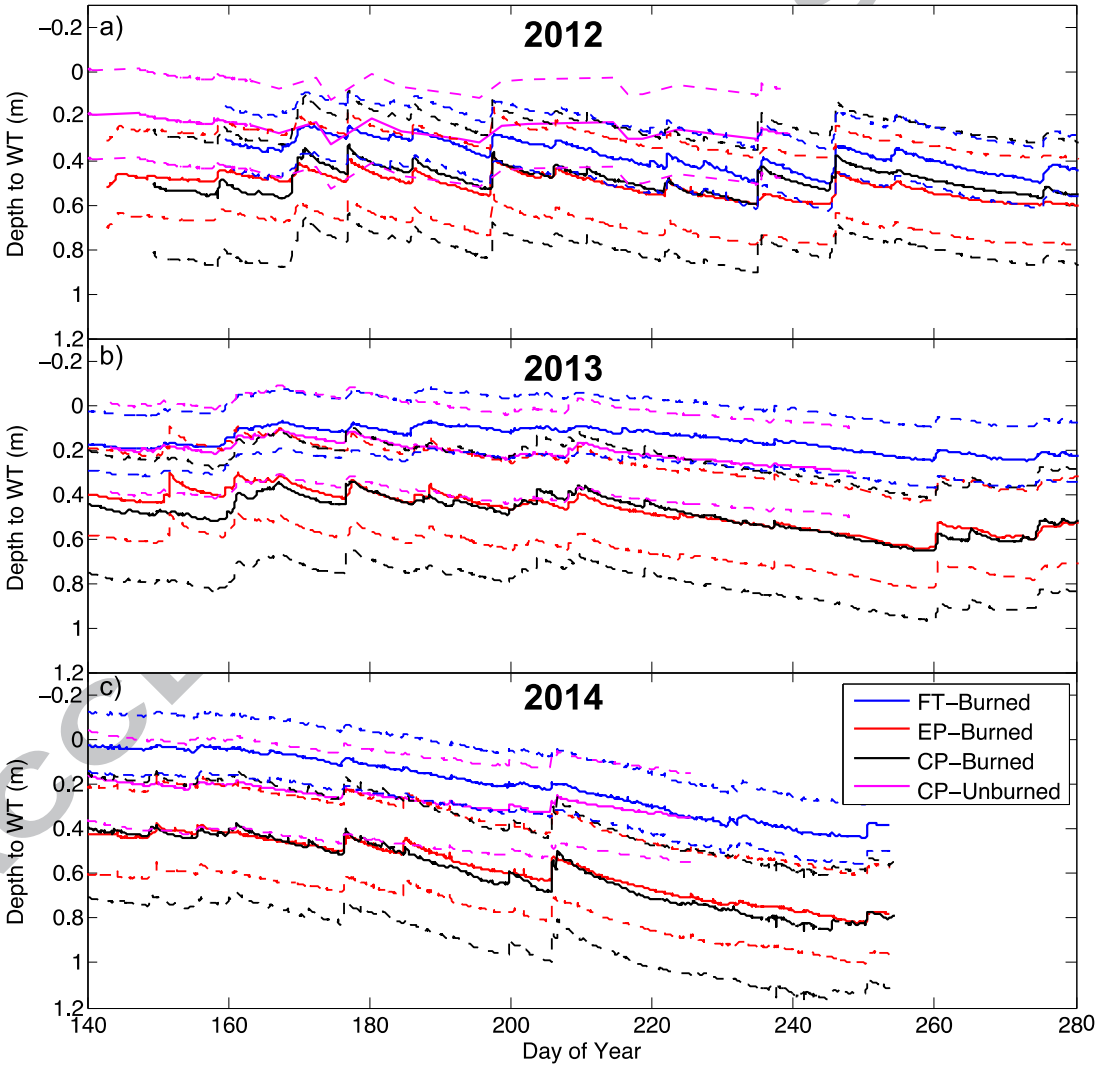

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Figure 4: Experiment 1: Soil tension (Ψ) measured at least once per week from May – September in 2012, 2013, and 2014 in three hummocks and hollows at both the burned and unburned sites. Because year was not a significant factor affecting Ψ, data are 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).

