

**ECOHYDROLOGICAL RESPONSE TO PEATLAND DRAINAGE
AND WILDFIRE**

ECOHYDROLOGICAL RESPONSE TO PEATLAND DRAINAGE AND
WILDFIRE

By

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ABSTRACT

Peatlands subjected to disturbance may undergo a dramatic alteration in ecohydrological conditions, potentially limiting the recolonisation of peat-forming species like *Sphagnum*. The aim of this thesis was to gain a better understanding of the ecohydrological response and *Sphagnum* recolonisation of a drained Alberta poor fen that also was impacted by wildfire. A portion of the poor fen was experimentally drained in the early 1980's and then both the drained and undrained portion of the peatland burned in 2001, providing an unique opportunity to examine the ecohydrological response to a peatland 'double disturbance'. The undrained site was characterized by a healthy recovery of peatland microform *Sphagnum* species, low soil water pressure (Ψ), high volumetric soil moisture (θ) content and high and stable water table position. However, the drained site showed no recolonization of *Sphagnum* with *Brome* grasses representing the dominant surface cover nine years post-wildfire. While the study period was generally wet and as such Ψ did not exceed thresholds limiting *Sphagnum* growth (≥ -100 mb) during the study period, a series of ecohydrological influences were found to be operating that limits *Sphagnum* recolonisation at the drained site. The physical structure of the peat substrate following drainage and wildfire has been considerably altered, causing changes to the moisture retention and water storage properties of the peat, largely through substantive increases in bulk density (ρ_b). Moreover, storativity and specific yield (S_y) have also decreased while at the same time the drained peat has become more humified, leaving the drained site more vulnerable to unstable water table fluctuations. As such, this has

lowered the ecohydrological ‘buffer’ or resilience to drought. Smaller decreases in θ are only needed reach $\Psi \geq -100$ mb at the drained and impose ecophysiological stress on *Sphagnum* growth. Finally, a dense canopy cover (*Betula* and *Salix*) has limited available radiation at the surface to recolonisation, shading out the surface, further limiting *Sphagnum* recolonisation.

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PREFACE

The main chapters of this thesis (2 and 3) have been prepared as independent manuscripts for submission to refereed scientific journals seeking publication. Therefore, given the independent nature of each manuscript, and in making each chapter a stand-alone text, some repetition of information is present.

1.0 INTRODUCTION

Peatlands comprise ~30 % of the North American boreal landscape, and within Canada alone, this extends to some 114 million ha (Keys, 1992). Globally, approximately 30-50 % of the soil carbon pool is stored in peatlands (Batjes, 1996; Holden, 2005; Turunen *et al.*, 2002), forming a stock of ~180 Pg of carbon within North America (Bridgham *et al.*, 2006). Peatlands are wetlands within which peat moss or partially decomposed plant litter material is accumulated over time, the rate of accumulation marginally faster than the rate of loss (Turetsky, 2003). Within Canada, peatlands are classified according to a minimum thickness of peat accumulation; 40 cm (NWWG, 1988; Zoltai *et al.*, 1988). The most common of peatland types found within Canada are bogs and fens. Bogs are the most spatially expansive of peatlands in Canada (NWWG, 1988) and are largely ombrotrophic wetland systems. Fens generally have a water table near the peat surface (Zoltai *et al.*, 1998) and receive water from surrounding catchment basins or mineral soil landscape features and precipitation. Fen classification (poor or rich) is determined by the vegetation composition of the peatland. Both poor and rich fens are common throughout Canada with treed fens constituting some 60% (Golder, 2002) of the wetlands of north-central region of Alberta, where wetlands, overall, form 65% of the regional landscape.

Treed poor fens are denoted by their vegetation structure and water chemistry (Vitt, 1994), and are largely low production systems somewhere between oligotrophic and mesotrophic states of nutrient sequestration. Poor fens are usually *Sphagnum* moss-dominated with ericaceous shrubs, able to hold an open, wooded or shrubby canopy (Halsey *et al.*, 1997), and have a porewater pH values generally between 4.5 and 5.5 (Vitt, 1990). *Sphagnum*, while it can exist homogeneously, holds a preference for an

undulating microtopography of raised hummock and low hollow microforms. Different species of *Sphagna* have specific niches within the microtopography and these features are not static during the lifetime development of a peatland (Kettridge *et al.*, 2008).

Peatlands by their nature as terrestrial ecosystems are vulnerable to disturbance. From a Canadian perspective, ~20 million ha of peatlands have been lost since 1800 (Taillefer and Wheeler, 2010), the majority through agriculture, development, extraction, forestry and mining (Government of Canada, 1991; Kaunisto and Päivänen, 1985; Price *et al.*, 2010; Van Seters and Price, 2001; Williams, 1995) with many more peatlands suffering severe degradation. In spite of their vulnerability to disturbance, peatlands can prove remarkably resilient, possessing the capacity to recover from the effects of a perturbation by altering their functioning instead of their fundamental structure, while still maintaining their key ecohydrological processes (Holling, 1973). However, exceeding thresholds of resilience usually results in an ecosystem regime shift (Belyea, 2009), often by ‘transitioning’ to an alternative ecosystem type. This has consequences for peatlands not only as unique microhabitats (Moore, 2002), but also for the fate of the stored soil carbon in these northern ecosystems.

The resiliency of peatlands has been demonstrated through a number of responses to singular disturbance events, affecting the ecosystem across several spatial and temporal scales (Holling and Gunderson, 2002). Most peatlands maintain stable moisture levels, accessible to *Sphagnum* and other moss species that form the surface of peatlands. Numerous mechanisms exist, that serve to stabilise the water table and

general hydrology through the peat substrate across temporal scales. Depth-dependent relationships of both hydraulic conductivity (K_{sat}) and specific yield (S_y) exist in peat profiles, usually in a declining log-linear fashion on account of the more decomposed, compressed peat with small pore spacings at depth (Boelter, 1965; 1968; 1969; Ingram, 1983). There are more macropores and greater elasticity in the near-surface *Sphagnum* peat (Kellner and Halldin, 2002), this architecture helps store excess water following excess rainfall and rises in the water table. The higher porosity within these surface peats stores a greater unit-volume of water for the same rise in water table. This avoids over-saturation of surface mosses. This layered structure of complementary hydrological processes stabilises the water table so that in effect, the peatland self-regulates its water table position. The main causes of water table drawdown are evaporation and transpiration. These processes hold complex interactions with the surface moisture status but can be mitigated by surface moss cover during long drought conditions. A process whereby pigmentation loss whitens mosses, becoming ‘sheep’ moss leads to an increase in albedo (Gerdol *et al.*, 1996, Benschoter *et al.*, 2011), thereby reducing net radiation at the peat surface. Crusting of moss capitula to increase surface resistance is another process, both acting to counter excessive evapotranspiration. Spatial organisation of peatland moss species (Breeuwer *et al.*, 2009) and the microtopographical features of hummocks, hollows and lawns (Belyea and Clymo, 1998; Couwenberg and Joosten, 2005) all lend themselves in some way to enhancing peatland ecohydrology resilience. Even after fire disturbance with deep levels of peat consumption, peatlands have been shown to largely recover their *Sphagnum* surface cover within 25 years, resuming ecosystem functioning (Benschoter and Vitt, 2008).

However, these mechanisms that lend peatlands their resilient nature, may become overwhelmed. Loss of the waterlogged conditions that promote peat accumulation through ongoing abiotic disturbances as well as associated decreases in soil-carbon accumulation, (Matthews and Fung, 1987; Aselmann and Crutzen, 1989; Bridgham *et al.*, 2000), combined with the predicted warming and drying at both northerly latitudes and mid-continental areas of North America (Meehl *et al.*, 2007) could overcome these conservation mechanisms. This would leave peatlands vulnerable, providing substantial feedbacks able to significantly increase levels of atmospheric greenhouse gases (Bridgham *et al.*, 1995, Gorham, 1995, Nisbet and Ingham, 1995, Moore *et al.* 1998).

Scales of disturbances vary, with accurate area surveys of disturbed peatlands non-existent. However, estimates firmly place wildfire as causing the greatest areal extent of disturbance at $44,100 \pm 1764 \text{ km}^2$ (Turetsky *et al.*, 2002) with the majority of abiotic disturbances; reservoirs and peat extraction (agriculture excluded) relatively small-scale compared to natural disruptive counterparts like permafrost degradation ($2630 \pm 105 \text{ km}^2$). Drainage is a small but growing abiotic influence on peatlands, especially as development of Canada's resource-rich north has grown, oil-sands mining easily the fastest growing disturbance, where the peatland surface is strip mined and overburden removed down to the underlying deposit sought for extraction (Rooney and Bayley, 2011). Other mining operations extract very large quantities of water, drying peatlands significantly. Fire disturbed peatlands see hydrophysical and hydrological changes similar to cutover peatlands with a shift in the surface datum through the consumption of the original upper peat surface, and potentially smouldering combustion consuming peat much deeper in the profile.

The consequence of these disturbances, drainage in particular, is to dewater the peat and develop a larger unsaturated zone. This results in subsidence of the surface peat layer through consolidation of sub-surface peat layers (Laine *et al.*, 1992; Rothwell *et al.*, 1996; Price and Schlotzhauer, 1999). The impacts of subsidence on peat carbon storage (Laine *et al.*, 1992, 1994) and nutrient cycling (Laiho and Laine, 1994) in forested peatlands are known, but the degree to which drainage affects future *Sphagnum* recolonisation are less well known.

Peatland drainage alters the physical peat structure, leads to compaction and enhances peat decomposition, increasing peat bulk density and increasing small pore size distributions. This in turn reduces the specific yield (S_y), which is the relationship between the quantity of water added or removed from the soil and resulting change in water table (Boelter, 1964), this lowers overall water storage capacity (Schouwenaars and Vink, 1992) of the peat. As time since disturbance increases, and without remediation or an environment fostering recovery, the effects of ‘primary consolidation and secondary compression’ as well as oxidation of the aerobic peat and shrinkages in peat volume continue to change the peat structure (Hobbs, 1986; Price and Schlotzhauer, 1999; Schothorst, 1977). Deeper mean water tables arise from the decrease in S_y , caused by the collapse of macropores and overall decrease in mean pore size (Okrusko, 1995), the decrease in S_y allows for greater evaporation to the atmosphere, which can lead to more rapidly fluctuating water tables (Price, 1996). Moreover, the decreased storage capacity combined with the larger unsaturated zone, makes drying of the peat a more frequent occurrence (Price, *et al.*, 2003). Furthermore the loss of large diameter macropores, concurrent with an increase in smaller, narrower pores, increases the water retention (Boelter, 1968) and lowers K_{sat}

throughout the peat profile (Boelter, 1965; Päivänen, 1973). This decreases free-water accessible to *Sphagnum* through capillarity. Capillarity can be typified by the movement of water up a wick. It being the height of water rise in a capillary tube (or plant stem) due to water molecule adhesion to the sides of the tube, spreading along the sides of the tube, and thus up the tube, demonstrating the attraction (on surfaces with low contact angles). The rise height is inversely proportional to the tube radius, the density of the liquid, and directly proportional to the surface tension of the liquid. Rise occurs because the pressure acting on the meniscus of liquid in the tube is less than that on the surrounding free water.

Sphagnum is reliant on capillarity through its pendant stems and clusters, comprised of branches for drawing moisture (Thompson and Waddington, 2008) and ensuring a steady moisture supply. The capillary pressures generated are sufficient when moisture is at readily accessible pressures, but beyond -100 cm equivalent water table, or -100 mb soil water pressure (Ψ), the capillarity cannot overcome the external Ψ pressures (Hayward and Clymo, 1983) on account of the water film breaking within the internal branch-stem linkages of *Sphagnum*. This causes the intra-cellular hyaline cells to drain out, desiccating the plant. This readily available moisture supply is at risk from the aforementioned changes to the underlying hydrophysical peat properties. This may be compounded by the structural effects on the biology of *Sphagnum* that high (very negative) soil water pressure values may have. High Ψ values causing the menisci of water to expand inwards, to the centre of the hyaline cells, this can break down the cell walls and allow outside air in to the cell, dewatering and damaging the cells (Lewis, 1988).

Despite the effects disturbance has on peat substrate from either wildfire, cutover or drainage disturbance, both *Sphagnum* diaspores and fragments, i.e. *Sphagnum* lacking active capitula (Baker and Boatman, 1985; Rochefort *et al.*, 1995) can recolonise bare peat, often forming ‘cushions’ in areas able to sustain *Sphagnum* (Robert *et al.*, 1999; Girard *et al.*, 2002). However, for *Sphagnum* to recolonise there are hydrological requirements; the 2 cm of near-surface peat soil ideally maintaining the aforementioned low matric soil water pressure (Ψ). However, the time periods at which Ψ may drop below -100 cm (below -100 cm considered to be below the threshold at which recolonisation can occur) and recolonisation still occurs are unknowns. Nor is known to what degree seasonality influences the effectiveness of the -100 mb threshold. It has been shown in laboratory based studies that steady, medium-to-high soil moistures (~ 75 %) is ideal in facilitating recolonisation. Both lab and field studies have demonstrated that ideal conditions for *Sphagnum* reestablishment occurs when the water table remains within 0 and -10 cm of the surface.

Mosses are largely shade-adapted with low levels of photosynthetic light saturation, irradiances of 30 to 300 $\mu\text{mol m}^{-2}\text{s}^{-1}$ (Davey and Rothery, 1997) common among bryophytes. However, *Sphagnum* show greater saturation tolerances, *spp. rubellum* having a limit of around 550 $\mu\text{mol m}^{-2}\text{s}^{-1}$ (Marschall and Proctor, 2004) meaning a tolerance for more net radiation at the surface. All disturbed sites tend to undergo quick re-growth from vascular or woody species in the time post-disturbance; this can quickly shade out understory species at the peat surface. In studies of N-limited environments, such total shading of the surface has been shown to significantly reduce biomass of *Sphagnum (capillifolium)* mosses (Bonnet *et al.*, 2010). Moreover, if the

spontaneous vascular and woody vegetation that quickly colonise post-disturbance (Graf *et al.*, 2008) shade-out pioneer mosses like *Polytrichum strictum* (a species shown to regenerate more readily under partial shading; Graf and Rochefort, 2010), the likely success of *Sphagnum* returning as dominant surface cover is diminished further still (Groeneveld *et al.*, 2007). *Polytrichum* acts as a nurse-plant, especially in hydrologically stressed peatland environments like cut-over sites, stabilising the microclimate and peat substrate for other moss species.

Whether peatlands will be resilient ecosystems in the face of increasing severity and frequency of disturbances in the future and the role disturbance will play in altering the large sinks for terrestrial and atmospheric carbon in Canada are important questions to address in peatland ecohydrology. A better understanding of how multiple disturbances can compound their effects to alter hydrological thresholds, critical to *Sphagnum* species reestablishment, and how they may influence recovery in degraded peatlands will enable better prediction of peatland areas in decline or at risk of total loss. Furthermore, findings will aid active restoration literature with a baseline data of how double-disturbance change key hydrophysical properties, better directing management of restoration projects. *Consequently*, the objective of this thesis is to gain a better understanding of the processes limiting natural *Sphagnum* recolonisation by investigating the ecohydrological response of the drained and undrained portion of the Sauleaux River Peatland that burned in 2001. Specifically, the thesis objectives are to:

- 1.) Understand and quantify soil moisture content, water table position and soil-water pressure relationships of a drained and undrained portion of a burned, *Sphagnum*-dominated fen complex;
- 2.) Relate these hydrological conditions to existing patterns of *Sphagnum* recolonisation, together with other biotic factors and investigate the causal relationships between them;
- 3.) Understand the effects that differing peat hydrophysical properties between drained and undrained, burned peat substrate have on water availability for *Sphagnum* recovery;
- 4.) Use these findings to demonstrate how peatlands may be at risk of decline or loss from the hydrological changes brought about by consecutive or double disturbances.

2.0 THE EFFECT OF PEATLAND DRAINAGE AND WILDFIRE ON PEAT HYDROPHYSICAL PROPERTIES: IMPLICATIONS FOR *SPHAGNUM* MOSS RECOVERY

ABSTRACT

Peatlands are generally resilient to natural and human disturbances, but concurrent multiple disturbances can dramatically impact peat hydrophysical properties that greatly reduces *Sphagnum* moss recolonisation success. Loss in *Sphagnum* cover in bogs and poor fens can lead to a loss in hydrological and ecosystem services such as long-term carbon sequestration. We determined the hydrophysical and moisture retention properties for peat cores removed from a partially drained and burned, forested poor fen in Alberta, Canada to examine how double disturbance impacts peatland ecohydrological resilience. The compounded effect of drainage followed by wildfire was to shift the surface datum down the peat profile, revealing denser peat to combustion, losing less-dense near-surface peats that regulate water table position and near-surface moisture content, both favourable to *Sphagnum* recolonisation. Specifically, bulk density (ρ_b), humification class and (surface) water retention at the drained peatland post-fire were 14.1% greater, two humification classes higher, and 15.6% greater respectively, compared to the adjacent undrained, burned portion of the fen. We present a conceptual depth-of-burn model using post-fire water retention data to demonstrate the existence of a smaller “hydrological buffer” (lower ecohydrological resilience) at the drained portion of the fen, the denser peat requires 15 % less water loss to reach a critical soil water pressure threshold for *Sphagnum* recolonisation. We conclude that multiple disturbances reduce peatland ecohydrological resilience, lose ecosystem services, and induce hydrological

limitations inhibiting of *Sphagnum* recovery. This potentially renders peatlands more vulnerable to further disturbance – specifically wildfire, owing to a larger fuel-to-energy sink ratio (peat:water), for a given water table position.

2.1 INTRODUCTION

Northern peatlands represent a long-term sink of atmospheric carbon storing ~ 220 – 460 Pg (Gorham, 1991; Turunen *et al.*, 2002), accounting for about one-third of the global soil carbon store (Batjes, 1996; Turunen *et al.*, 2002) and therefore play an important role in the global carbon cycle. The response of these ecosystems to natural or human disturbances has recently received increasing attention as this long-term carbon sequestration ecosystem service may be vulnerable (e.g., Waddington *et al.*, 2002). In a Canadian context, peatland drainage for silviculture (Kaunisto and Päivänen, 1985; Silins and Rothwell, 1999), peat extraction (Van Seters and Price, 2001), agriculture (Williams, 1995) and mining (Price *et al.*, 2010), represents the largest human disturbance to Canadian peatlands, while wildfire represents the largest natural disturbance to Canadian peatlands by area (Turetsky *et al.*, 2004). However, while some research suggests that peatland ecosystems are sensitive to disturbances (e.g., Ise *et al.*, 2008), peatlands appear to be predominantly resilient to disturbance (Robroek *et al.*, 2010; Strack and Waddington, 2007) due to strong negative feedbacks that are often manifested by changes in the hydrophysical properties of the peat (e.g., Thompson and Waddington, *submitted*; Whittington and Price, 2006). For example, following peatland drainage, peat previously stored under predominantly anaerobic conditions is exposed to aerobic conditions (Girard *et al.*, 2002; Price, 1997). This expedites structural changes to the peat hydrophysical properties such as bulk density

(ρ_b), porosity (ϕ), specific yield (S_y) and water retention which results in reduced lateral water losses (Whittington and Price, 2006) and higher water retention (Silins and Rothwell, 1998). Following drainage, peat oxidation (Waddington and Price, 2000), shrinkage of the peat structure (Kennedy and Price, 2005) and compression (especially at depth), from the overburden weight of unsaturated peat (Schothorst, 1977; Silins and Rothwell, 1998) all act to increase peat ρ_b . This is a finding not too dissimilar to wildfire where several cm of the near-surface peat is lost to combustion, resulting in peat with higher ρ_b (and hence moisture retention) at the surface (Thompson and Waddington, *submitted*). Nevertheless, these changes in the peat properties also serve to maintain the high near-surface moisture contents necessary for *Sphagnum* moss re-establishment (Joosten, 1992; Petrone *et al.*, 2004; Price and Whitehead, 2004) following peatland disturbance. Bogs and poor fens, in particular, are highly dependent upon *Sphagnum* being able to recolonise post-disturbance to maintain their high ecohydrological resiliency, this function itself controlled by the peat possessing suitable hydrophysical properties. *Sphagnum spp.* comprise around ≥ 70 % of surface vegetation in peatlands (Benscoter and Vitt, 2008), fixing carbon (Andrus, 1986), facilitating new peat formation (Turetsky, 2003) and maintaining northern peatlands status as carbon sinks. Hence *Sphagnum's* ability to recover post-disturbance and initiate recolonisation is essential (Waddington *et al.*, 2002).

