HYDROLOGICAL AND CHEMICAL DYNAMICS OF A CONSTRUCTED PEATLAND

HYDROLOGICAL AND HYDROCHEMICAL DYNAMICS OF A CONSTRUCTED PEATLAND IN THE ATHABASCA OIL SANDS REGION: LINKING PATTERNS TO TRAJECTORY

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LAY ABSTRACT

A better understanding of the hydrological functioning of reconstructed peatlands in the Athabasca oil sands region is required as it is a novel approach in this region and there is potential for thousands of hectares of land that will require this reclamation in the future. Due to their recent establishment potential trajectories of constructed peatlands have yet to be fully analyzed as only recently has sufficient data been collected to evaluate the hydrological and hydrochemical functioning and provide insight on its overall success. While design strategies may seem sound, these constructed systems are completely human-made and it is unclear how they will develop and function in a highly disturbed landscape. Thesis results suggest that current conditions are not favourable to sustain a peatland as marsh-like conditions have developed which will limit its ability to persist long-term in a dry and changing climate. It is recommended that design strategies shift to incorporate characteristics found in undisturbed saline peatlands that are capable of supporting peat-forming vegetation in a saline environment. Due to the many challenges associated with reclamation in this region, lessons learned from this pilot project will help guide future peatland construction.

ABSTRACT

Peatlands comprise of approximately half of the Athabasca oil sands region, many of which overlay some of the world's largest bitumen deposits where surface mining for this resource has permanently altered the landscape. By law, companies must reclaim disturbed landscapes into functioning ecosystems including integrated upland-wetland systems with the objective of forming sustainable peat-forming wetlands. This thesis presents six years (2013 – 2018) of water balance and associated salinity data from one of the two existing constructed upland-wetland systems, the Sandhill Fen Watershed (SFW), a 52-ha upland-wetland built on soft tailings to evaluate the hydrological and hydrochemical performance and its potential to be self-sustaining.

Following a considerable decrease in hydrological management, the dominant water balance components changed from primarily horizontal (inflow and outflow) to vertical fluxes (precipitation and evapotranspiration) which increased inundation, encouraged salt accumulation and changed plant communities. Results suggest that current conditions are not favourable for fen-peatland development as marsh-like conditions have developed, limiting water conserving functions and the ability to persist long-term in a changing climate.

In terms of winter processes, topography currently controls snow accumulation, redistribution and melt at SFW while the role of vegetation in these processes is expected to increase as it continues to develop. Runoff ratios of snowmelt from hillslopes were drastically different than those previously reported for reclaimed peatland watersheds highlighting the influence of different soil materials used during construction. Under various climate change scenarios of a warmer and wetter climate, results from the Cold Regions Hydrological Model indicate that the influence of winter processes will decrease, potentially putting reclaimed systems at greater risk of moisture stress.

Substantial hydrochemical changes have occurred as salinity was relatively low at the study onset as high volumes of inflow and outflow prevented ion accumulation. Over time, salinity continued to increase year-over-year throughout SFW from 2013 to 2018 in the wetland and margin areas. This increase in site-wide salinity was attributed to the shift in dominant water balance fluxes, changes in water table position and increased mixing of SFW waters with deeper saline groundwater that underlies the system. Based on its current conditions, it is unlikely that SFW will support peat-forming vegetation. It is recommended that design strategies shift to incorporate characteristics found in undisturbed saline peatlands that are capable of supporting peat-forming vegetation in a saline environment.

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PREFACE

This Ph.D. dissertation is a total of six chapters. Chapter 1 provides a literature review of the general thesis topic and highlights the contribution of this thesis to the field of research. Chapters 2, 3, 4 and 5 constitute the main body of the dissertation where chapters 2 and 3 focus on system hydrology of a constructed peatland-watershed and chapters 4 and 5 link the hydrology to the hydrochemical trends and processes. These main body chapters are presented as a journal article for peer-review. Chapter 2 assesses the hydrological development and functioning of a constructed peatland-watershed in the Athabasca oil sands region. Chapter 3 focusses specifically on the role of winter processes in the overall system hydrology and examines the influence of a various future climate scenarios on these winter processes. Chapter 4 examines the early hydrochemical changes following a shift in management and hydrologic regime. Chapter 5 focusses on linking system hydrology to observed hydrochemical evolution six years since construction. Finally, Chapter 6 provides a summary of the key findings from each chapter presented in the thesis and suggests recommendations for future research.

DECLARATION OF ACADEMIC ACHIEVEMENT

The main body of this thesis is contained in Chapters 2, 3, 4 and 5 which each represent a journal article that is published, submitted, or in preparation for submission to a peer-reviewed academic journal. Permission letters from publishers permitting the reuse of published material in this thesis can be found in Appendix C.

In additional to the work presented here, I have also been involved in the publication of one other journal article (currently under review) in collaboration with Dr. M. Graham Clark of the optical properties of dissolved organic matter at the same site where my PhD research occurred:

Clark, M. G., Biagi, K. M. & Carey, S. K. Optical properties of dissolved organic matter highlight peatland-like properties in a constructed wetland. *Science of the Total Environment. Submitted (STOTEN-S-21-05441)*.

Chapter 2:

Biagi, K. M., Clark, M. G. & Carey, S. K. Hydrological functioning of a constructed peatland-watershed in the Athabasca oil sands region: potential trajectories and lessons learned. *Ecological Engineering*. *Under Review (ECOLENG-D-20-01243)*.

Kelly Biagi (dissertation author) is the main researcher, first author and corresponding author of this paper. Kelly Biagi collected and processes the field data and samples and acknowledges field work assistance from Dr. Scott Ketcheson, Supriya Singh, Sarah Fettah, Hannah Ponsonby, Sean Leipe, Angus MacDonald, Keegan Smith, Ian

Martin, Dr. Michael Treberg and Dr. Gordon Drewitt. Data analysis and writing was primarily completed by Kelly Biagi with additional analysis and writing from M. Graham Clark and Dr. Sean Carey. Data for this work was collected from 2013 to 2018 where Kelly Biagi was absent during the 2013 and 2015 field seasons and relied on field assistants and student colleagues mentioned above.

Chapter 3:

Biagi K. M. & Carey S. K. (2020). The role of snow processes and hillslopes on runoff generation in present and future climates in a recently constructed watershed in the Athabasca oil sands region. *Hydrological Processes*, *34* (17): 3635–3655 DOI: 10.1002/hyp.13836

Kelly Biagi (dissertation author) is the main researcher, first author and corresponding author of this paper. Kelly Biagi collected and processes the field data and samples and acknowledges field work assistance from Keegan Smith, Ian Martin, Dr. M. Graham Clark, Dr. Michael Treberg and Dr. Gordon Drewitt. Data analysis and writing was primarily completed by Kelly Biagi with additional assistance from Angus MacDonald for help with model setup and Sean Leipe for help with the LiDAR data and ArcGIS. Manuscript editing and discussion guidance was provided from M. Graham Clark and Dr. Sean Carey. Data collection for this chapter was focussed during the 2017-2018 winter season where additional meteorological measurements were provided by O'Kane Consultants from 2013 to 2018.

Chapter 4:

Biagi, K. M., Oswald, C. J., Nicholls, E. M., Carey, S. K. (2019). Increases in salinity following a shift in hydrologic regime in a constructed wetland watershed in a post-mining oil sands landscape. *Science of the Total Environment 653*, 1445−1457 DOI: 10.1016/j.scitotenv.2018.10.341

Kelly Biagi (dissertation author) is the main researcher, first author and corresponding author of this paper. Kelly Biagi collected and processes the field data and samples and acknowledges field work assistance from Dr. Michael Treberg, Dr. Gordon Drewitt, Haley Spennato, Erin Nicholls, Chelsea Thorne and Dr. M. Graham Clark. Data analysis and writing was primarily completed by Kelly Biagi with additional assistance from Dr. Carl Mendoza with groundwater data as well as detailed comments and suggestions to the manuscript from Carla Wytrykush and Dr. Scott Ketcheson. Manuscript editing and guidance with the discussion was provided from Dr. Sean Carey.

Chapter 5:

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Kelly Biagi (dissertation author) is the main researcher, first author and corresponding author of this paper. Kelly Biagi collected and processes the field data and samples and acknowledges field work assistance from Dr. Michael Treberg, Dr. Gordon

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF ABBREVIATIONS

AOSR	Athabasca oil sands region
B1	Boardwalk 1
B2	Boardwalk 2
B3	Boardwalk 3
CRHM	Cold Regions Hydrological Model
DEM	Digital elevation model
ΔS	Change in storage
EC	Electrical conductivity
ET	Evapotranspiration
HRU	Hydrological response unit
I	Inflow
Ksat	Saturated hydraulic conductivity
LC-excess	Line conditioned excess
LEL	Local evaporation line
LFH	Litter-fibric-humic material
LMWL	Local meteoric water line
LOI	Loss on ignition
MR	Margins
NF	Nikanotee Fen
O	Outflow
OSPW	Oil sands process water
P	Precipitation
PET	Potential evapotranspiration
Pf sand	Pleistocene fluvial sand
RC1	Runoff Collector 1
RC2	Runoff Collector 2
SAR	Sodium adsorption ratio
SFW	Sandhill Fen Watershed
Si	Initial soil saturation
SWE	Snow water equivalent
T	Temperature
TR	Margins/transition areas
UP	Uplands
VWC	Volumetric water content

WBP	Western Boreal Plains
WSP	Water storage pond
WT	Water table
WTD	Water table depth

CHAPTER 1

INTRODUCTION

1.1 Reclamation in the Athabasca oil sands region

Landscapes in the Athabasca Oil Sands Region (AOSR) undergo considerable disturbance from surface mining activities to access the bitumen-rich McMurray formation (Government of Alberta, 2020), which permanently alters the existing vegetation communities, soil layers and hydrology (Price, McLaren, & Rudolph, 2010; Rooney & Bayley, 2011; Rooney, Bayley, & Schindler, 2012; Trites & Bayley, 2009b). In addition to this physical disturbance, excessive salts are ubiquitous in the post-mining landscape from the tailings material that are used to backfill open pits. As part of their legal requirements, oil companies must reclaim disturbed landscapes into functioning wetland and forest ecosystems (Province of Alberta, 2020). Since half of the pre-disturbance landscape was comprised by wetlands, >90 % of which were peatlands, constructed peatlands will be an important component of the reclaimed landscape. Constructing peatlands presents many challenges as there is limited previous reclamation work in this region and these systems must be completely reconstructed to re-establish suitable hydrological conditions on a relatively short timescale, compared to natural peatlands that form over a millennia (Clymo, 1983). Furthermore, the variable sub-humid climate and salinity of waste-materials used in reclamation are additional obstacles to success (Wytrykush, Vitt, Mckenna, & Vassov, 2012). It is expected that the hydrology and hydrochemistry of reclaimed peatlands will differ considerably from natural peatlands as a result of this disturbance, which will be reflected in the ecosystem functioning as hydrology drives solute distribution, vegetation and microbial communities and peat accumulation (W.-L. Chee & Vitt, 1989; Leung, MacKinnon, & Smith, 2003; Nwaishi, Petrone, Price, Ketcheson, et al., 2015; Vitt, House, & Hartsock, 2016a). Effective management of constructed peatlands to ensure their long-term success requires a better understanding of water and solute cycling processes in these systems and how they respond to various climatic conditions.

1.2 Undisturbed peatlands in the AOSR

Undisturbed peatlands play a critical hydrological role in the AOSR as the climate is usually in a state of water deficit where PET generally exceeds P (Devito, Mendoza, & Qualizza, 2012). These peatlands are capable of storing and transmitting water to adjacent landscape units during these seasonal and decadal wet and dry cycles (Devito, Creed, Gan, et al., 2005; Lapen, Price, & Gilbert, 2000; Smerdon, Devito, & Mendoza, 2005; Smerdon, Mendoza, & Devito, 2007; Thompson, Mendoza, Devito, & Petrone, 2015; Wells, Ketcheson, & Price, 2017). Although only ~25% of annual precipitation falls as snow (Environment Canada, 2020), winter processes play an important role in peatland hydrology (Price, 1987; Price & Fitzgibbon, 1987; Rouse, 2000; Woo & Winter, 1993) especially in the WBP as snowmelt is often the only replenishment of storage deficits from the previous growing season (Redding & Devito, 2011; Smerdon, Mendoza, & Devito, 2008). Complex feedback mechanisms also exist in undisturbed peatlands that enable them to maintain wetness in a variety of climate conditions, providing potential resilience to wildfire and climate change (Kettridge et al., 2017; Waddington et al., 2015). Disturbances to the natural hydrologic regime not only affect these feedback mechanisms, but also influence solute and nutrient distribution (Hayashi, van der Kamp, & Rudolph, 1998; Kelln, Barbour, & Qualizza, 2008; Macrae, Devito, Strack, & Waddington, 2013; Nwaishi, Petrone, Price, Ketcheson, et al., 2015; Simhayov et al., 2017), water fluxes (Faubert & Carey, 2014; Ketcheson & Price, 2011; Ketcheson et al., 2017b; Nicholls, Carey, Humphreys, Clark, & Drewitt, 2016; Price, 1997; Price, Heathwaite, & Baird, 2003), vegetation communities (Vitt et al., 2016a), peat properties and formation (Amon, Jacobson, & Shelley, 2005; Ketcheson & Price, 2016a; Meiers, Barbour, Qualizza, & Dobchuk, 2011; Nwaishi, Petrone, Price, Ketcheson, et al., 2015; Price, 1997, 2003), microbial activity (Nwaishi, Petrone, Price, & Andersen, 2015), thermal regimes (Petrone, Price, Waddington, & von Waldow, 2004) and gas exchange (Waddington & Price, 2000).

1.3 Existing constructed peatlands in the AOSR

Currently, two pilot-scale peatland-watersheds exist in the AOSR, the Sandhill Fen Watershed (SFW) (Syncrude Canada Ltd.) and Nikanotee Fen (NF) (Suncor Energy), that were constructed to test the success of various design and construction practices, and to gain an understanding of the resulting system functioning. The design and conceptual models developed for peatland creation in this region stemmed from a multitude of research on natural peatlands in the region as well as test cell research (Faubert & Carey, 2014) and numerical modelling (Price et al., 2010). The SFW was designed to mimic the water conserving functions observed in natural WBP peatlands that allow them to persist long-term in a sub-humid climate. This includes contributions from regional and local groundwater flows (Devito, Creed, & Fraser, 2005; Smerdon et al., 2005) and large storage capacities that can sustain wetlands through periods of drought (Barr, van der Kamp, Black, McCaughey, & Nesic, 2012; Devito et al., 2012; Redding & Devito, 2010). To achieve this, fine- and coarse-grained upland hillslopes (referred to as hummocks) were incorporated into the SFW design to promote groundwater recharge, increase the system's storage capacity (Wytrykush et al., 2012) and limit the upward flux of saline tailings water (BGC, 2019).

In contrast, the Nikanotee Fen design was guided by numerical modelling that was used to determine the appropriate upland to peatland ratio that would provide sufficient groundwater flow from the upland aquifer required to maintain fen wetness conditions (Ketcheson et al., 2017a; Price et al., 2010). These two constructed peatlands vary considerably in design as the SFW was built directly atop ~40 m of saline composite tailings (soft tailings) while the NF is a hydrologically isolated system that is underlain by a geosynthetic clay liner and has one large upland aquifer that is ~3 m thick and slopes towards the peatland (Ketcheson et al., 2017a). The general overarching challenge in peatland reclamation in the AOSR is to design and construct peatland-watershed systems atop mine waste material (e.g. soft tailings) that can be resilient to the highly disturbed and saline post-mining landscape, with the capability to mimic the functioning capacity of undisturbed peatlands within the AOSR under a variable and uncertain future climate.

