

Using a water balance approach to evaluate peatland restoration in South-Eastern Manitoba

By

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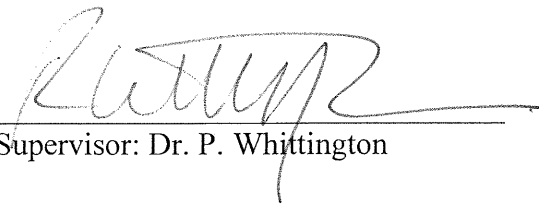
**Using a water balance approach to evaluate peatland restoration
in South-Eastern Manitoba**

Submitted by: **Haley Nelson**

In partial fulfillment for the requirements for the degree of

MASTER OF SCIENCE (ENVIRONMENTAL AND LIFE SCIENCES)

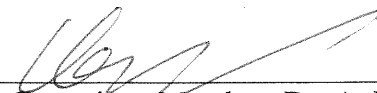
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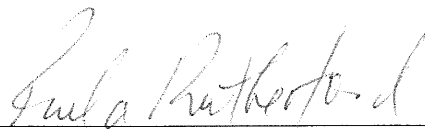
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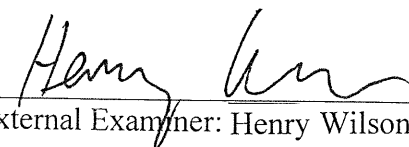
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Abstract

Prompt and active restoration of extracted peatlands back to a carbon accumulating ecosystem is a key component in the sustainable management of the growing media. Water availability is usually the limiting factor of success, thus quantifying a water balance is critical to understanding new restoration methods, especially in regions where water availability is a particular concern, such as the sub-humid climate in Manitoba. Typically, harvested peatlands are restored to bogs, but in Manitoba there has been a recent effort to restore remnant sites to fens. At this site three different surface recontouring techniques were used: Finished (flat, two passes with bulldozer), In-Progress (bunds created from one pass with the bulldozer) and Remnant (no passes, left untouched after extraction), each instrumented with wells, lysimeters, and tensiometers to examine hydrologic differences with surface roughness. Evaporation is the dominant water loss from the system with an average loss of 4.6 and 3.2 mm/day from May to August 2020 and 2021. In spite of recontouring restoration efforts, the water table dropped more than a metre below the surface, resulting in very low soil water pressure (-28.3 mb in May 2021 to as low as -334 mb in July 2021, with an average of -124 mb from May to August 2021) and conditions too dry to support fen bryophyte vegetation. By understanding the magnitude and distribution of water fluxes across the different restoration techniques, restoration efforts can be modified to reduce major water losses in future planning.

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1 Introduction

1.1 Thesis Structure

This thesis comprises two manuscripts with the overall purpose of calculating a water balance to further understand the hydrology of the Moss Spur 2 peatland, a previously harvested, now restored, peatland. The first manuscript (Chapter 2) focuses on the snow accumulation and melt dynamics, often omitted or ignored in water balance studies, to fully understand the recharge during the spring melt period from the previous year's losses. The second manuscript (Chapter 3) is the more traditional summer water balance (May to Sept 2020 and 2021) approach to quantify if the hydrology of the site is suitable to support fen bryophyte vegetation and assess the difference in surface recontouring techniques.

A brief overview of peatland restoration in Canada is needed to place the aforementioned manuscripts into context. This is largely because Manitoba's hydrogeomorphic setting and climate is quite different than that found in Quebec, where much of the research that informed the Peatland Restoration Guide (Rochefort, L., Quinty, F., Campeau, S., et al., 2003) was completed.

1.2 What is a peatland?

Wetlands are some of the most productive ecosystems on Earth providing habitat to many organisms and plant species, filtering and purifying water, flood and drought mitigation, and carbon sequestration (Environment Canada, 2016). In Canada, peatlands are the dominant wetland type (90%), covering ~113 million hectares (Daigle and Gautreau-Daigle, 2001). Peatlands in Manitoba cover ~19 million ha or 29.5% of the provincial land area, which represents ~17% of Canada's total peatlands (Daigle and Gautreau-Daigle, 2001). Wetlands in Canada are subdivided into two broad categories: organic wetlands and mineral wetlands

(NWWG, 1997). Organic wetlands, commonly referred to as peatlands, are wetlands where organic soils have accumulated to a minimum thickness of 40 cm (NWWG, 1997). Peatlands are generally separated into two classes, bogs and fens, on the basis of peatland's water source (NWWG, 1997), with bogs being ombrogenous (precipitation fed only) and fens also known as minerotrophic peatlands, receiving water from terrestrial sources (surface water and groundwater) in addition to precipitation inputs (NWWG, 1997). Some swamps are also peatlands (NWWG, 1997), but these are not prevalent in Manitoba (Halsey et al., 1997) and not discussed further in this thesis. Peatlands typically form one of two ways, either terrestrialization (infilling of a shallow lake) or by the plaudification of poorly drained land, where the rate of production of biomass exceeds the rate of decomposition resulting in an accumulation of organic matter (Quinty and Rochefort, 2003). With the accumulation of organic matter a true moss and/or sedge dominated fens can form that has access to the mineral rich groundwater, but over time this can transition into an ombrogenous bog with the continuous accumulation of peat, resulting in a domed surface elevation disconnecting the wetland from the mineral-rich groundwater, thus receiving nutrients from precipitation only leading to a more acidic environment more favorable for *Sphagnum* mosses (Borkenhagen and Cooper, 2016; Quinty and Rochefort, 2003; Rydin et al., 1999); thus, bogs succeed from fens.

1.3 Roles of peatlands

Peatlands are a valuable terrestrial ecosystems that play an important role in the regulation of climate due to their effect on the global biogeochemical cycles (Chapman et al., 2003; Harenda et al., 2018; Lazcano et al., 2018). Peatlands act as the landscape's water buffer providing water storage, filtration and supply to stabilise the surrounding macroclimate, to help mitigate floods and minimize erosion (Harenda et al., 2018). The water-logged soil conditions

found in peatlands limit the amount of available oxygen, creating anerobic conditions which slow decomposition rates (Lamers et al., 2015), resulting in the accumulation of soil carbon, leading peatlands to be one of the largest terrestrial soil carbon sinks (Lazcano et al., 2018; Rochefort et al., 2012). Peatlands can also support a large biodiversity of species, housing many endemic organisms specifically adapted to the unique characteristics (acidic, water-logged, nutrient poor) of a peatland ecosystem (e.g. *Sarracenia purpurea*) (Quinty and Rochefort, 2003; Urák et al., 2017).

1.4 Disturbing Peatlands

When looking at peatland disturbances it is important to define what type of disturbance has occurred (Connolly and Holden, 2013) since peatlands can be naturally disturbed through extreme climatic and geological events, lightning fires, pathogen outbreaks and natural colonization of invasive species, but more frequently are disturbed anthropogenically (Urák et al., 2017). Anthropogenic disturbances occur at various spatial and temporal intensities, all for economic gain to humans (Harenda et al., 2018). In Europe, the main anthropogenic activities disturbing peatlands include afforestation, conversion for agriculture, livestock overstocking, drainage and mining for fuel, burning and wind-farm development (Connolly and Holden, 2013). In North America, the main anthropogenic activities disturbing peatlands are flooding during the construction of hydro-electric dams, open-pit mining for oil sands and through the drainage and extraction for horticulture substrate and fuel source (Rochefort et al., 2012). But for the purpose of this thesis, we will be focused on the drainage and extraction of peatlands for horticulture purposes.

Peat extraction has been ongoing in Canada since the 1860s in the eastern provinces (Government of Manitoba, 2022) but only in Manitoba since the 1940s, mainly from bog

peatlands (Swystun et al., 2013) with mechanical extraction as the preferred extraction method since the 1960s (Robert et al., 1999). The main type of mechanical extraction is vacuum extraction (Robert et al., 1999). In preparation for vacuum extraction a network of drainage ditches are installed (~30 m apart and ~1 m deep) to lower the water table, allowing the heavy machinery access to the field (Petrone et al., 2003; Robert et al., 1999). After drainage, the surface vegetation is removed (Petrone et al., 2003) leaving the deeper catotelmic peat exposed. Through drainage and the removal of the upper layer (acrotelm) during peatland harvesting the hydrological properties of the extracted bog are altered by disrupting the hydraulic conductivity below (Price, 1997) severely limiting water availability, inhibiting the re-establishment of the non-vascular *Sphagnum* mosses (Campeau and Rochefort, 1996). Peatland sites can be harvested for 30+ years, with an average of 7 cm from a 6 m peat profile extracted per year (Radjevic, 2008). Unlike extracted bogs, undisturbed peatlands have self-regulating abilities (shrinking/swelling) to respond to moisture conditions throughout the season (Ingram, 1983), shedding water when the water table is high, and preventing drastic water table fluxes in drought periods (Price and Ketcheson, 2009). As mentioned above in extracted peatlands the water table is lowered, through a series of drainage ditches to allow the harvesting of the peat substrate. The change in the water table position affects the moss structure which in turn lowers the peats transmissivity, specific yield, and increases the peat's decomposition rate, and causes peat deformation as peat is highly compressible and when it dries out the substrate contracts and deforms which changes the hydraulic properties of the peat (Waddington et al., 2015).

1.5 Restoring Peatlands

Successful peatland restoration post-harvesting aims to return the extracted bog to a carbon accumulating ecosystem by restoring a suitable hydrology and by successfully

reintroducing target species (Bérubé and Rochefort, 2018; McCarter and Price, 2013; Price et al., 1998; Rochefort et al., 2003). Revegetation paired with rewetting has been linked to a reduction of carbon emissions, with a net C uptake in the extracted peatland within 14 years following active restoration (Nugent et al., 2018; Strack and Zuback, 2013). Not restoring after extraction causes large CO₂ emissions released to the atmosphere initially (1 year) and a decade later (15 years) (Nugent et al., 2019) stressing the importance of prompt active restoration post extraction. In the late 1990s (González and Rochefort, 2014) the moss layer transfer technique (MLTT) was developed in North America, as an active restoration strategy to restore ombrotrophic conditions suitable for bog species (Nugent et al., 2019; Rochefort et al., 2003). The MLTT method involves 1) reprofiling the surface topography to create a more uniform surface to ensure an even distribution of water available to the non-vascular mosses, 2) diaspore collection from a donor peatland, 3) spreading donor material on the restoration site in a 1:10-1:15 ratio (1 m² plant material over a 10-15 m² area), 4) straw mulching to reduce evaporation losses by increasing the albedo and lowering the vapour pressure deficit, 5) fertilizing site with a low dose of phosphorus and finally, 6) blocking drainage network to raise the water table (Hugron et al., 2020) to restore the hydraulic regime characteristics of a peatland ecosystem (Rochefort et al., 2003). The MLTT method has a ~80% species transferred establish success rate (Hugron et al., 2020).

In 1996 (Price) researchers established a successful water table threshold of 0.4 m below the surface for *Sphagnum* growth, if it falls deeper than the threshold *Sphagnum* growth will not be sustainable due to lower pore water pressures (lower than -100 cm). Graf et.al (2008) discovered fen vegetation returned to abandon sites more readily than bog vegetation. Indicating under ideal conditions (high water table and thin layer of residual peat) that remnant sedge peat can establish more rapidly than remnant *Sphagnum* peat (Poulin et al., 2005, Graf et al., 2008).

LaRose et al. (1997) discovered by leaving open water ponds in the restoration plan aided in the rewetting process. Campeau et al (2004) experimented with the remnant surface topography by enhancing the microtopography before the diaspore spreading. They found that the higher relief areas of the recontoured site not suitable for moss recolonization (Campeau et al, 2004).

The first ecosystem scale bog restoration project was undertaken at the Bois-des-Bel (BdB) peatland located in Eastern Quebec (Shantz and Price, 2006a). Restoration of this site began in 1999 after being left remnant for 19 years with minimal natural revegetation (Shantz and Price, 2006a) and after 10 years of restoration efforts McCarter and Price (2013) found a well-established *Sphagnum* moss cover, reduced runoff at the site and the water table was higher than the unrestored site indicating that the hydraulic processes seemed to be recovering. However, McCarter and Price (2013) also found the post-restoration regenerated moss to be hydraulically disconnected from the remnant cutover peat from the vertical growth of the regenerated moss layer developing more rapidly than the rise in the water table. These findings suggest that there is more to the restoration process than vegetative recovery, and the hydraulic connectivity between the remnant and regenerative peat layers is required for long term success to a functional carbon sequestration ecosystem.

1.6 Future of peatland restoration

Vacuum extraction can expose minerotrophic peat richer in minerals with a higher pH, thus creating conditions which may be more suitable for restoration towards a fen (Graf and Rochefort, 2008), rather than a bog but restoring a site to a fen is likely more complicated due to the differences in their hydrologic patterns and processes (Chimner et al., 2017). The major driving force to successfully restore the extracted peatland to a fen is a high water table (Priede et al., 2016), which makes the manipulation of the remnant hydrology a key step in restoration

(Gorham and Rochefort, 2003) through rewetting the field to improve remnant hydrological conditions for successful vegetative recovery (Malloy and Price, 2014). Rewetting is achieved through the hydrological management of the site, which involves blocking and infilling the active drainage network and reprofiling the field to achieve an even distribution of water by leveling sloping fields, creating holding basins and building berms (Quinty et al., 2020).