Cutover peatlands, where the peatland is first drained and the upper few dm of *Sphagnum* and surface peat are harvested, often result in dramatic changes in ecosystem form and function (e.g., Waddington *et al.*, 2002) with the ecosystem switching to a large and persistent source of carbon dioxide. Moreover, these cutover peatlands require active restoration efforts (e.g., LaRose *et al.*, 1997; Price *et al.*,

2002; Wind-Mulder *et al.*, 1996) to enable *Sphagnum* moss recolonisation and to return these ecosystems to net carbon sinks (Waddington and Warner, 2001). In effect this ‘double disturbance’ (drainage and extraction of surface peat) leads to much more dramatic changes in the hydrophysical properties of the peat with a doubling of ρ_b and an order of magnitude decrease in S_y (Kennedy and Price, 2005) generating near-surface soil-water pressures, often in excess of the ‘wilting point’ of *Sphagnum*; -100 mb, (Price and Whitehead, 2001), thereby inhibiting *Sphagnum* recolonisation (Price and Whitehead, 2001). In effect, the double disturbance exceeds the ecohydrological resilience of the peatland.

The frequency of these “double disturbances”, where the ecosystem services of peatlands are severely degraded or lost altogether are highly likely to increase in many regions as climate change and land-use change disturbances, either in quick succession or simultaneously, act together to decrease peatland ecohydrological resilience. The wildfires that consumed many drained Russian peatlands in 2010 (c.f. Zaidel’man, 2011) represent an example of this double disturbance. Once again in a Canadian perspective, future boreal wildfire are predicted to intensify considerably (Flannigan *et al.*, 2001; Bergeron *et al.*, 2004) with a greater likelihood of more severe and persistent drought conditions (Turetsky, 2010). We suggest therefore, that drier peatlands are more vulnerable to wildfire, with similar implications for other combinations of disturbance, though none on the same spatial scale as wildfire. Recent research suggests that peat that is well-humified and of high bulk densities, are likely more vulnerable to fire smouldering and consumption (e.g. Benscoter *et al.*, 2011; Thompson and Waddington, *submitted*), and less likely to enable *Sphagnum* recolonisation post-fire due to their unsuitable moisture retention properties

(Thompson and Waddington, *submitted*). We hypothesized that peatlands that have been drained and subsequently burned would be less likely to maintain hydrological and hydrophysical properties that would allow or maintain *Sphagnum* recolonisation. To examine this, we took advantage of a unique opportunity to study the combined effects of drainage and wildfire as a double disturbance at the Saulteaux River peatland complex (SRP), a partially drained poor fen in Alberta, Canada. This peatland has a large background of hydrophysical data on peat properties (Hillman, 1987; Tóth and Gillard, 1988; Swanson and Rothwell, 1989; Rothwell, Silins and Hillman, 1996; Silins and Rothwell, 1998; 1999) both before and after drainage. SRP later burned during an expansive wildfire event in 2001 and eight years after burning, no *Sphagnum* has recolonised at the drained portion of the complex. Previously undisturbed peatlands have shown recolonisation occurring within a year of fire and can be considered recovered within 25 years (Benscoter and Vitt, 2008). *Sphagnum* species have recovered at the undrained portion of the SRP post-fire. Consequently, the aim of this research is to examine the cumulative effects of peatland drainage and wildfire on peat hydrophysical properties at an Albertan peatland with known pre-fire hydrophysical properties in order to identify hydrophysical controls on stressors to *Sphagnum* recolonisation in highly disturbed peatlands. Determining the changes in post-fire peat hydrophysical properties (e.g., ρ_b , S_y , moisture retention, humification and hydraulic conductivity) will help gain a greater understanding as to whether peatlands can recover from double disturbances. This will aid in identifying peatlands at risk of ecosystem collapse post-disturbance and potentially direct the development of remediation plans to mitigate the loss of important peatland ecosystem services.

2.2.1 STUDY SITE

This study was conducted in the Sauleaux River peatland (SRP), in the Lesser Slave Lake region of north-central Alberta, Canada (55°8'N, 114°15'W). The SRP is a 100-ha treed poor fen and sits atop shale and glauconitic silty shale formations in the Lesser Slave Lake Lowlands (Swanson and Rothwell, 1989). Peat depth varies across the site between 0 to 4 m deep (Swanson and Rothwell, 1989), with western-boreal peatlands having a burn frequency of 70-100 years (Kuhry, 1994), it may have burnt as many as 40 times over the course of its formation. In 1987 a 50-ha portion of the SRP was drained (Figure A1.2) through the construction of 0.9 m deep ditches separated by 40 m or 25 m (Hillman, 1987) resulting in central plateaus in-between ditches (Figure A1.2) These plateaus do not have baulk ridges that are common in harvested peatlands (see Van Seters and Price, 2001).

Both the drained and undrained portions of the fen complex burned in 2001 during a large-scale fire event (LWF-063) (Quintilio *et al.*, 2001) resulting in the consumption of 19 ± 3 cm of surface peat in the drained portion of the peatland (Turetsky *et al.*, 2011) and 7 ± 1 cm at the undrained portion, determined by the adventitious root method of Kasischke *et al.* (2008). The undrained portion of the SRP is dominated by *Aulacomnium palustre*, *Sphagnum fuscum*, *S. magellanicum* and *S. angustifolium*. *Sphagnum* mosses are absent from the drained site but existed post-drainage and pre-fire. *Brome* grass species are the dominant surface cover at the drained site, with a dense woody overstory canopy of *Salix* and *Betula* species (Figure A1.5).

2.2.2 METHODOLOGY

In late August, 2010 we randomly removed nine 0.5 m deep soil cores from each of the drained and undrained sites. Cores were removed using sharpened PVC cylinders

and stored in capped, 10.1 cm (o.d.) PVC cylinders and frozen before transport thereby minimising impact following return to the lab. Minimal compression was observed in the field while sampling the cores. Each core was sectioned into 50 mm tall ‘peat pucks’ while frozen with a surgical band saw. Peat pucks were then thawed and retained within a 50 mm thick PVC ring with a secured porous muslin cloth base to minimise handling losses, maintain puck structure and aid in hydraulic contact between sample and the porous plate (see below) (Klute, 1986).

The peat pucks were saturated in de-aired de-ionised water for 24 hrs to allow the peat to fully saturate before being subject to negative suction pressures for determination of the moisture retention characteristics. This used a system of daisy-chained Soilmoisture Equipment Corp. (Santa Barbara, CA) model 1725D22 saturated porous plates with an air-entry pressure of 1 bar connected to a central constant -300 mb vacuum reservoir. Pressure was maintained via a hydrostatic buffering system to minimise variation in set pressures. The peat pucks were subject to constant negative pressures of 5, 10, 15, 20, 30, 40, 50, 75, 100, 150 and 200 mb for 24 hrs or until water outflow had ceased at each pressure step to derive moisture retention curves. Peat pucks were weighed following equilibration at each pressure step, the plates misted with water to re-establish hydraulic contact between pucks and the plate surface at start of each pressure increment in a method modified from Cagampan and Waddington (2008). Sealed acrylic chambers held both porous plates and samples, open Tupperware containers of water within the chambers to eliminate evaporation potential by maintaining a high RH% environment, the chamber sides and lid were misted to reduce condensation potential. Peat pucks were measured for volumetric shrinkage both after saturation and after oven drying after completion of all pressure

steps. Volumetric moisture content (VMC) was determined by weighing each peat puck on a laboratory scale at the end of the equilibration period of each pressure step and from the measured volume of the peat pucks. Water content at saturation (θ_s) ($\Psi = 0$ mb) was determined by calculating the porosity of each puck sample from the bulk density and using a particle density value of 1.47 (Päivänen, 1973). Following all moisture retention measurements the peat pucks were dried for 72 hours in an 85°C drying oven to constant mass to determine bulk density. Peat pucks woody content (volume) was then determined by manually separating peat mass from remnant wood debris. Moisture retention curves were corrected for their non-peat volume.

Peat decomposition was classed using the Von Post decomposition scale for each peat puck according the method prescribed by Stanek and Silc (1977) based on discernable remnant plant material, homogeneity of structure and free water content categorises degree of composition into ten identifiable classes (H₁-H₁₀). Upon completion of drying for bulk density, the pucks were broken up with the aid of a pestle and grain-size sieves to separate woody fraction from true peat. Woody fraction by volume was determined through displacement in a measuring cylinder using a sieved plunger. Each sample was also then classified as per Von Post's wood content classes (Stanek and Silc, 1977), root quantity was also classed at the same time to Von Post's root-thread classes. There are four possible categories for wood content and root thread classes from 'none' (V₀/R₀) to 'very many' (V₃/R₃). Specific yield ($S\gamma$) (\leq total porosity) was also determined through the equation:

Equation 2.1. Specific yield ($S\gamma$)

$$S\gamma = \frac{(M_s - MD)/\rho_w}{M_s/\rho_w}$$

Where M_s is the saturated sample weight, M_D the drained sample weight, and ρ_w is the assumed density of water (1 g cm^{-3}).

K_{sat} (saturated hydraulic conductivity) was determined using two methods. Shallow K_{sat} (0-50 cm) was determined in the lab using two 0.5 m core profiles for each site, the peat pucks derived from the profiles were saturated and then encased in wax, held within a 20 cm tall section of ‘Sonotube’ using a variation of the modified cube method (Beckwith *et al.*, 2003) for constant hydraulic head. A constant head apparatus was used to maintain constant head. High peat bulk densities allowed for proper encapsulation of the peat puck and meant minimal risk of preferential flow along the sides of the Sonotube. The tests were carried out before the peat pucks were placed under negative pressure on the plates for water retention so as to avoid altering the internal pore and pathway structure that would change K_{sat} . Deep K_{sat} values (depths of 0.3, 0.5 and 0.7 m) were measured in the field with piezometers using slug tests and logging pressure transducers. Piezometers for deep K_{sat} were constructed of 29 mm (i.d.) PVC conduit based on the method design from Surridge *et al.* (2005). A 29 mm (i.d.) PVC plug conduit capped the conduit at the base of the routed intake slots (75 mm in length), intakes were stacked in pairs, with a 10 mm gap between them to maintain rigidity in the conduit, this design ensured at least 70 % of the conduit intake was open to water flow. Piezometers were installed using a sharpened steel tube of slightly smaller diameter of the piezometer conduit. Infill was removed to the depth of measurement (0.3, 0.5 and 0.7 m) before fully developing the piezometers to unblock pores clogged through smearing and remnant debris from installation. K_{sat} was

measured at both sites on 26th August 2010, with the piezometers installed 4 m parallel to existing instrumented transects.

2.2.3 RETC model

The RETC soil water retention model (van Genuchten *et al.*, 1991) was used to estimate the van Genuchten parameter estimates (α , n and m) from the moisture retention curves, these derivations the shape fitting parameters for the unsaturated retention curve. The parameters α and n control the shape of the retention curve, where α effectively moves the curve up and down the suction axis, with great α values meaning the same moisture content at a smaller suction value. N controls the slope of the retention curve, a larger N means for a very large change in moisture content with a small change in suction above suction at air entry (van Genuchten *et al.*, 1991). The parameters can indicate tendencies of hydrophysical properties within the sampled soil, specifically pore size distribution. Each pressure step and corresponding θ value was assigned an equal weighting of 1.

2.2.4 Statistics Rationale

Statistical tests varied according to the parameters of the data tested; continuous, discrete, categorical and whether the data was normally distributed or not. The majority of tests of statistical difference were either one-way ANOVA or t -tests for continuous datasets, usually with Mann-Whitney U test for discrete data. Categorical data such as von Post decomposition class was tested with χ^2 tests. Tests were run within MINITAB 15 or Sigmaplot 11.0 according to the dataset format. Cause and effect were determined either through logistic regression for discrete data or linear regressions for continuous data. Multiple and stepwise regressions was used where

there were >1 effect but 1 cause. Multivariate tests (mainly Principle Component Analysis) were used to explore the variables within the datasets.

RESULTS

2.3.1 Porosity, bulk density and specific yield

Surface peat samples (0-5 cm) had significantly higher porosity (ϕ) ($p = < 0.001$) than deeper samples (5-50 cm) at both sites. Surface ϕ ranged between $96.3 \pm 3.4 \%$ for undrained and $92.8 \pm 2.2 \%$ for the drained site. There was no significant difference ($p = 0.375$) in 25-50 cm ϕ between the two sites.

Bulk density at the surface (0-5 cm) depths ($79 \pm 49 \text{ kg m}^{-3}$) was significantly lower ($p = < 0.001$) than greater depths (5-50 cm) ($141 \pm 30 \text{ kg m}^{-3}$) for both the drained and the undrained site. Near-surface (5-25 cm) bulk density ($119 \pm 29 \text{ kg m}^{-3}$) was significantly lower ($p = < 0.001$) at the undrained site and had lower median and maximum values ($124, 164 \text{ kg m}^{-3}$) than the drained site which averaged $146 \pm 32 \text{ kg m}^{-3}$ and had a median and maximum of 145 and 197 kg m^{-3} , respectively. At depths deeper than 25 cm, the differences were not significantly different ($p = 0.1$) with the drained site ($149 \pm 22 \text{ kg m}^{-3}$) only marginally greater than the undrained ($141 \pm 17 \text{ kg m}^{-3}$). Overall, across all depths (0-50 cm) the sites were significantly different ($p = 0.001$) with the 25% percentile values at 109 and 123 kg m^{-3} for the undrained and drained sites respectively and the 75% percentile values at 148 and 164 kg m^{-3} , respectively.

Surface (0-5 cm) Sy was significantly higher ($p = < 0.001$) than the rest of the profile (5-50 cm) at both sites (undrained = $0.54 \pm 0.16 \text{ m}^3 \text{ m}^{-3}$, drained = $0.40 \pm 0.13 \text{ m}^3 \text{ m}^{-3}$).

At the undrained site average upper (0-25 cm) S_y ($0.22 \text{ m}^3 \text{ m}^{-3}$) was significantly greater ($p = <0.001$) than average lower (25-50 cm) S_y ($0.13 \text{ m}^3 \text{ m}^{-3}$). At the drained site the difference between average upper ($0.17 \text{ m}^3 \text{ m}^{-3}$) and lower S_y ($0.08 \text{ m}^3 \text{ m}^{-3}$) was significant ($p = 0.001$). The average (0-50 cm) undrained S_y ($0.20 \pm 0.16 \text{ m}^3 \text{ m}^{-3}$) was significantly greater than the drained site ($0.16 \pm 0.13 \text{ m}^3 \text{ m}^{-3}$). The drained site had lower minimum S_y of $0.004 \text{ m}^3 \text{ m}^{-3}$ compared to $0.007 \text{ m}^3 \text{ m}^{-3}$ at the undrained site.

2.3.2 Von Post class

Von Post decomposition classes of low humification (H_1 - H_5) formed just 5.3 % of peat pucks at the drained site, but 29.5 % of peat pucks at the undrained site (Figure 2.1). For the most decomposed classes (H_9 , H_{10}) only 9.5 % of peat pucks were classed as H_9 at the undrained site, with none classifiable to the H_{10} class (Figure 2.1). However, at the drained site, 48.6 % of all peat pucks were classifiable into the H_9 and H_{10} classes, approximately four times greater than that of the undrained. For wood content class, the sites were significantly different in frequency distribution ($p = < 0.001$). The drained site had higher frequency of the V_3 (very woody) category than the undrained site throughout all depths within the peat puck profiles with V_3 category occupying 28.9 % of the observed counts against only 8.3 % at the undrained site. The undrained site had the greatest percentage of V_0 counts at 45.2 % against 23.6 % at the drained. For root thread classes the drained site was approximately double the equivalent percentage at the R_1 , R_2 and R_3 classes only being less at the R_0 (no root threads) class, 73.8 % at the undrained to 52.6 % at the drained site.

2.3.3 K_{sat}

There was no significant difference in K_{sat} between the drained and undrained sites for any soil depth. Surface (0-5 cm) K_{sat} was marginally greater at the drained site ($5.1 \times 10^{-4} \text{ m s}^{-1}$) compared to the undrained site ($3.7 \times 10^{-4} \text{ m s}^{-1}$). While K_{sat} for depths 5-30 cm were consistently lower at the drained site ($5.7 \times 10^{-5} \pm 5.0 \times 10^{-5} \text{ m s}^{-1}$) compared to the undrained ($1.0 \times 10^{-4} \pm 6.0 \times 10^{-5} \text{ m s}^{-1}$). At depths below 30 cm the two sites were similar with minimal standard deviation between values at either site (drained = $1.0 \times 10^{-4} \text{ m s}^{-1}$, undrained = $2.0 \times 10^{-5} \text{ m s}^{-1}$), with the deepest peat pucks having the lowest K_{sat} values (drained = $9.9 \times 10^{-6} \text{ m s}^{-1}$, undrained = $2.2 \times 10^{-5} \text{ m s}^{-1}$).

Piezometers located at 0.3, 0.5 and 0.7 m depths in the peat determined lower K_{sat} at the drained site for 0.3 m depth, 3.4×10^{-5} compared to $3.9 \times 10^{-4} \text{ m s}^{-1}$ at the undrained. K_{sat} at 0.5 and 0.7 m depths was found to be lower at the undrained than at the drained site, 7.6×10^{-5} and $4.0 \times 10^{-7} \text{ m s}^{-1}$ at the undrained to 1.1×10^{-4} and $2.7 \times 10^{-6} \text{ m s}^{-1}$ at the drained site. Both sites saw K_{sat} decline at lower depths ($> 0.5 \text{ m}$).

2.3.4 Soil moisture retention

Residual moisture content at -200 mb (θ_r) differed significantly ($p = 0.006$) between sites for surface (0-5 cm) peat pucks, $28.3 \pm 12 \%$ and $13.3 \pm 8.3 \%$ at the drained and undrained, respectively. The sites did not differ significantly when considered over all sampled depths. Between upper depths (5-25 cm) the two sites did not differ significantly ($p = > 0.05$) nor did they differ significantly ($p = > 0.05$) at lower depths (25-50 cm). At θ_{-100} (θ at -100 mb), over all sampled depths (0-50 cm) the two sites did not differ significantly ($p = > 0.05$), while for surface (0-5 cm) peat pucks at θ_{-100} the two sites were significantly different ($p = 0.008$). Drained site θ_{-100} mean for surface peat pucks was $30.5 \pm 12.7 \%$ with the undrained mean at $14.9 \pm 9.5 \%$. At

neither the upper (5-25 cm) nor lower (25-50 cm) depths did the two sites differ significantly ($p = > 0.05$). Saturated water contents (θ_s) between sites for all measured depths were significantly different ($p = 0.006$) with values of $91.5 \pm 2.8 \%$ and $90.3 \pm 2.0 \%$ at the undrained and drained sites, respectively. The undrained site had higher maximum θ_s values ~ equal to maximum porosity at 99 % where as the drained sites maximum was lower at 95.4 % but with a higher minimum θ_s than the undrained site (86.6 % vs. 81.9 %). θ_s was significantly greater at upper depths between 0-25 cm, at both sites. Soil moisture retention curves for surface and upper (≤ 25 cm) depths show greater ranges of wetting and drying for the same change in pressure against peat pucks from ≥ 25 cm in the core profiles (Figure 2.2). Differences in retention variation were greatest in surface (0-5 cm) peat pucks, ~ 100% greater than that of peat pucks from deeper in the profile. Undrained θ_{-100} differs more between shallow (0-25 cm) and deep (25-50 cm) ($p = 0.041$) than drained site ($p = 0.08$), with mean % moisture loss needed to equilibrate at θ_{-100} at 0-5 cm 18 % greater at the undrained than drained, $81 \pm 11 \%$ and $63 \pm 17 \%$ respectively.