1.4 Objectives

With the potential for tens of thousands of hectares of disturbed land requiring reclamation, a better understanding the evolution of water pathways, connections and fluxes between upland-lowland and surface-groundwater is required. However, due to their recent establishment, the hydrological and hydrochemical evolution of constructed peatlands have yet to be fully analyzed. Only recently have sufficient data been collected to evaluate the hydrological and hydrochemical functioning of constructed peatlands to provide insight on potential trajectories and overall success of the systems. The chapters presented in this thesis aim to:

1) Evaluate the hydrological performance and initial success of SFW by assessing the functionality of the original hydrological design components of the SFW that were intended to support a self-sustaining upland-peatland watershed. (Chapter 2)
2) Examine the influence of winter processes on the hydrology of a constructed watershed and apply a physically-based model to evaluate the impact of different reclamation soils and climate projections on these winter processes (Chapter 3)
3) Provide an early assessment of the hydrochemical response of a constructed wetland to variations in hydrologic management with respect to water sources, flow pathways and major chemical transformations within the first three years following commissioning (Chapter 4)
4) Build upon Chapter 3 and use the first six years of hydrochemical data to understand the development of salinity and hydrochemical patterns in a constructed peatland

watershed following a heavily managed hydrologic regime to provide insight on its potential for long-term success as a peat-forming wetland. (Chapter 5)

Hydrology and water management exerts a major control on ecosystem functioning and is therefore a critical component to fully assess if reclaimed systems are performing as they were designed to. While design strategies may seem sound, these constructed systems are completely human-made and it is unclear how they will develop and function in a highly disturbed landscape. Due to the many challenges associated with reclamation in this region, lessons learned from this pilot project will help guide future peatland construction. Chapters presented in this thesis provide an assessment of the hydrological and hydrochemical development of the SFW as well as provide insight on potential trajectories and changes to design strategies that could increase the long-term sustainability for future peatland construction in this region.

CHAPTER 2

HYDROLOGICAL FUNCTIONING OF A CONSTRUCTED PEATLAND WATERSHED IN THE ATHABASCA OIL SANDS REGION: POTENTIAL TRAJECTORIES AND LESSONS LEARNED

The chapter provides a summary of the overall hydrological functioning of a constructed peatland watershed six years after being constructed. The paper quantifies the changes in water balance and fluxes and uses that information to provide insight on the overall system function and potential for long-term success.

This chapter was submitted as Biagi, K. M., Clark, M. G. & Carey, S. K. Hydrological functioning of a constructed peatland-watershed in the Athabasca oil sands region: potential trajectories and lessons learned. *Ecological Engineering*. *Submitted (ECOLENG-D-20-01243).* This chapter has been printed with permission or from Elsevier (Appendix C).

ABSTRACT

Mine reclamation in the Athabasca oil sands region is legally required as companies must reconstruct disturbed landscapes into functioning ecosystems, which previously existed in the Boreal landscape. Upland-wetland systems are relatively new in the constructed landscape and only two exist to date. The objective of this work is to understand the key hydrological changes post-management of a constructed peatland watershed and provide insight on the overall system function. Six years of hydrometric data are presented from the Sandhill Fen Watershed (SFW), a 52-ha upland-wetland catchment built on soft tailings with a pump system to provide fresh water, support drainage, and limit salinization. Wet years (seasonal precipitation > evapotranspiration) occurred in 2013, 2016 and 2018 and dry years (seasonal precipitation < evapotranspiration) occurred in 2014, 2015 and 2017 where wet years had large 5-, 10- and 100-year storms which were absent in dry years. Surface conductance and solar radiation explained most of the variation in ET fluxes. Changes in management practices drove many of the observed hydrological changes. Heavy management in 2013 muted water table (WT) responses to climate as inflow and outflow (via pumps) controlled WT response. After 2014, management efforts declined and hydrological exchanges were predominantly vertical, and saturated storage across the wetland increased. As a result, WT variability was tightly coupled to ET regardless of WT position relative to the ground surface, with greater changes related to deeper water tables, suggesting the absence of water conserving feedback mechanisms. Intra-watershed water movement was primarily towards the wetland from recharge areas in the upland swale, whereas surface runoff was rare and only occurred during extreme rain events and spring snowmelt. Peat properties were degraded compared to those observed in undisturbed peatlands, and natural stratification of the peat profile was absent. Results suggest that current conditions are not favourable for fen-peatland development as marsh-like conditions have developed, limiting conserving functions and the ability to persist long-term in a changing climate.

2.1 INTRODUCTION

Disturbed landscapes in the Athabasca oil sands region (AOSR) require complete reconstruction following surface mining activities as these ecosystems are permanently altered in terms of their vegetation communities, soil layers and hydrology (Price et al., 2010; Rooney et al., 2012; Trites & Bayley, 2009b). To mitigate this disturbance, which is projected to reach up to 4800 km^2 in area, oil companies are legally required by the Environmental Protection and Enhancement Act to reclaim disturbed landscapes into functioning ecosystems such as forests, wetlands and lakes, which previously existed in the Western Boreal Plains (WBP) (Province of Alberta, 2020). Recent reclamation efforts have focused on fen peatlands (Daly et al., 2012; Wytrykush et al., 2012) due to their widespread presence in the pre-disturbed landscape (Vitt, Halsey, Thormann, & Martin, 1996) and the important ecosystem functions they provide (Nwaishi, Petrone, Price, & Andersen, 2015; Waddington et al., 2015). However, success of constructed peatland-watersheds in the AOSR is uncertain, as it is a relatively new concept in this region and peatlands are complicated systems that typically take thousands of years to form naturally (Clymo, 1983; Nwaishi, Petrone, Price, & Andersen, 2015). Additional challenges for peatland construction include the complete re-establishment of hydrologic functioning, soil layering and vegetation communities while maintaining sufficient water quality, as excessive salts are ubiquitous in the post-mining landscape from the mine waste materials that are used to backfill open pits (Chapter 4).

Undisturbed peatlands play a key role in water regulation within the WBP as atmospheric fluxes create a state of water deficit, where potential evapotranspiration (PET) generally exceeds precipitation (P) (Devito et al., 2012). These peatlands can store and transmit water to adjacent forest uplands during seasonal and decadal wet and dry cycles (Devito, Creed, Gan, et al., 2005; Elmes & Price, 2019; Smerdon et al., 2005, 2007; Thompson et al., 2015; Wells et al., 2017). Peatlands also have complex feedback mechanisms that enable them to maintain wetness in a variety of climate conditions, which acts as a potential resilience mechanism to wildfire and climate change (Kettridge et al., 2017; Waddington et al., 2015). Long-term organic accumulation in undisturbed peatlands creates the unique peat structure that drives the hydrologic functioning of peatlands and the formation of the water conserving feedback mechanisms (Clymo, 1983). The degree of decomposition and compression varies considerably in a natural peat column (Rezanezhad et al., 2016a; Wilkinson, Moore, & Waddington, 2019), which enables the distinct water storage and transmission features of peatlands. Typically, upper layers are characterized a high hydraulic conductivity (coupled with a low bulk density and high specific yield) and can shed water easily during wet periods, whereas lower layers consist of highly decomposed peat (low hydraulic conductivity, higher bulk density and lower specific yield), which impedes flow when the water table (WT) recedes and reduces further water loss (Nwaishi, Petrone, Price, & Andersen, 2015). Furthermore, the complex gradient in pore structures found in undisturbed peatlands provide two key regulations to the WT. First, the pore spaces provide resiliency as the pore spacing itself regulates decomposition of the pore structure (Waddington *et al.*, 2015). Second as the water level drops, decomposition increases and the gas content of pore spaces increase, which decrease the effective porosity and limit additional water losses (Waddington et al., 2015).

Currently, two pilot-scale peatland-watersheds exist in the AOSR, the Sandhill Fen Watershed (SFW) (Syncrude Canada Ltd.) (Wytrykush et al., 2012) and Nikanotee Fen (Suncor Energy) (Daly et al., 2012; Price et al., 2010), that were constructed to test the success of various design and construction practices and to gain an understanding of the resulting hydrological functioning. While there is considerable research of the immediate changes (<3 years) within the SFW including system hydrology (Nicholls *et al.*, 2016; Spennato *et al.*, 2018; Lukenbach *et al.*, 2019; Twerdy, 2019; Chapter 3), hydrochemistry (Chapter 4; Oswald & Carey, 2016; Twerdy, 2019), vegetation communities (Vitt, Glaeser, House, & Kitchen, 2020; Vitt, House, & Hartsock, 2016b) and carbon dynamics (Clark, Humphreys, & Carey, 2019, 2020), only recently has sufficient data been collected to evaluate the hydrological functioning and provide insight on overall success of the SFW. The overarching challenge in peatland reclamation in the AOSR is to design and construct upland-peatland systems atop mine waste material (e.g., soft tailings), that are resilient to the highly disturbed and saline post-mining landscape, while mimicking the hydrological functioning capacity of natural analogues in a variable and uncertain future climate. The objective of this paper is to evaluate the hydrological performance and initial success of SFW by assessing the functionality of the original hydrological design components of the SFW that were intended to support a self-sustaining upland-peatland watershed. Specifically, six years of hydrological data are used to: 1) quantify variability of water balance components and intra-watershed fluxes, 2) gauge the system's trajectory and its ability to persist in a sub-humid climate, encourage salt flushing and support a fen-peatland and lastly, 3) provide guidance for future construction.

2.2 SITE DESCRIPTION

The study site is located ~40 km north of Fort McMurray, Alberta in western Canada. This region is situated in the WBP ecoregion, which has a cold and sub-humid climate where mean annual PET (607 mm) is greater than mean annual P (456 mm) (Environment Canada, 2020). Thirty-year climate normals (1971 – 2010) for the Fort McMurray Airport indicate that most of the annual P falls as rain (342 mm) with the majority delivered during May to August and the remainder falls as snow (114 mm of water equivalent) (Environment Canada, 2020). The mean annual daily temperature is 0.7 ± 1.2 °C with maximums typically occurring in July when daily temperatures average 16.8 ± 1.1 °C (Environment Canada, 2020).

The SFW was designed to mimic the water conserving functions observed in natural WBP peatlands that allow them to persist long-term in a sub-humid climate. This includes contributions from regional and local groundwater flows (Devito, Creed, & Fraser, 2005; Smerdon et al., 2005) and large storage capacities that can sustain wetlands through periods of drought (Barr et al., 2012; Devito et al., 2012; Redding & Devito, 2010). The SFW was constructed in the northwest corner of a previously mined area (1977-1999) referred to as East-in-pit that is part of Syncrude Canada Ltd.'s Base Mine (57°02'N, 111°35'W). The SFW was built over four years (2009 – 2012), where the entire East-in-pit was first filled with 35 m of inter-bedded composite tailings and tailings sand layers followed by a 10 m tailings sand structural cap. SFW was then constructed on top of these materials and is 52 ha total with a 17 ha fen wetland and 35 ha uplands that include a gently sloping (0.1 – 0.5 %) upland swale area in the south of the watershed (Figure 2.1) as well as 20 ha of 3 – 8 m tall upland hillslopes (referred to as hummocks). These hummocks were constructed of a combination of finer- and coarse-grained soils to promote groundwater recharge, limit the vertical flux of saline tailings water, and increase the system's storage capacity (BGC Engineering Inc., 2019; Lukenbach et al., 2019; Wytrykush et al., 2012). Soil materials used during construction were salvaged from previously existing landscapes or are by-products from the mining extraction and treatment process. The hummocks were constructed of tailings sand that was mechanically placed and capped with Pleistocene fluvial sand (Pf sand) (5 hummocks) or salvaged clay till (2 hummocks) and covered with salvaged forest floor material (Lukenbach *et al*., 2019). A peat-mineral mix was placed in the lower upland areas while 0.5 m of clay followed by 0.5 m of donor peat were placed in the wetland area. More details of watershed design, construction materials and soil stratigraphy are outlined in Chapter 4. The SFW was vegetated in 2011 with species native to this region (Nicholls et al., 2016; Wytrykush et al., 2012), but species composition has changed considerably throughout the watershed with time (Vitt et al., 2016b). Inflow and outflow of water can be managed via a pump system that was installed during construction (see Chapter 4 for details). Fresh water can be supplied from a near-by, undisturbed lake (Mildred Lake) to the Water Storage Pond (Figure 2.1) which flows eastward through the wetland area towards the outlet where water can be pumped out of the SFW and back into East-In-Pit. It should be noted that aside from deeper groundwater flow paths, the outflow of surface and near-surface water can only occur when the outflow pump is activated. These pumps were installed with the intention to maintain wetness and mitigate the elevated salinity levels from construction materials (Wytrykush et al., 2012).

2.3 METHODS

The study period spanned from October 2012 – October 2018. The surface water balance can be represented as:

$$\Delta S = (P + I) - (ET + O) + \varepsilon \quad (1)$$

where ΔS is the change in saturated wetland storage, P is precipitation (both rain and snow inputs), ET is the evapotranspiration, I is inflow, O is outflow and ε represents the error term which is calculated as the residual. Deep groundwater hydraulic gradients are low (Twerdy, 2019) and subsurface inputs and outputs to SFW were not considered. The water balance was calculated for both the growing season (1 May to 30 September) and for the annual water year (1 October of previous year to 30 September of analyzed year).

2.3.1 Meteorological measurements

Three meteorological towers (Figure 2.1) were used to measure air temperature, windspeed and direction, relative humidity, short, and long-wave radiation. Details on these instruments can be found in Nicholls *et al.*, (2016). A tipping bucket rain gauge (Model CS700, Campbell Scientific Inc. (CSI), Logan UT, USA) was used to measure rainfall and a CSI CS725 gamma ray spectrometer and a CSI SR50A sonic ranger were used to measure snow water equivalent (SWE) and snow depth, respectively. All measurements were recorded on a half-hourly basis with CSI CR1000 data loggers since 2013. Eddy covariance towers in the wetland and upland (instrumented in 2013) measured turbulent fluxes of energy and water, one of which operates year-round (Clark et al., 2019; Nicholls et al., 2016). Instrumentation, methods for flux computation by the eddy covariance technique, as well as the gap filling of flux data is described in detail in Clark *et al.,* (2019). Measurements were absent at the upland eddy covariance tower in 2016 as site access was restricted during the Fort McMurray Horse River Wildfire. Daily average fluxes of actual evapotranspiration (ET) were calculated by converting half-hour latent energy fluxes into mm of equivalent water using and summing over the whole day (24 hours). Surface conductance was calculated from the eddy covariance data with the methods described in Spank *et al.,* (2016). Pearson correlation coefficients were then calculated between ET fluxes and surface conductance, incoming shortwave radiation and average soil temperature to evaluate causes of seasonal differences in ET as well as spatial differences between the two towers. Methods for annual snow water equivalent (SWE) used in the water balance are outlined in detail in Chapter 3.