Extensive research on fen restoration techniques has been conducted in Europe but these techniques are not suitable in North America since they focus on creating a *managed* fen ecosystem rather than restoring natural fen functions (Graf and Rochefort, 2008; Lamers et al., 2015). In North America, several studies have examined restoration techniques to restore minerotrophic conditions suitable to support fen species on remnant cutover peatlands (Borkenhagen and Cooper, 2016; Cobbaert et al., 2004; Graf and Rochefort, 2008; Lazcano et al., 2018; Malloy and Price, 2014; Rochefort et al., 2016). They found that the techniques used in diaspore collection and reintroduction of the MLTT was effective at establishing vascular fen species and true mosses found in natural fens but not successful at establishing fen bryophytes (Cobbaert et al., 2004; Graf and Rochefort, 2008). Fen bryophytes are more difficult to establish, but are important to the overall ecosystem function of a natural undisturbed fens (Graf and Rochefort, 2010). Fen bryophytes are capable of regenerating vegetatively and succeed the creation of microhabitats with increased shade and moisture (Graf and Rochefort, 2010).

1.7 Restoration in Manitoba

With the growing peat industry in Manitoba came the Peatland Stewardship Act in 2014 (Manitoba Government, 2014). The Peatland Stewardship Act prohibits exploration and harvesting of peatlands without a permit or licence, license holders must comply with the Act which states that any altered peatland must be restored, rehabilitated or reclaimed post extraction

(Manitoba Government, 2014). The MLTT was developed in Quebec for bog restoration, but in Manitoba there exists a different climate (drier with high evapotranspiration demands) and hydrogeomorphic setting, which links the hydrology and geomorphology of the area (Sidle and Onda, 2004). Many sites found in South-Eastern Manitoba have been extracted to the underlying sedge-peat layer where the remnant peat conditions resemble closer to a minerotrophic peatland than a ombrotrophic peatland leading restoration aimed towards the return of a fen ecosystem (PERG, 2017). Though the first fen restoration project in Canada, conducted in Quebec, showed the method used to restore fens required improvement (Rocheffort et al., 2016) the sites found in Manitoba (listed below) provided the perfect opportunity to try different techniques, gain knowledge on fen hydrology and compare the carbon flux differences in natural and restored ecosystems (PERG, 2017). With the decision to restore sites to fens, and the different hydrogeomorphic setting, a simple application of the MLTT may not be appropriate, and it would need to be adapted for the conditions present in the Prairies to develop suitable management practices.

1.7.1 South Julius

South Julius is a 237 ha active peat extraction site with a 35.5 ha experimental site found in the southeastern tip of the field with two distinct areas 1) bare peat, sparsely revegetated left since 2014, 2) regenerated sedge covered fen, rewetted in 2006 (15 years ago) (PERG, 2017). Researchers wanted to investigate if simply blocking the remnant drainage network to raise the water table could restore vegetative and ecosystem function to the site (Courchesne et al., 2023). The study found that rewetting was successful in the re-establishment of fen vegetation and it restored carbon sequestration comparable to the undisturbed fen but restoration outcomes were strongly influenced by hydrological management as excessive inundation limited vegetation recovery and substantially increased the methane emissions (Courchesne et al., 2023).

1.7.2 *Elma North*

Elma North is a 7.5 ha field that was last extracted in 2013 and restoration efforts began in September 2015 (Hawes 2018). Researchers wanted to investigate if active restoration focused on infilling the remnant drainage ditches and reprofiling the field could improve remnant hydraulic conditions of the extracted site (Hawes, 2018). The site was reprofiled to a checkerboard pattern to retain more snowmelt and rainfall. The study found that active restoration by recontouring the site raised the water table by 1 metre with the restored sites water table maintaining within the acceptable threshold for *Sphagnum* establishment and soil moisture conditions improved with the average soil water tension at -14.5cm (Hawes 2018).

1.7.3 *Moss Spur 2*

A part of the restoration efforts at the Moss Spur 2 site, which is the study site for the chapters below, looked at constructed ecotones and different water retention treatments (5-pond, 3-pond, Berm and Control) to improve hydraulic conditions to restore extracted peatlands to fen ecosystems (Yamoah and Whittington, submitted). They found that out of the four water retention treatments the 5-pond maintained the wettest conditions with a higher water table and greater soil moisture, the 3-pond configuration showed moderate improvements, the Berm was generally less effective than both pond treatments and the Control (no treatment) retained the least moisture and experienced the greatest drying (Yamoah and Whittington, submitted). By recontouring the research area they saw an average water table rise of ~30 cm. Yamoah and Whittington (submitted) concluded that by constructing ecotones and incorporating a water-retention features the hydraulic conditions were significantly improved to restore the site to a fen ecosystem.

1.8 Objective of the Thesis

The overall objective of this thesis is to calculate the water balance of an extracted bog to see if the hydraulic conditions are favourable to restore the site to a fen by examining the effects of different surface microtopographies. With specific objectives for each of the two manuscripts. The objective of chapter two is to evaluate how peatland harvesting and restoration efforts affect snow distribution and melt, while the objective of chapter three is to determine the water balance of a 80-ha remnant peat field research site that was last extracted in 2018 and examine if the different surface microtopographies (Remnant – no reprofiling, In Progress – 1 pass with the dozer creating linear mounds, and Finished – leaving a smooth surface, closing all drainage ditches) had any impact on remnant hydraulic conditions.

2 Recontouring techniques and their impact on snow accumulation and melt in an extracted peatland in SE Manitoba

2.1 Introduction

Snowmelt runoff is an important component of the annual water budget in both disturbed (Ketcheson et al., 2012; Metcalfe and Buttle, 2001) and undisturbed peatlands (Price & Ketcheson, 2009). However, there is little information on the variation of snow accumulation and melt between different peatland types (Whittington et al., 2012). Additionally, while peat harvesting has been ongoing in Canada since the 1860s (Government of Manitoba, 2022) in the eastern provinces (and Manitoba since 1940) (Swystun et al., 2015) only a few studies have focussed on the impact of peatland harvesting on snow and melt dynamics (e.g., Ketcheson et al., 2009); fewer studies have explored how restoration following harvesting influences snow accumulation and ablation dynamics, and differences among the range of current peatland restoration techniques have yet to be investigated. Further, much of the peat extraction and restoration techniques and associated research, including snow dynamics, stems from other parts of Canada, mainly Quebec (Rocheffort et al., 2003).

The main type of mechanical extraction used in Canada is the vacuum extraction method (Robert et al, 1999), which is also used in Manitoba (Offman, 2011). Vacuum extraction begins by installing drainage ditches and removing the surface vegetation from a natural peatland (Price et al., 2003) leaving behind the dark brown colour of the exposed catotelm; this decreases the surface roughness and increases the amount of incoming shortwave radiation incident reaching the surface, thus potentially altering snow accumulation patterns and the rate of snowmelt (Ketcheson et al., 2012). By removing the upper acrotelm layer (Petrone et al., 2003) the typical

diplotelmic structure is altered (Ingram, 1978); this affects the ability of the peat to regulate surface runoff, evaporative losses and water table fluctuations (Van Seters and Price., 2001) because the remnant catotelmic materials lack self-regulating characteristics (Ketcheson and Price., 2011).

The Canadian peatland restoration approach, also known as the moss layer transfer technique (MLTT) (LeBlanc et al., 2012), was created at the Sainte-Marguerite-Marie peatland in Quebec, Canada (Rocheffort et al, 2002). This is a 6-step approach including 1) field preparation, 2) diaspore collection, 3) diaspore introduction, 4) diaspore protection, 5) fertilization, and 6) rewetting/ditch blocking (Rocheffort et al, 2002). Field preparation requires site specific approaches with the intention of: 1) retaining as much water on site as needed, 2) evenly distributing water, 3) limiting duration of time with standing water at surface, and 4) avoiding flooding of large areas (Quinty and Rocheffort, 2003). These objectives can be achieved by filling drainage ditches, creating berms (including waffle patterned berms) and scraping the surface of the peat (Quinty and Rocheffort, 2003). Although these approaches have been successfully applied within restored peatlands in Quebec (McCarter and Price, 2013; Andersen et al, 2013), Manitoba has a different climate than Quebec and the peatlands within Manitoba have a distinct hydrogeomorphic settings separating them from peatlands found in Quebec. The hydrogeomorphic setting links the hydrology and geomorphology of a site (Sidle and Onda, 2004). For example in Quebec, more specifically at the Ste-Marguerite-Marie peatland located in the Lac-St-Jean region, has a mean annual precipitation of 820 mm with 25% falling as snow (LaRose et al., 1997). The site is found within the humid continental climatic region and is underlain by sedimentary rock (Ordovician limestone) covered with a thick layer of marine clay from glacial sediments (LaRose et al, 1997). Comparing this to the research at the Medika bog,

found within the sub-humid transitional low Boreal ecoclimatic region in South Eastern Manitoba that is characterized by short warm summers and long cold winters (Smith et al., 1998) where the mean annual precipitation is 570 mm with 20% falling as snow (Environment Canada, 2016) and the average evapotranspiration between 400 mm to less than 500 mm (Statistics Canada, 2017). The soils found within this depressional lowland are complex and poorly drained overlying loamy to clayey glaciolacustrine sediments (Smith et al., 1998). Due to the sub-humid climate, capturing and using the water released during the spring snowmelt is important due to limited water availability associated with the warm dry summers. Snow melt is a critically important input when calculating a complete water balance, but it is often omitted in peatland studies due to the difficulty/timing of the snow melt. Understanding how a complex surface topography, including microtopographic (swales and hummocks) and microtopographic features, (surface tiling) could better evenly distribute snow or increase retention of snowmelt water and promote infiltration (Ketcheson and Price, 2016) would be valuable in the restoration landscape design process.

During peatland restoration the largest challenge to overcome is usually the limited availability of water (Petrone et al., 2003). The distribution of water is highly influenced by the surface topography (Quinty and Rochefort, 2003) yet the effects of varying surface preparations and reprofiling in extracted peatlands remains poorly understood (Price et al., 1998). By increasing the surface roughness of extracted peatlands it could lead to enhanced moisture conditions of the peat surface (Eggelsman, 1988); for example, light ploughing of the surface peat to create depressions may hold back water generated in the spring snow melt period (Ferland and Rochefort, 1997). However, fewer studies have characterized the effects of snow distribution and ablation as a result of different surface re-profiling techniques. In reclaimed

sites, cleared of vegetation stands, surface topography influences the accumulation and ablation of the snow pack (Ketcheson and Price, 2016). The objective of having multiple surface topographies is to investigate how the microtopography influences the surface hydrology and snow accumulation on the surface. By increasing surface topography there is an increased retention of snow, causing a deeper snowpack (Ketcheson and Price, 2016). With a deeper snowpack there exists an increased thermal insulation leading to a decrease in soil frost (Dyke and Sladen, 2010), promoting the spring snowmelt to infiltrate into the soil rather than runoff.

Therefore, the objective of this study is to address the influence of peatland disturbance and subsequent restoration technique on snow dynamics, and to evaluate the importance of snowmelt on water availability during the springtime melt period and summer season water budgets. Specifically, this research aims to characterize snow dynamics associated with various peatland restoration surface treatments (flat, linear) and states of extraction (extracted, remnant), comparing the findings to natural (bog, fen) peatland ecosystems.

2.2 Study site

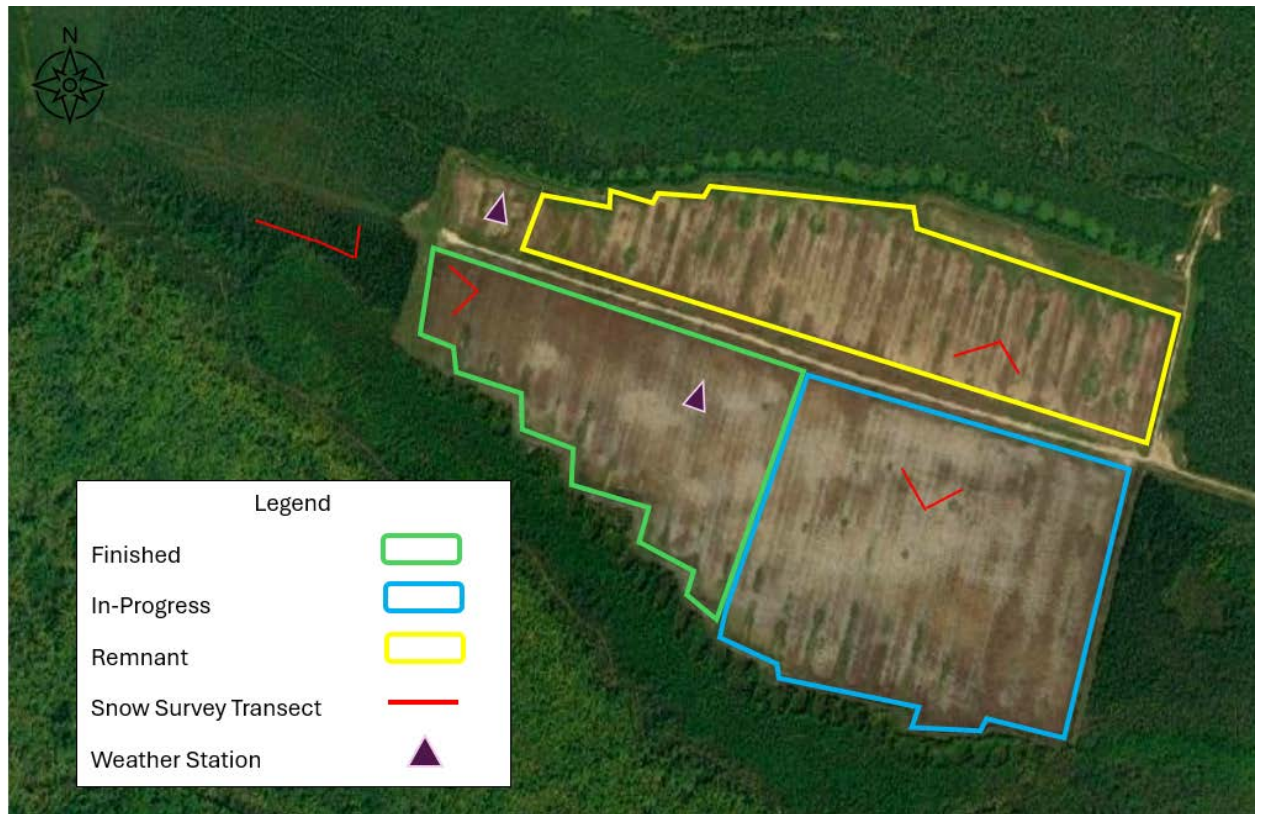


Figure 2-1 Study site map



A. Finished



B. Remnant



C. In Progress

Figures 2-2 A, B, and C are aerial images of the different surface topographies from above using a drone

The Moss Spur 2 peatland (49°58'51" N 96°09'53" W) is found ~90 km east of Winnipeg, Manitoba. The site is an 80-ha peat field (~ 1.5 km long, 300 – 800 m wide) with an access road separating the field into north (1/3) and south (2/3) sections. The field was last

extracted in 2018, and reprofiling began in late summer/early fall 2019 using a bulldozer. Reprofiling of the field could not be completed due to a series of rainfall events in September 2019 where ~ 204 mm was recorded at the Pinawa Canwarn (roughly 35 km from the site) (Environment Canada, 2019), leaving 3 different surface microtopographies; a “Remnant” which had no passes (i.e., untouched) (Figure 2-2, B), “In Progress” which had one bulldozer pass leaving linear mounds (~ 20-30 cm high and wide) from the sides of the plow roughly 4 m apart (Figure 2-2, C), and a “Finished” topography leaving a “smooth” surface (no mounds) (Figure 2-2, A). Restoration efforts were resumed in fall 2021 and completed by mid-October to a “Finished” topography (Figure 2-2, A).