The van Genuchten parameters (α and n) that control retention curve shape, varied considerably between sites and depth but only n as a parameter maintained a significant relationship, decreasing with increasing depth ($p = < 0.001$) at the drained site only, due to high values (1.1) in the 0-10 cm depth. A decline in n was observable at both sites, lower at the drained (mean $n = 1.09$) than the undrained (mean $n = 1.11$). Alpha decreases monotonically with depth ($p = 0.01$) at the undrained site but only when treated as 10 cm increment profiles as per Silins and Rothwell (1998). No significant relationship is observable when sampled as 5 cm peat pucks. No statistically significant depth dependent relationship exists at the drained site, with

mean α lower at the drained site (mean = 213.8, mean α at undrained = 399.3), both sites skewed by the very high surface (0-10 cm) α values skewing depth profiles for α . The α values maintained their pre-fire status of the drained site having lower air-entry matric potential than the undrained, but this potential saw a (relative) ~ 1200 % increase post-fire at the drained and ~ 270 % increase at the undrained. High r-squared values (≥ 98 %) indicated good fit between the shape parameters and the drying portions of the moisture retention curves when derived using RETC.

2.4 DISCUSSION

Physical peat properties

Surface and near-surface peat properties differed significantly between sites, specifically; ρ_b , ϕ and S_y that pertain to peat matrix structure and thus water retention and holding capacity (Clymo, 1983). Average ρ_b was higher, but not significantly so ($p = > 0.05$), at the drained site, likely due to the absence of recolonised *Sphagnum*. *Sphagnum* moss at the surface is characterized by low bulk density values (Hayward and Clymo, 1982). The surface of the undrained site, comprising of *Aulacomnium* and *Sphagnum spp.* from hummock with lower ρ_b values (Figure 2.3). The ρ_b values in the study fell within the ranges observed for fibric peats in natural fens (Boelter, 1968); and are consistent with *Sphagnum* dominated peatland microtopography having slower rates of decomposition (Johnson and Damman, 1991; Verhoeven and Toth, 1995).

Following the research conducted by Silins and Rothwell (1998), mean ρ_b in the upper 40cm of the peat profile has increased by 9.4 % and 55.7% at the drained and undrained SRP sites, respectively. ρ_b was lower at the surface and higher at depth;

consistent with widely held assumption that ρ_b generally increases with depth (Zoltai, 1991). However, completely monotonic increases of ρ_b with increasing depth were absent, with all sampled profiles showing some non-linearity of ρ_b with depth. This is likely in part due to wildfire altering the profiles through combustion and consumption of surface peat layers (Benscoter *et al.*, 2011), shifting the surface datum to a new position previously lower in the depth profile. As shown (see Figure 2.1; 2.3), this new datum is comprised of higher bulk density and decomposition class, consistent with the findings of Silins and Rothwell (1998) but with greatly exaggerated values post-fire. Such depth-wise transitions in the profile may be created by hummocks reverting to hollows following fire disturbance (Benscoter *et al.*, 2005). This introduces zones of lower density peat into the profile between peat layers of higher densities. Presence of woody debris at the drained site may also account for this variance, aerating the drained site peat and adding to the depth-wise variation in ρ_b already shown (Figure 2.3), though regression analysis found no causation effect between the two variables. Moreover, large rooting zones from greater tree and vascular growth are also contributing to changes in the peat structure. Vigorous growth of *Picea* post-drainage as well as a *Betula spp.* understory was noted by Silins and Rothwell (1999) increasing this woody fraction within the peat soil (Rothwell and Silins, 1990) through litter, and root casts of remnant root networks remaining in the soil after the stands burned.

Bulk density and humification show significant linear relationships ($p = < 0.001$) consistent with other drainage experiments (Silins and Rothwell, 1998; Hillman and Roberts, 2006). ρ_b behaves as a function of mean pore size (Boelter, 1968), with changes to humification classes a direct influence on this relationship, linear

regressions ($p = < 0.001$) showing this influence on the functional relationship significantly at both SRP sites. We found that humification classes in the SRP increased post-fire to a mean of H₈ (drained), H₆ (undrained) from H₅ and H₄ respectively, post-drainage (Silins and Rothwell, 1998). The post-drainage H_{class} values derived from rubbed fibre % through conversion tables (Green *et al.*, 1993; Luttmerding *et al.*, 1990; Soil Classification Working Group, 1998). While classified as very humified, the drained site peat pucks also possessed properties of earthified peat akin to European fen peat soils after drainage and intensive usage (high degree of decomposition, muck-like consistency, with fine granular (crumb) structure when dried forming a dusty consistency) (Schwärzel *et al.*, 2002). These changes may occur because the peat dewateres with shrinkage, compression (Price and Schlotzhauer, 1999; Silins and Rothwell, 1998) and subsidence observed post-drainage and then subsequently the consumption by fire, operate to increase bulk density. Consequently, mean H_{class} increases, through the introduction of aerobic conditions to the peat structure (Wind-Mulder *et al.*, 1996) by both drainage and root networks, the mean seasonal aerobic limit depth measured at 60 cm below surface post-drainage in 1992 (Silins and Rothwell, 1999). While no aerobic limit was measured in 2010, we hypothesise that fire may act counter to drainage, through the creation of a new surface datum, lowering the limit.

Moisture retention properties

Boelter (1968) demonstrated that for equivalent pressures of θ_{-100} , moisture contents of un-decomposed *Sphagnum* carpets were around 10 % with well-decomposed peat retaining up to 75 % moisture, the hyaline cells in live *Sphagnum* draining below these

pressures a major controlling influence (Hayward and Clymo, 1982). Our post-fire drained and undrained site data from the SRP show a linear relationship between ρ_b and θ_{-100} derived from multiple depth-wise profile distributions which is contrary to the curvilinear relationship shown by Päivänen (1973). Pre-fire, moisture retention was greater at the drained site (Silins and Rothwell, 1998) and this remains the norm post-fire. Water is not readily draining from the higher density peat at the drained, in part because of a finer pore structure (Boelter, 1968; Orkuszko, 1995) formed from an increase in air-filled pore spaces subsiding and a loss of larger $>600 \mu\text{m}$ pore spaces (Silins and Rothwell, 1999; 1998). While linear, the relationship is non-uniform with increasing variance and residuals in moisture retention as ρ_b increased (see Figure 2.3), demonstrating an effective ‘optimal moisture retention’ line for a given ρ_b .

These changes to pore-size distribution are evident from mean retention data at the SRP sites; between depths 0-0.5 m, to reach the same drawdown condition (θ_{-100}), the drained site needed to lose only 46 % of held moisture, the undrained site requiring a 61 % loss of held moisture to achieve equilibrium at θ_{-100} . Required losses at surface (0-5 cm) depths are greater, 63 % at the drained, 81 % at the undrained, both sites requiring a smaller change in θ (Figure 2.5) to reach θ_{-100} than was needed before fire (Silins and Rothwell, 1998), indicating a decrease of water loss necessary to generate conditions inhibiting *Sphagnum* recolonisation. Undrained surface losses are on account of high specific yield of the *Sphagnum* (Price, 1996) and the presence of *Aulacomnium* surface at the undrained site. Denser, more degraded peat with stronger moisture retention only present deeper in the profile at the undrained site. Greater homogeneity of peat structure at the drained site accounts for a lack of significant differentiation between shallow and deep retention values. Well-decomposed peats,

with lower ρ_b values (between 0.075 – 0.20 g cm⁻³) have been shown to hold on to more moisture; around 50-70 % at θ_{-100} (Schlotzhauer and Price, 1999; Mannerkoski, 1985; Päivänen, 1973; Boelter, 1968, 1969). Data from the SRP shows moisture retention values to be at the lower end of this range for peats within these ρ_b values.

The aforementioned peat properties interact across different depths, altering the peat structure that influences retention (Wallage and Holden, 2011). ρ_b is the overriding control on θ_r at the SRP ($p = < 0.001$), while depth and H_{class} also interact greatly, only with a regression model inclusive of all major peat properties ($\rho_b, H_{class}, V_{class}, S_y$) can the majority of variance be accounted for. Depth and H_{class} act as secondary ($p = 0.002$) influences at the undrained site, whereas at the drained, H_{class} was not significant in affecting θ_r outcomes ($p = 0.042$), depth operating as the secondary influence. Variance in retention curve shape is greater ($p = 0.001$) at the undrained site, plausibly due to greater consumption in hollows as noted by Benschoter and Wieder (2003), the characteristics of the curve shape a function of peat water retention properties (Weiss *et al.*, 1998). The drained site retention curves are tightly banded (mean SD ± 2.9 %) across all pressures, reiterating greater homogeneity of pore structure at the drained site, the live *Sphagnum* surface of the undrained site draining out at $\Psi \leq$ air-entry pressure, to a mean 42 % moisture content for θ_r .

A conceptual model (Figure 2.6) was derived from mean bulk densities of peat pucks for given depths from both the drained and undrained sites at the SRP. The model demonstrates a “depth of burn marker”, based on present-day peat properties, effectively showing a decreasing hydrological buffer at the drained site with greater depth of burn. The model is made possible because the values for θ_{-100} and θ_r behave as

a function of the pore size distribution. For this model, ρ_b was used as an analogue for that distribution. In effect the model showed that there is a considerably lower “buffer” of water content at the drained SRP site before the pressure threshold that limits recolonisation is reached. This greater depth of burn is achieved though drained or dried peat lacking moisture saturation, therefore holding denser organic fuel for the fire to combust directly or by smouldering.

Significant ($p = 0.019$ at drained, $p = 0.007$ at undrained) correlations exist between ρ_b and R_{class} , with negative coefficients, a function of greater root thread prevalence near the surface ($p = < 0.001$), in less dense peat puck samples. Particularly at the drained site where vascular and grass root systems now exist. Drainage condition was the largest determinant of R_{class} ($p = 0.006$). Root threads did not significantly influence moisture retention, which were shown to lower water retention in nearby burned forested bogs (Thompson and Waddington, *submitted*). A significant relationship was not found for woody debris (V_{class}) and retention. The woody debris serves as an inert fill material, regression analysis showed no significant influence of woody debris on θ_{100} or θ_r , it does reduce the overall quantity of peat material for a given soil volume. However, correction of peat pucks for wood moisture retention did not significantly alter retention curves.

Expedited tree growth post-drainage at the SRP, increased quantities and quality of *Larix* and *Picea* woody debris at the drained site with significant correlations of V_{class} to ρ_b ($p = 0.001$) at the undrained site, but not at the drained site despite the greater number of ‘very woody’ classifications. These remnant root casts and subsequent macropore development from woody fraction may aid pedogenical change of drained

site peat towards an earthified state through alteration of pore size distribution. While not suggestive of a moisture retention influence, the changes to flow pathways are noted (see *Impacts on moisture transport and storage*) in the K_{sat} data.

van Genuchten parameters

The parameters showed a reversal of pre-fire conditions and are indicative of considerable decline in mean slopes of the water retention curves in the time post-fire from values determined post-drainage (Silins and Rothwell, 1998). Inverse determination of the parameters shows the drainage conditions differing between the drained and undrained site condition, with shape determining functions; α and n , varying amongst depth and drainage condition. Alpha, which was lower at the drained site, is indicative of the lower air-entry matric potentials at the drained site, while not significant ($p = > 0.05$) an interaction effect of depth and drainage was evident for α , decreasing more rapidly with depth in undrained peat than in drained peat along a roughly logarithmic profile. The $\ln(\alpha)$ correlates moderately ($R_{sq} = 56\%$) to S_y specific yield, suggesting modelled van Genuchten parameters are a convenient tool in modelling real-world moisture retention values of disturbed peatlands.

As noted by Whitehead (1999), low ρ_b and high S_y are indicative of more fibric peats, losing water more readily to atmosphere at a given pressures, versus a characteristically smaller pore structure of more decomposed (high ρ_b , low S_y). However, the quantity of water needed to be lost to reach a given pressure, is considerably less so at the drained site, leaving it more vulnerable to drying events (see Figure 2.6).

Humification can control S_y (Boelter, 1964; Päivänen 1973) linear regressions demonstrating this influence significantly ($p = < 0.001$) at both SRP sites. H_{class} acting as a depth dependent function across the SRP. Specific yield values at the undrained site were typical of values for moderately decomposed fen peats measured in Finland and Canada (Päivänen 1973; Price, 1992; Ronkanen and Kløve, 2005), but with greater means than expected when compared to S_y values (0.07-0.1) observed in cutover peatlands (Van Seters and Price, 2001; 2002).

Impacts on moisture transport and storage

Saturated vertical hydraulic conductivity changed significantly ($p = 0.028$) from values measured post-drainage at both sites. At the undrained site shallow depth (0-10 cm) K_{sat} values were lower compared to pre-fire data (104.1 vs. 551.3 cm hr⁻¹) with a difference in total range an order of five-to-tenfold (Figure 2.4), with parallel ρ_b increases up to 300 %. Shallow drained site K_{sat} was lower (Figure 2.4), and this was also true of the 10-20 cm depths at the undrained (33.8 vs. 340.7 cm hr⁻¹), likely due to the loss of *Sphagnum* and the large pore sizes from combustion, and subsequent ‘exposure’ of the relatively higher decomposed peat that previously existed lower in the peat profile. As an average for all sampled depths, K_{sat} decreased at the drained site by 41.5 %. However, this decrease was not uniformly distributed over the profile. Measured K_{sat} at 20-30 cm was faster than the pre-fire K_{sat} values, potentially caused by root growth and greater woody fraction (factors significantly correlated with depth, $p = < 0.01$). These factors serve to increase preferential flow through the peat. This was clearly occurring at deep depths, particularly 30-40 cm depth where mean K_{sat} increased by ~ 320 fold from a mean of 0.19 ± 0.13 to 64.47 ± 76.4 , which may in

turn increase saturated flow to drainage ditches, further drying the site and intensifying the desired drainage effect of increased soil aeration for tree growth. The reduction in tortuosity between connecting pores and introduction of large pores to the compacted, humified peat may change the nature of water movement in the peat. The slope of the K_{sat} (ρ_b) relationship declined post-fire at both the undrained and the drained site from pre-fire values, closer to the slopes shown by Päivänen (1973).

2.5 CONCLUSION

Tendencies of hydrophysical soil properties showed agreement with many of the findings found post-drainage, before the fire occurred (Silins and Rothwell, 1998), but with substantially greater variance at both sites. The inputs of woody debris and tree roots increased saturated hydraulic conductivity that would not be seen in unforested maritime peatlands after drainage, with the input of biological litter aiding earthification of the peat soils. The SRP, through double disturbance has undergone rapid decreases in the quality of the peat structure essential to maintaining water availability to *Sphagnum* to recolonise. The results show the effect of double disturbance altering the hydrological properties associated (high S_y , low near-surface ρ_b) with making peatlands resilient, adaptive ecosystems. The thresholds to induce moisture stress decreased with successive disturbance, causing the cessation of new peat formation entirely. Aeration (Silins and Rothwell, 1999) of the peat from dewatering and tree growth in the years post-drainage has had broader consequences, allowing rapidly establishing species like *Salix* and *Betula* observed at the SRP today to succeed in the hydrophysical niche created by the disturbances. This altered biodiversity has prevented (through direct competition for light) or greatly inhibited (through large transpiration losses) the recolonisation of *Sphagnum*.

3.0 ECOHYDROLOGICAL RESPONSE OF A PEATLAND TO DRAINAGE AND WILDFIRE

ABSTRACT

Peatlands ecosystems possess built-in resiliency to singular disturbances, but multiple disturbances can dramatically alter ecohydrological processes, placing limits on successful *Sphagnum* moss recolonisation. How boreal peatlands respond to multiple disturbances, following a loss of *Sphagnum* cover, has direct implications for localised hydrological regimes and regional scale carbon sequestration. We determined relationships between water table, soil moisture and soil-water pressure in conjunction with a basic water balance for a partially drained and burned, forested poor fen in Alberta, Canada. We examined how double disturbance has initiated a regime shift in species to those not native to fens (*i.e.*, grasses), though changes in site hydrology. The effects of drainage and wildfire have lowered storativity at the drained site, through the loss of less-dense surface peats to combustion. This has denied the peatland the ability to regulate water table from rapid fluctuations to atmospheric and vegetative fluxes. We present water table, soil moisture, soil water pressure and canopy openness as ecohydrological factors that have limited and later halt *Sphagnum* recolonisation, post-disturbance. Establishment of a dense woody canopy has likely initiated a positive-feedback loop, where both denial of light to the bare peat surface, and limitations in surface moisture by leaf litter, block capillary flow and shade-out *Sphagnum* fragments. This has denied *Sphagnum* opportunities to recolonise, even if favourable hydrological conditions exist. Such opportunities already limited by a lower hydrological buffer in the drained peat. We conclude that multiple disturbances

can cause the loss of peatlands' ecohydrological resilience. In doing so, valuable peatland ecosystem services may be lost and initiate regime shifts to species un-supporting of continual peatland function. This shift has potential to leave remnant peat more vulnerable to smouldering combustion, and rapidly alter northern peatlands mode of operation as carbon sinks.

3.1 INTRODUCTION

Northern and boreal peatlands store an estimated 220 - 460 Pg of carbon (Gorham, 1991; Turenen *et al.*, 2002), which represents ~ 30%, of the global terrestrial soil carbon pool (Batjes, 1996, Turunen *et al.*, 2002). However, there is a concern that natural and human disturbances are reducing this long-term atmospheric carbon sink ecosystem service (Turetsky *et al.*, 2011a). Wildfire represents the largest disturbance to peatlands in western Canada (Turetsky *et al.*, 2002), while drainage and prolonged drought is becoming an increasingly frequent disturbance in these ecosystems (e.g., Waddington and Strack, 2007). These landscape-scale disturbances can contribute to and directly cause the loss of *Sphagnum* moss in northern bogs and poor fens (Rydin *et al.*, 2006). Moreover, during and long after disturbance has occurred, these perturbations can lead to the direct release of CO₂ (Hilbert *et al.*, 2000). Consequently there is a need for a comprehensive understanding of the role multiple disturbances have on influencing the hydrological and carbon biogeochemical functions of northern and boreal peatlands. The aim of this research is to examine the ecohydrological response of a peatland to both wildfire and drainage.

Sphagnum mosses are the keystone species of northern peatlands (van Breemen, 1995; Turetsky *et al.*, 2008). *Sphagnum* mosses provide the foundation of peatland

resistance and resilience to disturbance and degradation (Rocheftort, 2000) and are critical for the maintenance of their long-term carbon sink function (Kuhry and Vitt, 1996; Chirino *et al.*, 2006). *Sphagnum* communities have been shown to be ecohydrologically resilient to peatland drainage (Strack and Waddington, 2007; Bridgham *et al.*, 2008) and wildfire (Benscoter, 2006). However, when disturbances are considered in combination such as drainage and extraction (Joosten, 1997) there is evidence that peatlands switch to a large and persistent source of atmospheric CO₂ (e.g. Waddington *et al.*, 2001). Moreover, they usually also undergo a regime shift (e.g., Hilbert *et al.*, 2000) with the complete loss of *Sphagnum* mosses (Lavoie and Rocheftort, 1996). While peatlands may be resilient to drainage or wildfire disturbance we suggest that the combination of drainage and wildfire disturbance in a peatland may also lead to a regime shift and persistent losses of CO₂. Turetsky *et al.* (2011) demonstrated that peat combustion was up to five times greater in a drained peatland than an adjacent undrained peatland. Given the expected future greater probability of multi-year drying events due to changes in boreal precipitation patterns (Soja *et al.*, 2007), such as enhanced drying of the mid-west regions of North America (Meehl *et al.*, 2007) fire return interval should decrease and burn severity in forested peatlands should increase (Flannigan *et al.*, 2001; Bergeron *et al.*, 2004). While wildfire consumes large expanses of Canadian peatlands each year (~1850 km²) (Turetsky *et al.*, 2004) and is recognised for its large contribution to atmospheric carbon (Amiro *et al.*, 2009) we are unaware of any studies that have examined the ecohydrological response of drained peatlands following wildfire

Sphagnum mosses lack the active water transport found in vascular plant species, reliant instead on passive capillary transport (Clymo, 1973; Hayward and Clymo,

1983; Thompson and Waddington, 2008). Consequently, maintaining a high and steady water table is essential for *Sphagnum* re-colonisation in disturbed peatlands (LaRose *et al.*, 1997; Schouwenaars, 1988; Cagampan and Waddington, 2008). More specifically, soil water pressure (Ψ) (also referred to as capillary pressure or soil tension) must not exceed -100 mb, not even if only occurring for short intervals during the growing season where otherwise favourable hydrological conditions i.e. high VWC (>50%) and high water table levels exist (e.g. Price, 1996; Price and Whitehead, 2001; Thompson and Waddington, 2008).