2.3.2 Inflow and outflow measurements

Water levels in the water storage pond (inflow) and V-notch weir (outlet) were measured using a pressure transducer (Rosemount Inc., Chanhassen, MN, USA). Lateral inflow volumes via pumping in the water storage pond was measured using a Model AT868 AquaTrans Ultrasonic Flow Transmitter. Further details on the pump system and other instrumentation are found in Nicholls *et al.* (2016). When pumps are on, water flows from the water storage pond, through a gravel dam into the wetland area and then moves eastward towards the outlet where near-surface water is pumped out of SFW back into East-In-Pit (Figure 2.1). Following 2014, the inflow pump remained off and only pre-planned outflow events occurred to mitigate high water levels in the wetland (Chapter 4; Nicholls et al., 2016).

2.3.3 Hydrometric measurements

Eleven near-surface wells were installed in 2013 followed by an additional 22 wells in 2014 to increase the spatial sampling distribution (Figure 2.1). Near-surface WT fluctuations were measured using PVC slotted wells installed to depths between 0.48 and 1.02 m which were instrumented with Solinst Junior Edge Levelogger pressure transducers from May to October each sampling year. The sampling program in 2016 was delayed due to the Fort McMurray Horse River Wildfire. Transducers measured water level and temperature every 15 minutes which were supplemented with monthly manual measurements of water level. Levels from transducers were baro-corrected using a Solinst Barologger and associated software. Wells were named according to their position within SFW, where the prefix B represents wells along boardwalks one (B1), two (B2) and three (B3), TR represents wells in the margins (transition) and UP represents wells in the uplands. Although B3W2 is on B3, it is categorized as a margin well based on its position in the watershed. Wells that were chosen for transects discussed in this paper are indicated on Figure 2.1.

Relationships between the ratio of daily change in WT (ΔWT) and ET were modelled as a function of water table depth by fitting a quadratic regression to determine if water conserving feedback mechanisms (outlined in Waddington *et al*., 2015) were developing in SFW. The expectation was that a decoupling between ΔWT:ET would indicate the presence of these water conserving feedback mechanisms with declining water tables.

Hillslope runoff during rain events was measured using constructed runoff collectors on north- and south-facing slopes (Figure 2.1). Details on runoff collector construction, data collection and analysis are outlined in Chapter 3.

2.3.4 Hydrophysical properties

Both in-situ field and laboratory methods were used to evaluate soil properties throughout SFW. Prior to this study in 2012, organic matter content and bulk density were the only soil parameters recorded (NorthWind Land Resources Inc., 2012). Soil pits were excavated on the hummocks, upland swale, margins and wetland to sample the soil profile for each landscape unit. Samples from soil pits were taken for laboratory analysis where each depth had replicates of three or more. Saturated hydraulic conductivity (Ksat) was determined using a KSAT instrument from METER Group Inc. The Hyprop instrument, from METER Group Inc., was used to produce soil water characteristic curves for the peat and peat-mineral-mix soils. Data from the Hyprop was also used to estimate porosity, which is automatically calculated within the HypropFit software. Specific yield was also estimated from Hyprop data by calculating the difference between the saturated volumetric water content (VWC) and the VWC at 330 mb; the equivalent to the sample field capacity. Samples were analyzed for bulk density using standard methods (Freeze & Cherry, 1979) with the exception that the samples were oven-dried at 80 °C to limit any loss of organic matter. Organic matter content was determined via loss on ignition (LOI) with samples placed in a muffle furnace at 550 °C for four hours. In situ measurements were made using a Guelph Permeameter for Ksat and single ring infiltrometers to estimate field Ksat and infiltration capacity. Ksat estimation from the Guelph Permeameter followed standard procedures (Soilmoisture Equipment Corp., 2012). Single ring infiltrometers were installed into the ground surface to depths of at least 1 cm and constant head tests were conducted until a steady state infiltration rate was reached. Ksat was assumed to be equal to the steady state infiltration capacity reached during each single ring infiltrometer test.

2.4 RESULTS

2.4.1 Meteorological variability

Wet and dry years during the study period were categorized when the difference between growing season (1 May to 30 Sep) P and actual ET (P - ET) were positive (P > ET) and negative (P < ET), respectively (Table 2.1). In drier years (2014, 2015 and 2017), ET exceeded rainfall totaling -32, -123 and -25 mm, respectively whereas wet years (2013, 2016 and 2018) had P-ET totaling 45, 116 and 147 mm, respectively. Additionally, the regional climate normal for growing season P (308 mm) was comparable or higher than dry year totals (299, 231 and 264 mm, respectively) and was lower than wet year growing season totals (375, 350 and 407 mm, respectively) (Table 2.1) (Environment Canada, 2020). The wet years also corresponded to the darkest June 1 to October 1 period (2.24, 2.22, and 2.10 GJ of incoming solar radiation for 2013, 2016, and 2018 respectively) where the dry years were brighter (2.29, 2.32, and 2.31 GJ for 2014, 2015, and 2017 respectively). Although the wet years 2016 and 2018 were coldest (mean June 1 to October 1 air temperatures of 12.1 and 13.6 °C), the dry years 2014 (14.3 °C), 2015 (14.1 °C), and 2017 (14.3 °C) were cooler than 2013 (14.6 °C).

Rainfall distribution was variable among years at SFW. Dry years (2014, 2015, 2017) had smaller monthly totals with one month considerably higher than others whereas wet years (2013, 2016, 2018) had multiple months with higher total rainfall (Figure 2.2). The highest rainfall in all years typically occurred in June or July with the exception of 2014 where most rain fell in May (Figure 2.2).

In general, rain events were typically small and rarely exceed 10 mm/day (439 events, 87% of events over all years) (Figure 2.3). Dry years (2014, 2015 and 2017) had a higher number of rain events <10 mm/day (93, 93 and 74 events, respectively) whereas wet years (2013, 2016 and 2018) had a lower number of <10 mm rain events (48, 77 and 54 events, respectively), but a greater number of heavy rain events (Figure 2.3). Rain events up to 20 mm were observed several times in in both wet and dry years (Figure 2.3) whereas rain events >20 mm/day were rare but occurred at least once each year with a higher frequency in wet years (in both cases) (Figure 2.3). Six of the observed rain events during the study period classified as return period storms for the Fort McMurray region (Golder Associates Ltd., 2010), including 100-year (92 mm on 21 Jul 2018), 10-year (58 mm on 3 Sept 2016), 5-year (54 mm on 9 Jul 2016) and 2-year storms (45, 50 and 42 mm on 29 May 2014, 9 Jun 2013 and 28 Jul 2013, respectively) (Figure 2.3). All these events were >40 mm and typically occurred during wet years except for one event in 2014 (Figure 2.3). Comparing to the regional 30-year climate normal, more rain was delivered in fewer days at SFW from 2013 – 2018, averaging 321 mm in 62 days, compared to 308 mm in 96 days (Environment Canada, 2020). The occurrence of the high return period storms likely contributes to this 34-day difference between the current study period and the climate normal.

Snow water equivalent (SWE) was greatest in 2013 and 2018 (two of the three wet years) totaling 126 and 133 mm, respectively, which was greater than the 30-year climate normal of 113 mm whereas 2014 – 2017 were considerably lower totaling 57, 60, 61 and 46 mm, respectively (Table 2.1). Snow season duration followed SWE patterns, where the longest duration occurred in 2013 and 2018 (166 and 163 days, respectively) while 2015 – 2017 were less (159, 125 and 139 days, respectively). The 2014 snow season was longer than other years (211 days) due to early season snowfall. On average, SWE at the SFW was lower than the reported 30-year climate normal of 133 mm while average snow season duration was the same (161 days) (Table 2.1). In general, snow accumulation was similar across landscape units where any variation was driven by topography (Chapter 3). While snowmelt was also driven by topography, variation in hillslope aspect strongly influenced the timing and rate of melt where south-facing slopes melted one month earlier and twice as fast as north-facing slopes (Chapter 3).

Monthly wetland and upland ET were similar from 2013 – 2015 where peak seasonal ET occurred in May or June (Figure 2.2). In later years, ET totals between the wetland and upland diverged. While peak monthly ET in the upland was maintained in 2017 at 85 mm and increased considerably in 2018 to 130 mm, peak wetland ET declined by ~25 mm and totaled 72, 76 and 60 mm from 2016 – 2018, respectively (Figure 2.2). Growing season totals were similar where upland ET showed an increasing trend over the study period, totaling 329, 333, 369, 356 and 372 mm, respectively for 2013 – 2015 and 2017 – 2018. Upland ET was not recorded in 2016 (see methods). Wetland ET totals were similar to upland values in early years totaling 330, 331 and 354 mm, respectively for 2013, 2014 and 2015, but then decreased during 2016, 2017 and 2018 totaling 233, 289 and 260, respectively (Table 2.1).

Surface conductance (inverse of surface resistance) explained a large proportion of the variance in ET fluxes at both sites ($r = 0.79$, p-val <0.001 and 0.70, p-val <0.001 for wetland and upland towers respectively), but was stronger at the wetland site, particularly in July, August, and September ($r = 0.89$ vs 0.62 in the upland, both p-val <0.001), when the canopy was fully developed. Incoming shortwave radiation also explained a large proportion of the daily variation in ET fluxes ($r = 0.84$, p-val <0.001 and 0.71, p-val <0.001 for wetland and upland respectively), which suggests why the wetland had a greater reduction in ET during months with high P (Figure 2.2), as months with higher than average P also correspond to reduced incoming solar radiation. There was a positive relationship between ET fluxes and soil temperatures ($r = 0.63$, p-val <0.001 and $r = 0.65$, p-val < 0.001, for the wetland and upland, respectively) however, in July and August there was a decoupling between soil temperature and upland ET fluxes that was not as pronounced in the wetland ($r = 0.32$, p-val < 0.001 and $r = 0.17$, p-val $= 0.036$ for the wetland and upland, respectively).

2.4.2 Soil hydrophysical properties

Soil properties of the salvaged peat, placed exclusively in the wetland area, had limited depth-dependent relationships (Figure 2.4). Organic matter content of the peat was the only parameter that significantly correlated with depth (p-value < 0.05, $r = 0.6$) whereas porosity, bulk density, specific yield and Ksat were not significantly correlated with depth (p-value > 0.05) (Figure 2.4). Organic matter content, porosity, bulk density, specific yield and Ksat of the salvaged peat samples were taken in 2017 and 2018 and averaged 59 %, 0.7, 0.4 g/cm^3, 0.3 and 5.7 x 10^{-5} m/s, respectively throughout the soil column (0 to 30 cm) over both years (Table 2.2). Organic matter content and bulk density (the only properties measured prior to this study) of the peat material increased since 2012, one year following peat placement, averaged 41 % and 0.3 g/cm^3, respectively.

Peat mineral mix, a combination of salvaged peat and mineral soil (clay), was placed in the margins and upland swale, which had variable soil properties as the mixing of the two soil types was uneven across the landscape (Table 2.2). Organic matter content, porosity, bulk density, specific yield and Ksat of the peat mineral mix material averaged 15 %, 0.6, 0.6 g/cm^3, 0.3 and 6.4 x 10^{-5} m/s, respectively throughout the soil column (0 to 30 cm) from both 2017 and 2018 samples. The hummocks were constructed of coarse soil materials and are considerably more conductive with higher infiltration capacities than the peat and peat mineral mix materials. Material properties of the hummocks are reported in Chapter 3 and Lukenbach *et al.,* (2019).

2.4.3 Water table dynamics and intra-watershed exchanges

2.4.3.1 WT dynamics

Pump activity strongly influenced WT position throughout SFW, particularly in the wetland where much of the inflow and outflow water was added to/drawn from. Due to the exceptionally large volume of water that moved through the wetlands in 2013 (Table 2.1), wetland and margin WTs primarily responded to pump activity instead of climate (Figure 2.5). WT positions during May to September 2013 at B1, B3 and the margins averaged 0.04, -0.01 and -0.3 m, respectively where negative values indicate that the WT was below the ground surface (no measurements were taken at B2 in 2013) (Table 2.3). With limited pumping from 2014 - 2018, WTs were largely mediated by P and ET and in general increased across the wetland (Figure 2.5c). Average wetland WT positions increased from 2014 – 2018 and were highest in 2018 averaging 0.06, -0.08 and 0.2 m, respectively for B1, B2 and B3 while margin WTs were similar among years averaging –0.4 m in 2018 (Table 2.3). It should be noted that pump activity in 2017 and 2018 lowered the wetland WTs considerably, which would have been much higher without these outflow events.

Following the variable WT in 2013 from the pump activity, wet and dry years dictated wetland WT behaviour. In drier years (2014, 2015, 2017), WTs were highest in May and decreased throughout the growing season (Figure 2.5). This signal was not as evident during wet years (2016 and 2018), as several heavy rain events maintained high WTs during the growing season (Figure 2.5). Over time, two areas in the wetland became increasingly distinct, as B2 WTs fluctuated closer to the ground surface while B1 and B3 WTs largely resided above the ground surface (>0.2 m) (Table 2.3; Figure 2.6). For most years, the WT in the B3 was well above the ground surface year-round, even during drier years whereas WTs surrounding B1 and B2 typically fell below the ground surface by the end of the growing season (Figure 2.5c). The margin WTs were typically below the ground surface for much of the season averaging -0.3 m from 2014 – 2018. The upland swale WTs were not measured in 2013, but remained consistent during 2014 – 2018, averaging -0.6 m (Table 2.3; Figure 2.6).

The ΔWT:ET ratio demonstrated a non-linear quadratic response depending on depth of the water table (Figure 2.7). The shape of the function was similar at all three boardwalks, with greater ratios (i.e., a greater water table response to atmospheric forcing) attributed to low water tables. Since soil hydrophysical properties have little relationship with depth, this change in response of the water table to atmospheric forcing is unlikely to be due to differences in the soil matrix composition. As the water table approached 0.2 m above the surface, the ratio reached an asymptote, increased in variance, and may have even decreased but there are limited data to fully evaluate.

2.4.3.2 Intra-watershed fluxes

Water was added to the wetland area primarily via lateral groundwater contributions from the margins and uplands while surface overland flow from hillslopes was rare. The uplands acted as recharge areas where the hydraulic gradient was consistently towards the margin and wetland areas throughout the growing season (Figure 2.5b). Flow reversals between the wetland and margin occurred in early years (2013 – 2015) (Figure 2.5) (Spennato *et al.,* 2018), but declined considerably starting in 2016 where the majority of flow occurred from the margins to the wetland even with the exceptionally high WTs in 2018 (Figure 2.5b). The SFW system was designed for water in the wetland to flow from B1 eastwards towards the outlet near B3. This flow system was maintained in the earlier years (2014 and 2015), even when the outflow pump was largely inactive. From 2016 onwards, the direction of flow was outwards from B2 (mid-fen) towards B1 (west) and B3 (east) (Figure 2.5c). The original gradient from B1 to B3 was only restored during outflow pump events.

Surface overland flow from hummock hillslopes, which was measured in 2018, only occurred during heavy rain events (Figure 2.8) and spring snowmelt (Chapter 3). The hummock hillslopes have a large average infiltration capacity ($2 \times 10^{-4} \pm 9 \times 10^{-5}$ m/s) and a high storage capacity, as they are constructed of highly conductive Pf and tailings sand (Ksat = 1×10^{-4} and 6×10^{-5} m/s, respectively) (Chapter 3). These capacities are rarely exceeded as the majority of rain events are low intensity and low volume (typically <10 mm). Only high intensity rain events exceeded the infiltration capacity (Figure 2.8) such as the 100-year rain event on 21 July 2018 where a total of 92 mm in 31 hours produced 27 mm of runoff.