Limited vegetation is present at the site other than cattails (*Typha latifolia*) and horsetail (*Equisetum spp.*) found in remnant drainage ditches, Canadian thistle (*Cirsium arvense*), sow-thistle (*Sonchus arvensis*), and bog birch (*Betula pumila*) found along the access road and some cotton grass (*Eriophorum brachyanterum*) found throughout the field. A “Natural” bog site, located at Moss Spur 2, was measured to compare and contrast the differences found between extracted and natural, as well as act as a reference. The site has typical bog species such as *Sphagnum* moss, tamarack (*Larix laricina*), black spruce (*Pinus mariana*), jack pine (*Pinus banksiana*), Labrador tea (*Rhododendron groenlandicum*), leather leaf (*Chamaedaphne calyculata*), bog rosemary (*Andromeda polifolia*), pitcher plants (*Sarracenia purpurea*), and bog cranberry (*Vaccinium oxycoccos*).

2.3 Methods

The landscape classification is based on location and re-surfacing technique. At each site snow surveys were conducted to evaluate their surface treatments on snow accumulation and melt patterns using pre-melt data. Snow surveys were conducted at Moss Spur 2 along

established research transects to gather snow depth, snow water equivalence (SWE), and snow pit data. Measurements in 2020 were limited to February 18th and March 19th due to COVID-19 and limited field access, in 2021 they were limited to March 3rd, 8th and 17th due to limited snow fall and an early melt period, in 2022 they were limited to March 23rd, 31st, April 4th, 5th and 8th due to equipment shortages and limited field access.

2.3.1 Snow Depth and Snow Water Equivalence

Snow depth measurements were taken using a metal metre-stick ruler every 10 paces (~ 8 meters) and snow density measurements every 20 paces (~ 16 metres). Snow density was measured by using a standard ESC-30 (Eastern snow conference) plastic snow tube and the sample was weighed using a Brecknell Electro Samson hanging mass scale to calculate snow water equivalence (SWE). SWE was calculated using the mass of the snow within the tube, the density of water (1 g cm^{-3}), the depth of the snow, and the area of the snow tube ($\pi r^2 = 33.18 \text{ cm}^2$).

2.3.2 Snow Pits

Snow pits were completed within each treatment using a ProSnow Kit II to measure snow density and the temperature in depth profile within the snowpack near the ablation lines. Snow density was determined by using the mass of a sample of snow divided by the known volume of the sampler to determine the ripening of the snowpack.

2.3.3 Meteorological

A weather station was installed in the south-western part of the Remnant section consisting of a rain gauge (Texas Instrument 525), temperature/relative humidity sensor (HCS3) and a net radiometer (Kipp and Zonen NRLite), all connected to a Campbell Scientific CR1000 data logger recording 15-minute averages/totals of 60-second measurements. Both temp/relative humidity and net radiation were taken at 1 m heights.

2.4 Results

2.4.1 Meteorological

The Canadian climate normals (1981-2010) for Beausejour (located ~ 30 km from the research site) shows an average snow depth at the start of March of ~ 14 cm (Government of Canada, 2010.). As restoration resumed on the site in October 2021, the weather stations were removed and never reinstalled. Missed data was retrieved from the Pinawa weather station (Climate ID: 503BIER) located ~29 km North East of the site at the Canadian Nuclear Laboratories (Canada, 2022). This is the closest local weather station available with daily averages.

Table 2.4.1-1: Comparing the 30-year climate normals temperature (°C) to the meteorological data collected at the weather station in the field. The bolded data has been retrieved from the Pinawa meteorological station (Climate ID: 503B1ER)

	<i>1981-2010 Normals</i>	<i>2020</i>	<i>2021</i>	<i>2022</i>
<i>January</i>	-16.9	-12.5	-10.8	-20
<i>February</i>	-12.8	-13.3	-18.9	-20.5
<i>March</i>	-5.7	-5.2	-0.8	-7.6
<i>April</i>	4.2	1.2	3.0	-0.2
<i>October</i>	5.0	0.9	12.0	
<i>November</i>	-4.8	-3.2	-1.9	
<i>December</i>	-14.1	-9.4	-13.6	

Air temperatures varied year to year relative to the 1981-2010 climate normals (Table 2.4.1-1). Overall, 2021 was the warmest of the three field seasons, experiencing above normal temperatures in most months with October being 7.0°C warmer than the climate normals, except for February which was 6.1°C colder than the climate normals. 2020 experienced temperatures generally closer to the climate normals except for January and December being notably warmer (4.4°C and 4.7°C, respectively) and October was 4.1°C colder than the climate normals. In contrast, 2022 was consistently colder than all years with February having the largest difference being 7.7°C colder than the climate normals.

2.4.2 Snow Depth

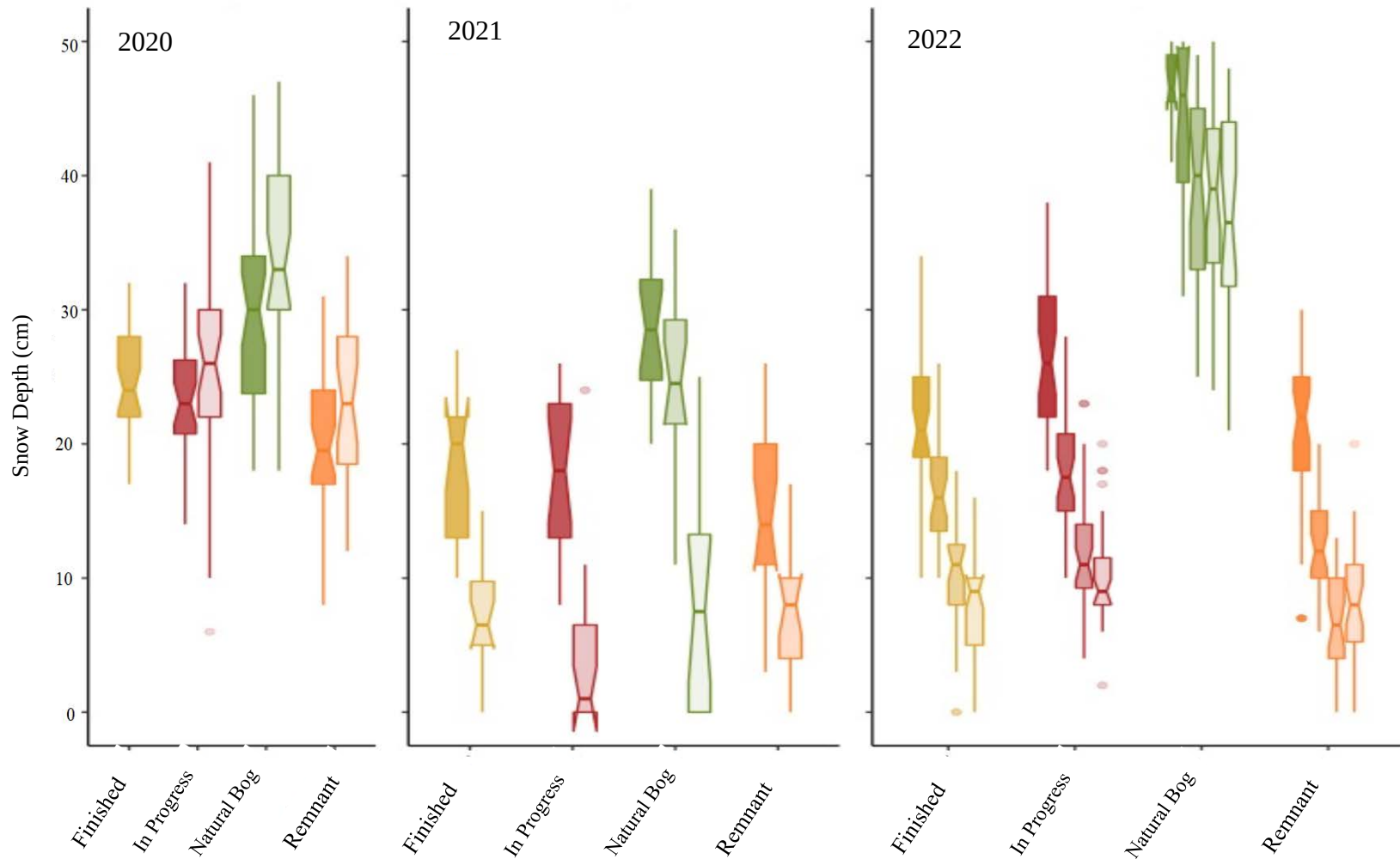


Figure 2-3: Snow Depth vs. Location box plot graph with measurements on February 18th and March 19th 2020 respectively, with measurements on March 3rd, 8th and 17th 2021 respectively, and with measurements on March 23rd, 31st and April 4th, 5th and 8th 2022 respectively. Note each of the colours represents each site (X-axis) and the different shading represents different sampling days. The natural bog is to be used as a reference, not in the comparisons between sites.

In 2020 there was a snow event that occurred between the two measurements, (Figure 2-3) showing an increase of the median snow depth by ~ 3 cm. Pre-melt measurements (February 18, 2020) show a variable snow cover was observed throughout the site, with the medians varying from ~ 20 to 30 cm deep with the deepest median snowpack observed at the in-progress sites (26 cm), while the shallowest snowpack was observed at the Remnant site (23 cm). In Progress and Remnant site indicate a greater variability in the snowpack depth compared to the Finished site.

A similar trend occurred in 2021 (Figure 2-3) where the re-surfaced sites (In Progress and Finished) exhibited a deeper peak median snowpack (March 3rd, 2021) ranging from 18 to 20 cm, with the deepest median snowpack was observed at the Finished site (20 cm), and the shallowest snowpack observed at the Remnant site (14 cm). Snow melt started prior to our arrival on March 3rd leaving the tops of some of the mounds found at the In Progress site bare. The site was practically snow free by March 17th, except for some snow buildup in the main drainage ditches and natural bog that surrounds the site.

In 2022 (Figure 2-3) the whole site was reprofiled to have a flat surface topography. The In Progress site exhibited the greatest median snow depth on March 23rd at 26 cm of all the resurfaced sites. As the melt period progressed the In Progress site maintained the greatest median snow depth. The Remnant site exhibits the lowest median snow depth on each day measured except on the first measurement on March 23rd where the Finished site snow depth was 21cm which was lower than the Remnant sites at 22 cm; not a significant difference. By the last field day, the In Progress site had the least variability within measurements and the greatest amount of snow present.