Thompson and Waddington (2008) suggest that in drier conditions, water table is relatively unimportant with actual water retention being more important given its direct influence in affecting the wilting point of *Sphagnum* (Thompson and Waddington, *submitted*). A consequence of peatland drainage or drought is peat surface subsidence (Silins and Rothwell, 1998) which results in higher moisture retention due to the higher proportion of small pore spaces (Price and Whitehead, 2004). While, drained peatlands should retain soil moisture more readily (Waddington and Price, 2000) than natural peatlands, drained peatland surface Ψ values may be too high to allow *Sphagnum* to access the water. This occurs because the amount of soil water deficit through water table draw-down, transpiration or drainage required to induce -100 mb Ψ value is greatly reduced (Sherwood *et al.*, 2011). Consequently, remnant *Sphagnum* communities remain moisture stressed, inhibiting future recolonisation.

Another consequence of peatland disturbance is a reduction in species diversity (e.g. Laine *et al.*, 1995), thereby potentially lowering the overall ecohydrological resilience

to further disturbance. This lower resiliency occurs due to reduced microform variability due to subsidence (Whittington and Price, 2006) and an absence of species resistant to combustion (Shetler *et al.*, 2008). This therefore decreases the range of hydrological conditions that *Sphagnum* communities can operate within (Rydin, 1993). Combinations of disturbances (e.g., wildfire and drainage) may push peatland ecosystems beyond their ecohydrological resilience thresholds (i.e. Ψ), inhibiting *Sphagnum* species from recolonising for the majority of summer growing seasons in northern peatlands and potentially leading to a regime shift (e.g., Hilbert *et al.*, 2000). A regime shift would fundamentally change the carbon storage function (e.g., Waddington *et al.*, 2002), capability and resilience of these wetland ecosystems to future disturbances. To examine this effect of peatland drainage and wildfire on ecohydrological response we took advantage of a unique opportunity presented by the Saulteaux River peatland (SRP). The SRP is a drainage experiment in central Alberta, Canada where the combined effects of drainage and wildfire have disturbed a poor fen complex. The SRP has a large research background with data on the hydrology and hydrophysical properties from both before, and after drainage (Hillman, 1987; Tóth and Gillard, 1988; Swanson and Rothwell, 1989; Rothwell *et al.*, 1996; Silins and Rothwell, 1998; 1999). The site later burnt in the Chisholm fire (LWF-063) event in June, 2001 and since then, *Sphagnum* has failed to recolonise the drained portion of the fen complex. The general objective of this study is to evaluate and understand the ecohydrological response of a peatland to drainage and wildfire, specifically to (i) assess the post-drainage, post-fire state of hydrologic relationships of water table; soil-water pressure and soil moisture (ii) evaluate their effect on *Sphagnum* recolonisation at a burned fen complex for a drained and undrained site locale.

3.2.1 STUDY SITE

The study was conducted at the Sauleaux River peatland, 37 km south-east of the town of Slave Lake in the Lesser Slave Lake region of north-central Alberta, Canada (55°8'N, 114°15'W). The study site forms part of a 100-ha treed poor fen (Hillman, 1987), located atop shale and glauconitic silty shale formations in the Lesser Slave Lake Lowlands (Swanson and Rothwell, 1989). Peat depth varies across the site between 0 to 4 m deep (Swanson and Rothwell, 1989). The 1970-2010 mean annual temperature was 1.8°C with average July temperature 15.8°C and annual precipitation 503 mm, 385 mm falling as rainfall from April to October (Environment Canada, 2011). Measured site data for the period of study 2009-2010 showed a mean annual temperature of 0.6°C, with an average growing season (May-August) temperature of 15.3°C. In 1987 a 50-ha portion of the SRP was drained through the construction of 0.9 m deep ditches separated by 40 m or 25 m (Hillman, 1987) resulting in central plateaus in-between ditches (Figure A1.2). These plateaus do not have baulk ridges that are common in harvested peatlands (see Van Seters and Price, 2001). Both areas burned as part of the fen complex during the large-scale fire event (LWF-063) in 2001 (Quintilio *et al.*, 2001). The fire consumed 19 ± 3 cm of surface peat in the drained portion of the peatland (Turetsky *et al.*, 2011) and 7 ± 1 cm at the undrained, determined by the adventitious root method of Kasischke *et al.* (2008).

As of 2010 the surface vegetation in the undrained site is dominated by *Sphagnum fuscum*, *S. magellanicum*, *S. angustifolium*. Dwarf rosemary (*Andromeda polifolia*) is the primary vascular species. *Sphagnum* mosses are absent from the drained site, glow moss (*Aulacomnium palustre*) the only moss species present. *Brome* grass species dominate surface cover, interspersed with dwarf raspberry (*Rubus acaulis*), fireweed

(*Chamerion angustifolium*) and three-leaved false Solomon's seal (*Smilacina trifolia*). Canopy cover at the drained site is comprised of competing willow and birch species including *Salix bebbiana*, *S. pedicellaris* and *Betula pumila*. Above the canopy, remnant fire-charred black spruce (*Picea mariana*) trunks (Figure A1.4) with little to no interception value remain.

Prior to fire, *Sphagnum fuscum* was the dominant surface cover across both sites; post-drainage surface cover at the drained site was in a moisture stressed state, evidenced from desiccated appearance (Silins, *per. comm.*).

3.2.2 METHODOLOGY

Microclimate

The study period was late May to end of August, 2010. Bulk rainfall was measured using tipping bucket rain gauges 1 m above surface at the two sites. An additional tipping bucket rain gauge, placed beneath the drip line of a tree measured throughfall of the drained site tree canopy. Climate normals (1971-2000) were taken from the Slave Lake A, meteorological station (55.30°N, 114.78°W). At the drained and undrained sites, daily evapotranspiration (mm day⁻¹) were calculated with the Penman-Monteith (1965) combination equation model (Equation 3.1).

$$ET = \frac{\Delta(K + L) + c_a \rho_a \text{Cat} \{e_a^* - e_a\}}{\lambda_v \rho_w \{\Delta + \gamma (1 + \text{Cat}/\text{Ccan})\}}$$

Where, Δ is the slope of the saturation vapour pressure versus the temperature relationship at T_a , K is incoming shortwave radiation, L is incoming longwave radiation, c_a is the atmospheric heat capacity, ρ_a is the density of air, Cat is the

atmosphere conductance, e_a^* is equal to the saturation vapour pressure of atmosphere, e_a is the water vapour pressure in atmosphere, λ_v is equal to the latent heat of vaporization, γ the psychrometric constant, ρ_w the density of water and C_{can} is the canopy conductance. A tower mounted Kipp and Zonen NR Lite net radiometer recorded net radiation (Q^*), at 10 m above the surface. Ground heat flux (Q_g) was calculated from soil temperature arrays using the ‘GradCal’ method (Liebethal *et al.*, 2005), where Q_g is equal to the temperature gradient between 10–20 cm depths, and the change in energy storage between 0–10 cm. The latter calculated from thermocouple (Type-T, 20 ga, Omega Engineering, USA) measurements at 2 and 5 cm. Q_g was averaged from 3 soil temperature arrays at each site.

Air temperature and relative humidity was measured with a shielded, Campbell Scientific CS215 Temperature and Relative Humidity Probe. Measurements of stomatal conductance (C^{*leaf}), for deriving the canopy conductance term were made using a Delta-T Devices AP4 (Cambridge, UK) porometer. The samples were taken in early July 2010 on 20 specimens. Sampling was either on predominantly woody or vascular species, according to dominant canopy vegetation at each site.

Water table position

Sampling transects at both sites were established at the same locations to that of the original studies (Hillman, 1987; Silins and Rothwell, 1998; 1999) to allow directly comparable datasets. Groundwater wells, constructed of 50 mm (i.d.) ABS, slotted along their 2 m length and wrapped in geotextile screen, were organised at a 1m spacing in 10m transects at both sites. The transect at the drained site was arranged perpendicular to the ditch extending to 10m from the ditch edge and located 1 m

parallel to a previously studied research transect (Silins and Rothwell, 1999). The transect at the undrained site was organised to bisect several *Sphagnum* hummocks, in north to south orientation. Water level was continuously measured using an ‘Odyssey’ capacitance-based level recorder (Dataflow Systems Pty Ltd, NZ) within one well at each site, and manually every 7-10 days at the remaining wells. The ditches appeared non-active at the time of measurement.

Soil-water pressure

Soil-water pressure (Ψ) was measured with 2.2 cm (o.d.) porous ceramic cup-type tensiometers (Soil Measurement Systems, Tucson, AZ, USA) at the drained and undrained sites, at 5, 15 and 30 cm below surface, at the 0, 5 and 10 m points along the transects at both sites. Tensiometers were installed by inserting into undisturbed peat along the vertical face of a small pit, or downward into the peat horizon for deeper (straight type) tensiometers. The pit was then backfilled with peat. Both elbow and straight tube type tensiometers were used. Internal and outside atmospheric pressure was measured using a Tensicorder (Soil Measurement Systems) meter ± 1 mb accuracy. Measurements were calibrated for differing water column heights within the respective tensiometers used.

Soil moisture

Soil volumetric moisture content (VMC) was measured at 15 and 30 cm depths using CS615 and CS616 (Campbell Scientific) water content reflectometer probes at 0, 5 and 10 m intervals on the transect for both sites. Triplicates of near surface soil moisture at depths of 2, 5 and 7 cm was measured approximately every week, for all transect points (1-10 m), at both sites using a Delta-T WET Sensor. The probe was

inserted at set angles to measure the diffuse soil moisture at relevant depths at each transect point.

Plant species identification

Plant species identification and % cover were determined at eight sampling points randomly distributed in the vicinity of each well transect. Each point consisted of a 0.5 m² quadrat plot and an assessment of species present by % cover, including both ground surface and understory was carried out. Sampling occurred while herbaceous and sedge species were flowering to ensure species were correctly identified. Indices of similarity were calculated according to both Jaccard's index (Jaccard, 1902) and Sørensen's quotient of similarity (Q/S) (Sørensen, 1948). Both are indexes devised for comparing species diversity between communities. Jaccard's index (Equation 3.2) divides the number of species found in both samples (j), by the number of species found in only one sample or the other (r). This is then multiplied by 100 to find a percentage for floral similarity.

Equation 3.2 Jaccard's Index
$$\frac{j}{r} 100$$

Sørensen's quotient (Equation 3.3) was used alongside Jaccard's as it essentially has the same properties as Jaccard's, expressing 100% similarity when all species are common to both samples. The terms are denoted by (j) the number of species common to both samples, (a) is the total number recorded in the first sample, (b) is the number recorded in the second sample.

Equation 3.3 Sørensen's quotient of similarity

$$[Q/S] = \frac{2j}{a+b} 100$$

Both indexes have empirical foundations, giving less weighting to outliers while retaining sensitivity within more heterogeneous datasets.

Light transmissivity, canopy openness and leaf area index

Light transmissivity, canopy openness and leaf area index (LAI) were calculated using gap light photography using a Nikon D60 camera and a Sunex 185° Super Fisheye lens. On top of a flexible tripod, the camera body was levelled to sit flat with the ground surface, the lens aimed skyward at the immediate canopy. A high-resolution digital photograph was taken of the canopy and the visible light penetration through it. The photograph was taken using a time delay allowing the operator to ensure they would not obstruct the margins of the lens's viewing extent. Photographs were captured across two transects at each site, situated 20 m parallel to the well transects established at each site. This was done to avoid the well heads potentially obstructing the lens's viewing margins. They were also taken at the centre of the eight species identification plots at each of the drained and undrained sites. The photographs were processed using GLA v.2 software for total light transmissivity, calculated LAI and canopy openness (Frazer *et al.*, 1999), following calibration to the lens and site location specifications.

3.2.3 Statistical Rationale

Statistical tests varied according to the parameters of the data tested; continuous, discrete, categorical and whether the data was normally distributed or not. The majority of tests of statistical difference were either one-way ANOVA or *t*-tests for continuous datasets, usually with Mann-Whitney *U* test for discrete data. Tests were run within MINITAB 15 or Sigmaplot 11.0 according to the dataset format. Cause and effect were determined either through logistic regression for discrete data or linear regressions for continuous data. Multiple and stepwise regressions was used where there were >1 effect but 1 cause. Multivariate tests (mainly Principle Component Analysis) were used to explore the variables within the datasets.

3.3 RESULTS

3.3.1 Species cover and diversity

A total absence of *Sphagnum* moss cover was observed at the drained site species plots. Moreover only minimal remnant *A. palustre* moss was detectable on *Picea* nurse logs and associated detritus. In contrast, twelve bryophyte species were observed at the undrained site, including three *Sphagnum spp.* (Figure 3.1). These were located among a recovering hummock and hollow surface topography which is absent at the drained site.

Shannon diversity indices for all functional groups were 2.03 and 2.15, at the drained and undrained sites, respectively. Values for bryophyte taxa were 0.22 and 1.37, respectively. Sørensen's quotient of similarity was calculated as a measure of similarity between zones in the same area i.e. drained and undrained, essentially β -diversity or the species unique to a community. The quotient value was calculated at

8.82, indicative of species similarity of 8.8% between sites. Jaccards index (flora similarity as a percentage) value was $\pm 0.8\%$ to that of the Sørensen's quotient.

Aulacomnium palustre dominated ground cover, forming 10.2 and 55% ($\pm 5\%$ SE) at the drained and undrained sites respectively. Other mosses (Figure 3.1), including *Sphagnum*, *Campilium* and *Tomenthypnum* species formed the majority of the undrained sites remaining ground cover. *S. fuscum* was the most common species type after *A. palustre* at the undrained, forming 10.6% ($\pm 6.0\%$ SE) of total ground cover. At the drained site *Brome* grass species formed the majority of ground cover. Amongst lower-overstory vascular species, *A. polifolia* and *S. trifolia* dominated at the undrained site (Figure 3.2) with *C. interior* and *diandra* sedge species formed the remaining cover. The mature canopy of the drained site was comprised of seven different *Salix* (willow) species and two different types of *Betula* (birch). Within the undrained site, only young saplings of bog willow and birch were identifiable, with a total population count of ≤ 10 from within the species quadrats.

3.3.2 Light availability

Mean total light transmissivity at the species plot quadrats was significantly lower ($p < 0.001$) at the drained site ($20.7 \pm 26.7\%$) than the undrained site ($87.3 \pm 5.9\%$). Transmissivity values derived from the transect points were also significantly lower ($p = < 0.05$) at the drained site ($49.8 \pm 24.7\%$) compared to the undrained site ($84.8 \pm 13.5\%$). Peaks in transmissivity at the drained site (Figure 3.3) corresponded with natural openings within the *Picea* and *Salix* overstory canopy. Canopy openness above species quadrats varied considerably between sites. Openness averaged $15.0 \pm 18.3\%$ at the drained site, with a minimum of 7.2%. Canopy openness averaged $68.3 \pm 6.1\%$

for values from the quadrat samples. Maximum values at the undrained site were 93.5% and 77.6%, from transect and quadrat points, respectively (Figure 3.3). At both the drained and undrained it was found that median values taken from their respective transect and quadrat points did not differ significantly ($p > 0.05$). Calculated effective 4-Ring LAI was significantly greater ($\sim 3x$ greater, $p < 0.001$) at the drained site versus the undrained site (Figure 3.4). Values at the drained site species plots ranged from 0.4 to 3.1 (mean = 2.6 ± 1.1) and the distribution of values were highly negatively skewed. Undrained quadrat values had a peak value of 0.3, and averaged 0.2 ± 0.1 . For values calculated for transect points, t-tests showed variance in LAI to be significantly higher ($p < 0.001$) at drained site (1.0 vs. 0.1).

3.3.3 Water balance

Total precipitation (P) recorded during the 2010 study period was 280 mm (Figure 3.5). Precipitation for June, July and August was 108, 79 and 122% of the long-term climate normals, respectively (Environment Canada, 2011). Average rainfall interception for the study period from willow and bog birch trees (I_{tree}) at the drained site was 63%, while rainfall interception from the shrub and sedge canopy (I_{shrub}) at the undrained site (in the absence of a true overstory canopy) was only 10% for the same period (see Figure 3.6). Daily evapotranspiration ranged between 0.7 to 4.6 mm d^{-1} , and 0.5 to 4.9 mm d^{-1} at the drained and undrained sites, respectively, with daily averages of 2.6 mm d^{-1} and 2.8 mm d^{-1} . Despite the higher transpiration, total study period evapotranspiration (ET) was higher at the undrained site (272 mm), than the drained site (254 mm).

While the absolute water table position was essentially flat at the undrained site (no hydraulic gradient), a slight gradient (0.008) existed at the drained site away from the ditch towards the centre of the plateau. Using saturated hydraulic conductivity values from Sherwood *et al.* (2011) we calculated a lateral groundwater loss (Q_{seepage}) from the drained site of 1.1 mm d^{-1} , totalling a 103 mm input over the study period. Q_{seepage} at the undrained site was 0 mm given no discernable hydraulic gradient. No surface runoff was observed at either SRP site.

3.3.4 Water table, soil moisture and pressure

Mean study period water table position at the drained site was -25.9 cm and -12.4 cm at the undrained site. Maximum water table drawdown was -35.9 and -12.4 cm and the drained and undrained sites, respectively. The drained site water table response to rain events was more responsive, with a larger magnitude in water table change (Figure 3.5, 3.7) than at the undrained site. Peat storativity (S) was higher at undrained site (0.59), than the drained site (0.18).

Average θ at the surface (-2 cm) of the drained site was $25 \pm 5\%$, slightly lower than the undrained site ($28 \pm 12\%$). The lowest θ values were recorded at the surface (-2 cm) of the undrained site (Figure 3.8). Mean θ within shallow depths ($\leq 7 \text{ cm}$) was 44.1% and 43.3% for the drained and the undrained sites, respectively. Near surface (2 to 7 cm depth) θ varied considerably between sites (Figure 3.8), the range of variation $\sim 200\%$ greater at the undrained, than for the same wetting or drying events at the drained site. Temporally, the undrained site displayed the most rapid gains and losses in near-surface soil moisture. Duration analysis showed soil moisture remained above 40% in near surface (2 to 7 cm depth) for 48 and 54% of the study period, at the

drained and undrained sites, respectively. At lower depths (15, 30 cm) below-surface (Figure 3.9), the drained site was consistently drier ($64 \pm 1.9\%$) at 15 cm and ($71 \pm 1.2\%$) at 30 cm respectively. Undrained θ values averaged $74 \pm 4\%$ at 15 cm and $76 \pm 2\%$ at 30 cm. Within the drained site at depths below 7 cm, mean θ was greater towards the ditch boundaries. At both sites, of the lower depths measured, -15 cm showed the greatest variance in θ . While linear relationships exist between θ and water table, they were relatively weak (i.e. $R^2 = 0.22$). Logistic regressions showed surface (-2 cm depth) θ was not significantly related ($p = > 0.05$) to changes in water table.