2.4.4 Wetland water balance

Inflow and outflow pumps were heavily used in 2013 and dominated the water balance, totaling 870 and 883 mm, respectively, which is more than double the input from

P (Table 2.1). Following a small inflow event in 2014 (15 mm), the inflow pump remained off for the remainder of the study period. After 2013, outflow events were reduced and only occurred to mitigate exceptionally high wetland WTs. Outflow was relatively small in 2014 and 2015 totaling 17 and 55 mm, respectively, but made up a considerable component of the water balance in 2017 and 2018 totaling 102 and 210 mm, respectively (Table 2.1). The outflow pump events in 2014 and 2015 lasted only several hours compared to 2017 and 2018 which lasted for many days (Table 2.1). The influence of the outflow pump was evident in later years with wetland WTs lowering by an average of 0.2 m in four days both in July and August 2018 (Figure 2.5c).

Once pump activity decreased after 2013, vertical fluxes were the largest component of the water balance for the remainder of the study years (Table 2.1). Pronounced wet years occurred in 2013, 2016 and 2018, all of which had several large rain events that contributed to the high cumulative annual totals (390, 372 and 436 mm, respectively) compared to ET (330, 268 and 304 mm, respectively) (Table 2.1). In contrast, 2014, 2015 and 2017 were drier, where annual ET (387, 419 and 316 mm, respectively) exceeded that of annual rain (331, 285 and 326 mm, respectively). SFW is part of a larger groundwater flow system that originates from south and moves north and northeast through the SFW, yet this contribution was considered small in the overall water balance. Annual saturated storage changes (calculated as the difference over the water year) were considerably higher than zero in wet years except 2013 (-88, 141 and 199 mm for 2013, 2016 and 2018, respectively) which is likely a result of the high pump activity. In drier years, storage changes were closer to zero or negative in drier years (29, -141 and -139 mm for 2014, 2015 and 2017, respectively). The residual term represents the deviation from closing the annual water balance (Table 2.3).

2.5 DISCUSSION

While constructed ecosystems are extremely young on peatland-formation timescales, assessing early changes as hydraulic and physical properties re-establish is critical to understand potential trajectories and help guide future reclamation practice. Undisturbed peatlands are complex environments that perform a variety of important ecosystem services and have unique feedback mechanisms that make them resilient to change (Waddington et al., 2015). The variability of WT position across the SFW wetland in response to ET does not indicate a widespread capacity to regulate changes in WT within the wetland region (Figure 2.7). It was expected that regions developing these feedback mechanisms in SFW would show a decoupling between ΔWT and ET, but this relationship only intensified with WT depth. These results provide evidence that these water conserving feedbacks are absent in the central wetland region (B1, B2 and B3) of SFW and its current hydrologic conditions are considerable barriers to their development of peatland functioning. As a result, overall ecosystem functions in constructed peatland-watersheds will differ considerably from their natural analogues if current conditions persist over time (Ketcheson et al., 2016a; Nwaishi, Petrone, Price, & Andersen, 2015).

2.5.1 What were the major hydrological processes and changes at SFW?

The major hydrologic change observed at SFW was the shift in dominant water balance fluxes from inflow and outflow in 2013 to vertical fluxes 2014 and onwards (Figure

2.9). Inflow and outflow were disproportionately high in 2013, as both were more than two-fold the annual P for SFW (Table 2.1). Following the heavy management, wet and dry years emerged where wet years (2013, 2016, 2018) were characterized by heavy and intense rain events while the majority of events in dry years (2014, 2015, 2017) were small (<10 mm). ET was the largest loss of water after 2013 where annual and seasonal ET differences were largely controlled by climate conditions and changing vegetation (Table 2.1). While upland and wetland ET were similar in the first three years, patterns diverged in the last three years where upland ET continued to increase and wetland ET decreased (Figure 2.2). The continued development of upland vegetation in addition to consistent water availability accounted for the observed year-over-year increase in upland ET which is expected to plateau overtime as vegetation matures (Strilesky, Humphreys, & Carey, 2017). A lower surface conductance in the wetland may explain the lower ET compared to the uplands due to ponded conditions and the presence of *Typha*. Lower than expected ET has been observed in *Typha* ecosystems (Goulden, Litvak, & Miller, 2007). Additionally, 2018 and 2016 had the lowest levels of solar radiation and the coldest growing season which inhibited evaporation from the ponded surface waters. Cold temperatures disproportionately suppressed ET from the wetland since, unlike in the upland, soil temperatures are significantly correlated to ET throughout the whole growing season.

This shift in the water balance fluxes changed wetland WT behaviour including the observed increase in wetland saturated storage. Pump activity in 2013 drove changes in WT position which were drastic as large fluctuations (>0.6 m) occurred in a matter of days (Figure 2.5). As a result, persistent ponded areas did not form throughout the wetland. With the shift to predominantly vertical fluxes after 2013, the limited outflow increased the saturated storage in the wetland as inputs from surrounding watershed and P events were unable to drain. The overall low conductance of SFW soils also impede water movement and drainage (Figure 2.4). Water is primarily supplied to the wetland via P, but is also supplied via near-surface (Figure 2.5b) and deeper groundwater from the south uplands (Twerdy, 2019) while surface runoff input from hummock hillslopes is rare and only occurs during heavy rain events (Figure 2.8) and spring snowmelt (Chapter 3). These hillslopes are constructed of tailings sand material capped with coarse and fine Pleistocene fluvial sand which have large infiltrations capacities (Chapter 3) that encourage recharge (Lukenbach et al., 2019). Hummock recharge may initially increase as soil covers weather (Kelln, Barbour, & Qualizza, 2007; Meiers et al., 2011; Sutton & Price, 2019), however, changes in climate and vegetation development may act to decrease percolation and groundwater recharge. This contrasts results from the Nikanotee Fen, the other constructed peatland in the AOSR, where hillslopes comprise of finer soil textures, higher antecedent moisture contents and low infiltration capacities, thus producing large volumes of surface runoff to the wetland (Ketcheson & Price, 2016c, 2016b).

The peat properties in SFW are considerably degraded (Figure 2.4 and Table 2.2) and will impact the potential for peatland development (Rezanezhad et al., 2016a). Natural stratification through the undisturbed peat profile, where humification typically increases with depth (Rezanezhad et al., 2016a; Waddington et al., 2015; Wilkinson et al., 2019), is absent in salvaged peat used in reclamation which is considerably degraded when peat is drained, stockpiled, transported and compressed during placement. The weak positive relationship of increased percent organic content with depth at SFW (Figure 2.4) is likely a result of increased aerobic decomposition when the WT was below the ground surface and soils were exposed to air, particularly in 2013 (Figure 2.5c). This relationship of increased organic content with depth is opposite of what is found in undisturbed peatlands, as deeper peat typically has less organic matter and a lower porosity (Clymo, 1983). Bulk density values are orders of magnitude higher than those reported in undisturbed peatlands while Ksat and specific yield within the top 0.3 m resemble those deeper in the natural peat column (>0.4 m) (Chason & Siegel, 1986; Wilkinson et al., 2019). This indicates the loss of stratification in the reclaimed peat column as the highly decomposed peat typically found at depth in undisturbed peatlands is close to the surface in reclaimed peatlands. Peat properties in SFW are similar to those reported for the Nikanotee Fen, where stratification throughout the peat profile was lost, thus limiting the ability to regulate changes in WT (Figure 2.4) (Nwaishi, Petrone, Price, Ketcheson, et al., 2015; Scarlett & Price, 2019). This is evident in SFW as WT fluctuations were drastic in the wetland in response to drainage and large P inputs (Figure 2.5) indicating little capacity to regulate changes to WT position.

2.5.2 Will SFW be a successful peatland watershed?

The overall goal of both pilot peatland-watersheds in the AOSR was to test the viability of various design and construction practices and gain an understanding of the resulting system functioning. To promote overall ecosystem success, SFW was designed to: 1) persist in a sub-humid climate, 2) encourage salt flushing and 3) support a fen ecosystem.

2.5.2.1 Persist in a sub-humid climate

In its current managed form, SFW can persist throughout wet and dry periods (as observed) with limited outflow management as sufficient water is provided from P and lateral drainage from upland areas to maintain an area of saturation. However, the lack of an outlet resulted in water accumulation year-over-year that has caused persistent standing water (>0.2 m) in the wetland (Figure 2.5), which is not typically observed in undisturbed peatlands. Occasional very high WTs require outflow management which will likely persist until a defined sill is established for surface water outflow, which is planned for SFW when it is further integrated into the surrounding reclamation landscape. The establishment of a sill will place a long-term upper limit on WTs, yet allow the wetland to remain saturated. However, this may increase the risk of drying in the future as an upper limit on saturated storage will be set. While the WBP region experiences decadal dry cycles (7 – 10 years), there has not been an extremely dry year since the construction of SFW. While climate change scenarios for this region predict increases in both temperature and precipitation, this may promote overall drier conditions at SFW (Helbig et al., 2020). While an increase in rain will deliver more water to SFW, an increase in temperature is expected to lengthen the growing season, enhance ET (Ireson et al., 2015; Thompson, Mendoza, & Devito, 2017) and decrease the replenishment of groundwater stores via snowmelt as the length of the winter season and snowpack SWE are expected to decline (Chapter 3). Changes in the rate, timing and magnitude of both rain and snowmelt will have important yet uncertain influences on future hydrological conditions at SFW. Finally, the ongoing development of surrounding upland forest vegetation may act to reduce water for recharge from enhanced

ET and could eventually use the wetland as a source of water during periods of drought (Strilesky et al., 2017; Thompson et al., 2015). The ET of a reclaimed forest in the same region increased by ~200 mm within the first ten years since construction (Strilesky et al., 2017), indicating the potential for increased water use for the upland regions of SFW and reduction in water partitioned for recharge.

2.5.2.2 Encourage salt flushing

Due to the well documented ubiquitous salinity in reclaimed systems in the AOSR (Chapter 4; Kelln et al., 2008; Kessel, Ketcheson, & Price, 2018; Kessler, Barbour, van Rees, & Dobchuk, 2010; Simhayov et al., 2017), it is critical that design features of these landscapes have an effective flushing capability. While the upland swale and hummocks provide recharge water to the downgradient margins and wetland throughout the growing season (Figure 2.5b) (Lukenbach et al., 2019), there is still persistent ion accumulation, and changing chemical composition, despite the increasing WT position (Chapter 4). Without a free-flowing outlet, saline water is not effectively flushed out of the system and P inputs, including large events, do not provide sufficient dilution and deeper groundwater flow pathways do not act as an effective flushing mechanism (Twerdy, 2019). As a result of the rising salinity, the SFW wetland chemistry is most similar to undisturbed brackish marshes and saline fens that exist in the AOSR (Hartsock, Piercey, House, & Vitt, 2021). In contrast, the Nikanotee Fen has a naturally-flowing outlet and while wetland salinity is still relatively high (~2500 μS/cm), year-over-year increases are considerably less (Kessel et al., 2018) than at SFW (Chapter 4).

2.5.2.3 Support a fen-peatland

The SFW peatland was constructed in a hydrogeological setting to maintain adequate wetness with a donor peat base with the goal that, over time, the self-design of nature (Mitsch & Wilson, 1996) would encourage peatland development. However, the degraded peat properties and increase in saturated storage are substantial barriers to successful peatland development.

The degraded peat will limit the ability of SFW to provide ecosystem functions that undisturbed peatlands provide, particularly the water conserving functions. The typical large pores and resultant high Ksat and specific yield within the top of an undisturbed system's peat column (Chason & Siegel, 1986; Price, 1992) allow for the rapid shedding of water, which decreases with depth where deeper layers have a higher proportion of small pores which effectively retain water (Waddington et al., 2015). The undisturbed peat column can also minimize WT decline from increased ET as water transfer to the surface via capillarity is greatly reduced once large pores are emptied (Waddington *et al.*, 2015). Additionally, peat plays an important role in reducing thaw rates of seasonal ground ice which prevents early season ET from reaching potential rates (VanHuizen *et al.*, 2020). The SFW does not appear to have widely established feedbacks which moderate the WT as it declines in response to ET (Figure 2.7). This inability to conserve water and mitigate WT variability could exacerbate the predicted lower WTs and overall potentially drier environment with the onset of climate change (Ireson et al., 2015; Thompson et al., 2017) where runaway drying could considerably lower the WT in the wetland and promote afforestation (Waddington *et al.*, 2015). Recovery of these peat soils will depend on the system's ability to maintain near-surface WTs for peat-forming vegetation to establish and decompose slowly under anaerobic conditions, which could take decades.

The lack of outflow and resultant saturated peat column and persistent standing water in the wetland areas is another barrier that limits the capability of SFW to regulate WT fluctuations. Pump use is not reasonable for larger scale reclamation and induces rapid WT fluctuations that typically do not occur in natural systems and can stress vegetation (Vitt et al., 2016b). However, the increased saturated storage in the wetland has reduced viable conditions for peat-forming vegetation that require a consistent near-surface WT (Trites and Bayley, 2009b, 2009a). A gradient of standing water has developed throughout the wetland (Figure 2.6) which has influenced wetland salinity patterns (Chapter 4) and vegetation distribution (Vitt et al., 2016b). The B3 area resembles marsh-like conditions where the WT is above the ground surface throughout the season and salinity is moderate as a result of the ponded conditions (~1500 μS/cm) (Chapter 4; D. H. Vitt et al., 2016b). B1 and B2 maintain a near-surface WT (Table 2.3; Figure 2.6) however, salinity is typically higher at B1 (>3000 μS/cm) than at B2 (~2000 μS/cm) (Chapter 4). The combination of near-surface WTs and intermediate salinity has made B2 the only area that can support peat-forming vegetation (Vitt et al., 2016b), while B1 and B3 have non-peat-forming vegetation typically found in marshes and uplands such as *Typha latifolia*, *Carex aquatilis* and *Calamagrostis canadensis* (Vitt et al., 2016b). Similar patterns have been observed at the Nikanotee Fen where *Typha latifolia* is a dominant species in wetland areas with persistent standing water (Borkenhagen & Cooper, 2019).

Culminating the above barriers, the trajectory of the SFW is likely towards a non-peat-forming wetland due to the high WTs, elevated salinity and resultant shifts in vegetation. This will influence the long-term sustainability of this system on the landscape as non-peat-forming wetlands cannot perform the same water conservation and transmission functions that sustain undisturbed peatlands and surrounding ecosystems during periods of drought in this sub-humid climate.