2.4.3 Snow Water Equivalence

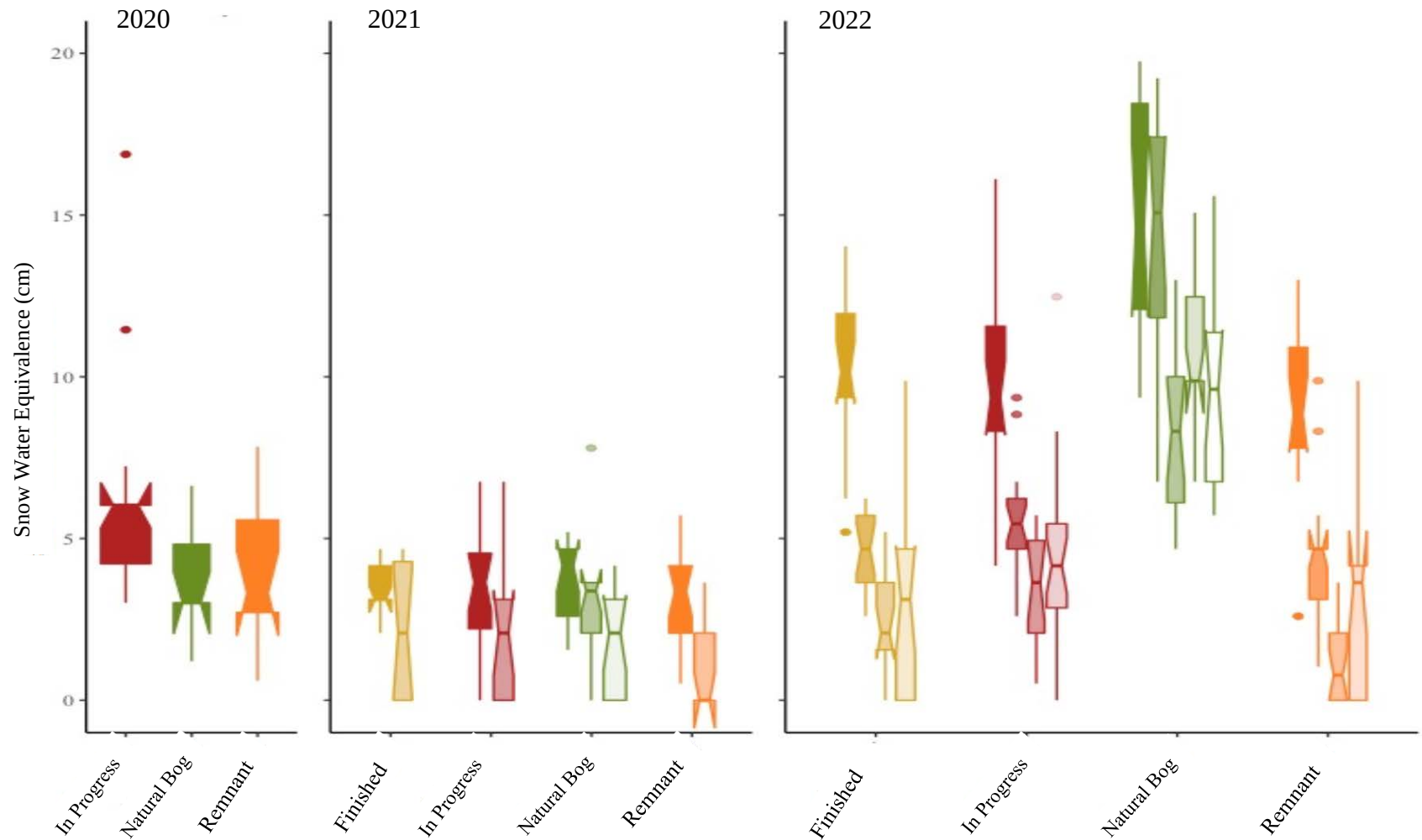


Figure 2-4: Snow Water Equivalence box plot graph for with measurements on March 19th, 2020, respectively, with measurements on March 3rd, 8th and 17th 2021 respectively, and with measurements on March 23rd, 31st and April 4th, 5th and 8th 2022 respectively. Note each of the colours represents each site (X-axis) and the different shading represents different sampling days. The natural bog is to be used as a reference, not in the comparisons between sites.

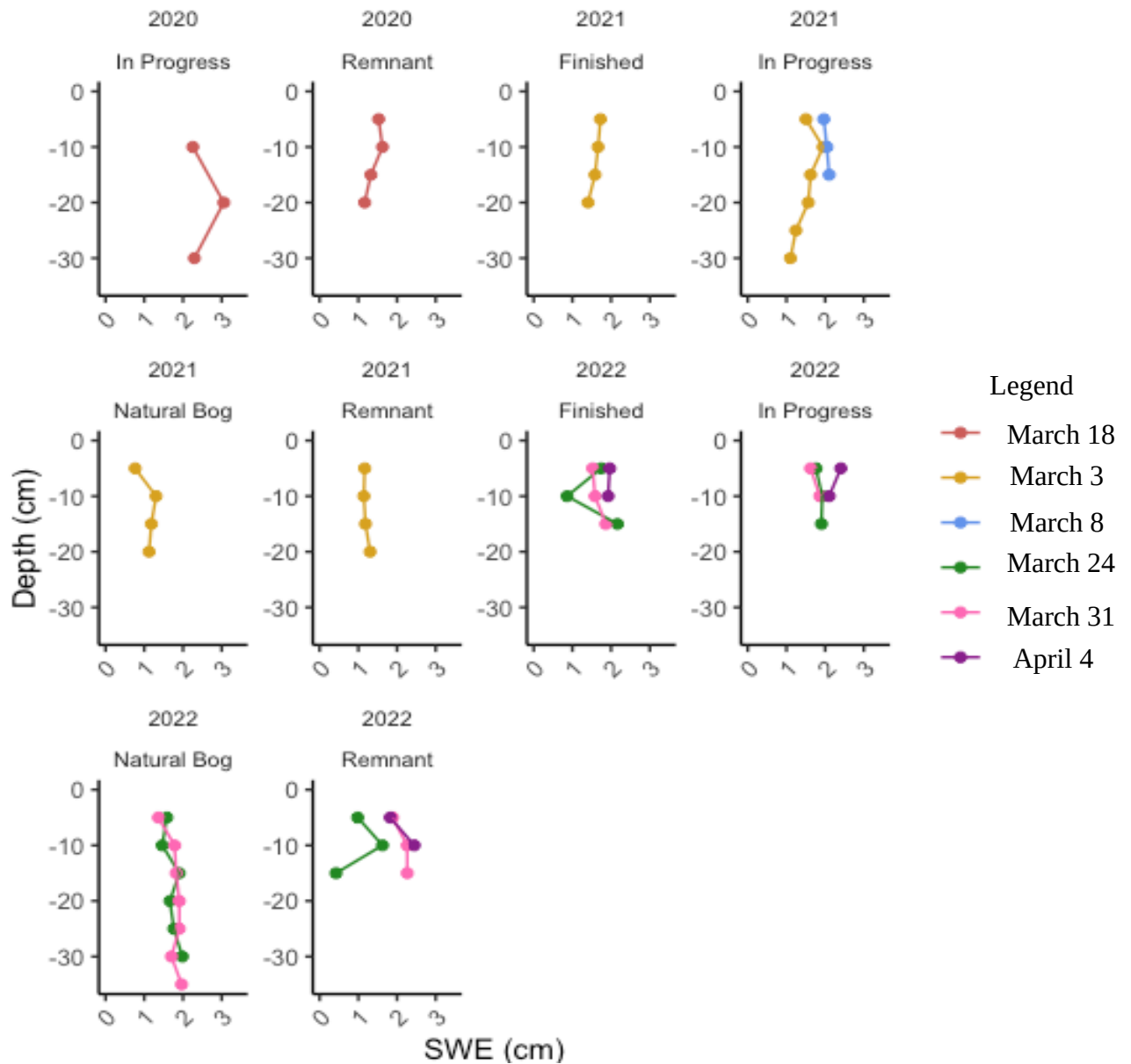
The 2020 snow water equivalent (SWE) trend (Figure 2-4) is opposite from snow depth (Figure 2-3) with the area of deepest median snowpack found in the Natural Bog (~20 cm) demonstrating the lowest median SWE (~3 cm), and the shallower median snowpack found in the In Progress (~22-25 cm) has the greatest non-outlier median SWE values (~6 cm). The Finished site could not be measured due to limited daylight during the field visit.

The 2021 trend (Figure 2-4) found the greatest SWE at the In Progress site where the median SWE was ~2 cm and ~4 cm respectively. The In Progress site also exhibited the greatest variability between measurements. The Finished site had the least variability, and the Remnant site had the lowest median SWE.

The 2022 SWE trend (Figure 2-4) is very similar to the trends found in snow depth (Figure 2-3) with the area with the deepest median snowpack found in the Natural Bog (~45 cm) demonstrated the deepest median SWE (~15 cm) and the area with the shallowest median snowpack found in the Remnant site (~5 cm) has the lowest median SWE (~1 cm).

2.4.4 Snow Pit Data

Table 2.4.4-1: Snow Pit Linear Graph; each colour represents a different field day.



In 2020 the highest SWE from the snow pits was ~3.0 cm located at the In Progress site at 20 cm. Typically, as the sample depth increases the SWE decreases, however, at the In Progress site, SWE is the same at the shallowest and deepest point, with the middle having the

greatest SWE value. The Remnant site exhibits lower SWE values compared to the In Progress site.

In 2021, the In Progress site had the highest SWE values with a gradual decrease with depth. The other three sites all had lower SWE with uniform values throughout the snow profile. The In Progress site was the only site to get two measurements in 2021 (March 3rd and 8th) as when we returned to the site the snow was mostly water due to the early melt.

In 2022, differences in the SWE values were less pronounced. the Natural Bog and In Progress sites maintained the most consistent SWE, compared to the Remnant and Finished sites which show a greater spatial variability, suggesting uneven SWE patterns.

In all three study years, the highest SWE values were measured at the In Progress site in 2020 and 2021. SWE profiles among the four sites were more similar in 2022. This could be attributed to more sampling days, rather than just one as found in 2020 and 2021.

2.5 Discussion

In 2020, the Finished site exhibited the highest median snow depth (24 cm) and least variability within the snowpack whereas the In Progress site has exhibited an increased variability within the snowpack and a decreased median snow depth (23 cm) (Figure 2-3). This could be attributed to the more uniform smooth surface cover of the Finished site compared to the linear mounds present on the In Progress site. Both re-surfaced sites (Finished and In Progress) exhibited higher median snow depths (24 cm and 23 cm respectively) than the unrestored (Remnant) site with no restoration efforts in place (median snow depth of 19.5 cm) suggesting that resurfacing the peat surface has a positive impact on snow accumulation, thus resulting in a deeper snowpack. Some ripening of the snowpack was apparent in the middle of the sample, leading to denser snow (Figure 2-4). This is caused by wind compaction of snow

particles paired with a warm period causing melt during the day and re-freeze at night (Ketcheson and Price, 2016).

In 2021, March temperatures were above the climate normal (Table 2.4.1-1); pairing this with the below average snowfall this area experienced a quick melt period that was difficult to capture. We found the Finished site exhibited the deepest median snow depth (20 cm) and the least variable SWE, whereas the Remnant site exhibited the shallowest median snow depth (14cm) with the lowest median SWE (2 cm) (Figure 2-4) and the In Progress sites median snow depth was similar to the Finished sites at 18 cm but exhibited the greatest median SWE (4 cm). This supports the findings from the previous year that any re-surfacing of the peat surface has had a positive impact on snow accumulation.

In 2022, all sites were re-profiled to ‘Finished’, removing any vegetative regrowth, leaving snow distribution to be influenced primarily by wind speeds and direction (Ketcheson and Price, 2016). The Remnant site’s median snow depth (on March 21st) varied from 21 cm (Finished site) to 26 cm (In Progress site) with the greatest variability found in the In Progress site. As the melt period progressed the Remnant sites median snowpack was the shallowest between all sites and the In Progress sites median snowpack remained the deepest. This could be attributed to the wind blowing from the remnant site, over the road and depositing to the In Progress site. The SWE trend mimicked the snow depth and the areas with the deepest snowpack exhibited the greatest median SWE. By April 8th the site was mostly standing water covered by a thin ice layer with patches of bare peat.

With a lack of surface vegetation, surface topography becomes the dominant control on snow hydrology affecting snow accumulation patterns (Ketcheson and Price, 2016). The Moss Spur 2, due to its infancy stage of restoration, vegetation cover has minimal impact on snow

distribution and thus differences in the snowpack could be driven solely by enhanced wind erosion and differences in surface topography (Ketcheson and Price., 2016). In open environments spatial variations in SWE data are a result of varying topography, wind deposition and vegetation stands, if present (Pomeroy et al., 1998). Due to this, open environments typically have a higher SWE compared to adjacent forested sites (Buttle et al., 2005) The SWE is how much water will be available during the melt period; this source of water represents the initial re-wetting of the site and any that gets retained replenishes storage deficits and helps to partially offset the challenges of limited water availability associated with being in a sub-humid climate. The SWE was more variable on the resurfaced sites than the natural (Figure 2-4) and the greatest median snowpacks were found on the resurfaced sites (Figure 2-3) suggesting any resurfacing of the peat surface aids in retaining more of the snowpack contributing to less ground frost, increasing retention of melt water, and promoting infiltration to recharge underlain aquifer limiting the effects of low water availability during the long warm summers.

2.6 Conclusion

Due to the limited field assess from the Covid-19 pandemic further sampling (e.g., repeated snow depth/SWE surveys) during the 2020 melt period was not possible. In a sub-humid environment, the spring freshnet contributes a substantial amount of water important in capturing to fully understand the annual water budget. Since water availability has been found to be the largest challenge in the restoration of peatlands, including and retaining the water released during the melt period will aid in the development of effective water management strategies. Surface variation paired with other factors (present surface vegetation, wind erosion and deposition) contributes to how much snow is retained directly affecting the amount of water available during the spring melt period (Landry and Rochefort, 2012).

With the limited number of observations, it is difficult to evaluate substantial differences, but the results indicated that the Finished and In Progress sites generally exhibited a greater median snow depth and SWE compared to the Remnant site. Suggesting that resurfacing of the peat surface has a positive influence on the snow accumulation and SWE. Increased snow retention would help to restore the extracted peatland's water storage and help offset water limitations from the previous year's deficits. These findings highlight the importance of restoring hydraulic conditions following peat harvesting to increase the potential success in restoring the site to a carbon accumulating system. Continued monitoring of the snowpack conditions, particularly comparisons between the Finished and In-Progress sites surface micro-topographies would be required to determine if one restoration approach is more efficient than the other at improving remnant hydrologic conditions.