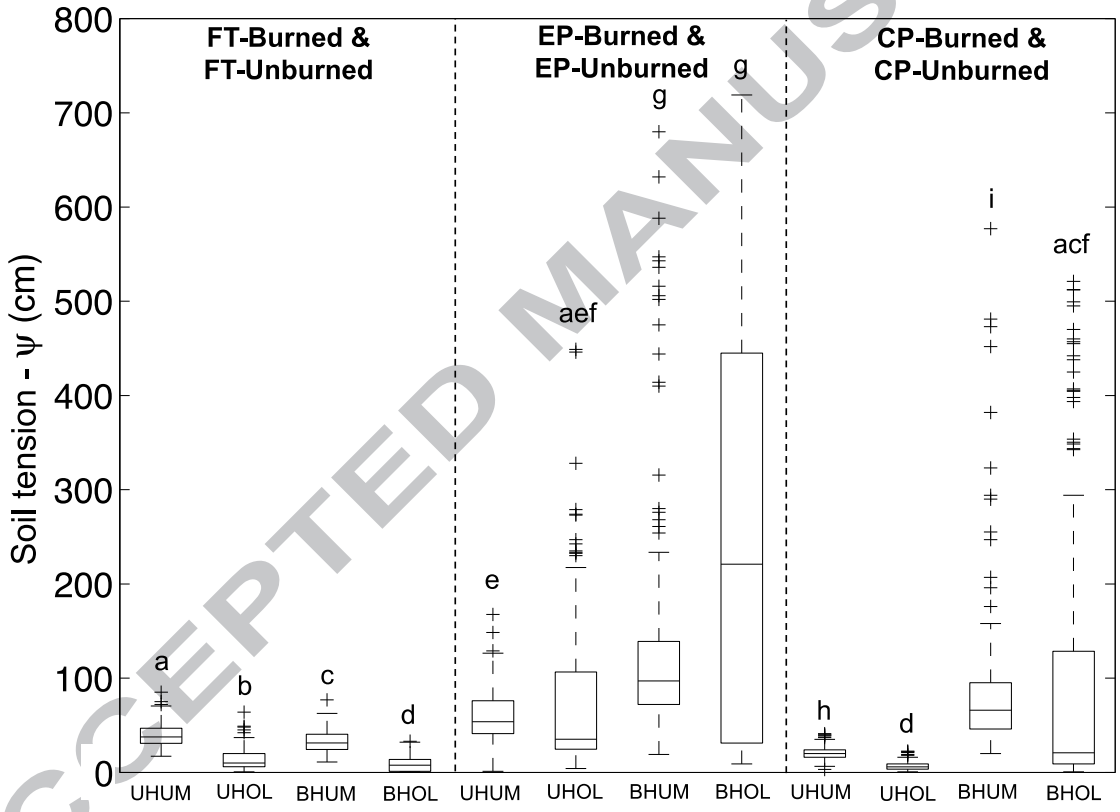

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Figure 5: a) Experiment 1: Surface volumetric moisture content (θ, top 0.03 m) in 2014 in three hummocks and hollows at both the burned and unburned sites. UHUM, UHOL, BHUM, and BHOL represent unburned hollows, unburned hummocks, burned hummocks, and 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 2. Surface θ with the same lowercase letter are not statistically different (least significant difference α=0.05).