2.5.3 Recommendations for future construction

The original concept of SFW included outflow via a pump and underdrain system to remove saline water via outflow pumping. In addition, hummock hillslopes were constructed to encourage recharge, form groundwater mounds and prevent the upward movement of salts at depth into the wetland (Figure 2.9a). However, the unnatural nature of the pumps was not ideal for SFW and results presented here indicate that without consistent outflow, the saturated conditions and subsequent accumulation of salts has created unfavourable conditions for peatland vegetation (Figure 2.9b). Additionally, while the hummocks effectively promote recharge, the base of the hummocks are the primary region where OSPW discharges at the surface (Figure 2.9b). While completely eliminating the intrusion of salts is unlikely, several design changes could help limit saturated and saline conditions and promote more favourable conditions for peatland vegetation development. Firstly, ensuring consistent flushing via recharge from higher elevation areas (by increasing their area) as well as a constructed outlet that allows for natural drainage will increase freshwater supply and help mitigate the high WTs and elevated salinity in the wetland Secondly, extending the clay liner beneath the hummocks will help limit the formation of

Na^+ enriched areas at the base of the hummocks where OSPW from depth is discharging at the surface (Figure 2.9c). This will also help direct recharge water directly to the margins and wetland and limit recharge water lost to deep percolation. Lastly, a combination of marsh and fen wetlands comprised of salt-tolerant vegetation may be more appropriate based on the vegetation shifts in current constructed peatlands as a result of high WTs and elevated salinity (Figure 2.9c) (Borkenhagen & Cooper, 2019; Vitt et al., 2016b) with the potential to maintain wetness through dry periods. A fen-peatland constructed upgradient of marsh areas would have more access to input of freshwater via P and reduced interaction with more saline groundwater while slightly downgradient marsh areas could retain large inputs of water and dilute salinity while acting as a potential water source for the peatland (Figure 2.9c). Additionally, salt-tolerant species capable of enduring inundated conditions (Borkenhagen & Cooper, 2018; Vitt et al., 2020) and forming peat (Trites & Bayley, 2009a; Volik, Petrone, Wells, & Price, 2017; Wells & Price, 2015a) that have been observed in natural saline peatlands in this region may be more appropriate vegetation choices considering the likelihood of saline conditions and the resultant incursion of non-peat forming salt-tolerant vegetation.

2.6 CONCLUSIONS

With the potential for surface mining activities to affect thousands of square kilometers in the AOSR, it is critical to determine the best way to construct self-sustaining ecosystems that can be resilient to a changing climate. The critical ecosystem functions that undisturbed peatlands provide are challenging to replicate as consistent near-surface WTs and stratified peat column are required. However, these functions are required for their long-term success as they act to conserve water during drought conditions. This research aimed to assess the overall hydrologic functioning of a constructed peatland-watershed as well as provide insight on its potential trajectory within a disturbed landscape.

Shifts in the dominant water balance fluxes controlled much of the change observed at SFW within the first six years since commission. In the first year, inflow and outflow were the greatest fluxes as a result of heavy management where WTs were variable in response to pump activity. In the years following, management of inflow and outflow was targeted and as a result, P and ET were the largest gains and loss of water annually and changes to WT position were gradual in response to vertical fluxes. ET was typically greater in the uplands and increased over time while wetland ET has declined since the early years. During wet years, wetland WTs were maintained above the ground surface as large rain events replenished water losses from ET whereas WTs continuously declined in drier years as small rain events could not offset ET losses. Regardless of wet or dry climate conditions, WTs were significantly coupled to atmospheric fluxes, with a stronger relationship with WTs below the ground surface, indicating the absence of water conserving functions typically observed in undisturbed peatlands. Additionally, the peat in SFW is considerably degraded, as it lacks the stratification that defines undisturbed peat columns and enables them to perform the key water conserving feedback mechanisms that allows them to persist in a sub-humid and changing climate.

Cumulatively, the high and unconstrained WTs, degraded peat column and elevated salinity are considerable barriers towards long-term peatland development within SFW. Evidence that the SFW has shifted away from a peatland has emerged as marsh-like areas have developed. A constructed outlet that allows for natural drainage is a key design feature to limit high WTs and provide a flushing mechanism for the elevated salinity. For future landscape design, a combination of up-gradient fens and down-gradient marsh wetlands comprising of salt-tolerant (and peat-forming) species within the reclaimed landscape may be more appropriate as this has occurred naturally in the SFW wetland.

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Data Availability Statement

Data is available from the corresponding author upon request.

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Tables

Table 2.1. Annual and seasonal wetland water balance summary from 2013 – 2018. Annual values represent the water year (1 Oct of the previous year to 30 Sep of following year). WTL and UPL represent the wetland and upland, respectively. All water balance calculations are for the wetland only.

	Growing Season (1 May to 30 Sept)				
	Rain	**Rain Days**	**WTL ET**	**UPL ET**	**P-ET**
	mm	**days**	**mm**	**mm**	**mm**
2013	375	50	330	329	45
2014	299	75	331	333	-32
2015	231	75	354	369	-1238
2016	350	62	233	*	116
2017	264	53	289	356	-25
2018	407	55	260	372	147
Average	321	62	300	352	
Climate Normals	308	96			

	Annual (1 Oct – 30 Sep)											
	Rain	**SWE**	**Snow Season**	**WTL ET**	**UPL ET**	**Inflow**	**Inflow Volume**	**Total Inflow Time**	**Outflow**	**Outflow Volume**	**Total Outflow Time**	**ΔSat Storage**	**ε**
	mm	**mm**	**days**	**mm**	**mm**	**mm**	**m³**	**hours**	**mm**	**m³**	**hours**	**mm**	**mm**
2013	390	129	166	330	344	870	147838	3003	883	150162	2058	-88	-264
2014	331	59	211	387	406	15	2470	10	17	2907	13	+29	28
2015	285	62	159	419	426	0	0	0	55	9334	60	-141	-14
2016	372	62	125	268	*	0	0	0	0	0	0	+141	-25
2017	326	95	139	316	403	0	0	0	102	17362	122	-139	-142
2018	436	119	163	304	487	0	0	0	210	35770	276	+199	158
Average	357	88	161	337	378								
Climate Normals	342	113	161**	607 (PET)									

*Upland eddy covariance instruments not set up in 2016 due to Horse River fire

**When snow depth >= 1 cm

Table 2.2. Peat and peat-mineral mix (PMM) properties.

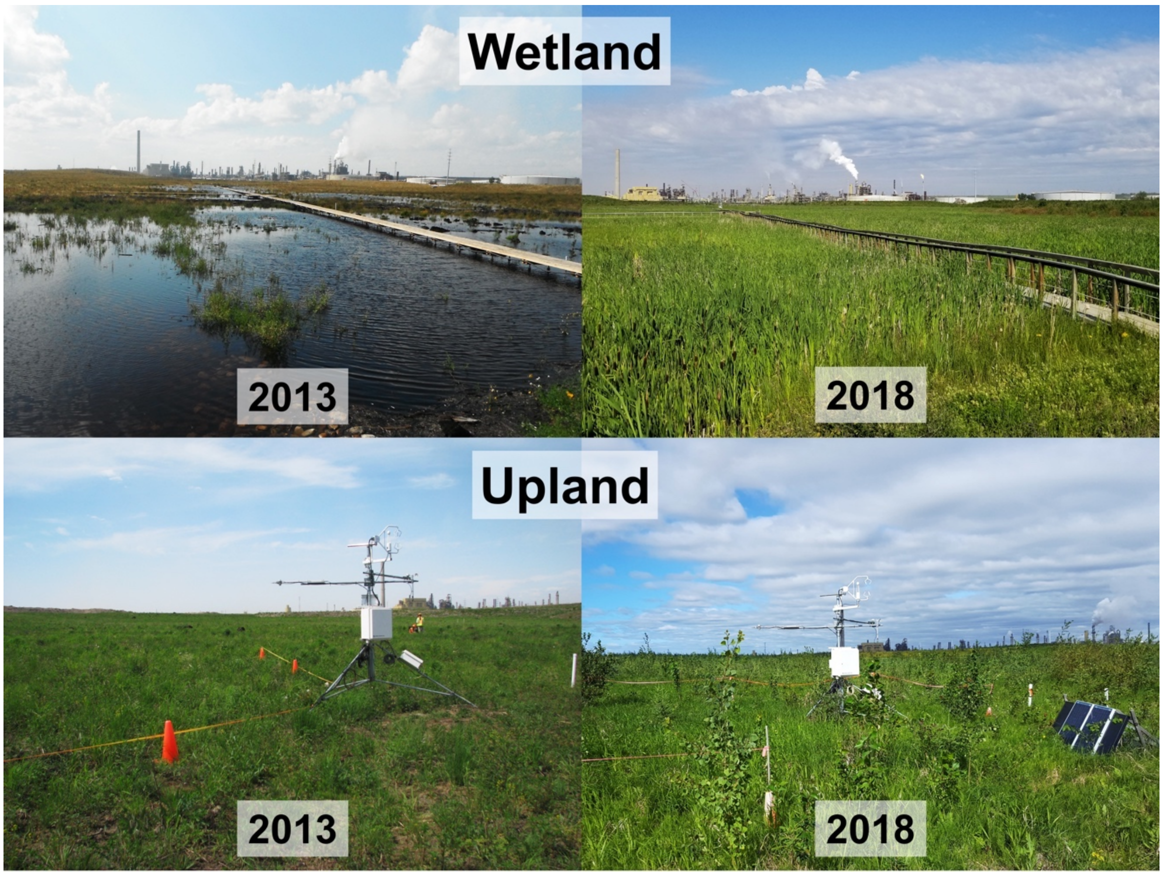
	Parameter	Units	Mean	Max	Min
Peat	Bulk density	g/cm^3	0.3 ± 0.1	0.6	0.2
	Organic content	%	56 ± 21	89	23
	Specific yield	-	0.4 ± 0.06	0.5	0.2
	Porosity	-	0.7 ± 0.2	0.9	0.3
	Ksat	m/s	5.7×10^{-5} ± 1E-4	3.9×10^{-4}	2.7×10^{-6}
	Infiltration capacity	m/s	6.4×10^{-5} ± 5E-5	1.8×10^{-4}	3.9×10^{-6}
Peat Mineral Mix	Bulk density	g/cm^3	0.6 ± 0.3	1.0	0.2
	Organic content	%	17 ± 19	76	0.6
	Specific yield	-	0.3 ± 0.08	0.4	0.2
	Porosity	-	0.6 ± 0.2	0.9	0.3
	Ksat	m/s	6.4×10^{-5} ± 1.4E-4	6.9×10^{-4}	9.2×10^{-8}
	Infiltration capacity	m/s	1.7×10^{-4} ± 1.5E-4	6.1×10^{-4}	6.1×10^{-6}

[1]Soil properties taken from NorthWind Land Resources Inc. (2012).

Table 2.3. Water table positions at Boardwalks 1, 2, 3, margins and the upland swale. Seasonal WT change is from May to Sept and annual WT change is from September of the previous year to September of current year.

	Boardwalk 1				Boardwalk 2			
	Δ Seasonal WT Position	Avg Seasonal WT position	Time WT above ground surface	Δ Annual WT Position	Δ Seasonal WT Position	Avg Seasonal WT position	Time WT above ground surface	Δ Annual WT Position
	m	m	%	m	m	m	%	m
2013	0.02	0.04	69	-	-	-	-	-
2014	-0.2	0.2	96	0.18	1.0	-0.2	23	-
2015	-0.6	0.09	69	-0.35	-0.3	-0.4	17	-0.14
2016	0.1	0.06	73	0.31	0.03	-0.1	89	0.11
2017	-0.6	0.1	57	-0.31	-0.4	-0.1	38	-0.11
2018	-0.2	0.06	42	0.41	-0.1	-0.08	37	0.3
	Boardwalk 3				**Margins**			
	Δ Seasonal WT Position	Avg Seasonal WT position	Time WT above ground surface	Δ Annual WT Position	Δ Seasonal WT Position	Avg Seasonal WT position	Time WT above ground surface	Δ Annual WT Position
	m	m	%	m	m	m	%	m
2013	0.2	-0.01	45	-	0.4	-0.3	4	-
2014	-0.1	0.03	57	-0.17	-0.2	-0.4	5	0.27
2015	-0.4	0.1	57	-0.04	-0.4	-0.4	3	-0.08
2016	0.04	0.1	94	0.25	-0.3	-0.03	41	-0.05
2017	-0.1	0.2	77	-0.18	-0.3	-0.3	16	-0.08
2018	0.2	0.2	89	-0.06	-0.1	-0.4	11	0.22
	Upland swale							
	Δ Seasonal WT Position	Avg Seasonal WT position	Time WT above ground surface	Δ Annual WT Position				
	m	m	%	m				
2013	-	-	-	-				
2014	-0.7	-0.6	0	-				
2015	-0.2	-0.6	0	-0.001				
2016	-	-	-	-				
2017	-0.2	-0.6	0	-				
2018	-0.2	-0.6	0	0.17				

Figures



Graphical Abstract. Development of the wetland (top) and upland (bottom) in the Sandhill Fen Watershed from 2013 to 2018.

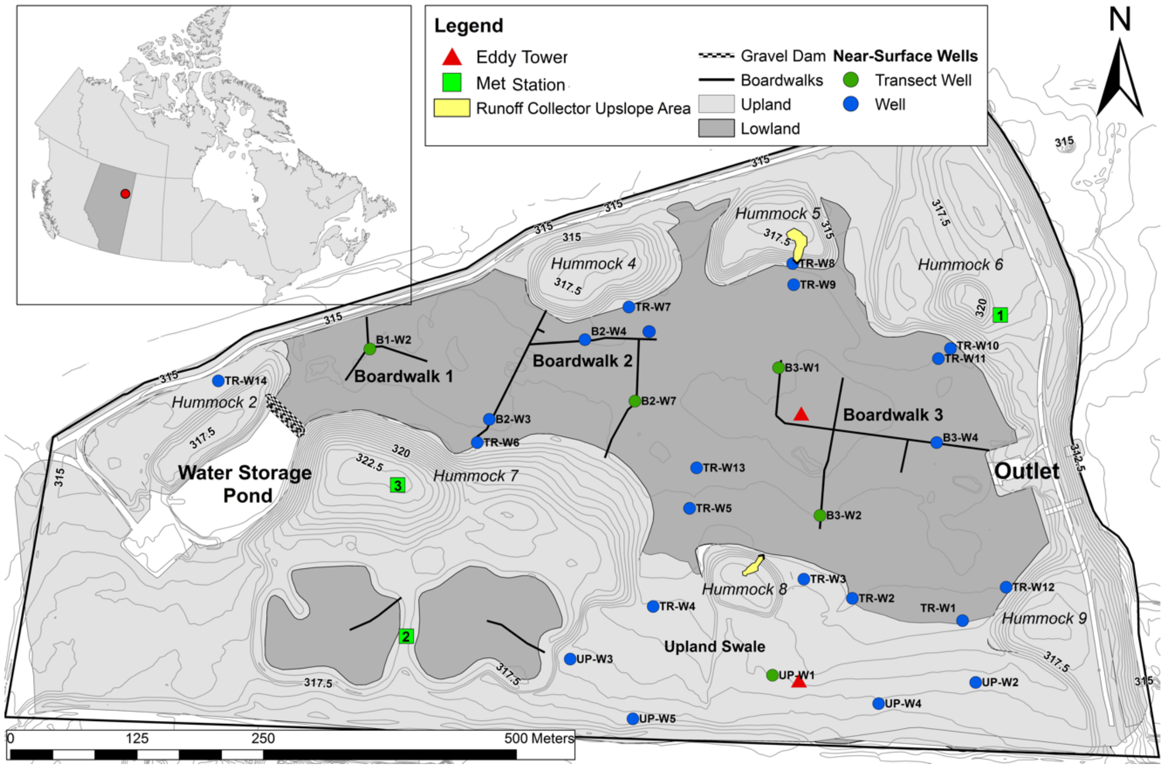


Figure 2.1. Instrumentation of the SFW. Dark and light grey areas represent the wetland and uplands, respectively. The margins are considered to be the area along the upland-wetland border.