3 Extracted bog water balance to restore site to a fen with the use of different surface microtopographies

3.1 Introduction

Canadian peatlands cover ~12% of the country's land area or 1.1 million km² (Tarnocai, 2009), which is the largest area of peatlands of any country across the globe (Tarnocai et al., 2011). Canadian peatlands represent a significant portion of the global terrestrial carbon storage containing ~147 Gt of organic carbon, which is equivalent to 56% of all stored organic carbon in Canadian soils (Tarnocai, 2009). In North America, two broad types of peatlands are recognized based on vegetation, water regime and chemistry (Bérubé and Rochefort, 2018): bogs and fens. Bogs, or ombrogenous peatlands are solely precipitation fed, resulting in an acidic, low mineral nutrient environment often dominated by *Sphagnum* mosses (Rochefort et al., 2012; Zoltai and Vitt, 1995). Fens, or minerogenous peatlands have a more complex hydrology encompassing both precipitation and drainage from the surrounding watershed, which can result in an alkaline, mineral rich environment with a diverse plant community consisting of brown mosses, sedges and fen bryophyte species (Rochefort et al., 2012; Sampath et al., 2016; Zoltai and Vitt, 1995). North American peatlands are an important source of peat moss as a growing medium in the horticulture industry (CSPMA, 2014; Rochefort et al., 2012). Canada has been classified as the largest producer and exporter of horticulture peat in the world (CSPMA, 2014), with the majority extracted from bog peatlands (Swystun et al., 2013). Peatlands rely on many processes (water table position, evapotranspiration rate, runoff, soil moisture content, etc.) to form a water-logged environment, slowing biomass decomposition, for peat to accumulate and sequester carbon (Waddington et al., 2008, 2001).

In preparation for peat extraction, the drainage and removal of the upper layer (acrotelm) alters the hydrological properties of the remnant bog severely disturbing the natural peatland ecosystem (Price, 1997; Rochefort et al., 2016) from a carbon sink to a source of carbon (Waddington et al., 2001) constraining the re-establishment of *Sphagnum* mosses (Campeau and Rochefort, 1996). Restoration projects in North America aim to return the extracted bog to a carbon accumulating ecosystem (Price et al., 1998; Rochefort et al., 2003). During the 1990s in North America, the Peatland Ecology Research Group (PERG) developed the moss layer transfer technique (MLTT) to restore ombrotrophic conditions suitable for bog species (Rochefort et al., 2016, 2003) and it has been successful over the last 20 years (González and Rochefort, 2014). But vacuum extraction techniques can extract the remnant bog to the underlying sedge-peat, exposing the deeper minerotrophic fen peat that is richer in minerals with a higher pH, creating conditions which may be more ideal for fen restoration over bog (Graf and Rochefort, 2008; Rochefort et al., 2016).

Fen restoration is likely more difficult due to their complex, under-studied hydrological system that encompasses the surrounding watershed (Chimner et al., 2017; Sampath et al., 2016). Several North American studies have examined factors that influence fen restoration, focusing on specific fen species plant assemblages to restore a ‘natural’ functioning fen ecosystem (Bérubé and Rochefort, 2018; Borkenhagen and Cooper, 2016; Chimner et al., 2017; Graf and Rochefort, 2010, 2008; Malloy and Price, 2014; PERG, 2017). But most of these studies have taken place in the maritime provinces (Quebec) where there exists a surplus of annual precipitation and due to the new endeavour of fen restoration in North America, it makes it difficult to relate findings to the continental climate found in the prairie provinces (Manitoba, Saskatchewan and Alberta in Canada) since they are drought prone.

Bérubé and Rochefort (2018) and Chimner, et al., (2017) emphasized the importance in restoring the eco-hydrological connectivity to the surrounding watershed for a successful fen restoration, since both the hydrology and water chemistry influence the vegetation composition. This can be achieved by creating a transitional zone from the natural undisturbed peatland to the extracted peatland through infilling the drainage network, and eliminating any blockages that would affect the connectivity to the watershed (Chimner et al., 2017; PERG, 2017). Water table depth has been found to be one of the main factors influencing fen species establishment (Borkenhagen and Cooper, 2016), especially when looking at fen bryophyte species that require a water table 0 to -10 cm below the surface with a soil water potential of -4.3 to -8.6 units (Graf and Rochefort, 2010). Graf and Rochefort (2010) state the increasing importance of including and focusing on the success of fen bryophytes species during fen restoration projects, these species are important in the overall function and composition of fen systems.

The goal of this research is to evaluate the remnant hydrological processes found in the cutover area found at the Moss Spur 2 peatland in hopes to restore the site to a fen. Due to a series of rain events in the fall of 2019, restorative measures were brought to a halt leaving three surface topographies: 1) Finished, which had two passes with the bulldozer and ecotone creation, 2) In Progress, which has one pass with the bulldozer creating ~20-30 cm linear mounds and; 3) Remnant, which had no restoration. This gave an unplanned opportunity to study the before, during and after-effects of restoration. The specific objective is to calculate the water budget of the extracted peatland before, during and after restorative measures to evaluate and further understand the effects of site reconfiguration on the remnant hydrologic processes. This understanding of the remnant hydrologic processes and patterns will provide essential information for the next steps in the restoration plans, especially for plant reintroduction.

3.2 Study site

The Moss Spur 2 peatland (49°58'N, 96°1'W) is located approximately 90 km east of Winnipeg, Manitoba, Canada (Figure 3.2.1). Moss Spur 2 is found in the Boreal Shield ecozone, within the Whitemouth ecodistrict where the climate is characterized by short, warm summers and long cold winters (Smith et al., 1998). The closest weather station with historic normals (1981-2010) is found in Beausejour, with a mean annual temperature of 2.8 °C, and January and July temperatures of -16.3 °C and 19.3 °C respectively. Precipitation totals of 570 mm with ~20% falling as snow (Environment Canada, 2021). Potential evaporation is between 400 mm and 500 mm (Statistics Canada, 2017). Vegetation was largely absent for the site, save for some cotton grass (*Eriophorum spp.*).

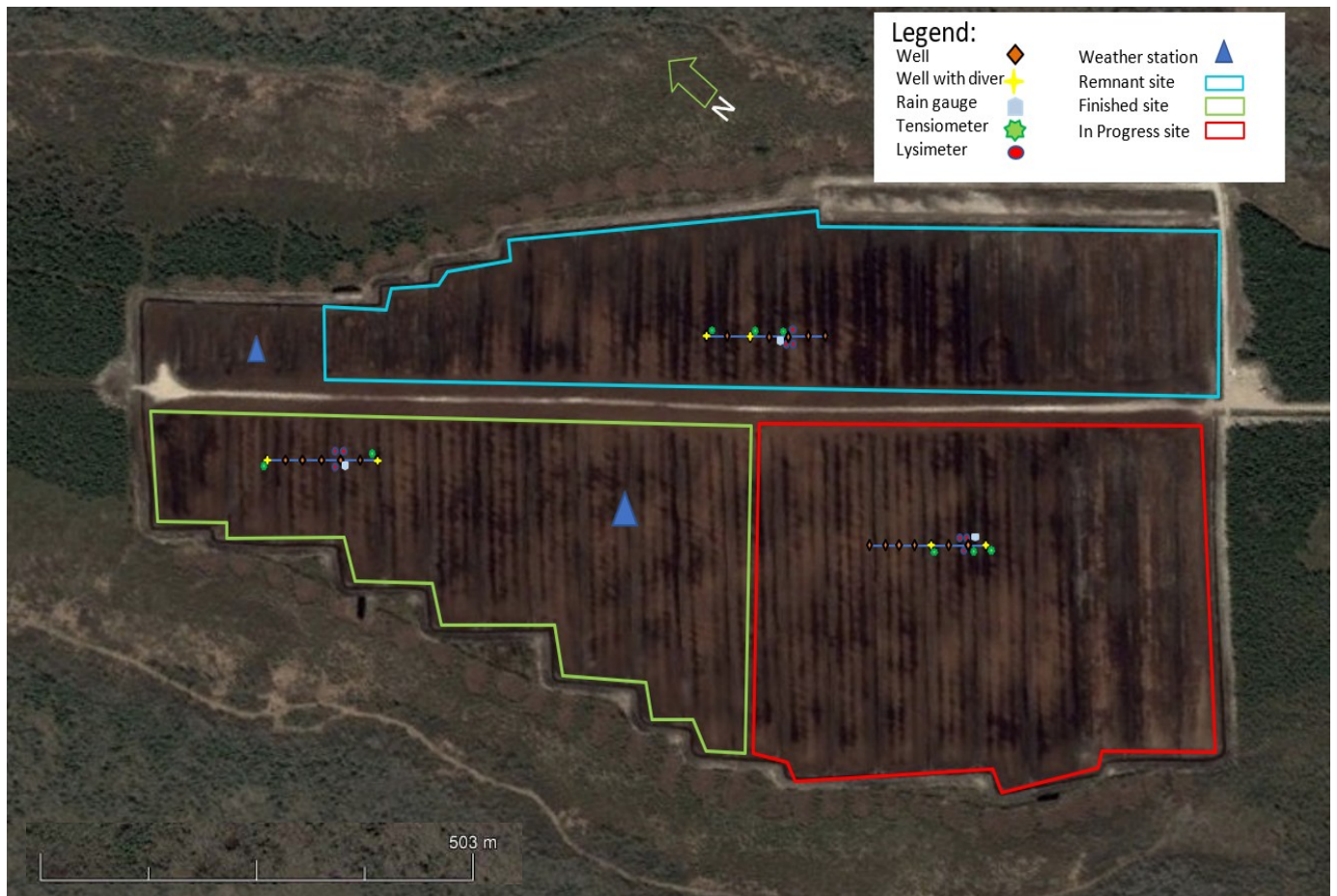


Figure 3-1 Study site map.

The site is an 80-ha peat field (~ 1.5 km long, 300 – 800 m wide) last extracted in 2018 with an access road separating the field into north (1/3) and south (2/3) sections. Pre-restoration data was collected summer 2019, and restoration efforts began in September 2019 which included reprofiling of the fields with a bulldozer. Unfortunately, due to a series of rainfall events (~ 243 mm fell in September 2019) (Environment Canada, 2019) restoration efforts were stopped (due to equipment shortages and COVID-19); they were not resumed until Fall 2021. This left three different surface microtopographies; 1) Remnant, which had no reprofiling and all drainage ditches were intact; 2) In Progress, which had one bulldozer pass to begin leveling the fields, leaving long linear mounds (~20-30 cm high and wide, 100s of meters long) from the sides of the bulldozer plow with parameter drainage ditch still open; and 3) Finished, leaving a “smooth” surface with no drainage ditches present, giving us the unique opportunity to study the before, during and after-effects of restoration. Reprofiling was Finished after this study was completed in October 2021.

3.3 Methods

3.3.1 Meteorological

A weather station was erected in the southern part in Finished section at Moss Spur 2 (Figure 3.2.1), collecting data (with a few interruptions) from June 2019 to October 2021. The weather station consisted of a rain gauge (Texas Instrument 525), temperature/ relative humidity sensor (HCS3), a net radiometer (Kipp and Zonen NRLite), and nine moisture content probes installed at different depths in the peat substrate (CS650 and CS655) all connected to a Campbell Scientific CR1000 data logger recording 15-minute averages/totals of 60 second measurements.

Another data logger was erected in the north-western part of the remnant section of Moss Spur 2 (see figure 3.2.1) collecting data from June 2019 to October 2021 with some interruptions. The data logger consisted of 5 Stevens moisture content probes installed at

different depths in the peat substrate connected to a Campbell Scientific CR200 data logger, recording 15-minute totals.

3.3.2 *Ground Water Flux, Storage Change*

Polyvinylchloride (PVC, 2.54 cm diameter) wells were installed to the underlying mineral layer to monitor the water table along ~ 75 m transects running east to west at each site (Remnant, In Progress and Finished) (Figure 3.2.1). Each transect consisted of 8 wells, installed ~ 10 m apart, with slotted intakes up the length of it. The wells were installed into pre-augured holes at various, site-determined depths based on remnant peat depths (Remnant: 0.5 – 0.9 m, In Progress: 0.5 – 1 m, Finished: 0.9 – 1.4 m). The wells were measured manually every other day throughout the summer (2020 and 2021). To calculate the change of storage in the saturated zone we used the following equation;

$$\Delta WT = \frac{\text{Input of Water}}{Sy}$$

Equation 3-1: calculating the change in storage of the saturated zone

Where the input of water is the change in the water table from measurement to measurement, calculating for the change in water table position. Hawes (2018) calculated the specific yield (Sy) using a pressure transducer in a well to measure the rapid water table rise following a known precipitation input recorded at the weather station. Using 13 rain/water table rise events Hawes (2018) determined the Sy of the deep peat soil to be 0.06 for non-flooded conditions. As Hawes study site is close to this study site, and a part of the same peatland complex (Bannatyne, 1980) we assumed the same Sy.

When the South weather station was installed two pits with moisture probes were installed to measure the change in moisture within the remnant peat profile throughout the field season every 15 minutes. In one pit, probes were installed at 5 cm, 10 cm, 15 cm, 20 cm and 25

cm depths. In the other pit, probes were installed at 5 cm, 10 cm, 15 cm and 20 cm depths (only 4 probes were present).

To solve for the change in storage for the peat profile, the moisture content on Day 1 (May 28th 2020 and 2021) was subtracted from the moisture content on Day done (August 19th, 2020 and 2021) for each probe. These values were then multiplied by the thickness the probe is centered in to give the change in moisture content from day 1 until day done for each field season.