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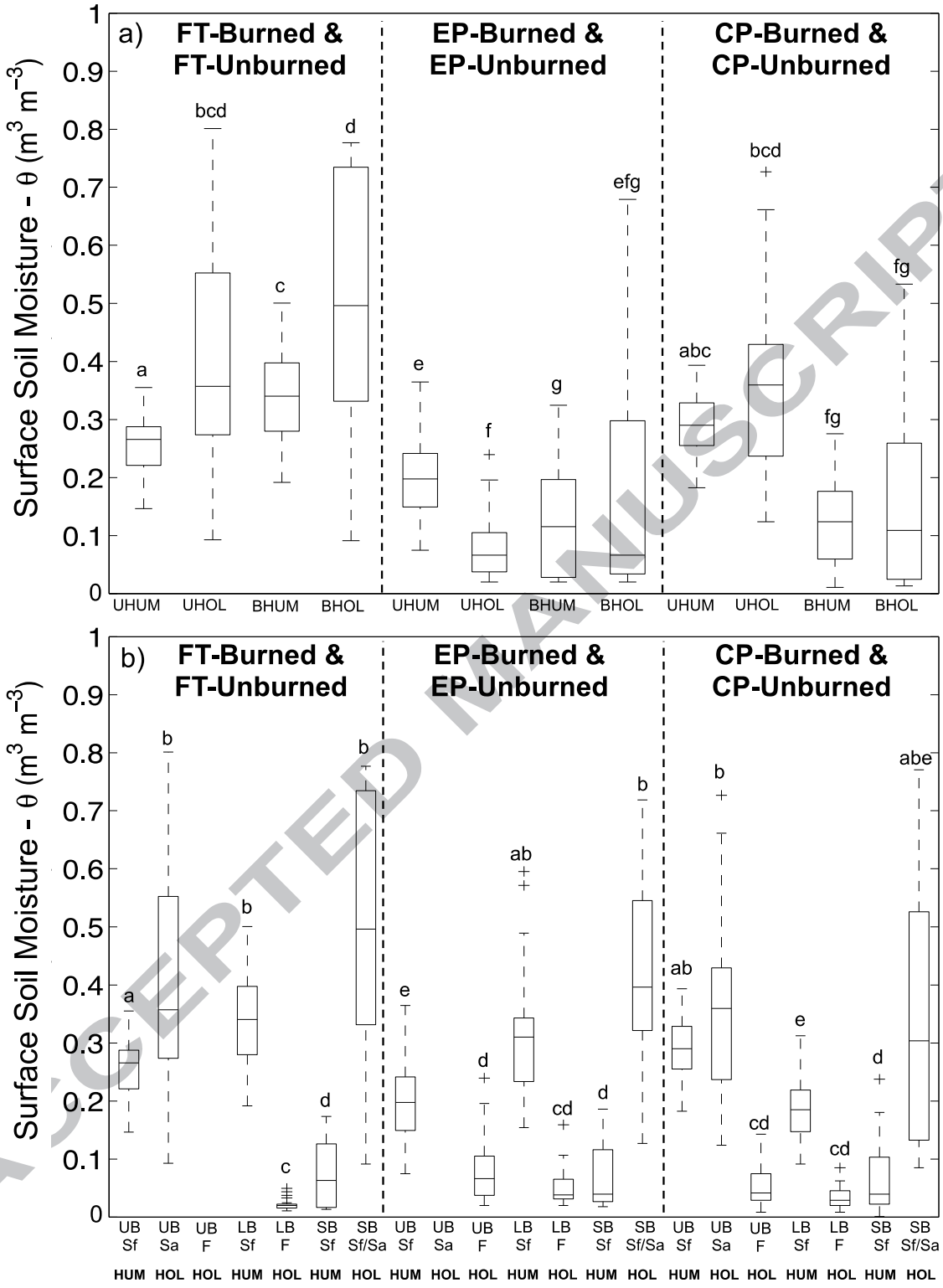

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Figure 6: Experiment 2: Soil tension (Ψ) in burn severity groups (microform also noted) at both the burned and unburned sites. The nomenclature for the burn severity groups is listed in Table 2. Ψ data from 2012 and 2013 was obtained by classifying the tensiometers placed in microforms from experiment 1 into burn severity groups. Because of the small sample sizes in 2012 and 2013, statistical analyses were not conducted. For 2014 ($n = 3$ per burn severity group), Ψ with the same lowercase letter are not statistically different (least significant difference $\alpha=0.05$).