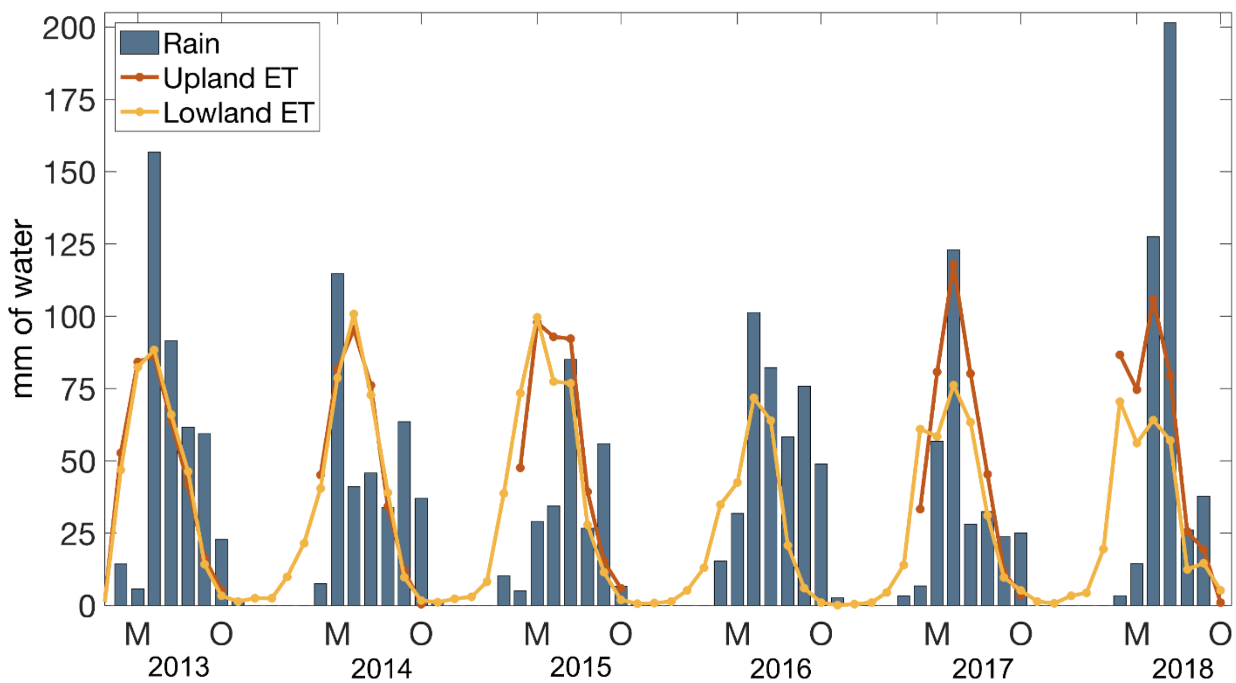


Figure 2.2. Monthly rain and evapotranspiration (ET) from 2013 to 2018 for the lowlands (yellow) and uplands (red).

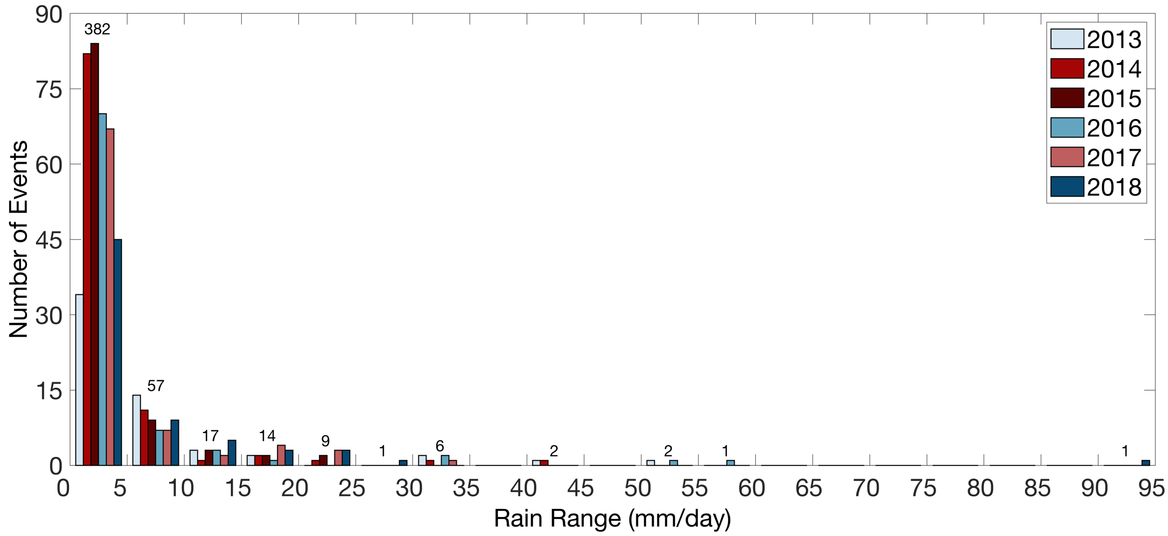


Figure 2.3. Frequency distribution of daily sum of rain events in 5 mm increments from 2013 – 2018 (based on the water year). The red and blue colouring highlight the dry and wet years, respectively where darker colours represent a greater dry or wet year. Numbers above each increment represent the total number of rain events of all years combined. Data is averaged from the three weather stations outlined in Figure 2.1.

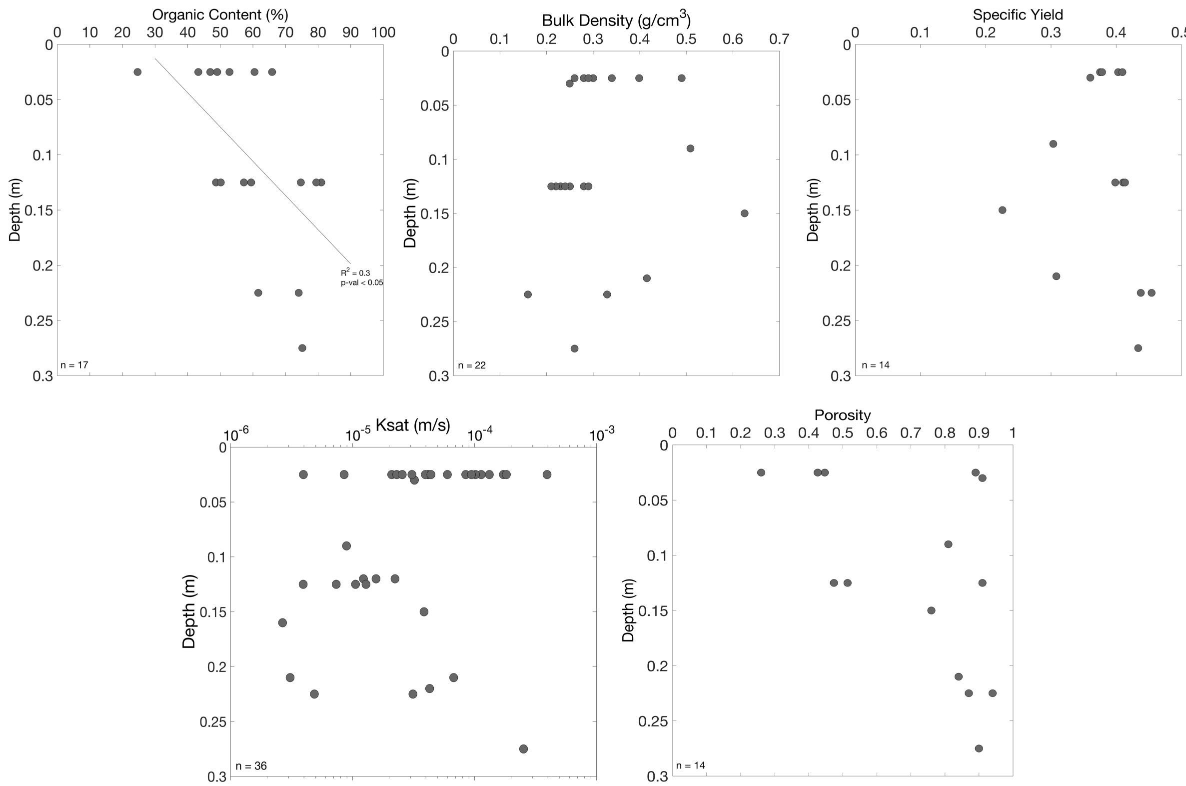


Figure 2.4. Hydrophysical properties of the peat profile. Properties with a significant relationship with depth have a line of best fit. Sample size (n) is noted in the bottom left corner of each figure.

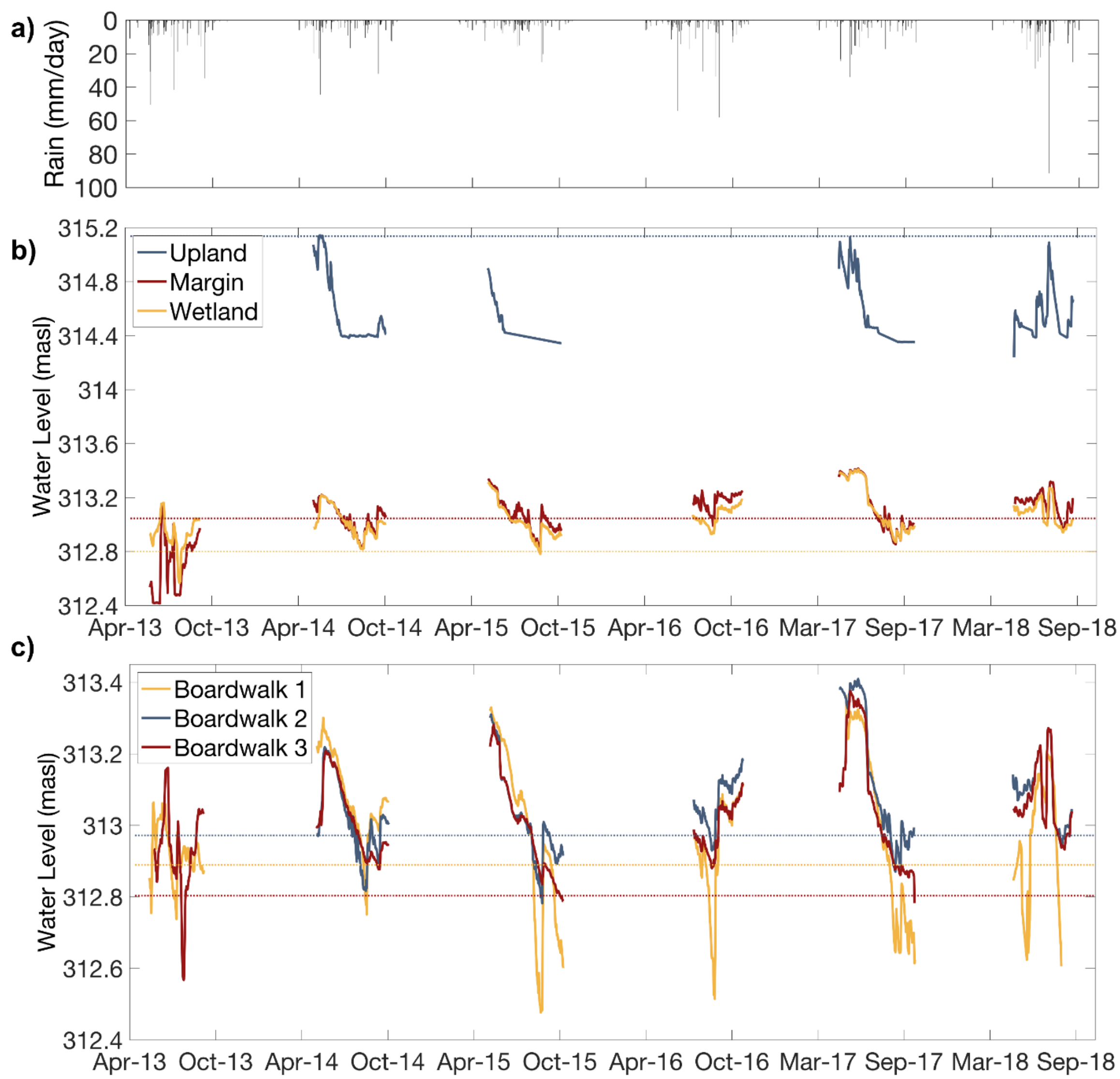


Figure 2.5. a) Rainfall hyetograph and water table position (masl) of the b) landscape transect (uplands, margins and wetland) and c) wetland transect (Boardwalks 1, 2 and 3) during April to October from 2013 – 2018. One well was selected to represent each landscape unit. The dashed lines represent the ground surface and is coloured to match its respective well. The sites selected to represent the wetland, margins, uplands, B1, B2 and B3 were B3W1, B3W2, UPW1, B1W2, B2W7 and B3W1, respectively.

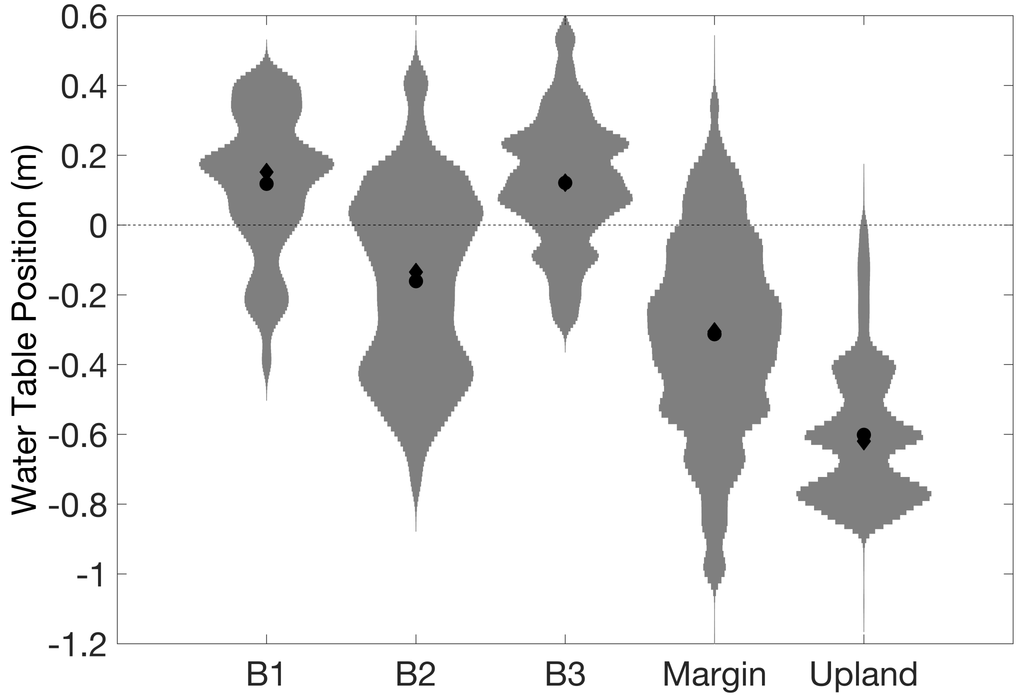


Figure 2.6. Violin plots of water table position at all sample locations at Boardwalks 1 (B1), 2 (B2), 3 (B3), margins and uplands during 2014 – 2018 under a more natural management regime (minimal pump use). The mean and median values are represented by the black circles and diamonds, respectively. Values are relative to the ground surface (dashed lined at zero). The sample size is indicated by the numbers above the x-axis labels.