Soil-water pressure showed similar seasonal trends to water table, lowest (i.e. least negative) at the undrained site in mid-July (Figure 3.10), at -2 mb, consistent with a peak in soil moisture. Pressure averaged -17 mb over the three depths measured, for the duration of the study period. While slightly higher soil moisture values were observed at the end of June, pressure values were around 10 mb higher than their mid-July low. Pressure was highest in early August at the drained site for all the measured depths at $-55.8 (\pm 8.4)$ mb, coinciding with a drop in water table following low precipitation over a three week period. With the exception of early June where ground-ice was still an influence; pressure was always lower at the drained site. Across the profile of 5, 15 and 30 cm depths measured for Ψ , the drained site showed very small upward vertical gradients (i.e. greater Ψ nearest to surface) between depths during water table drawdown > 5 cm. At wetter periods such as the end of June through to early July, the undrained site responded with decreases in Ψ not observed at the drained site, while during the early August dry period, Ψ at the undrained stabilised, not increasing like Ψ at the drained site.

3.4 DISCUSSION

Water table behaviour

A drought year did not occur in 2010 as initially predicted (Canadian Drought Monitor, 2010). This limited opportunity to observe dry conditions, wherein *Sphagnum* recolonisation would have been limited by known hydrological thresholds (i.e. $\Psi \geq -100$ mb). The water table in the drained site decreased to -35.9 cm in early August (Figure 3.5). This was less than maximum drawdowns measured in the years before fire, which ranged from -50 to ≥ -74 cm (Tóth and Gillard, 1988; Silins and Rothwell, 1999). The higher than expected water table, was in part, a result of changes to the surface datum and the changes in the hydrophysical properties of that surface. At Saulteaux, combustion of peat by fire of up to 19 (± 3) cm at the drained site was a strong driver of this change (Turetsky *et al.*, 2011). Combustion not only decreased the height between relative topographic surface to water table position, but also exposed previously saturated, denser peat from deeper in the profile as the new peat surface (Sherwood *et al.*, 2011). The lower specific yields of this peat serve to decrease the peat moisture storage term at the drained site.

The water balance showed a deficit of -47 mm and -20 mm for the 2010 study period at the drained and undrained sites, respectively. The efficiency of the ditches has been considerably reduced (Turetsky *et al.*, 2011), both from slumping of banks and latterly from beaver dams impeding discharge (Q) from the site, leading to water backup in the ditches and to function to recharge the plateaux between ditches. The ditches are effectively no longer active, ditch damming by beavers previously suggested as likely

to be problematic in affecting the drainage experiment in the time post-drainage (Hillman, 1987).

The storage changes at the drained site are further reflected in the flashy water table response (Figure 3.5) to drying and wetting periods seen at the drained site, similar to those seen in harvested peatlands (Price *et al.*, 2003). Stabilisation of the water table is a known requirement for *Sphagnum* to recolonise disturbed peatlands, as is the case at cutover sites, where the original peat surface has been lost (Girard, 1999; Grosvernier *et al.*, 1997; Campeau and Rochefort, 1996). Furthermore, the maximum water table drawdown at the drained site is close to the suggested limiting -40 cm water table. Water tables maintained above -40 cm are thought necessary for reestablishment of typical species in harvested bogs (Schouwenaars, 1988; Rydin, 1993). At the undrained site, a larger *Sy* exists (Sherwood *et al.*, 2011) as a consequence of the recovering *Sphagnum* and *Aulicomnium* surface cover, and in combination with minimal canopy interception (28 mm), retards flashiness in water table response. The magnitude of water table movement at the drained site places limitations on moisture availability to peatland species. This is evidenced by low near-surface θ values at the drained site. Surface (2cm) θ values averaged 25% over the study period at the drained site, considerably below the suggested 50-75% range for volumetric moisture contents shown to facilitate reestablishment (Price and Whitehead, 2001). θ values at the surface of the undrained were not significantly larger than those at the drained site. Variance in θ throughout the depth profile was significantly greater than the drained site for wetting and drying events (Figure 3.8, A1.1). The undrained site moss surface more susceptible to evaporative flux than the low porosity peat at the drained site.

Limits to moisture accessibility

Further to θ limitations, the deciduous nature of the birch and willows contribute a significant litter layer (see Figure 3.6), such that upward capillary flow may be blocked to areas of wet, bare peat. As previously observed in cutover peatlands, these are the locations where *Sphagnum* would most likely recolonise (Farrick, 2008, Price and Whitehead, 2004). So while VWC is generally greater at the drained site, it isn't readily accessible to recolonising diaspores or *Sphagnum* fragments. Low moisture values observed at the surface of an abandoned cutover bog shown to correlate with an absence of recolonising *Sphagnum* (Whitehead, 1999). This is compounded further still by the overwhelming requirement for high soil-water pressures to facilitate *Sphagnum* recolonisation at disturbed sites (Price, 1996; 1997; Price and Whitehead, 2001; 2004; Thompson and Waddington, 2008). Ψ was not found to be limiting at either of the SRP sites in 2010, reaching a maximum of -72 mb at 5cm at the drained site (Figure 3.10). However, laboratory based moisture retention curves derived from SRP peat cores suggest that Ψ would be limiting if rainfall values for the months of June, July and August had been those of a 'normal' year. Water-table drawdown estimates derived from the simple relationship of change in storage over the specific yield of the profile (Equation 5).

Equation 3.4 Water-table drawdown bucket model $\frac{\Delta S}{Sy}$

It suggests a water table up to 79 cm deeper than the 2010 maximum (-35.9 cm) had precipitation input been closer to long-term normals for drought years, like the

precipitation seen in 2009. Such changes to ΔS would leave the drained site very vulnerable, requiring smaller decreases of soil moisture to reach pressures ≥ -100 mb (Sherwood *et al.*, 2011). Moreover, Ψ may increase more rapidly, decreasing or stopping capillary rise to the surface from the saturated zone, should water table place lower in the peat profile (Mannerkoski, 1985; Ahti, 1974).

Several other studies have demonstrated a linear relationship between Ψ values for surface peats and water table (Ahti, 1975; Mannerkoski, 1985; Päivänen, 1973). At the SRP, this appears the case (Figure 3.5, 3.10), observable during wetting events at the undrained but not for drying events like those observable at the drained i.e., from the 9th – 18th August (221-230 DOY). Rank order correlations proved inconsistent in showing significance across all sampled depths and transect points. Furthermore, regression analysis found no causal link between the two variables, possibly on account of the small n value (17) from manual sampling of the tensiometers. This tends to suggest that changes in Ψ are influenced more by surface water fluxes, like moisture uptake by woody species as opposed to just water table alone. The lack of a strong linear relationship between water table and Ψ is likely on account of the drained SRP site possessing a peat matrix which isn't subject to significant volume changes (Price and Schlotzhauer, 1999). It has already undergone significant deformation by compression and combustion post-fire and drainage (Rothwell; Turetsky *et al.*, 2011).

Observed erratic variations (Figure 3.10) in Ψ values can be largely attributable to the temporary recharging of the unsaturated zone, by frequent and large-scale rain events

i.e. DOY 193; 224. This unsaturated zone is a function of the ditch network and aerated nature of the peat (Silins and Rothwell, 1999).

LAI and transmissivity to surface

Effective LAI values were likely underestimated at both sites, given the indirect hemispherical photo method used. The method is not fully able to take into account overlapping leaf canopy (as the direct method can) in particularly dense canopies (Wilhelm *et al.*, 2000). At the undrained site, LAI immediately above the peat surface was likely underestimated as it was not possible to sit the camera lens flush with the moss surface. As such the undrained evaporation terms were modified to reflect this, with a slightly higher LAI value used (1.0), given it better reflected the shrub canopy. Nevertheless, the drained and undrained sites differ significantly in their indices of LAI (Figure 3.4), which was significantly greater at the drained site. This leafy canopy means greater peat surface shading, impeding light required of *Sphagnum* diaspores recolonising. While some shade is deemed useful to the recolonisation of mosses in a restoration context (Price *et al.*, 2003), the near total shading of the ground surface (see Figure 3.6) at the drained through the canopy of both *Salix* and *Betula* is almost certainly inhibitive (Bisbee *et al.*, 2001; Benscoter and Vitt, 2008). The shading at the peat surface is in excess of 20% and thus favourable to feather moss establishment over that of *Sphagnum* (Bisbee *et al.*, 2001).

High stomatal resistances and shading from dense shrub cover have been shown to act as a water conservation element to surface moisture in the time after disturbance (Thompson *et al.*, 2011). However, at the SRP, the closed nature of the birch and willow canopy has stopped significant shrub growth within the understory. Shrub

positioning has largely been channelled towards natural openings at the drained site, reducing effectiveness of possible moisture conservation properties.

Species composition and implications for recolonisation of Sphagnum

The failure for *Sphagnum* to recolonise at the drained portion of the SRP post-fire is the result of several confounding variables. There is reduced near surface moisture availability, and enhanced variability with moisture stresses potentially enhanced through a positive feedback loop (Thompson *et al.*, 2011), where dry-adapted trees and shrubs out-compete the mosses. Birch, willows and tall grasses, species usually associated with moist upland sites, shade-out the understory surface during the growing season. A similar phenomenon has been noted in the crowding out of *Sphagnum* by dense shrubbery seen in cutover bogs (Farrick, 2008). The undrained site has next-to-no canopy, with the shrub growth that is present sufficient to allow enough light for *Sphagnum* growth (Bisbee *et al.*, 2001). The high water table and lack of aerated substrate (Silins and Rothwell, 1999) has halted rapid shrub and woody species growth that occurred at the drained; the hydrophysical conditions not conducive to species establishment. Of the woody species identified at the undrained site, black spruce seedlings were the most prevalent, suggesting total recovery of the undrained site to a true poor fen state will occur in the future.

Increased interception and evapotranspiration losses by shrubs and trees was previously suggested (Rothwell *et al.*, 1996) as amplifying decreases in water table levels post-drainage, and this appears to be the case now in light of the very dense overstory canopy. Post-fire the species composition of the drained site has shifted to varieties more associated with Albertan uplands; *Brome* (grass), *Betula* (birch) and

Salix (willow). The previously dense *Picea* (spruce) and *Larix* (tamarack) overstory was totally consumed during the 2001 fire. This was partially due to increased productivity of the trees following drainage, which would have lent more fuel to consumption. Furthermore, the fire occurred in spring when ground ice is present, this may have served as a barrier to upward moisture transport, artificially drying the peatland surface in time before fire. Willow species have previously been shown to establish rapidly in ‘pulses’ following drying periods and wildfire activity at wetland sites (Timoney and Argus, 2006). The species is able to dominate due to rapid rates of growth, as is apparent at the SRP. This very dense tree canopy at the drained site likely accentuates drying events, both through interception and transpiration, likely generating significant soil moisture deficits further reducing water tables (Ledger and Harper, 1987; Burt *et al.*, 1990).

3.5 CONCLUSION

Wildfire is a natural disturbance on the Canadian boreal landscape under which forested peatlands and their functioning are disrupted, and from which, they usually recover. Here we show the combined effects of drainage and fire at an Alberta poor fen have overwhelmed the hydrological mechanisms that lend peatlands their resiliency against disturbance. The drained site has lost its resilience through changes to the species dynamic, a consequence of altered ΔS and available moisture held at pore water pressures accessible to *Sphagnum* within dry years (Sherwood *et al.*, 2011). The regime shift to upland species within the drained site followed on from the failure of *Sphagnum* hummock microforms to survive combustion. These hummocks already observed experiencing significant moisture stress in the time post-drainage.

Following this loss of *Sphagnum* ground cover, new peat formation has halted, lowering the resiliency. A combination of low volumetric moisture at the surface, unstable water table and a Ψ threshold which is likely inhibitive of *Sphagnum* recolonisation during dry years, have halted this recovery. The Ψ threshold (-100 mb) was likely exceeded for significant portions of the growing seasons in the ten years post-fire. As suggested by peat moisture retention curves from the SRP (Sherwood *et al.*, 2011). Owing to the lack of a drought year in 2010, these thresholds were not observed to be exceeded in this study. However, as the water balance demonstrates, if the ΔS deficit was that of one associated with a normal year, with efficiently functioning drainage ditches, then water table place considerably lower in the peat. This would increase soil water pressure beyond the threshold that stops *Sphagnum* recolonisation.

Further to this, upland species communities that have had a competitive advantage over the native poor fen species, imposing limits moisture availability to any recolonising *Sphagnum*. In summary, the combined impact of drainage and fire have decreased potential moisture availability, destabilised water table regimes and altered storage characteristics. This has brought about a positive feedback whereby *Sphagnum* mosses have been outcompeted by fast-establishing species suited to the drier, aerobic substrate. Such regime shift may form a positive feedback loop, further denying *Sphagnum* the opportunity to recolonise through changes in both light accessibility and leaf litter impeding capillary flow to the surface. The woody and vascular species possessing active water transport mechanisms needed to be able to overcome inhibiting Ψ -thresholds associated with water table drawdown of dry, drought years. These changes have depressed much of the built-in resistance that peatlands possess to

disturbance. The Sauleaux River peatland has shown how double disturbance can develop hydrological thresholds that in natural peatlands are only occasionally limiting, into permanent barriers to dominant species recolonisation. This is then reinforced by the consequent changes to biodiversity that ultimately bring a regime shift to the ecosystem itself.

While understory species can contribute to large portions of boreal forest NPP (Chapin 1983; O’Connell *et al.*, 2003), it is unlikely the new upland species stand will sequester carbon dioxide at the same rate that the peatland did prior to drainage and fire. This is wider consequence to the carbon stocks of peatlands, given the potential risk of multiple disturbances. Greater understanding of temporal periods after double-disturbance, wherein inhibiting thresholds are preventative of *Sphagnum* species recolonisation is essential; both to predict where similar regime shifts may cause the loss of peatlands, and to preserve the ecosystem services that they bring to the boreal landscape.

4.0 THESIS SUMMARY

For the duration of the 2010 study period, peat Ψ remained above -100 mb at both the drained and undrained study sites, with surface peat (-2 cm depth) θ peaking at 34.1%. This is well below the 75% values suggested ideal to facilitate recolonisation (Joosten, 1992; Petrone *et al.*, 2004; Price and Whitehead, 2004). Overall, the undrained site maintained greater mean surface, and near-surface (-5 to -7 cm depth), peat moisture values than the drained site, though they showed greater variability over the study period. The variance observed was largely on account of periods of wetting and drying, the surface mosses drying during periods free of precipitation. Areas situated topographically lower along the sampled undrained transect, maintained higher θ values, while a near-surface water table allowed for surface moisture to meet atmospheric demands from evapotranspiration, supplying it through capillary rise. The *S. fuscum* found populating the undrained site was largely within the microtopographical confines of hummock extents. The species possessing greater capillarity (Rydin *et al.*, 2006), and thus withstanding slightly lower Ψ values associated with greater distance from the water table that comes with the elevation of hummock topography.

Sphagnum re-established on hydrologically suitable peat substrate at the undrained site, possibly helped by lateral expansion of remnant post-fire hummocks, a microtopographical feature absent from the drained site. Low water tables within the drained site, where θ is largely limiting throughout much of the unsaturated zone, may work in conjunction with extraneous biotic factors to limit recolonisation. Over-shading, a dense deciduous leaf litter inhibitive of capillary flow to the surface and

roots drawing moisture from near-surface peats may operate to further inhibit recolonisation. The shallow root balls of the birch and willow servicing their canopy transpiration demands via this moisture uptake. A comparison of peat hydrophysical properties from eighteen cores showed recolonised *Sphagnum* hummocks from the undrained portion of the Saulteaux fen were largely mimicking water storage functions (i.e. S_y) of an intact, natural peatland surface. Contrarily, the drained site has lower S_y over the entire depth profile, rather than a logarithmic S_y profile, as observed at the undrained site, traditional of natural peatlands. The former showing much larger fluctuations in water table as a consequence of this. The mean water table depths differentiated greatly between sites, with high and relatively stable at the undrained site and low and fluctuating at the drained site, respectively. The greater specific yields of peat that comprised the undrained surface suggest a water table that will remain close to the surface during dry periods. At the drained site, the earthified nature of the peat, high water retention properties and the very low S_y contribute to a peat substrate unlikely to support *Sphagnum* fragments long enough to re-establish *Sphagnum* cushions. Furthermore, greater bulk density of the drained site peats gives rise to a lower hydrological buffer, hydrological limitations strongly influence near-surface θ values. As well as contributing to a lower θ at which limiting Ψ would be reached.

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FIGURES 2.1 – 2.6

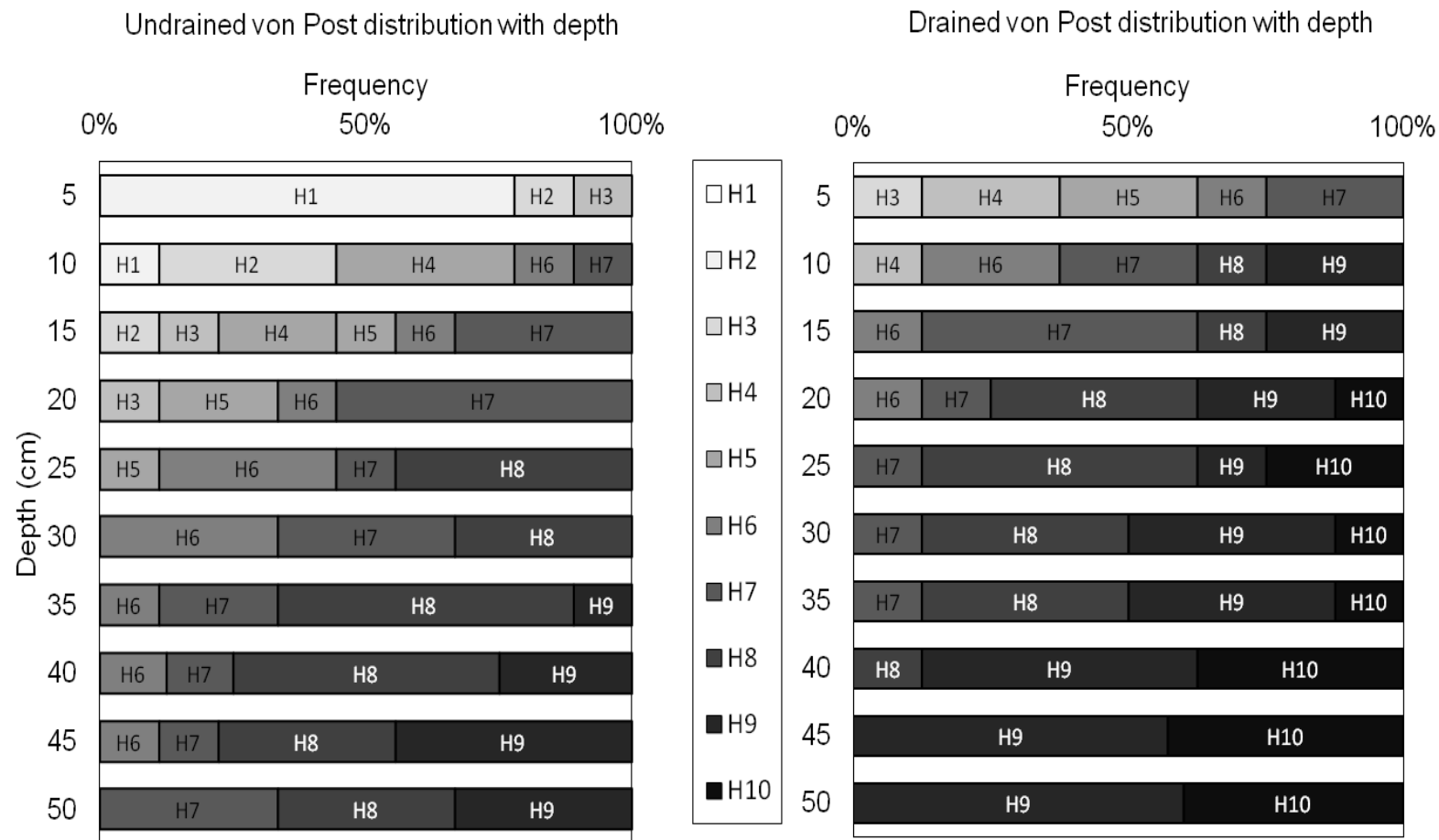
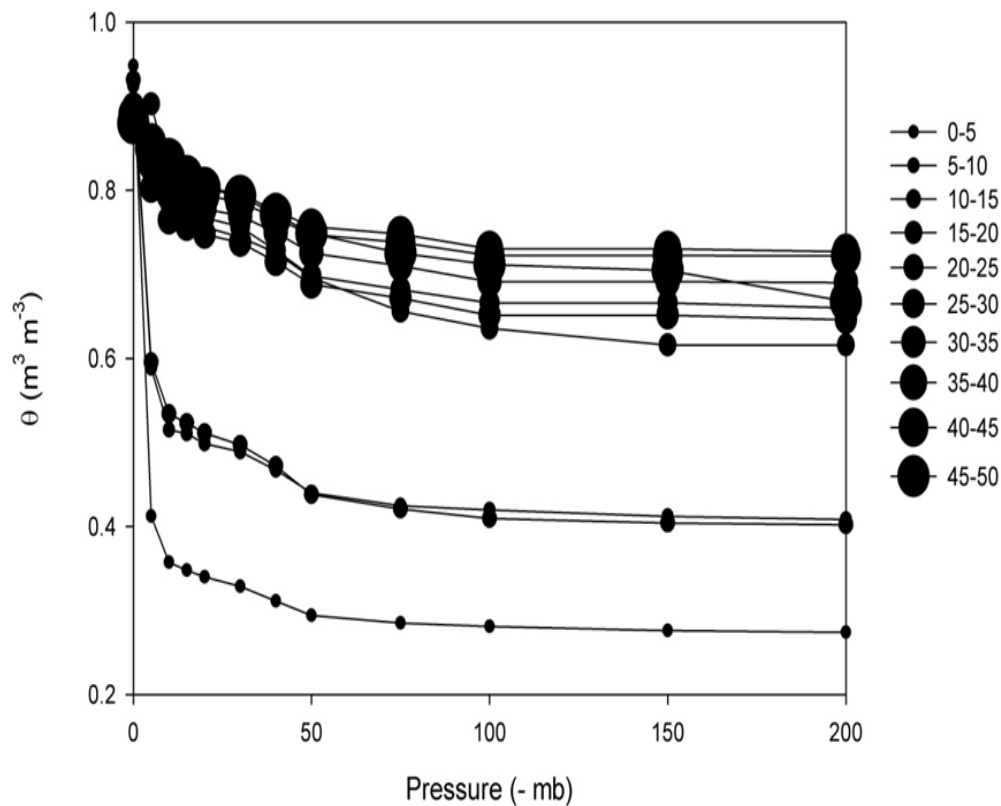


Figure 2.1 – von Post decomposition classification for the drained and undrained sites at the SRP.