3.3.3 Evapotranspiration

The Priestley-Taylor method (1972) was used to calculate evapotranspiration from each site using manual measurements (weighting lysimeter buckets) and potential evapotranspiration from the weather station data. It is important to note this method does not take into account surface roughness, stomatal resistance or other vegetative parameters, and these measurements are replaced using manual measurements of evaporation through lysimeters.

$$E_t = \alpha \left(\frac{s}{s + \gamma} \right) \left(\frac{Q^* - Q_g}{L_v \rho} \right)$$

Equation 3-2: Priestley-Taylor Method (PT method)

Where E_t as the potential evapotranspiration rate (mm d^{-1}), s as the slope of the saturation vapour pressure vs. temperature curve ($\text{Pa } ^\circ\text{C}^{-1}$), γ as the psychrometric constant ($0.0662 \text{ KPa } ^\circ\text{C}^{-1}$ at 20°C), Q^* as the net radiation flux (J d^{-1}), Q_g as the ground heat flux (J d^{-1}), L_v as the latent heat of vaporization (J kg^{-1}) and P as the density of water (kg m^{-3}). For our calculations Q_g was assumed to be 10% of Q^* (Hawes, M, 2018) Q^* (J d^{-1}), Q_g (J d^{-1}) and the air temperature (T_a $^\circ\text{C}$) were obtained from the meteorological station in addition to s , γ , L_v and ρ to calculate E_t . α represents the calibration coefficient and is the slope of the equilibrium evaporation line ($\alpha = 1$) which is plotted against the actual evaporation (ET_a) to correct the model's evaporation rate

(Priestley and Taylor, 1972). ET_a was measured using 24 lysimeters (open top 19-liter buckets filled with peat from surrounding area with holes drilled in the bottom to mimic surrounding peat) installed at 8 sites in triplicate: Remnant, Finished, south weather station, north station and natural bog except for the In Progress site has 3 more lysimeters than the other sites due to the hill/hollow topography left from the bulldozer pass. To encompass the variation of both the hills and the hollows 3 lysimeters (A, B, C) were installed in the hollow and the other 3 lysimeters (D, E, F) were installed into the hill. For our calculations we estimated the hills represented 10% of the total surface area found at the in-progress site. Buckets were weighed every other day from May 20th to August 19th 2020, and May 5th to August 20th 2021 using a Brecknell electro Samson 25 kg (+/- 10g) hanging mass scale. To monitor the water supply in the lysimeter and compare it to the surrounding area, the volumetric soil moisture (VMC) was monitored using a Hydrosense II moisture probe to ensure the lysimeter conditions were similar to the surrounding conditions. Water was added or removed accordingly to ensure the lysimeters were representative of the surrounding area.

3.3.4 *Soil Water Tension*

Tensiometers were installed in 2020 in the Remnant, In Progress and Finished sites, and again in 2021 in the Remnant, In Progress, Finished, south weather station and north data logger sites. The tensiometers were installed 10 cm below the peat surface, measured every other day from May 29th to August 19th 2020, and May 28th to August 18th 2021 using a Soil Measurement Systems Tensimeter. A known volume of water was added at the top of tensiometers when the water level fell below the calibrated clear plastic top.

3.4 Results

3.4.1 Meteorological

Table 3.4.1-1: 1981-2010 temperature normals (°C) compared to monthly temperature averages measured in the field. The bolded data has been retrieved from the Pinawa meteorological station (Climate ID: 503B1ER)

	1981-2010 Normals	2019	(+/-) Historical Difference (°C)	2020	(+/-) Historical Difference (°C)	2021	(+/-) Historical Difference (°C)
May	11.4	16.1	4.7	7.1	-4.3	10.4	-1
June	16.7	17.5	0.8	17.3	0.6	18.7	2
July	19.3	19.1	-0.2	20.0	0.7	21.0	1.7
August	18.5	17.2	-1.3	18.3	-0.2	18.2	-0.3

The average temperatures recorded at the weather station set up in the Finished section of Moss Spur 2 were similar between the three seasons (2019, 2020, 2021) except for the month of May. Comparing this to the historical normals data from the Beausejour meteorological station (climate ID: 5030160) there are a couple of outliers. May 2019 was 4.7°C warmer than the climate normals and May 2020 was 4.3°C colder than the climate normals. Save from those two outliers the temperature data was quite similar (within a degree) for most months.

Table 3.4.1-2: 1981 – 2010 Precipitation normals (mm) compared to cumulative monthly precipitation(mm) measured in the field, retrieved data is bolded.

	1981- 2010 Normals	2019	(+/-) Historical (mm)	2020	(+/-) Historical (mm)	2021	(+/-) Historical (mm)
May	58.1	30	-28.1	22.5	-35.6	54.7	-3.4
June	87.5	44.6	-42.9	132.8	45.3	38.4	-49.1
July	87.1	162.5	75.4	74.5	-12.6	32.4	-54.7
August	76.3	53.2 + 24.4	1.3	50.4	-25.9	79.1	2.8
September	65.1	74.3 + 144.2	153.4	1 + 20.3	-43.8	18.5	-46.6
October	46.0	64.6	18.6	1.4 + 20.5	-24.1	40 + 20.6	14.6
Totals	420.1	399.2 + 198.6	177.7	282.6 + 40.8	-96.7	263.1 + 20.6	-136.4

Missed rain data for August and September 2019, September and October 2020 and October 2021 was retrieved from the Pinawa Canwarn CNN (climate ID: 5032161) meteorological station. Missed data was only retrieved for entire missed rain days when the weather station onsite was down. Compared to the historic precipitation data (Figure 3-1), the 2019 field season had an above-average precipitation (+178 mm). This is mostly attributed to the series of rain events that accumulated ~243 mm of rain occurring at the end of August and throughout September 2019. The 2020 and 2021 field seasons had below-average precipitation (-97 mm and -136 mm respectively).

Although our field season was only from May – August, it is important to note the precipitation in the fall as this attributes to the recharge from the summer losses.

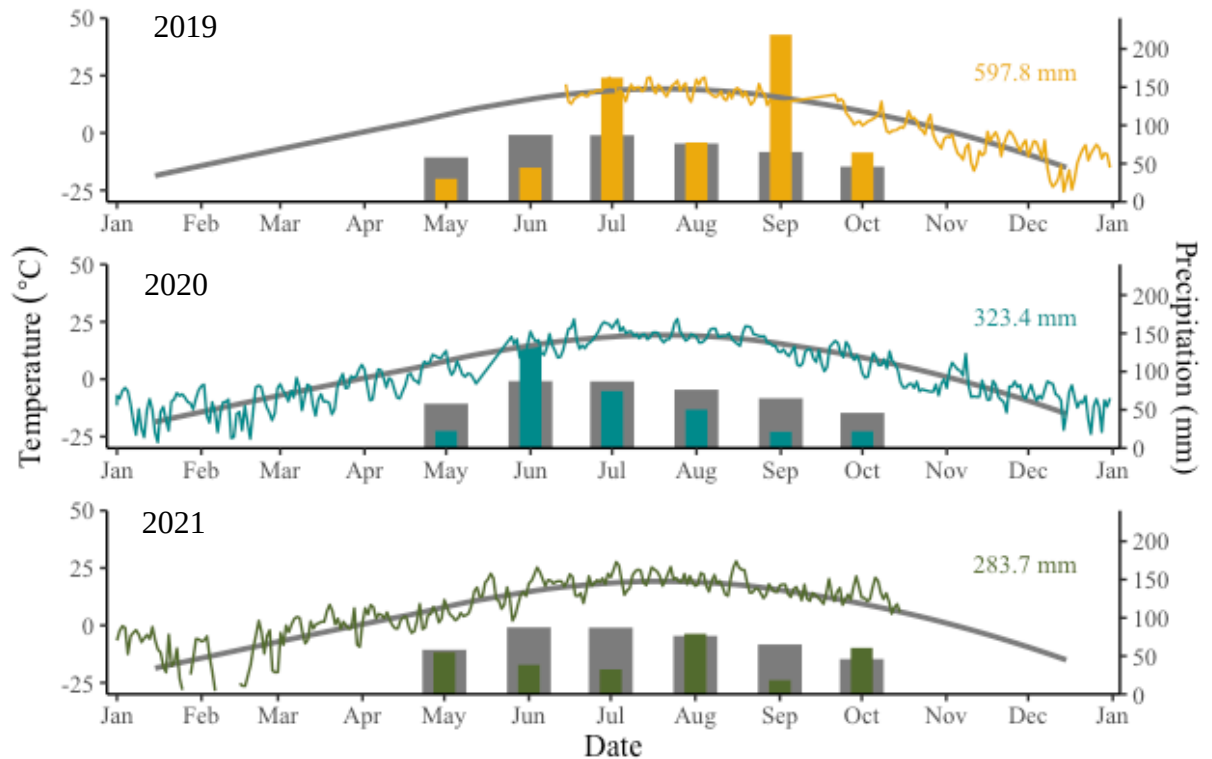


Figure 3-2: 2019-2021 Moss Spur meteorological data compared to 30-year climate normals from the Beausejour weather station. The grey represents the climate normals.

3.4.2 Evaporation

Evaporation data collected from the lysimeters had alpha values ranging between 0.3 (In Progress, Lysimeter E) to 1.42 (Finished, Lysimeter C) in 2020 and 0.33 (Mulch, Lysimeter C) to 1.76 (In Progress, Lysimeter C) in 2021 (Table 3.4.2-1). The alpha values were determined using the slope of best fit from each lysimeter, they were then averaged for the total evaporative losses. The slopes for each lysimeter in 2020 have the least variation at the north weather station (0.75-0.78) and most variation at the In Progress hills (0.51-0.76), and in 2021 the Finished slope has the least variation (0.88-0.89), and the north weather station has the most variation (0.61-0.92) (Table 3.4.2-1). In Figure 3.4.2 each of the coloured points represent a manual measurement at a specific lysimeter (each colour represents a single lysimeter), the model evaporation refers to the equilibrium evaporation, and the actual evaporation refers to the manual

measurements done using the lysimeters. The dashed line represents a 1:1 relationship. The average daily ET in 2020 varied from 3 mm/day (North Weather Station) to 6 mm/day (Remnant) (Table 3.4.2-2), and in 2021 it varied from 1 mm/day (mulch) to 4 mm/day (South Weather Station and In Progress) (Table 3.4.2-3). The total mm (Table 3.4.2-2 and Table 3.4.2-3) is areally weighted to better reflect the actual total evaporation for the site from May to September. Throughout the 2020 field season the site lost 353 mm, and in 2021 the site lost 303 mm (Table 3.5.2-2 and Table 3.5.2-3).

Table 3.4.2-1: Alpha and R² values for each lysimeter at each site

Site	Lysimeter	2020		2021	
		Alpha	R ²	Alpha	R ²
Finished	A	1.46	0.86	1.40	0.88
	B	1.62	0.81	1.51	0.89
	C	1.74	0.85	1.42	0.89
In Progress	A	1.31	0.92	1.51	0.80
	B	1.28	0.84	1.40	0.81
	C	1.25	0.90	1.76	0.88
	D	0.42	0.67	0.49	0.57
	E	0.30	0.51	0.52	0.63
	F	0.42	0.76	0.46	0.63
Remnant	A	1.66	0.94	1.23	0.83
	B	1.63	0.89	1.11	0.82
	C	1.43	0.89	1.01	0.72
North Weather Station	A	0.95	0.75	1.04	0.61
	B	1.10	0.77	1.13	0.87
	C	0.93	0.78	0.80	0.92
South Weather Station	A	1.35	0.90	1.24	0.69
	B	1.30	0.92	1.11	0.79
	C	1.51	0.88	1.19	0.78
Mulch	A	N/A	N/A	0.46	0.95
	B	N/A	N/A	0.52	0.92
	C	N/A	N/A	0.33	0.93

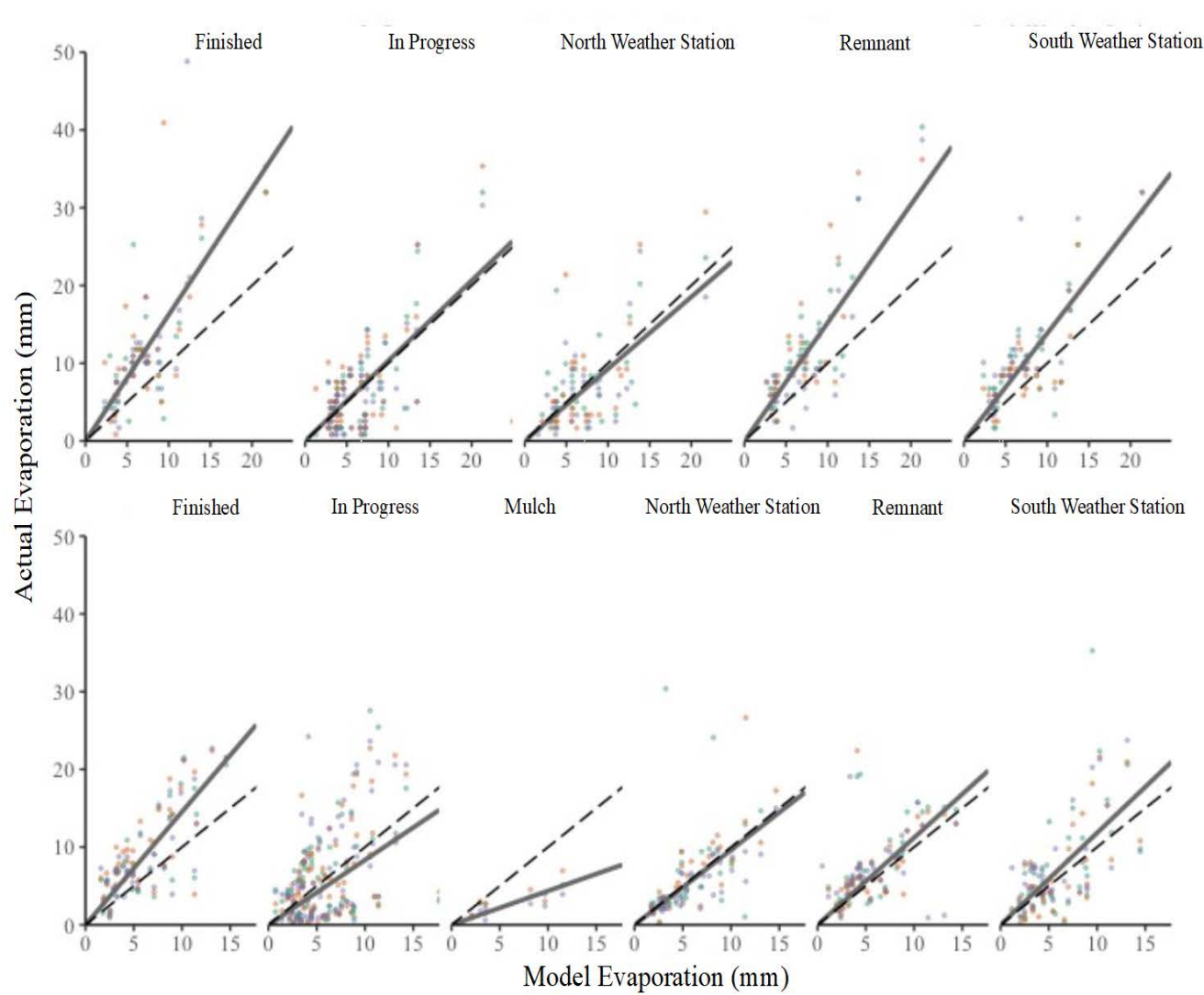


Figure 3-3: Alpha values determined by using lysimeter at each site. The alpha values were inserted into the P-T equation to determine total evaporation per site. The dashed line is $\alpha = 1$.