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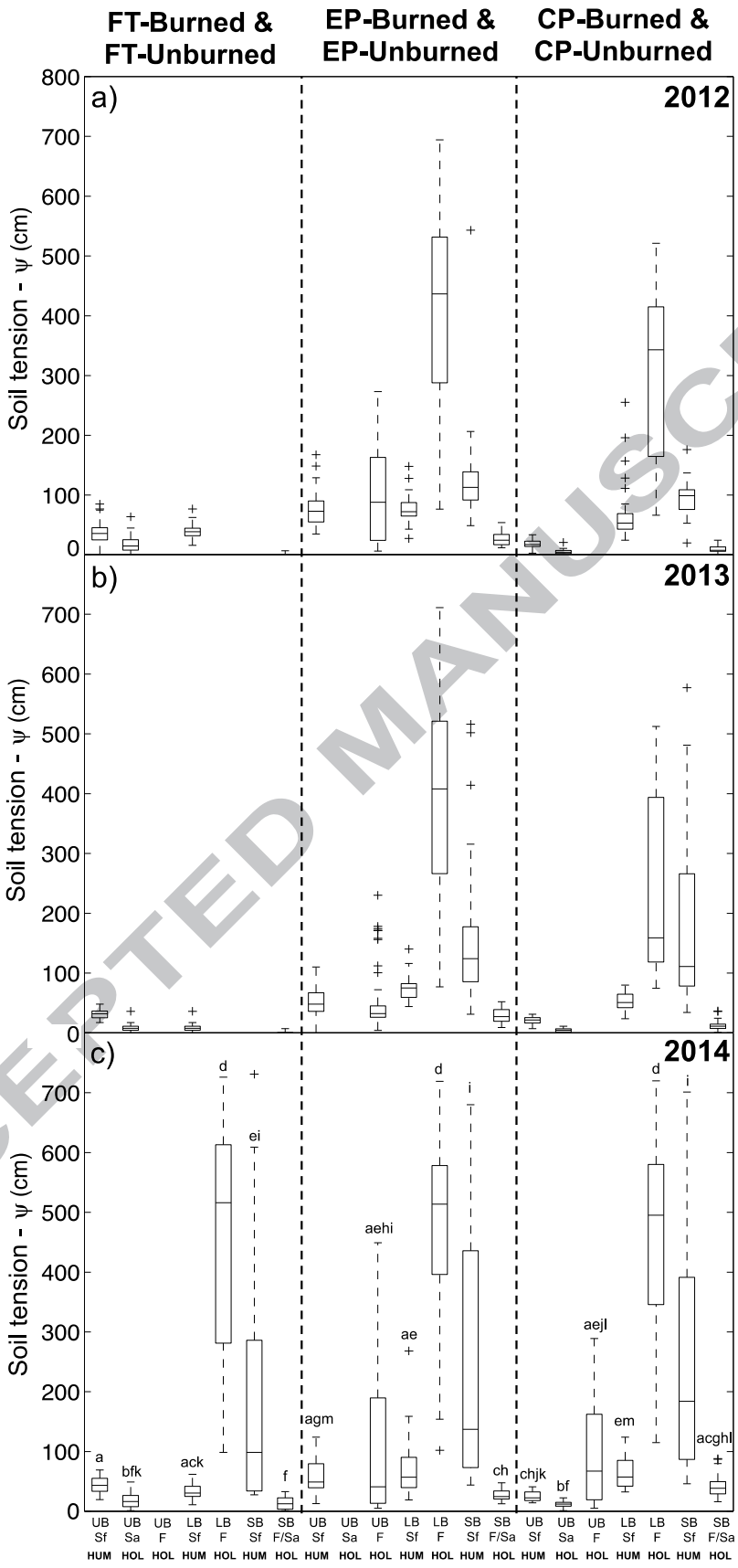

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Figure 7: a) Depth to WT versus soil tension (Ψ) at all sites for the 2012-2014 study period for burn severity groups that had significant WT-Ψ relationships (see Table 3). Colours denote hydrogeological setting, open/closed symbols denote burned/unburned, 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 < 10 cm between near-surface peat and the water table. Points plotting above the upper dashed line indicate that moss stress is more likely to occur because hydraulic head differences between near-surface peat and the water table exceed 10 cm. b) The same Ψ data as in panel a, but plotted as exceedance probability distributions.