Mean soil moisture retention to θ_r at 200 mb for core SRD 10M2 peat pucks at drained site of the SRP.



Mean soil moisture retention to θ_r at 200 mb for core SRUD 10M1 peat pucks at undrained site of the SRP.

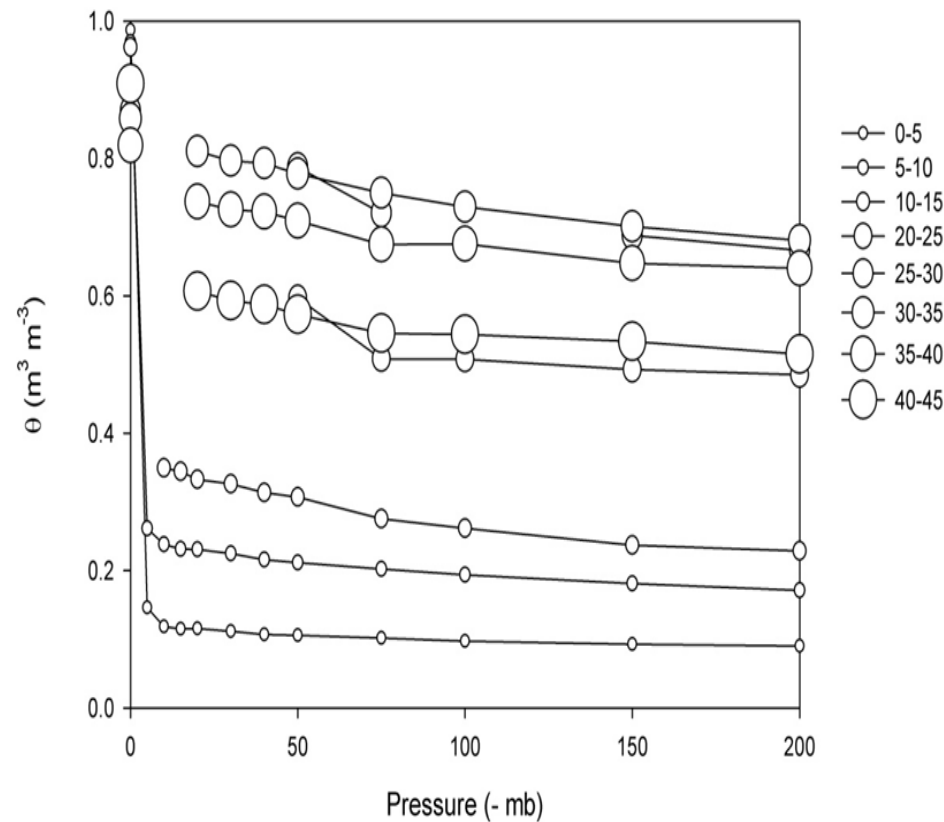


Figure 2.2 – Mean soil retention to θ_r at -200 mb for core ID 10M2 peat pucks from the drained site and core ID 10M1 peat pucks from the undrained site of the SRP.

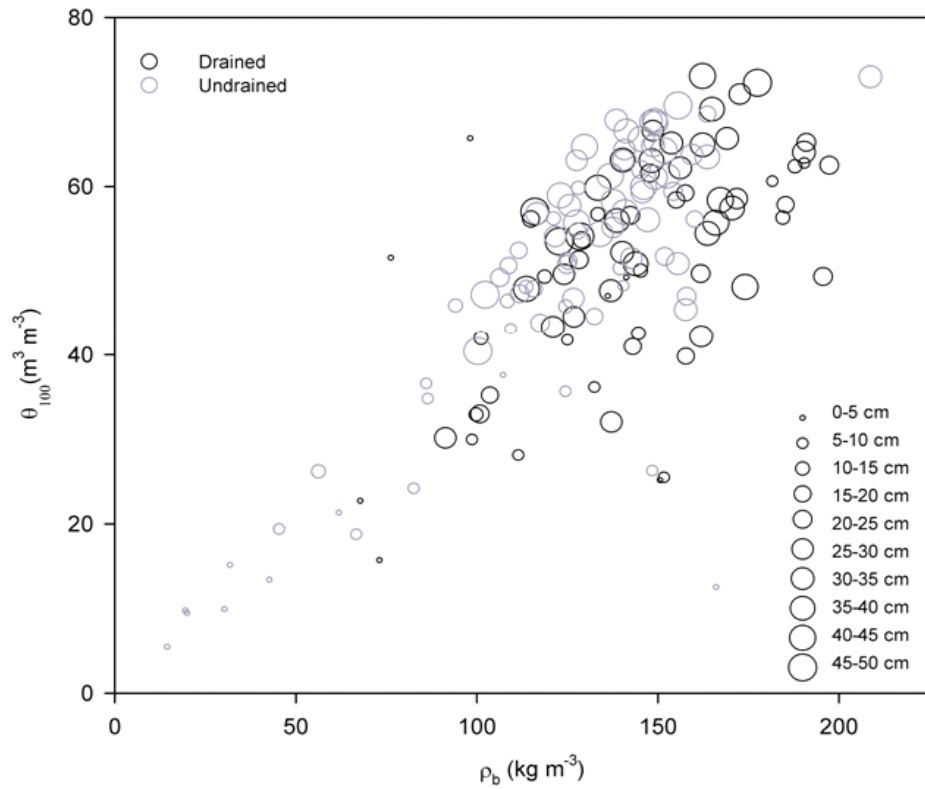


Figure 2.3 – Relationship between ρ_b and θ_r at -100 mb for drained and undrained peat from SRP for 5 cm peat puck increments between 0-0.5m depth.

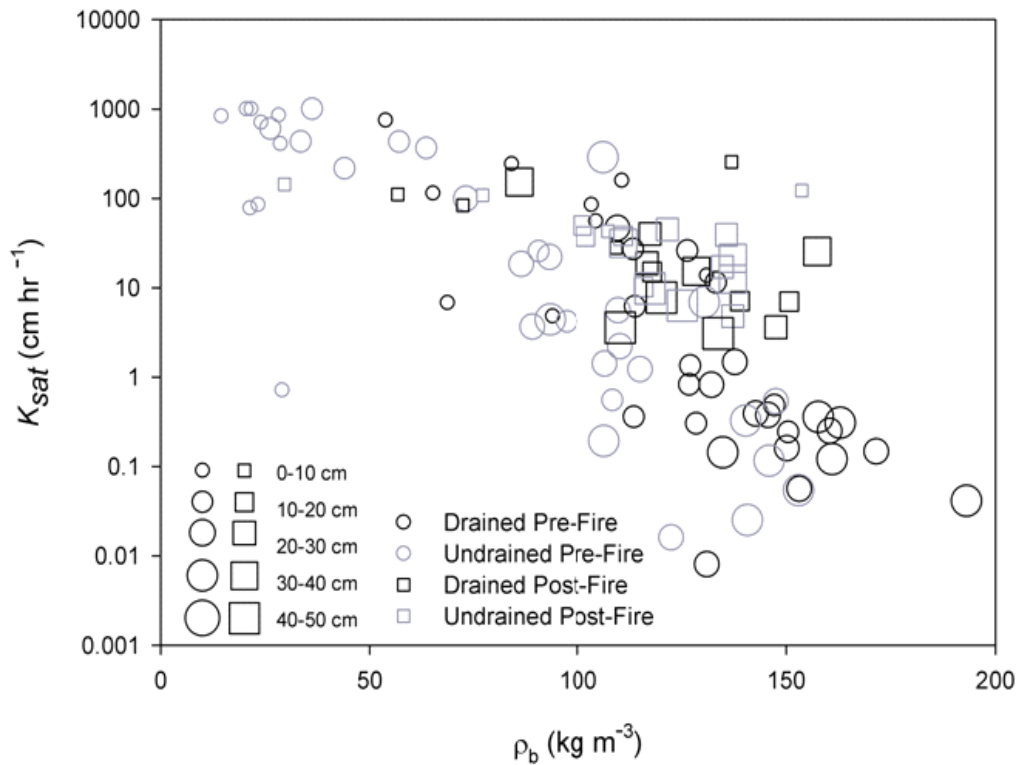


Figure 2.4 – Relationship between K_{sat} and ρ_b for drained and undrained peat from SRP both pre-fire and post-fire for 5 depths. Adapted from Silins and Rothwell (1998).

Figure 2.5 – Moisture retention curve for surface 0-10 cm peat pucks between drained and undrained sites at SRP. (B) = post-fire retention data, otherwise assume pre-fire data. Adapted from Silins and Rothwell (1998).

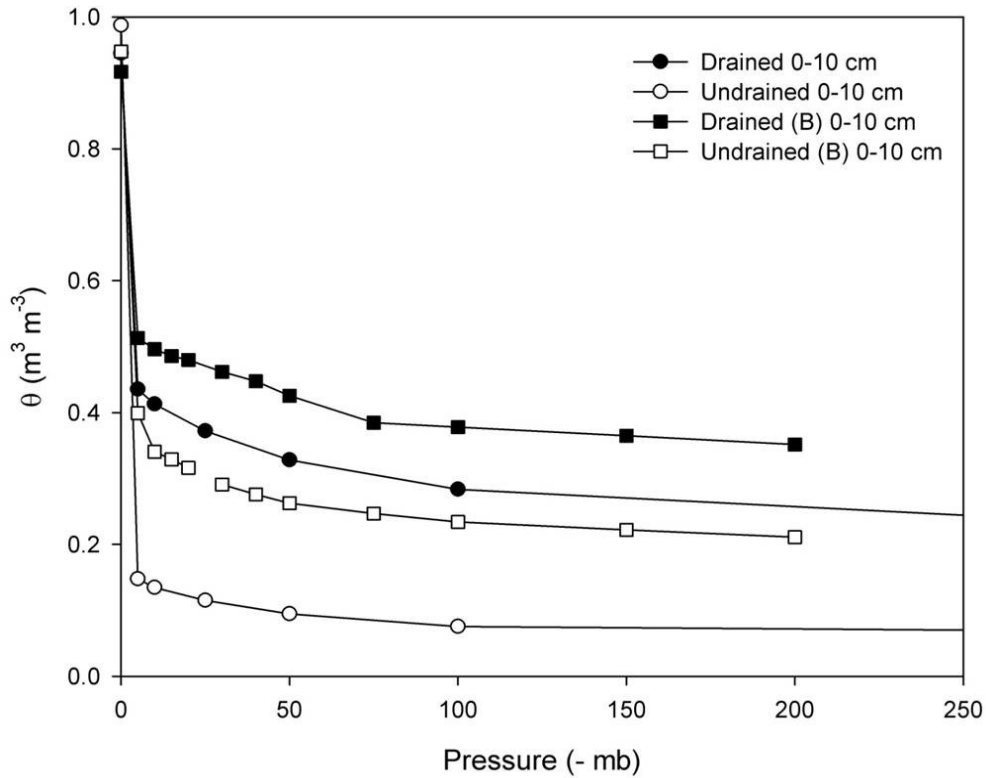
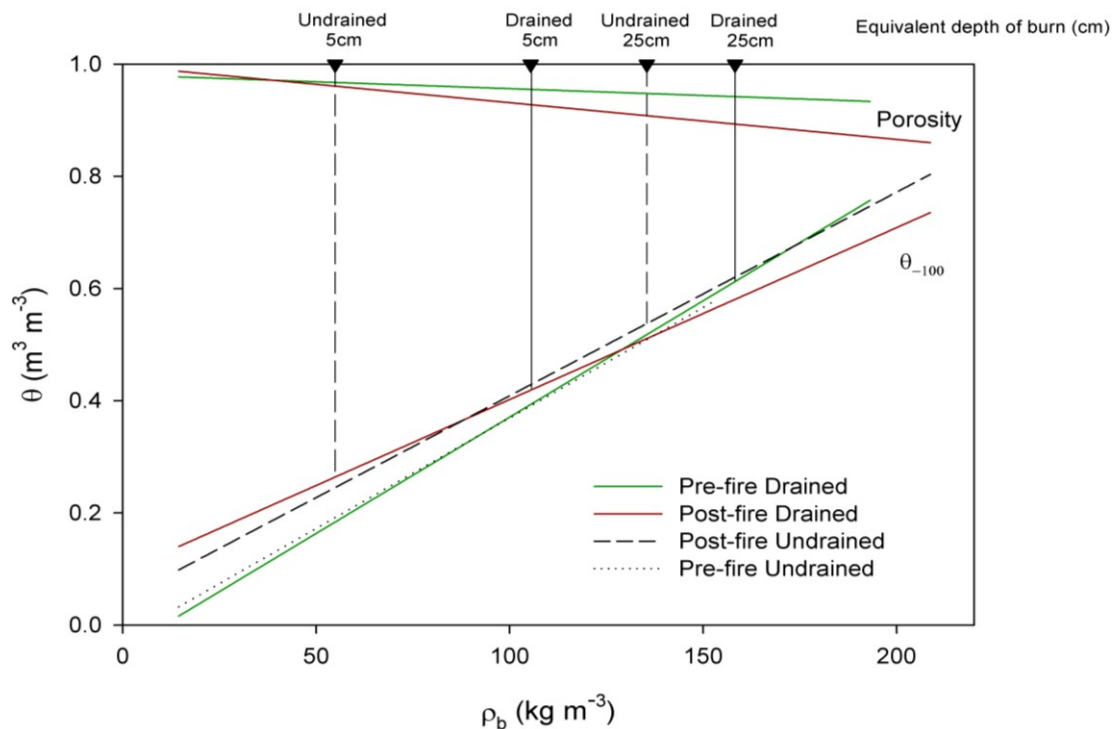


Figure 2.6 – Conceptual depth of burn changes of pre and post-fire θ_{-100} and porosity.



FIGURES 3.1-3.10

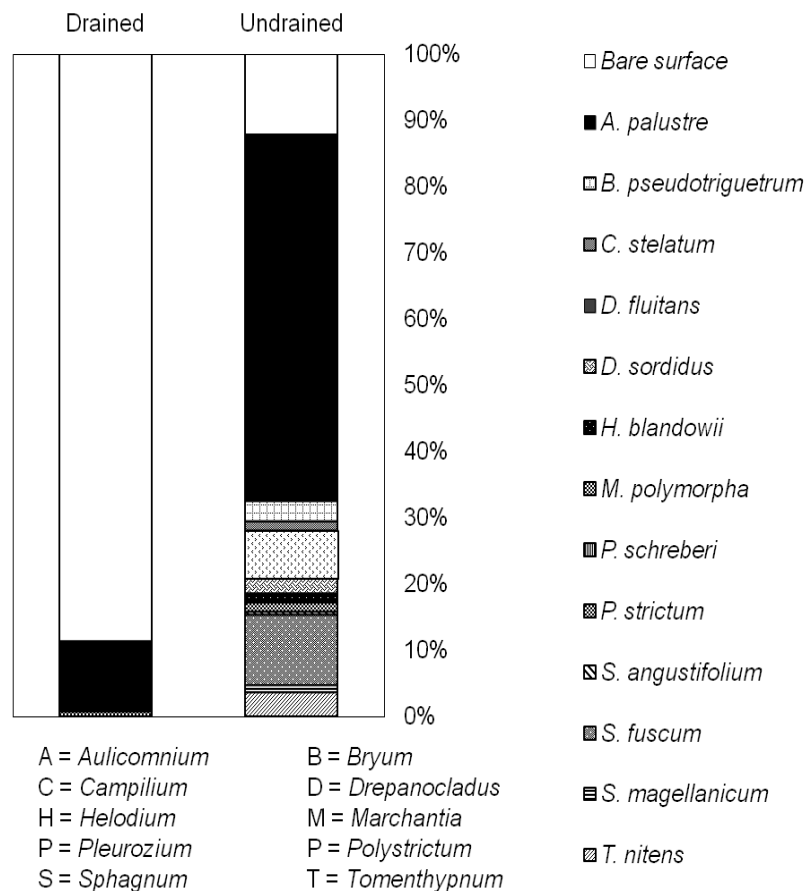


Figure 3.1 – Aggregated percent cover of mosses and true moss ground cover types within both SRP sites, July 2010 (N=100, 0.5 m² plots per site).