Table 3.4.2-2: 2020 Average alpha and ET per site aerially weighted. Total represents the areally weighted total mm of ET for the field season.

2020

<i>Site</i>	Finished	In Progress	Remnant	North Weather Station	South Weather Station
<i>Average Alpha</i>	1.60	1.32	1.57	0.99	1.39
<i>Average (mm/d)</i>	5	5	6	3	4
<i>Total (mm)</i>	109	96	116	5	27

Table 3.4.2-3: 2021 Average alpha and ET per site aerially weighted. Total represents the areally weighted total mm of ET for the field season.

2021

<i>Site</i>	Finished	In Progress	Remnant	North Weather Station	South Weather Station	Mulch
<i>Average Alpha</i>	1.44	1.61	1.12	0.99	1.18	0.44
<i>Average (mm/d)</i>	3	4	3	2	4	1
<i>Total (mm)</i>	83	110	77	5	27	1

3.4.3 Soil Water Tension

The average soil water tension varied depending on the stage of restoration for both study years. In 2020, the Finished section of the field had the highest soil water tension (-58.2 cm), and the In Progress section had the lowest soil water tension (-84.2 cm) (Table 3.4.3-1). In 2021, a similar trend showed the Finished section with a higher soil water tension (-92.2 cm) compared to the In Progress section (-120.6 cm), but the mulch section has the highest soil water tension at -36.3 cm (Table 3.4.3-1). It is important to note that the mulch section was created on August 6th,

2021, only consists of two weeks' worth of measurements (7 days in total) and it rained 56.5 mm during the measurements.

Table 3.4.3-1: Average soil water Tension (cm) grouped by site from May 29th 2020 – August 19th 2020 to May 28th 2021 – August 20th 2021.

<i>Site</i>	<i>Average Soil Water Tension (2020)</i>	<i>Average Soil Water Tension (2021)</i>
<i>Remnant</i>	-65.7	-85.6
<i>In Progress</i>	-84.2	-120.6
<i>Finished</i>	-58.2	-92.2
<i>South Weather Station</i>	N/A	-117.4
<i>North Weather Station</i>	N/A	-125.0
<i>Mulch</i>	N/A	-36.3

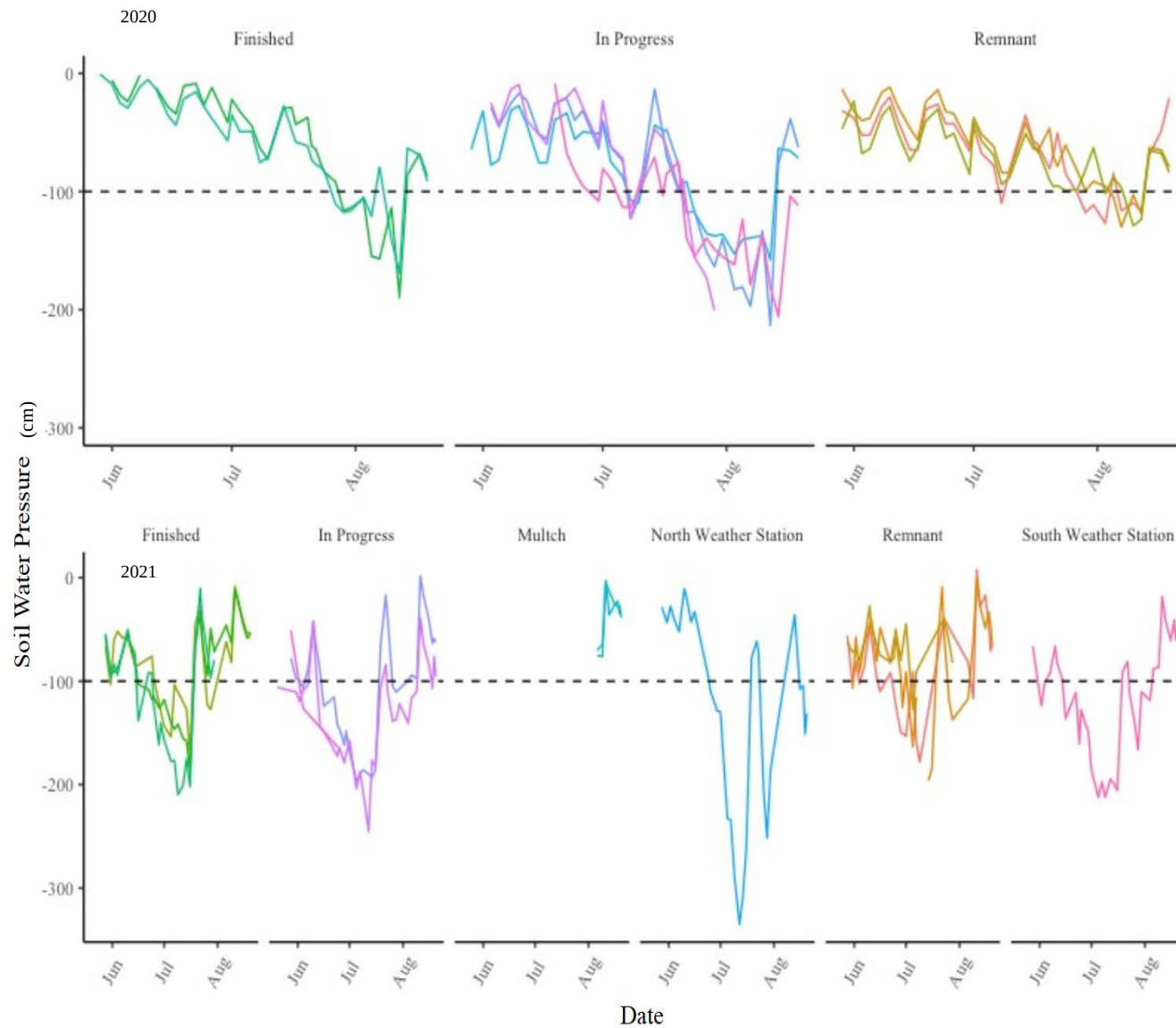


Figure 3-4: Soil water pressure (cm) measured using tensiometers installed -10 cm into the peat profile in the 2020 and 2021 field season, respectively. Each color represents a single tensiometer.

3.4.4 Water Table (Saturated Zone)

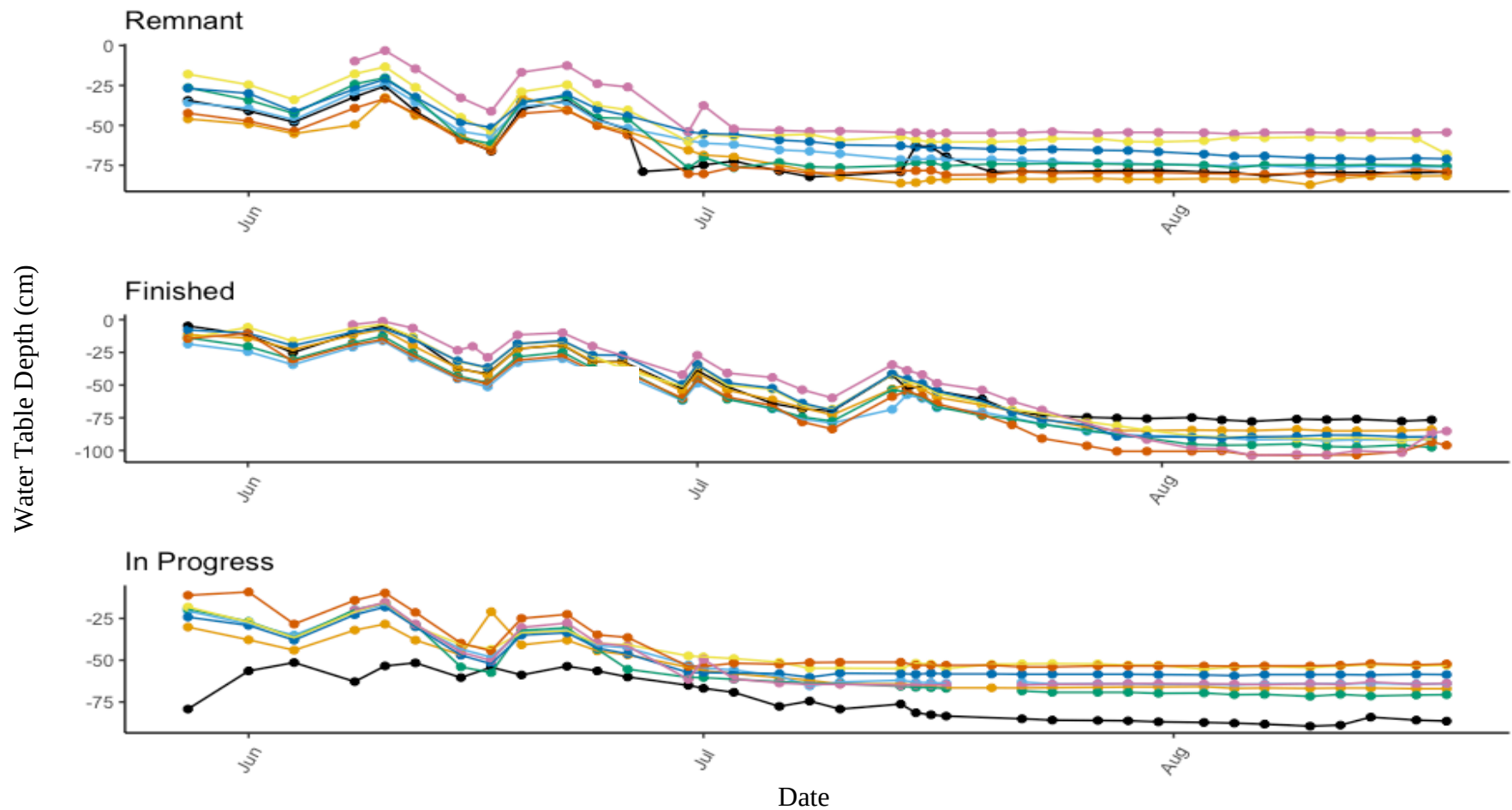


Figure 3-5 Water Table change over time in the 2020 field season. Each colour represents a different well within the research transect to show the variability across the transect.

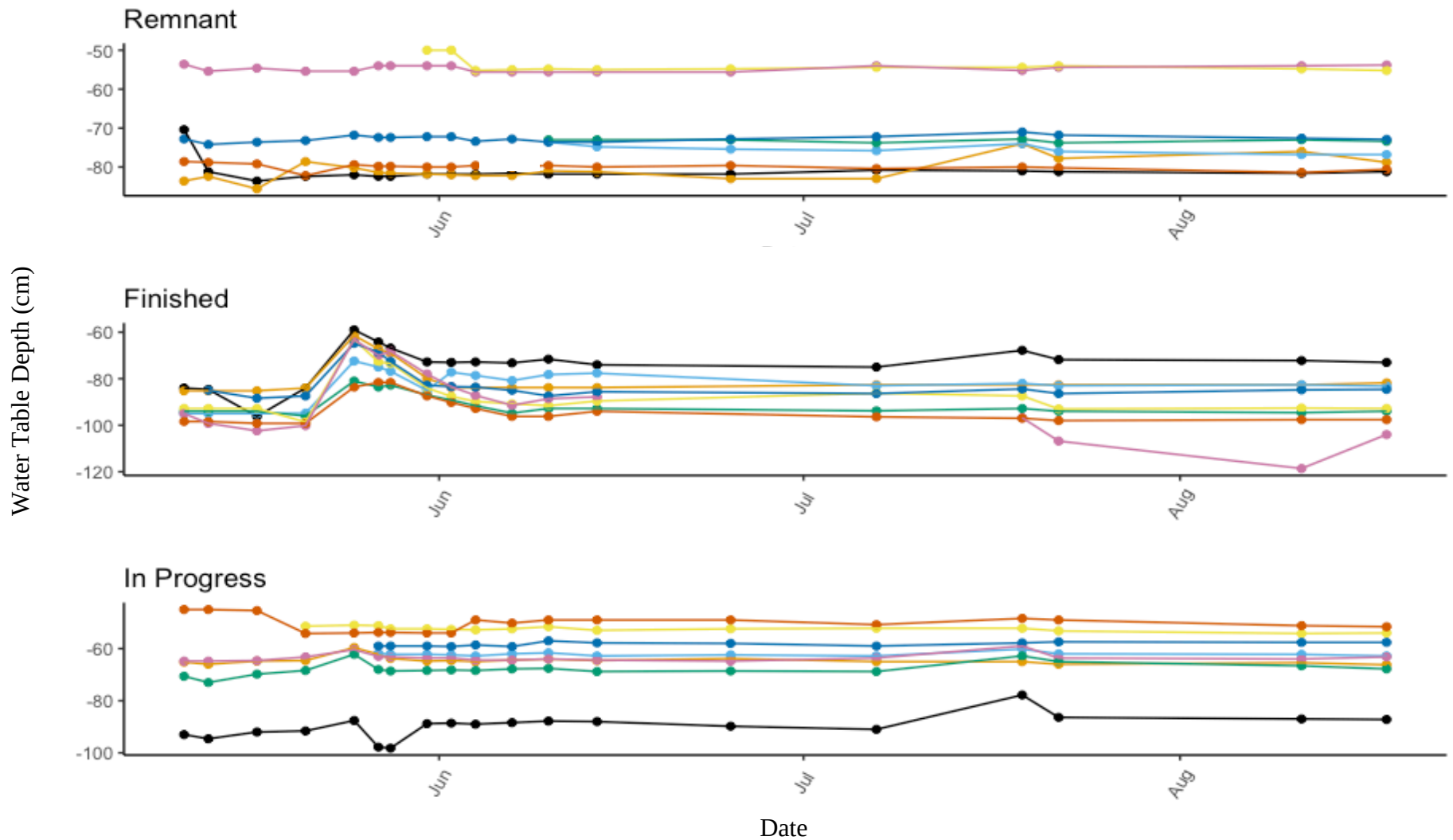


Figure 3-6 Water Table change over time in the 2021 field season. Each colour represents a different well within the research transect to show the variability across the transect. Where the line goes flat it indicates a dry well, showing just the depth of the peat profile at this point.