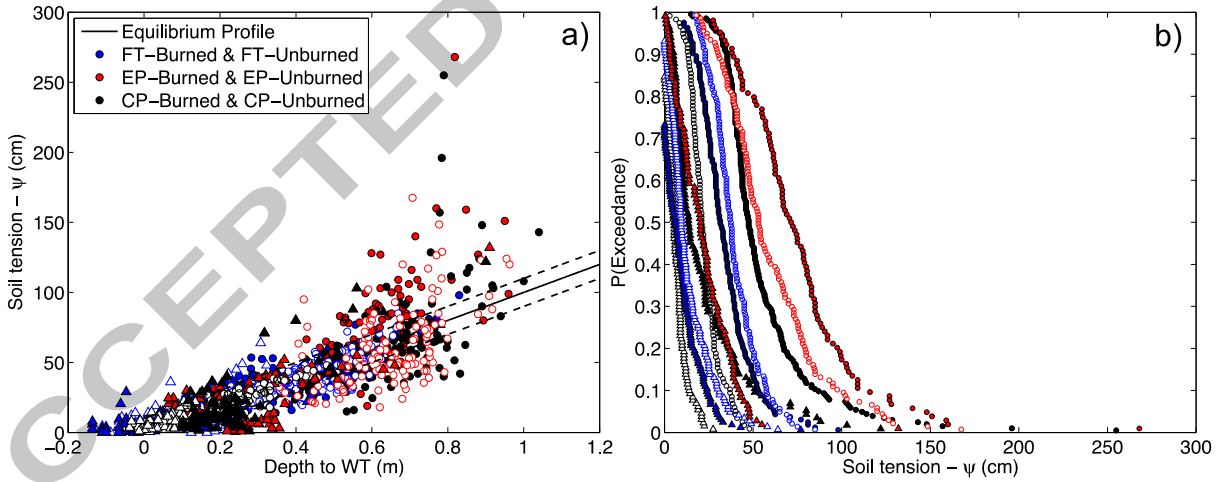

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Figure 8: a) Surface volumetric moisture content (θ, top 0.03 m) and b) soil tension (Ψ) in the burn severity groups (microform also noted) at only the burned sites (see Table 2 for nomenclature) collected during the intensive field survey (IFS) on 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).

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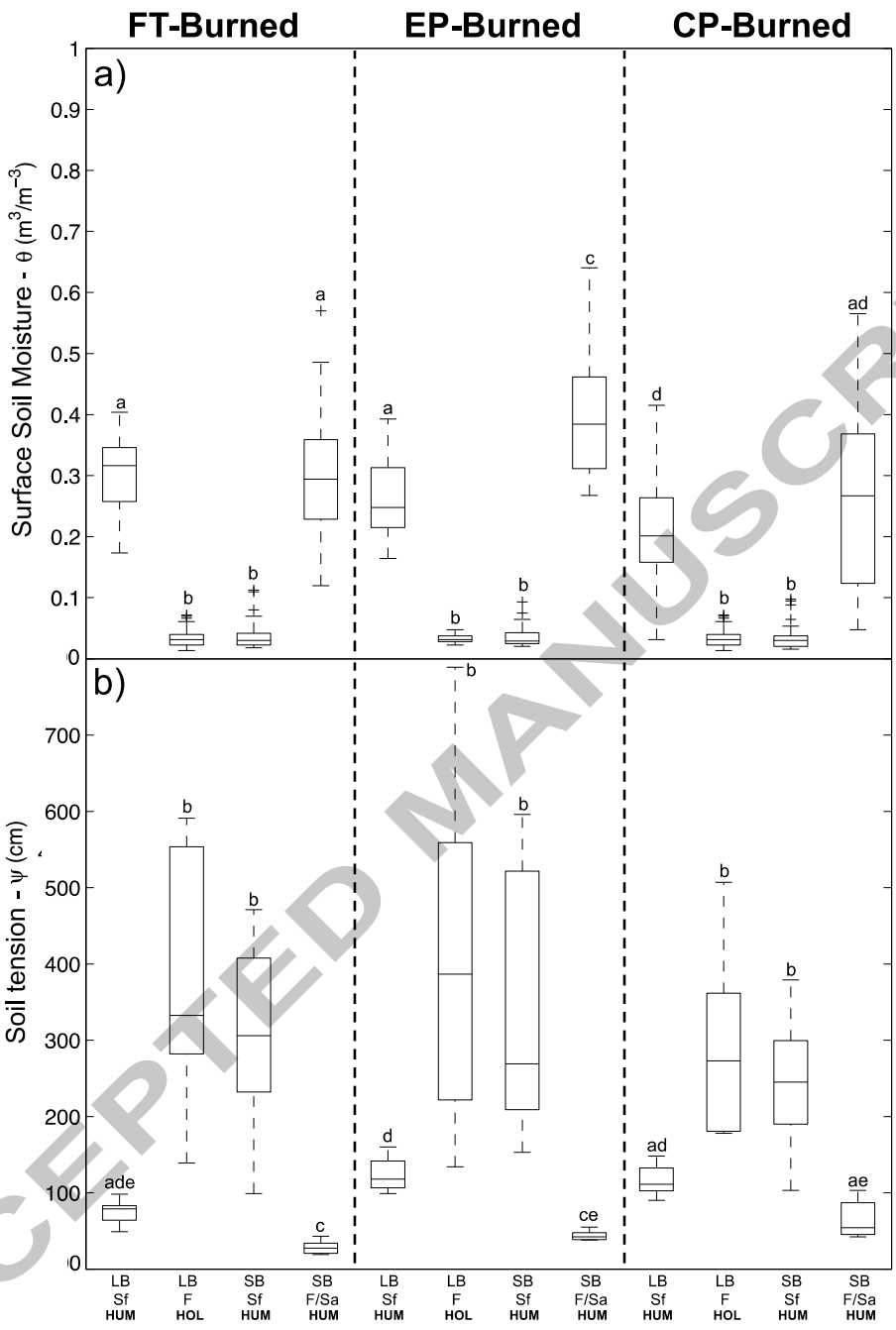