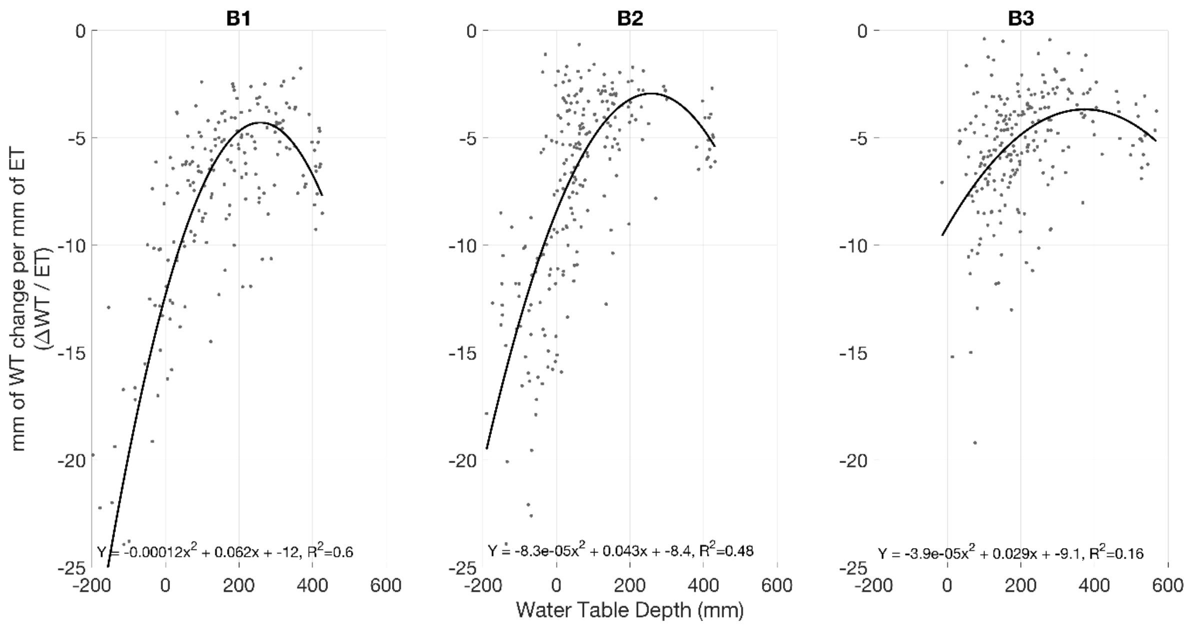


Figure 2.7. Scatter plots of the daily change in water table with respect to daily evapotranspiration (ΔWT:ET both in mm). Negative and positive water table depths indicate below and above surface positions, respectively. The relative influence of ET of the WT is greater with increasing depth to the water table.

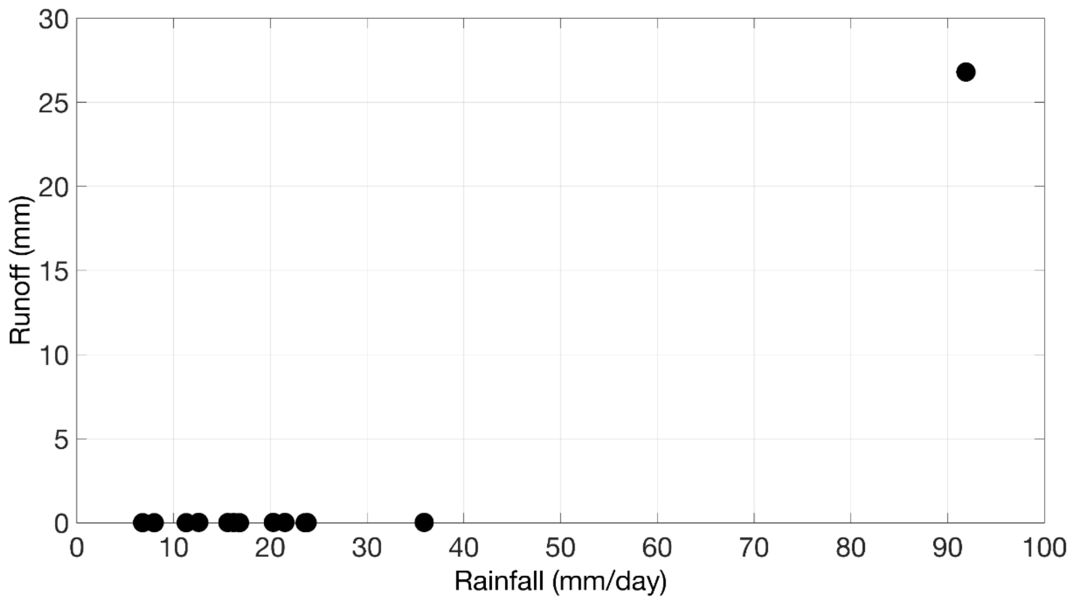


Figure 2.8. Rainfall-runoff relationship from the hummock hillslopes in 2018 (the only year recorded) from the runoff collectors.

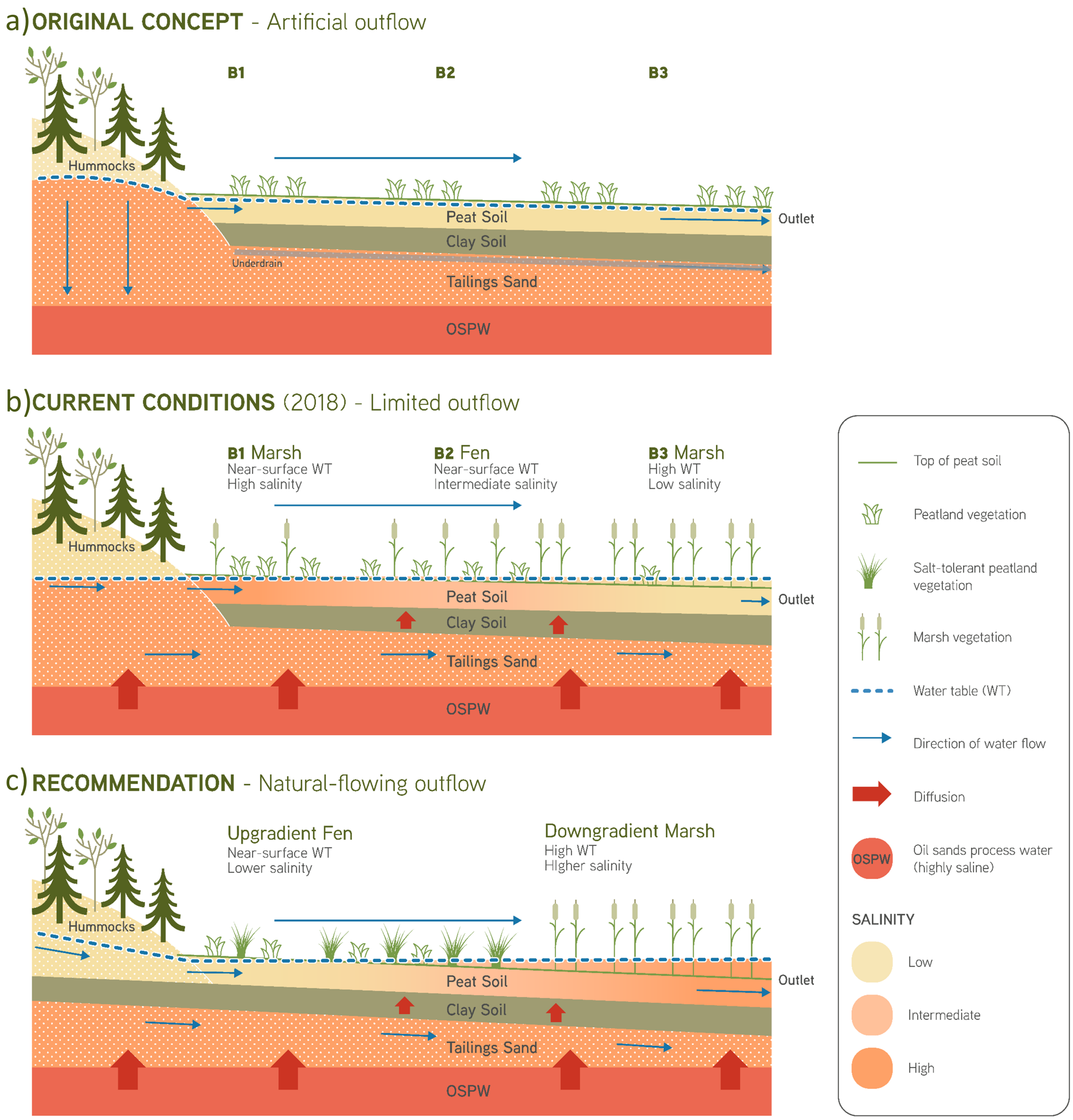


Figure 2.9. Conceptual diagram of the a) original design, b) current conditions (2018) and c) recommendations for future design strategies. B1, B2 and B3 represent boardwalks 1, 2 and 3 respectively currently present in SFW.

CHAPTER 3

THE ROLE OF SNOW PROCESSES AND HILLSLOPES ON RUNOFF GENERATION IN PRESENT AND FUTURE CLIMATES IN A RECENTLY CONSTRUCTED WATERSHED IN THE ATHABASCA OIL SANDS REGION

This chapter quantifies the snow component of the overall site water balance (in Chapter 2) as well as examines the role of winter processes on current and future hydrological conditions in a potentially warming climate.

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ABSTRACT

Mine reclamation in the Athabasca oil sands region Canada is required by law, where companies must reconstruct disturbed landscapes into functioning ecosystems such as forests, wetlands and lakes that existed in the Boreal landscape prior to mining. Winter is a major hydrological factor in this region as snow covers the landscape for 5 to 6 months and is ~25 % of the annual precipitation, yet few studies have explored the influence of winter processes on the hydrology of constructed watersheds. One year (2017-2018) of intensive snow hydrology measurements are supplemented with six years (2013-2018) of meteorological measurements from the constructed Sandhill Fen Watershed to: 1) understand snow accumulation and redistribution, snowmelt timing, rate and partitioning, 2) apply a physically-based model for simulating winter processes on hillslopes and 3) evaluate the impact of soil prescriptions and climate change projections on winter processes in reclaimed systems. The 2017-2018 snow season was between November and April and SWE ranged between 40-140 mm. Snow distribution was primarily influenced by topography with little influence of snow trapping from developing vegetation. Snow accumulation was most variable on hillslopes and redistribution was driven by slope position, with SWE greatest at the base of slopes and decreased towards crests. Snowmelt on hillslopes was controlled by slope aspect, as snow declined rapidly on west and south-facing slopes, compared to east and north-facing slopes. Unlike results previously reported on constructed uplands, snowmelt runoff from uplands was much less (~30 %), highlighting the influence of different construction materials. Model simulations indicate that antecedent soil moisture and soil temperature have a large influence on partitioning snowmelt over a range of observed conditions. Under a warmer and wetter climate, average annual peak SWE and snow season duration could decline up to 52 % and up to 61 days, respectively, while snowmelt runoff ceases completely under the warmest scenarios. Results suggest considerable future variability in snowmelt runoff from hillslopes, yet soil properties can be used to enhance vertical or lateral flows.

3.1 INTRODUCTION

Surface mining in the Athabasca oil sands region (AOSR) of northern Alberta, Canada, has disturbed >900 km^2 of forests and wetlands in the Western Boreal Plains (WBP), as entire landscapes are removed and excavated up to 75 m to access the bitumen-rich McMurray formation (Canada's Oil & Natural Gas Producers, 2017). These mining activities have permanently altered pre-disturbance vegetation communities, carbon storage capacity (Rooney et al., 2012), soil layers (Nwaishi, Petrone, Price, Ketcheson, et al., 2015) and hydrologic functioning of these natural systems (Price et al., 2010; Rooney et al., 2012; Trites & Bayley, 2009b). In addition to the physical disturbance, excessive salts are ubiquitous in the post-mining landscape from the tailings material that is used to backfill open pits (Chapter 4; Kessel et al., 2018; Kessler et al., 2010; Simhayov et al., 2017). As part of their legal requirements, oil companies must reclaim disturbed landscapes into a mosaic of functioning ecosystems including wetlands, peatlands, forests and pit lakes (OSWWG, 2008) however, reclamation involves the complete reconstruction of these ecosystems, which is complicated given the regional sub-humid climate, limited research and knowledge, ubiquitous salinity and the unprecedented scale of the disturbance (Daly et al., 2012; Elshorbagy, Jutla, Barbour, & Kells, 2005; Wytrykush et al., 2012). Currently, only two watersheds that include peatlands have been constructed and instrumented which vary considerably in design (Daly et al., 2012; Ketcheson et al., 2016a; Wytrykush et al., 2012).

The AOSR has a sub-humid climate which controls water availability primarily through precipitation and evapotranspiration (Devito et al., 2012). Most of the annual precipitation in the WBP occurs during the growing season when evapotranspiration is highest, and excess of soil water is minimal (Ferone & Devito, 2004). Only ~25% of annual precipitation falls as snow (Environment Canada, 2020), yet snowmelt produces a large, but variable, surplus of water in a relatively short amount of time before the growing season when evapotranspiration is low (Devito, Creed, & Fraser, 2005; Devito et al., 2012). Autumn rainfall events and snowmelt are often the only source to replenish groundwater stores depleted by evapotranspiration during the previous growing season (Redding & Devito, 2011; Smerdon et al., 2008). Additionally, snowmelt water is a critical source of freshwater to constructed watershed ecosystems that can have highly saline waters with elevated sodium concentrations which originate from the construction materials used in reclamation (Chapter 4). Winter processes therefore play an important role in reclamation hydrology as they influence snow storage, redistribution, melt and water transmission (Price, 1987; Price & Fitzgibbon, 1987; Rouse, 2000; Woo & Winter, 1993). Variations in blowing snow, sublimation, interception, frozen ground and vegetation communities among landscapes impact snow accumulation and melt regimes (Pomeroy et al., 2006, 1998; Price & Fitzgibbon, 1987; Spence & Woo, 2003; Whittington, Ketcheson, Price, Richardson, & Di Febo, 2012; Woo & Marsh, 2005; Woo & Winter, 1993). Soil ice formation is another critical component of winter hydrology as frozen soil water is unavailable in the early growing season for evapotranspiration (Devito et al., 2012; VanHuizen, Petrone, Price, Quinton, & Pomeroy, 2020). In addition, frozen ground can enhance snowmelt runoff to lowlands that may rely on this seasonal input of water prior to the growing season (Devito et al., 2012; VanHuizen et al., 2020).

Surface overland flow typically makes up a small component of the annual water balance in the undisturbed WBP due to the large storage capacity of soils even when frozen (Devito, Creed, & Fraser, 2005; Devito et al., 2012; Ferone & Devito, 2004; Redding & Devito, 2011). At the regional scale, a recent study reported that peatland-swamp ecosystems across the WBP are the primary producers of runoff (3 – 27 % of annual precipitation) and downgradient water transfer, while open-water wetlands and forestlands act as water sinks (Devito et al., 2017). Drivers of runoff generation are variable among WBP ecosystems and include snowpack depth (Devito, Creed, & Fraser, 2005; Ferone & Devito, 2004), concrete frost development (Redding & Devito, 2011), depth to confining layer and soil storage (Devito, Creed, & Fraser, 2005), snowmelt rate, autumn antecedent moisture conditions (Ireson et al., 2015; Redding & Devito, 2011) and to a lesser extent, vegetation canopy and soil type (Redding & Devito, 2011). Surface runoff also varies across the reclaimed landscape, where hillslopes with low antecedent soil water content and the presence of macropores only produces runoff once the high soil storage is exceeded (Kelln, Barbour, & Qualizza, 2009; Shurniak & Barbour, 2002), while hillslopes designed with a high antecedent soil moisture and low infiltration capacity (with finer textures) produces a high volume of surface runoff during snowmelt and heavy rain events (Ketcheson & Price, 2016c, 2016b). Soil materials used in reconstruction are harvested from the pre-mining landscape and can include coarse textured, fine textured and veneer-type soils, all of which represent different hydrologic response areas due to their unique soil properties (Devito et al., 2012). These soils are prescribed to reconstructed ecosystems based on desired hydrologic behaviour and storage capacity of the system and therefore construction practice can play a large role in dictating surface runoff from hillslopes in reclaimed watersheds in the AOSR (Kelln et al., 2009; Ketcheson & Price, 2016c; Shurniak & Barbour, 2002).