In Figure 3-6 all three sites experienced seasonal drawdown of the saturated water table. The In Progress and Remnant sites showed similar trends and their wells went dry in late June/early July. Whereas the Finished site wells held water till mid- to end-July. This difference is a result of the drainage network around the Finished site being closed and a deeper remnant peat profile found in the Finished site compared to the remnant peat depths in the In Progress and Remnant sites. In Figure 3-6, all three sites except for a brief rise in the Finished site wells at the end of May experienced no drawdown of the water table as there was no water table during the 2021 field season.

To calculate the change in storage from the water table we averaged the first field day measurements (Day 1 - May 28th, 2020, May 28th, 2021) to the final field day measurements (August 19th, 2020, August 18th, 2021) for each of the 8 wells at each site. After calculating each of the site's average outputs (or losses) they were then multiplied by the Sy value of 0.06 and areally weighted their value to better represent the field variance.

In 2020, when comparing the sites, it is noted that the Finished site has a greater saturated zone value indicating this site has more water available. Interestingly, the In Progress sites saturated zone value is less than both the Remnant and Finished sites but is comparable to the Remnant site with only a difference of 0.566 mm (Table 3.4.4.1).

2021 showed a similar trend to 2020 with the Finished site having the greatest saturated zone value at 3 mm (Table 3.4.4.1) and the Remnant and In Progress having comparable values with a difference of 0.2 mm (Table 3.4.4.1) with the In Progress having the lowest saturated zone value.

When comparing 2020 values to 2021 values it is apparent that 2021 was substantially drier than in 2020 with a difference of 84 mm between the totalled average output and 27 mm between the totalled weighted outputs (Table 3.4.4.1).

Table 3.4.4-1 Saturated Zone values (mm) from well transect at each site

2020	average output (mm) 2020	weighted output (mm) 2020	average output (mm) 2021	weighted output (mm) 2021
Remnant	25	8.0	2	0.6
In Progress	24	7.5	1	0.3
Finished	47	15	9.0	3
TOTAL	96	30.5	12	4

3.4.5 *Unsaturated Flux*

In 2020, the values of VMC increased with depth as the upper layers of the soil profile should be the first to dry out through the season, while the deeper soils remained moister as the sun and wind have less effect at depth. In 2021, the VMC values jumped from increased moisture at the top and drier conditions 15 cm in the profile back to increased moisture as it goes deeper (Table 3.4.5-1).

When comparing 2020 values to 2021 values the VMC of the soil profile is much drier in 2021 with relatively small differences as the soil column gets deeper. If we compare the 25 cm depth for both years, there is a 15.07 VMC difference (Table 3.4.5-1).

Table 3.4.5-1 Average Volumetric Moisture Content Values from soil probes at South Weather Station

2020		2021		Comparing 2020 to 2021 VMC
Depth (cm)	Volumetric Moisture Content (VMC)	Depth (cm)	Volumetric Moisture Content (VMC)	
5	9.48	5	4.68	4.8
10	10.59	10	4.07	6.52
15	10.89	15	3.77	7.12
20	11.86	20	4.06	7.80
25	20.77	25	5.70	15.07
5	10.98	5	4.06	6.92
10	11.99	10	4.64	7.36
15	15.18	15	3.90	11.32
20	22.50	20	5.73	16.77

3.4.6 The Water Balance

In 2020, the precipitation totalled 323.4 mm while the evaporation totalled -353.4 mm; flux contributions from the saturated zone were -30.45 mm and -62 mm from the unsaturated zone resulting in a total water balance deficit of -122.45 mm. In 2021, the precipitation totalled 283.7mm while the evaporation totalled 302.8mm; flux contributions from the saturation zone were -3.78 mm and -20.2 mm from the unsaturated zone resulting in a total water balance deficit of -39.3 mm.

Comparing both years, they both had water balance deficits. The values from 2021 appear “better” but this is due to the lack of moisture available.

Table 3.4.6-1 Water Balance for all years of study. The water balance was calculated by subtracting the ET from the total precipitation then +/- the flux.

	2019	2020	2021
<i>Precipitation (mm)</i>	+597.8	+323.4	+283.7
<i>Evaporation (mm)</i>	N/A	-353.4	-302.8
<i>Saturated Zone</i>	N/A	-30.45	-3.78
<i>Unsaturated Zone</i>	N/A	-62	-20.2
<i>Total (mm)</i>	N/A	-122.45	-39.3

3.5 Discussion

Peatland extraction at Moss Spur 2 started in 2007 and was last completed in 2018. The 11 years of substrate removal significantly disrupted the site's ecosystem dynamics, primarily through the lowering of the water table by a series of drainage ditches and the removal of vegetation, leaving behind a remnant sedge peat substrate typically found in fen peatlands. Restoration efforts began in the fall of 2019; however, due to a series of precipitation events and equipment shortages the site's restoration resulted in the development of three distinct surface microtopographies (Remnant – untouched, In Progress – linear mounds and Finished). As a result, the study's objective was revised to examine the hydrological impacts of these varying microtopographies, with the aim of restoring the extracted bog to a fen peatland system.

The meteorological data (Tables 3.4.1-1) compared the 30-year historical climatic normals to the observed conditions during the 2019-2021 field seasons. The average monthly temperatures across all months were within 1°C of the climate normals, except for May 2019, which exhibited an anomaly of approximately +4.7°C, and May 2020, which showed a decrease of approximately -4.3°C. The temperature spike in May 2019 may have resulted in an earlier

onset of the growing season and fast melt, whereas the temperature drop in May 2020 could have delayed the growing season's start slowing the melt. However, given that the remainder of the monthly averages remained within 1°C of the normals, the overall impact on seasonal dynamics was likely minimal. In terms of precipitation (Table 3.4.1-2), the 30-year normals for May to October were 420 mm. In comparison, the 2019 field season had a surplus of 177.7 mm, with a significant 218.5 mm falling in September alone. Conversely, the 2020 field season had a deficit of 96.7 mm, and the 2021 field season had a larger deficit of 136.4 mm. These consecutive annual precipitation deficits contributed to notably dry field conditions over the two-year period which is apparent in the field season measurements for the water balance calculations.

The change in storage (Table 3.4.4-1) gives the greatest insight into truly how dry the field conditions were at each site, with a difference of 26.8 mm between the years' weighted outputs. In 2020, the areally weighted output was 30.4 mm with 14.8 mm from the Finished site, and in 2021 it was 3.78mm with 2.85mm from the Finished site. The Finished site had more water available in both years, but it was not enough moisture to sustain fen bryophytes. All sites had water table levels below the surface for both years; however, in 2020, the wells went dry in early June and in 2021 the wells were dry most (99%) of the field season.

Evaporation is the greatest loss to the site and when comparing evaporation rates (Table 3.4.2-2 and Table 3.4.2-3) in 2020 the Remnant site had the highest ET rate at 6 mm/day, and 116 mm areally weighted total and both the Finished and In-progress sites had an ET rate of 5 mm/day with 109 mm and 96 mm total respectively. In 2021, the In-progress site had the highest ET rate 4 mm/day and 110 mm total and both the Finished and Remnant sites had an ET rate of 3 mm/day with 83 mm and 77 mm total respectively. Given that each site has minimal vegetation cover typical of an extracted bog, the ET variability between sites most likely is a result of direct

evaporation from the exposed peat and available surface moisture rather than plant transpiration. Lysimeters were placed in transects that most represented the entire site to ensure ET measurements reflected the individual site characteristics.

Price (1997) found that *Sphagnum* moss requires the soil water tension to be greater than -100 cm (i.e., less negative) for successful growth. Graf and Rochefort (2010) found that fen bryophytes require a soil water tension of -43.9 cm to -87.7 cm. In 2020, the In Progress site had the greatest soil water tension at -84.2 cm, then the Remnant site had a soil water tension of -65.7 cm, and the Finished site had the most ideal soil water tension average at -58.2 cm. In 2021, the In Progress site had again the greatest average soil water tension at -120.6 cm, followed by the Finished site with an average of -92.2 cm then the Remnant site with an average of -85.6 cm. Using the previous research findings, the data collected in 2020 at all sites it appears to be within range for a potentially successful *Sphagnum*, as well as fen bryophyte growth. In 2021, when comparing previous findings, it shows that all sites would be too dry to support fen bryophyte growth, and the In Progress site would also be too dry for *Sphagnum* growth. It should be noted that it was difficult to maintain the tensimeters for accurate readings in the 2021 field season due to the lack of precipitation and drying of the remnant upper peat layer.

The water balance results (Table 3.4.6-1) showed that the ET demands exceeded precipitation inputs in both 2020 and 2021, resulting in net water deficits of -122.45 mm and -39.3 mm respectively. This implies that in both years the atmospheric demand was greater than the precipitation inputs during the field season. It is evident when looking at the unsaturated zones losses that the water stored in the upper peat layers is being depleted, likely through surface evaporation and infiltration. When comparing 2020 to 2021 values we observed a substantially larger deficit in 2020, suggesting that the hydraulic conditions were drier that year,

but it was actually the opposite. In 2021 there was less moisture overall in the system and was very dry for the whole field season, showing a lesser deficit but it is due to lack of moisture in the system, not improved conditions. Although there was a moisture deficit in 2021, the Mulch site exhibited a much lower alpha (0.44) (Table 3.4.2-3) indicating that the evaporation rate was much less than the predicted evaporation. This suggests the mulch helps limit the amount of evaporation coming off the field, which is the largest demand of the system. The Mulch sites soil water tension (Table 3.4.3-1) was also the highest at -36.3 cm which is well above the -100 cm threshold and could sustain fen bryophyte species.

3.6 Conclusion

The results found that during the study period the site remained moisture-limited following peat extraction and partial restoration. Despite the use of three distinct surface microtopographies, all the sites experienced water table levels well below the peat surface, high evapotranspiration losses, and the soil water tensions exceeded the minimum thresholds required to sustain bryophyte establishment. The combined consecutive precipitation deficits experienced during the 2020 and 2021 field seasons further intensified the drought conditions, contributing to the depletion of moisture stored within the upper peat layers. Although the calculated water balance deficit was considerably smaller in 2021, field observations showed that the overall site moisture availability was lower due to the reduced precipitation and prolonged drying of the peat profile. Together these findings suggest that while restoration activities have begun to modify surface conditions, the hydrological regime has not yet recovered sufficiently to support fen bryophyte growth or the re-establishment of peat-forming processes. Continued hydrological management and monitoring will be necessary to help improve water retention and support vegetative growth for peatland recovery at the site, hopefully returning the site to a carbon

accumulating ecosystem. It is unfortunate that we were trying to calculate a water balance in a severe drought as it does not represent the normal moisture conditions that are found within eastern Manitoba.

4 Conclusion

This thesis evaluated the use of different surface microtopographies in both snow accumulation and during the growing season to capture a full picture of the remnant hydrological conditions at the Moss Spur 2 research site. Together, these two studies provide a comprehensive assessment on surface microtopographies and how they influence water availability throughout the annual water cycle.

We found that restoration activities (In Progress and Finished) positively influenced both the winter and growing season hydrology but due to the consecutive years of below-average precipitation they were not enough to restore pre-extraction hydrological conditions.

Highlighting the importance of studying the full hydrological cycle when completing a peatland restoration project, rather than a single season as findings in one season may not translate into sufficient moisture availability to sustain the hydrological conditions required for restoration.

The improved performance of the Finished site indicates that a more uniform surface topography enhanced both snow retention and growing-season water availability. However, the other two sites (In Progress and Remnant) still had active drainage ditches removing water from the sites. Due to equipment shortages, weather limitations and Covid-19 the main drainage ditch around the In Progress site and all the drainage ditches in the Remnant site remained in place, affecting the moisture conditions.

Future research should continue long-term monitoring to determine if the remnant drainage ditches being closed had an impact on improving the sites hydrological conditions. As climate conditions continue to be the limiting factor, understanding how natural peatland ecosystems respond to drought conditions will be important in improving restoration practices.

My recommendations for the peat industry and restoration practitioners from this research include ensuring all drainage ditches are closed after peat harvesting to retain water on the site. Where appropriate a culvert with a manual screw gate could be installed to provide a better management of water levels onsite. The manual screw gate would allow water to be retained during snow melt and periods of high precipitation to reduce excessive water loss, or it can release water in the case of a flood event. Despite the limited observations our results suggest the Mulch treatment was effective at reducing evaporation rates, which we found to be the greatest loss to the system. Therefore, once the drainage ditches are closed a mulch application on site may improve water retention, helping to restore remnant hydrological conditions, increasing the potential for a successful restoration of a carbon accumulating system.

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