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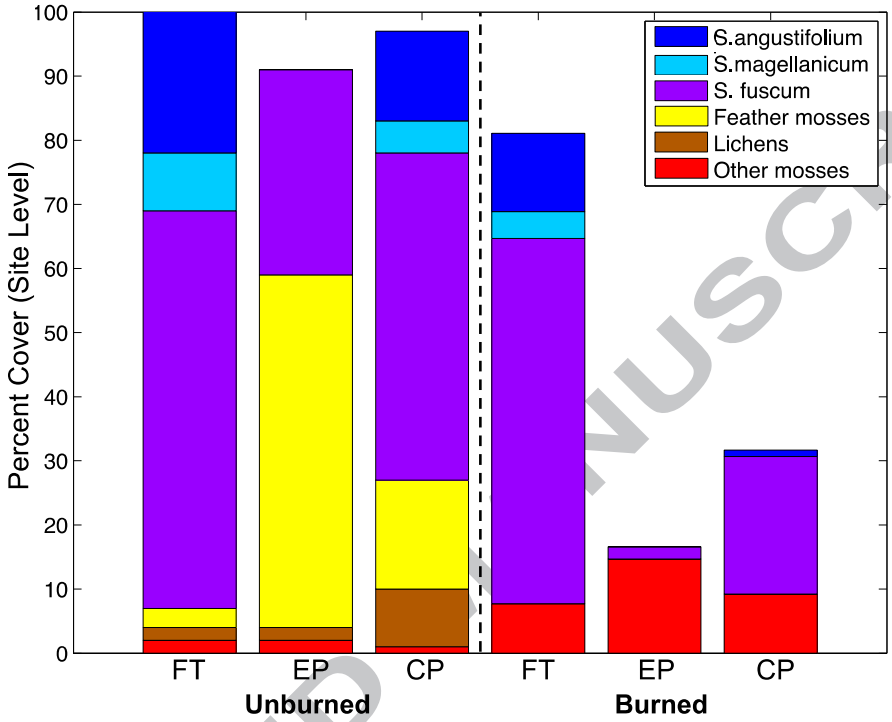
Figure 9: Percent cover of ground layer vegetation at the burned and unburned sites along the hydrogeological transect scaled to the site level by accounting for the coverage of each burn severity group (see Table 1).



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- Landscape-scale hydrological processes affect post-fire recovery in peatlands
- Hydrogeology affects peatland post-fire recovery by influencing water tables
- Hydrogeology affects peatland post-fire recovery by influencing pre-fire vegetation
- Peatlands isolated from groundwater exhibit lags in post-fire moss recolonization

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