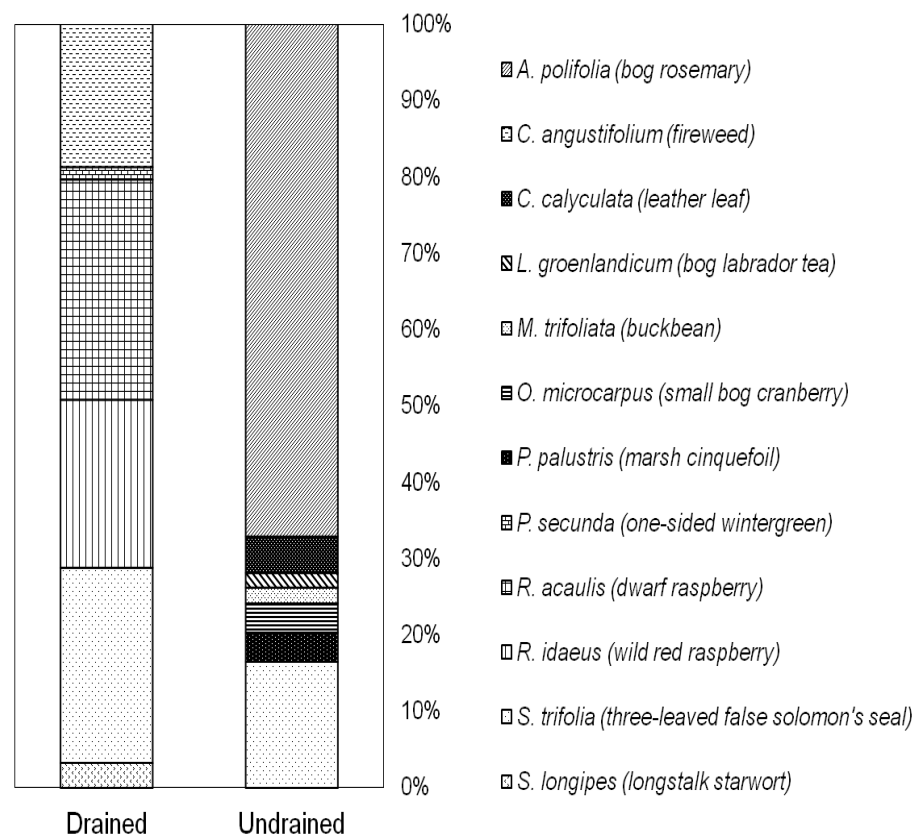


Figure 3.2 – Aggregated percent cover of understory cover types (exclusive of sedge) within both SRP sites, July 2010 (N=100, 0.5 m² plots per site).

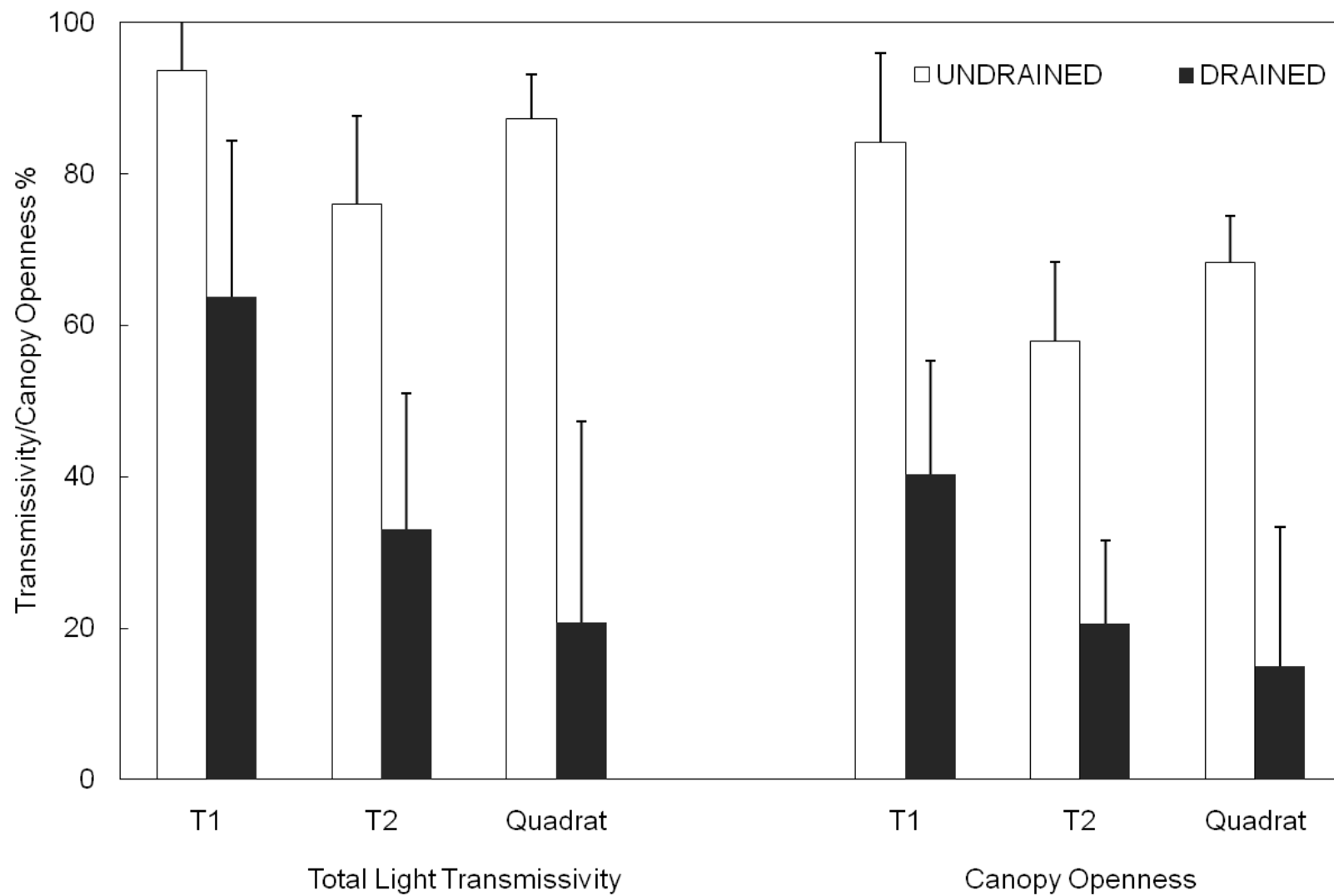


Figure 3.3 – Average total light transmissivity and canopy openness with undrained site (white bar) and drained site (black bar) for 2010 survey year.

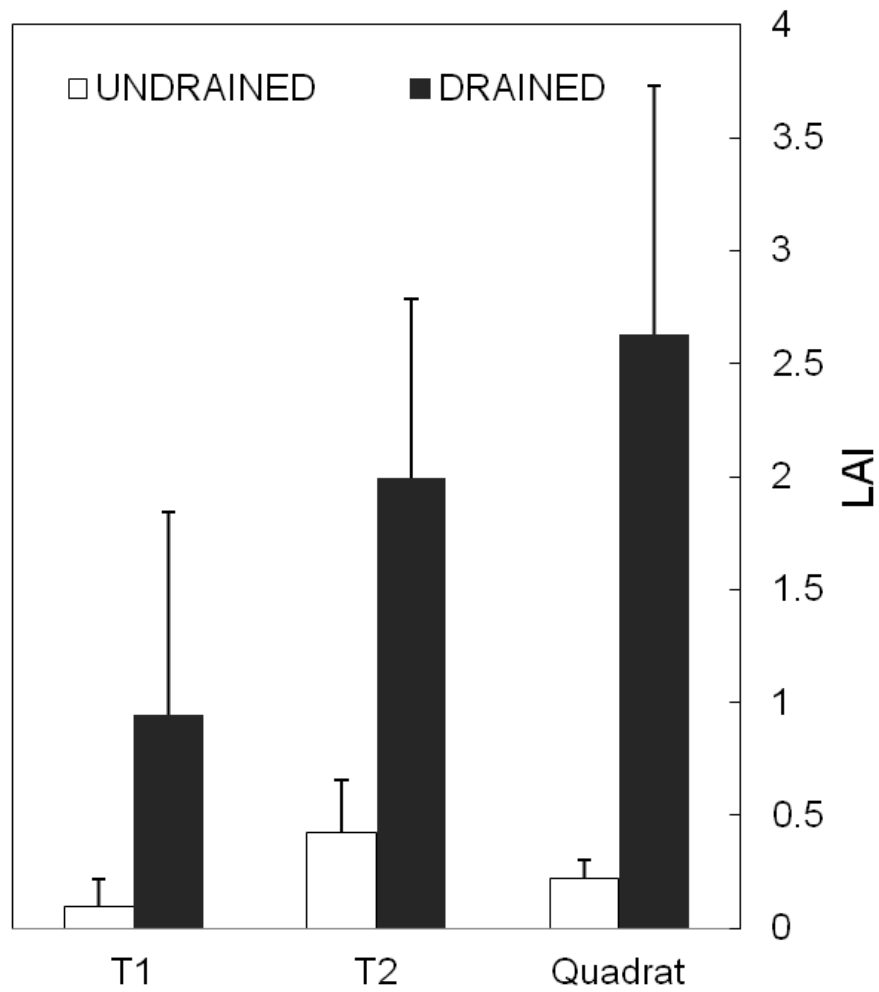


Figure 3.4 – Average calculated 4-ring LAI with undrained site (white bar) and drained site (black bar) for 2010 survey year.

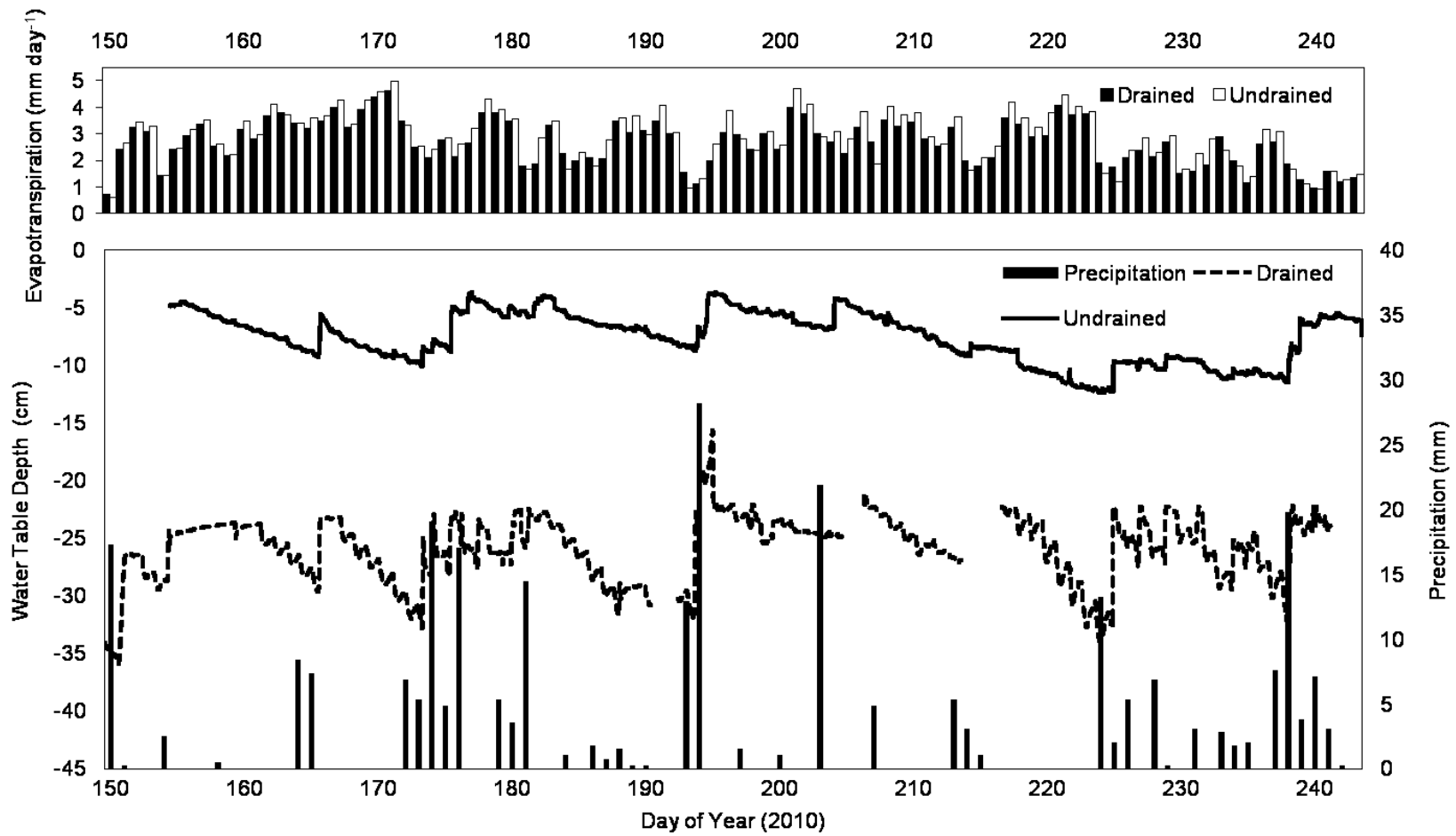


Figure 3.5 – Water table, precipitation and evapotranspiration for drained and undrained sites of the SRP.

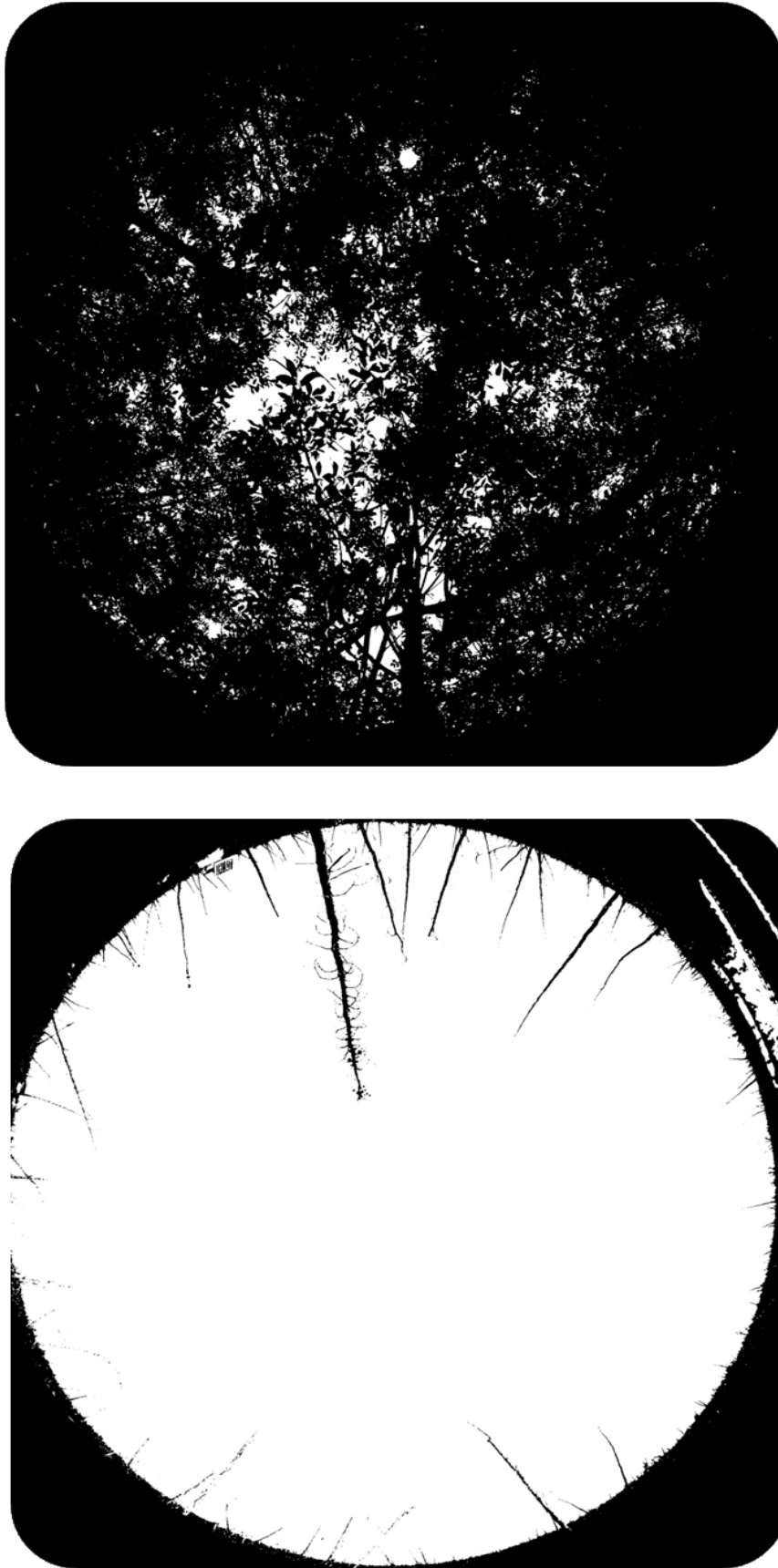


Figure 3.6 – Hemispherical photography showing canopy openness from drained (top) and undrained (bottom) at the SRP.

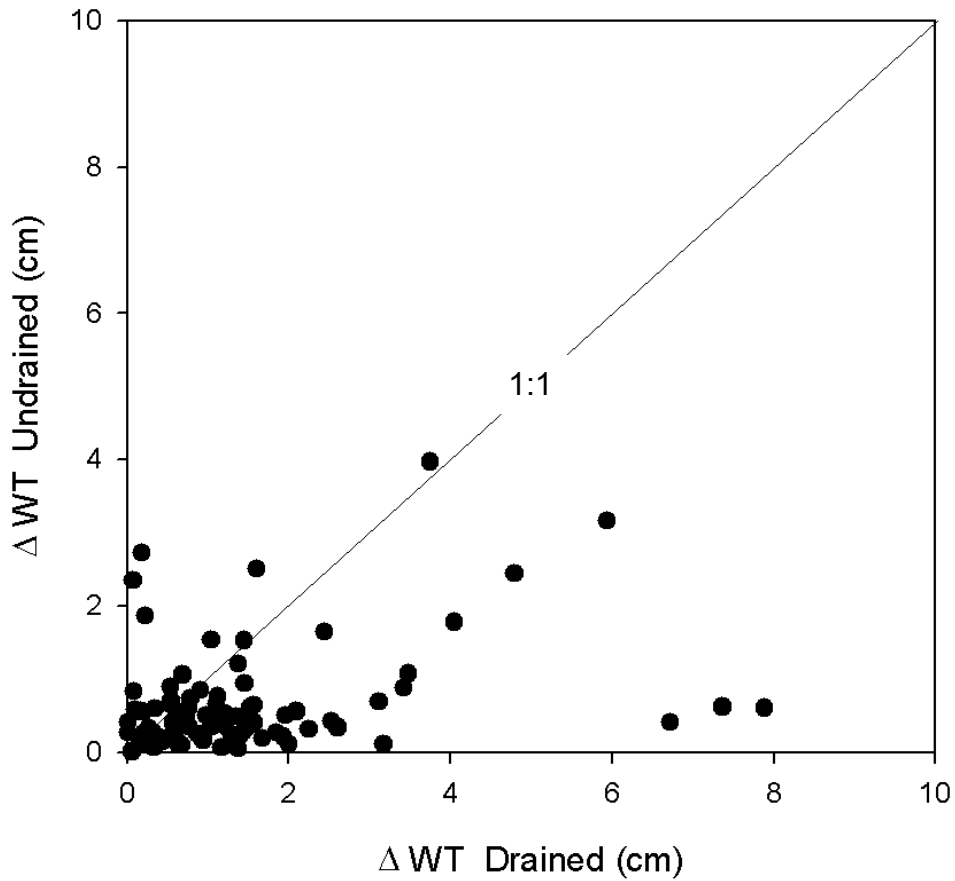


Figure 3.7 – Change in water table at drained and undrained SRP sites for rainfall events in 2010 study period

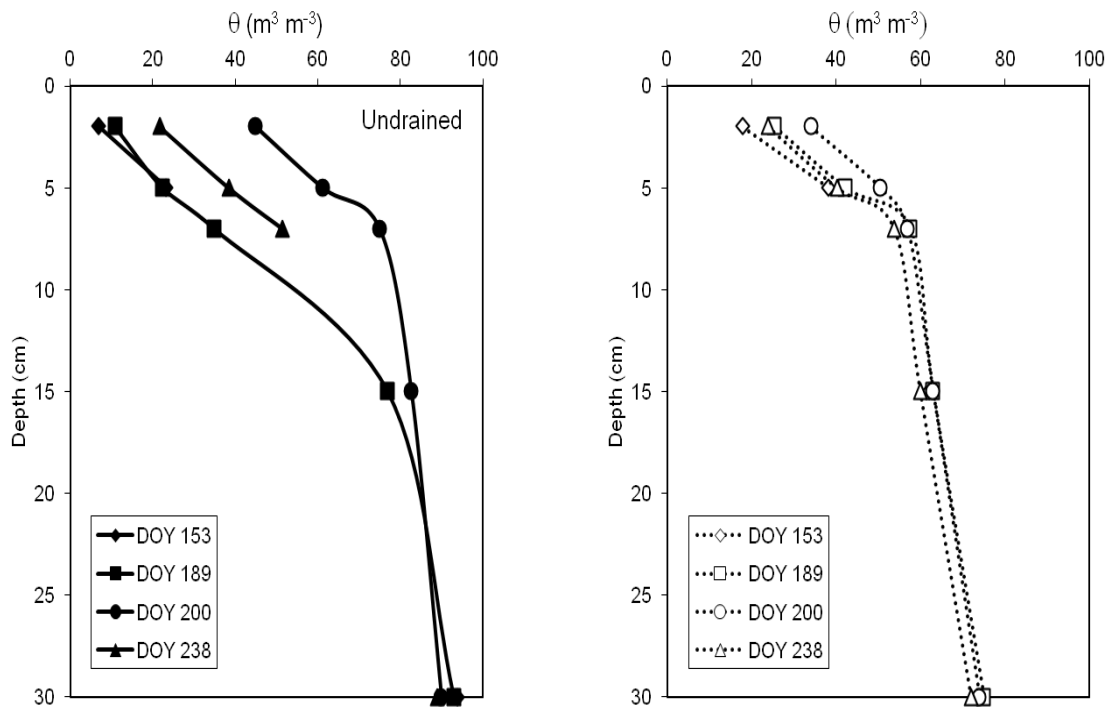


Figure 3.8 – Volumetric moisture (0.02-0.3 m depths) at SRP sites for days of year in 2010.

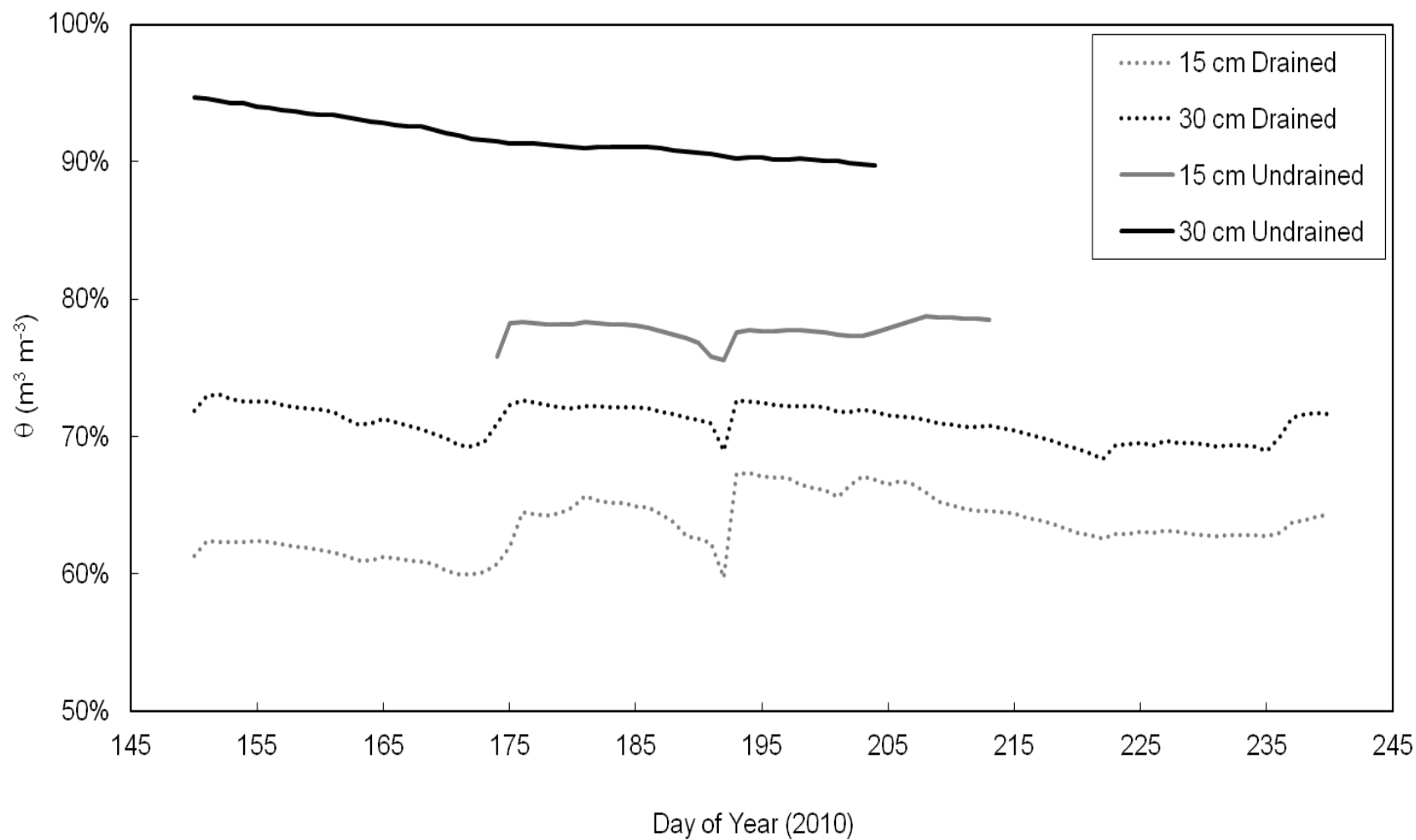


Figure 3.9 – Logged volumetric soil moisture (0.15-0.3 m depths) at SRP sites during 2010 study period.

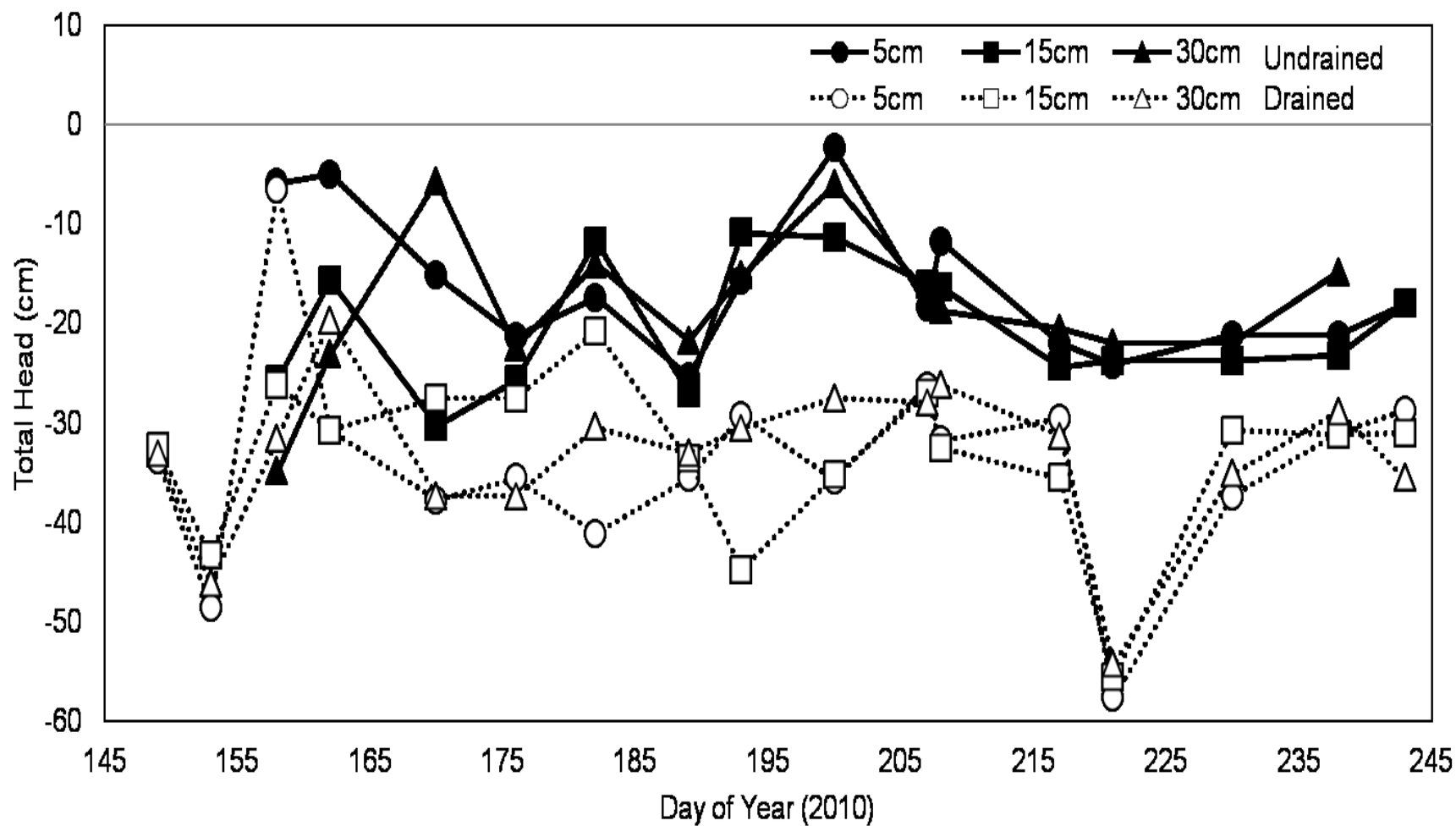


Figure 3.10 – Average soil water pressure Ψ (expressed as total head relative to surface) for 5, 15 and 30 cm below surface.

APPENDIX A.

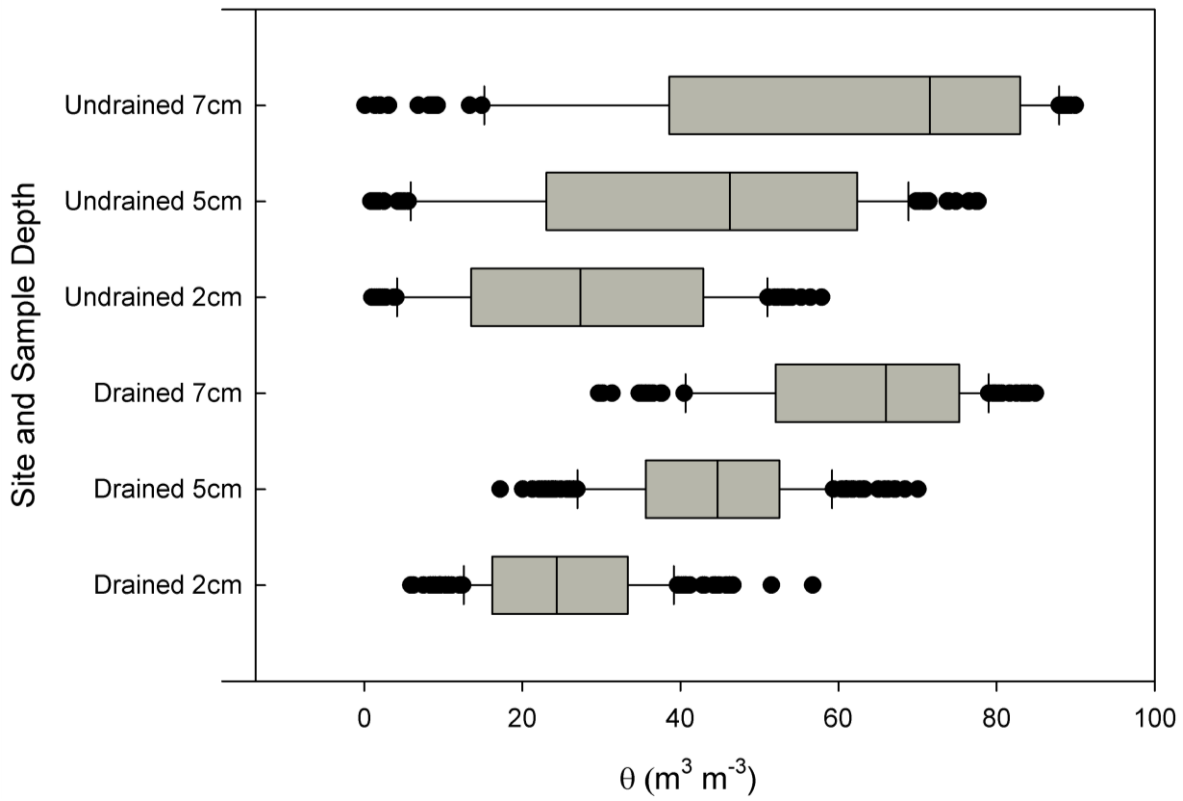


Figure A1.1 – Boxplots of surface (0.02m) and near-surface (0.05-0.07m depths) of volumetric moisture content for both drained and undrained SRP sites.

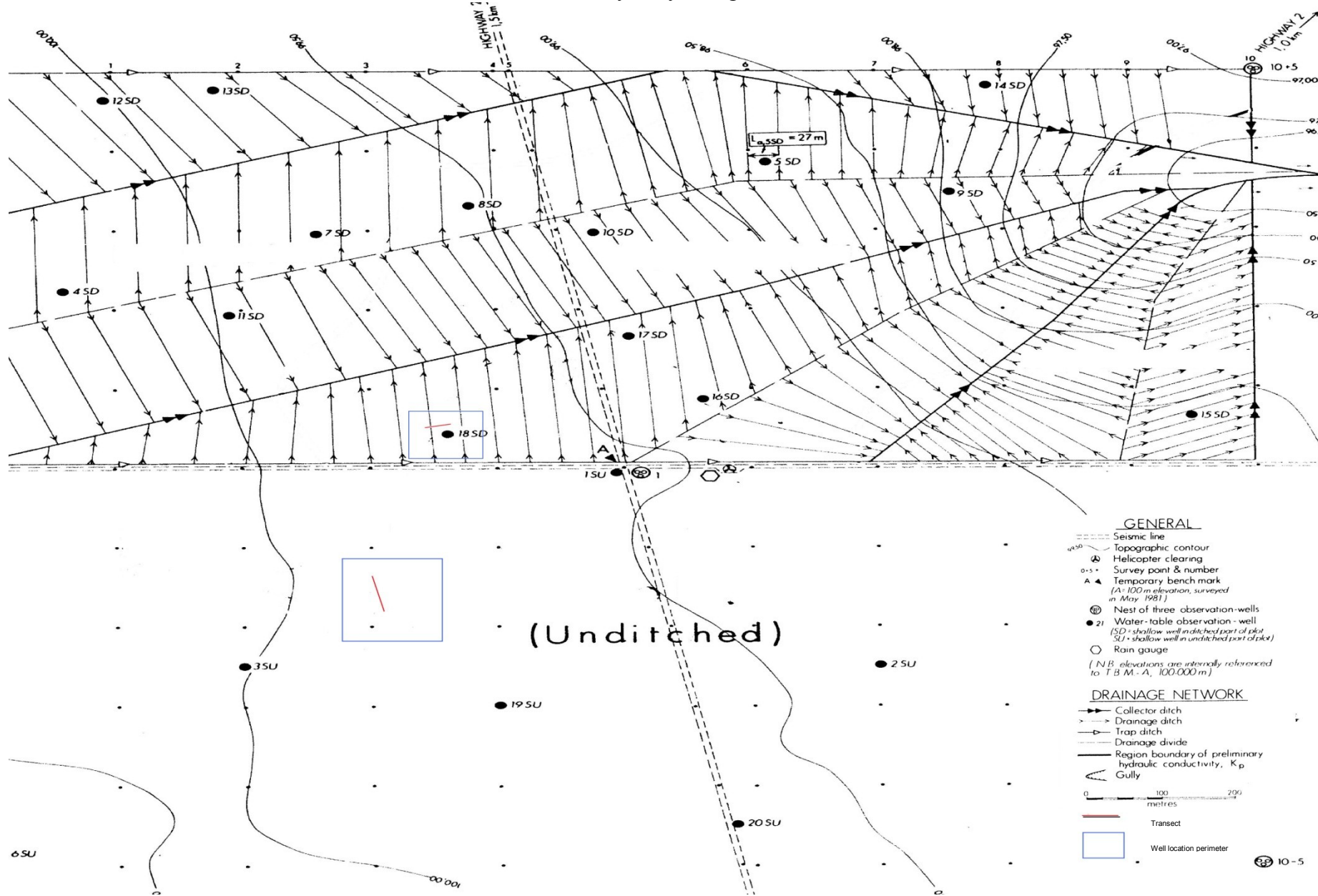


Figure A1.2 – Saulteaux River Site Map

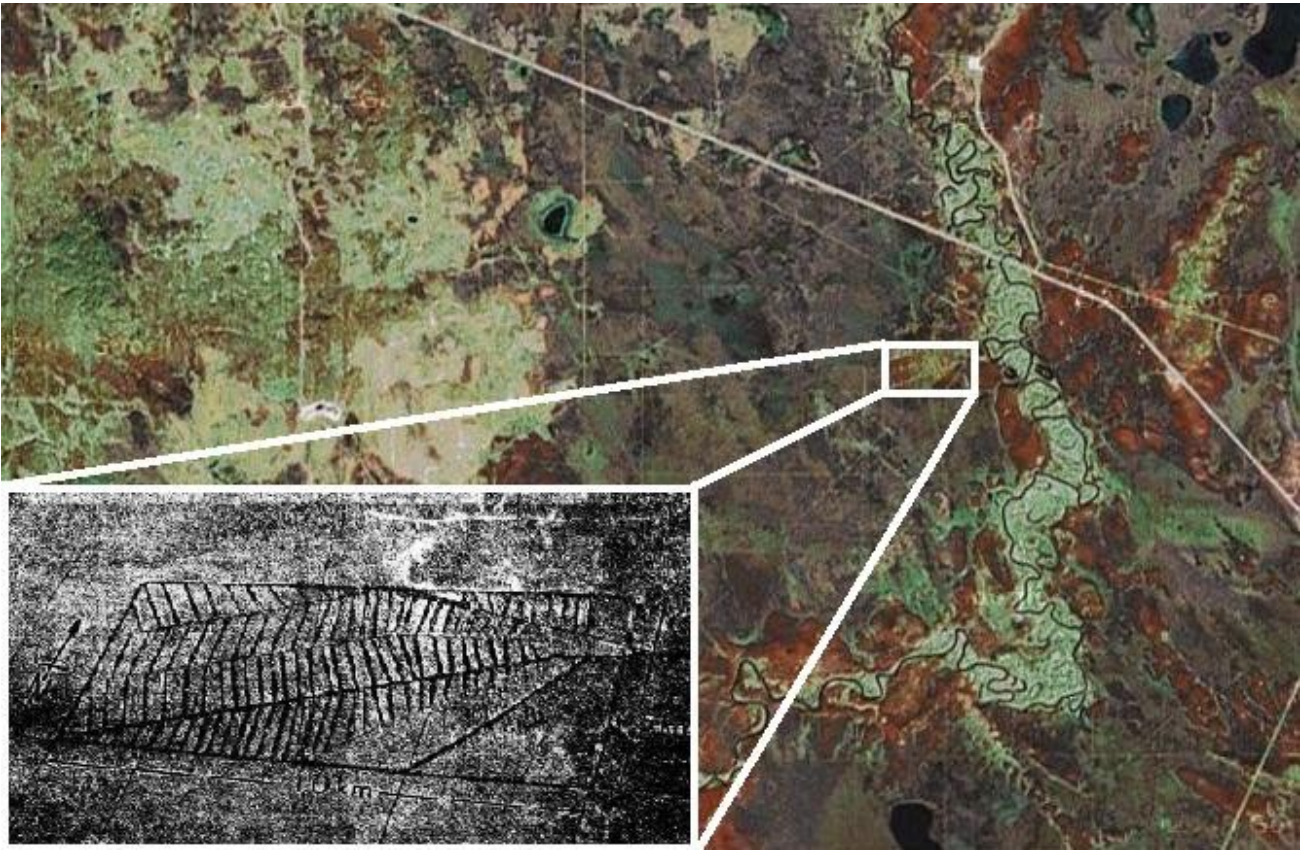


Figure A1.3 – Sauleaux River Aerial Photo



Figure A1.4 – Drained site overstory with black spruce remnant stand



Figure A1.5 – Saulteaux River drained site transect instrumentation (L) and representative surface cover (R)