Modelling hydrological processes in constructed landscapes, particularly those that include wetlands, is challenging because natural peatlands are complex heterogeneous systems that provide many key ecosystem functions and are resilient to change through feedback mechanisms (Waddington et al., 2015). Many studies have been successful at using models to highlight and explain hydrological processes observed in WBP, as well as potential ecosystem responses to disturbance or stress (Hilbert, Roulet, & Moore, 2000; Smerdon et al., 2007; Thompson et al., 2015). Modelling constructed systems in the AOSR presents an additional challenge because of the dynamic nature of these systems within the first several years of development. There is no consistency of models used to represent constructed systems in the AOSR, as models have been chosen and developed to examine a specific process or operational questions. Of the models used, many have focused on subsurface water fluxes (Carrera-Hernández, Smerdon, & Mendoza, 2012; Dobchuk, Shurniak, Barbour, Kane, & Song, 2013; Huang, Barbour, & Carey, 2015; Kelln et al., 2007, 2009; Lukenbach et al., 2019; Shurniak & Barbour, 2002; Sutton & Price, 2019) and/or inorganic solute transport (Huang, Hilderman, & Barbour, 2015; Kelln et al., 2008, 2009) with little focus on surface-atmosphere interactions, even though these are the dominant water fluxes in the WBP (Devito et al., 2012; Strilesky et al., 2017). There are limited studies that focus on overall system performance that incorporate multiple components of the hydrological cycle in addition to subsurface processes (Elshorbagy et al., 2005; Elshorbagy, Jutla, & Kells, 2007; Keshta, Elshorbagy, & Carey, 2009), but few have been conducted on constructed peatlands or focus specifically on winter processes. An additional challenge in building landscapes in the AOSR is the uncertainty associated with climate change, as the WBP is expected to experience increases in temperature and precipitation in the coming century (Ireson et al., 2015; Meehl et al., 2007). We have a limited understanding of the role of climate change on constructed systems in the AOSR that already face challenges associated with water supply and quality. Such influences are not fully understood but are important for reclamation and closure plans as they could significantly impact the outcome of constructed systems in the long-term.

Constructed ecosystems provide a unique opportunity to study hydrologic functioning and development as these systems mature and develop in a continuously changing landscape and will likely differ considerably from their natural analogues (Elshorbagy et al., 2005). Despite the importance of winter processes, few studies (Kelln et al., 2009; Ketcheson & Price, 2016c; Meier & Barbour, 2002) have examined their influence on system hydrology of constructed watersheds in the AOSR. In addition, a range of soil prescriptions are used in reclamation and snowmelt partitioning will vary greatly depending on the soil properties which will influence water availability and down-gradient water transfer prior to the growing season. As up to 4800 km^2 of the AOSR is suitable for surface mining, understanding how winter processes influence system hydrology and water availability is critical for the long-term success of current and future constructed ecosystems. To address this, the objectives of this research are to use data from a constructed upland-wetland watershed to: 1) quantify snow accumulation and redistribution, melt timing, rate and partitioning, 2) apply a widely used physically-based model for simulating winter processes and runoff generation on upland hillslopes, and 3) evaluate the impact of different reclamation soils and climate projections on winter processes. This information will help guide future landscape construction practice and provide information on the potential long-term importance of winter processes in a changing climate.

3.2 SANDHILL FEN WATERSHED

The Sandhill Fen Watershed (SFW) is a constructed peatland-upland system in the northwest corner of East-In-Pit, a previously mined area (1977-1999), and is part of Syncrude Canada Ltd.'s Base Mine (57°02'N, 111°35'W) which is approximately 42 km north of Fort McMurray, Alberta. This region is situated in the Western Boreal Plains ecoregion of western Canada which is characterized by a cold and sub-humid climate where mean annual potential evapotranspiration (607 mm) is greater than mean annual precipitation (456 mm) (Environment Canada, 2020). Climate normals (1971 – 2010) for the Fort McMurray Airport indicates that most of the annual precipitation falls as rain (342 mm) and the remainder falls as snow (113 mm of water equivalent) with a regional average peak snow depth of 31 cm (February). Winter temperatures are typically coldest in January averaging -17.4 ± 5 °C and averages -12.1 °C from November to March. The annual mean temperature is 1 ± 1.3 °C (Environment Canada, 2020).

Over the course of four years (2009 – 2012), East-In-Pit was filled with 35 m of inter-bedded composite tailings and tailings sand layers followed by a 10 m tailings sand structural cap as part of the reclamation strategy. The SFW was then constructed on top of these materials and is 52 ha total with a 17 ha fen wetland, 35 ha upland including 20 ha of upland hillslopes (referred to as hummocks) (Wytrykush et al., 2012). Due to the relatively low grade of the wetland and upland areas, (0.1-0.5 %), the hummocks were constructed to create distinct groundwater recharge areas as well as add topographic variation to the watershed. The hummocks were constructed of tailings sand that was mechanically placed and capped with Pleistocene fluvial sand (see Table 3.1 for specific soil properties). A peat-mineral mix was placed in the lower upland areas while 0.5 m of clay followed by 0.5 m of donor peat materials were placed in the wetland area. Details of watershed design, construction materials and soil stratigraphy are outlined in Chapter 4. The SFW was vegetated in 2011 with species native to this region (Nicholls et al., 2016; Wytrykush et al., 2012), but species composition has changed considerably throughout the watershed with time (Vitt et al., 2016b). Unique to the SFW, inflow and outflow can be managed via a pump system that was installed during construction. Fresh water can be supplied to the Water Storage Pond (Figure 3.1) from a near-by natural lake (Mildred Lake) which flows westward through the wetland area towards the outlet where water can be pumped out of the SFW and back into East-In-Pit. It should be noted that aside from deeper groundwater flow paths, the outflow of surface and near-surface water can only occur when the outflow pump is activated. The intention of these pumps was to mitigate the elevated salinity levels from construction materials by providing fresh water and flushing out saline water in the first few years while the system established within the disturbed landscape. These pumps have remained largely off since 2013 (Chapter 4; Nicholls et al., 2016).

3.3 METHODS

3.3.1 Meteorological measurements

Air temperature, windspeed and direction, relative humidity, short, and long-wave radiation were measured at each of the three meteorological towers (Figure 3.1) on SFW. Instrument details can be found in Nicholls et al. (2016). Precipitation was measured with a tipping bucket rain gauge (Model CS700, Campbell Scientific Inc. (CSI), Logan UT, USA) for rainfall and a CSI *CS725* to measure snow water equivalent (SWE) as well as a CSI SR50A sonic ranger to monitor snow depth. All measurements were recorded on an hourly basis with CSI CR1000 data loggers since 2013. In addition, one eddy covariance tower that was instrumented in 2013 measured turbulent fluxes year-round (Clark et al., 2019; Nicholls et al., 2016).

3.3.2 Snow accumulation and melt measurements

Intensive field measurements were made during the 2018 winter season (November 2017 – April 2018) and included snow survey transects starting on 15 February 2018 throughout the different landscape units within SFW including the wetland, uplands and hummocks (Figure 3.1). Snow survey measurements included snowpack depth every 10 m using an avalanche probe, snow water equivalent (SWE) every 30 m using a Mount Rose corer and ground ice presence every 30 m using a metal rod. To supplement SWE measurements, snow pits were completed in the upland and wetland to quantify snowpack density using standard approaches (Adams & Barr, 1974). SWE measurements on the hummocks were divided into slope position and aspect to assess differences in snow accumulation patterns.

Slope position was assigned using visual observations and contour lines where the bottom slope was within the first meter of the hillslope, the crest was the entire flat portion at the top and mid-slope was the remainder of the hillslope area in between. Slope aspect was assigned using the Aspect tool in the Spatial Analyst toolbox in ArcGIS. All statistical analysis of snow accumulation trends was completed using the R language for statistical computing (R Core Team, 2018).

To quantify and partition snowmelt from hummock hillslopes, surface runoff collectors were constructed on the north- and south-facing slopes that drained into v-notch bucket weirs with a pressure transducer. Runoff collectors, modified from Ketcheson and Price (2016b), were constructed prior to melt by digging shallow trenches (~15 cm) that extended out from a bucket weir in a "V" shape where each arm of the "V" was approximately 3-4 m. Flexible, plastic garden edging was sealed into the bottom of the shallow trenches using hydraulic cement to ensure meltwater could not flow across the trenches. At the base of the "V", a plastic eavestrough was used to direct surface runoff into the bucket weirs and was also sealed with hydraulic cement to limit leakage (Figure 3.2). Large patches of ground that were cleared for runoff collector construction were filled with snow after construction was completed to limit any change to ground surface albedo. A pressure transducer was installed in each bucket weir to continuously measure water level and discharge during melt. The pressure transducer was placed in a small latex bag filled with antifreeze to prevent any damage from water freezing which did not affect measurements (Figure 3.2). Discharge measurements for each bucket weir were made to create independent rating curves for each runoff collector. A one-meter resolution LiDAR digital elevation model

(DEM) of the study area was used to determine the drainage direction and delineate the contributing area for each runoff collector using the Hydrology toolset in ArcGIS. Calculated contributing areas of runoff collector one (RC1) and two (RC2) were 331 m^2 and 160 m^2, respectively (Figure 3.1). The horizontal and vertical accuracy of the LiDAR data was 30 cm and 15 cm, respectively at 95% confidence through comparison to independently surveyed ground points.

Snow surveys were also completed on older reclaimed sites on the Syncrude Canada Ltd. property that are in more advanced stages of regrowth to compare snow accumulation and melt patterns using the same methods as above. Sites were within 5 km from the SFW and included two mature reclaimed forests, South Bison Hill (~20-year-old mature aspen/white spruce stand) and Jack Pine (~30-year-old jack pine stand) have delayed melt compared with the newly reclaimed Coke Beach site (~10-year young aspen) and the SFW (~6 years).

3.3.3 Hydrophysical properties

Soil properties of hummocks have been measured as part of the active research at the SFW (2012 – 2018) and are summarized in Table 3.1. Both in-situ field and laboratory methods were used to estimate soil properties. Soil pits were constructed on hummocks to collect samples at various depths to capture all soil materials used in hummock construction. Saturated hydraulic conductivity (Ksat) values are averaged values from several data collection methods including in-situ using the Guelph Permeameter and single ring infiltrometer as well as laboratory methods using the KSAT and Hyprop instruments from METER Group Inc. Ksat estimation from the Guelph Permeameter followed standard procedures (Soilmoisture Equipment Corp., 2012). Single ring infiltrometers were used to estimate field Ksat and infiltration capacity. Single ring infiltrometers were installed into the ground surface to depths of at least 1 cm and constant head tests were conducted until a steady state infiltration rate was reached. Ksat was assumed to be equal to the steady state infiltration capacity reached during each test. Porosity is automatically calculated using the Hyprop software and specific yield can be estimated from the Hyprop data by the difference between the saturated volumetric water content (VWC) and the VWC at 330 mb which is equivalent to the field capacity of the sample. Samples were analyzed for bulk density using standard methods (Freeze & Cherry, 1979) with the exception that the samples were oven-dried at 80 °C to limit any loss of organic matter. Organic matter content was determined via loss on ignition (LOI) where samples were placed in a muffle furnace at 550 °C for four hours. LOI was calculated as the difference between the pre- and post-muffle furnace weight divided by the pre-muffle furnace weight. The remainder of soil properties in Table 3.1 were extracted from NorthWind Land Resources Inc. (2012).

3.3.4 Model setup and parameterization

The cold regions hydrological model (CRHM) (outlined in Pomeroy et al., 2007) provides a platform that can be used to assess potential hydrological responses of constructed peatlands in the AOSR as it continues to develop and in response to future climate change. CRHM is a physically based hydrological model that can simulate hydrological processes in a modular fashion and is particularly strong in representing winter hydrological processes (Pomeroy et al., 2007). Modules within CRHM are selected by the user based on what simulated processes are needed which cover a wide range of hydrological processes. CRHM can assign linked algorithms that simulate hydrological processes to different hydrological response units (HRUs) and can route water between HRUs through pathways such as blowing snow, overland flow, groundwater flow when specific thresholds for that unit are exceeded. This unique feature allows HRUs to be a series of cascades across the landscape (Pomeroy et al., 2007; Rasouli, Pomeroy, Janowicz, Carey, & Williams, 2014). Because CRHM is physically based and is a purpose-built model based on process understanding, typical calibration methods used in many other models are omitted. Divergence between model and observations are diagnosed based on process understanding and model structures changed until suitable dynamics are achieved (Cordeiro, Wilson, Vanrobaeys, Pomeroy, & Fang, 2017; Rasouli, Pomeroy, & Whitfield, 2019). CRHM has been successfully applied to a variety of catchments including the Canadian Prairies (Fang & Pomeroy, 2007, 2009; Fang et al., 2010; Shook, Pomeroy, Spence, & Boychuk, 2013), agricultural catchments (Cordeiro et al., 2017), arctic regions (Krogh & Pomeroy, 2018; Quinton & Baltzer, 2013; Quinton, Bemrose, Zhang, & Carey, 2009; Quinton, Carey, & Goeller, 2004), peatlands (Knox, Carey, & Humphreys, 2012; Quinton & Baltzer, 2013), and mountainous regions (Ellis, Pomeroy, Brown, & MacDonald, 2010; Fang et al., 2013; Pomeroy, Fang, & Ellis, 2012; Weber et al., 2016).

Several physically based modules were used (detailed in Pomeroy et al. (2012) and Fang et al. (2013)) to examine the hydrological controls on winter processes in the SFW and include the following:

1) Observation module: imports and reads observed meteorological data which include continuous, hourly time-steps of air temperature, precipitation, relative humidity, wind speed, incoming short-wave radiation and incoming longwave radiation which are used as forcing data to drive other CRHM modules.

2) Radiation module: global radiation, direct and diffuse shortwave radiation based on site latitude, elevation, slope and azimuth (Garnier & Ohmura, 1970). Radiation from this module is used in the sunshine hour module, energy budget snowmelt module and net all-wave radiation module.

3) Sunshine hour module: used shortwave radiation and maximum sunshine hours to estimate total sunshine hour. Estimates from this module are used in the energy-budget snowmelt module and net all-wave radiation module.

4) Slope correction for the shortwave radiation module: the incoming shortwave radiation at a level surface to estimate the incident shortwave radiation on a slope. The module uses the measured incoming shortwave radiation from the observation module as well as the calculated direct and diffuse solar radiation from the radiation module to calculate the adjustment ratio for the shortwave radiation on a slope.

5) Longwave radiation module: uses the measured shortwave radiation to estimate incoming longwave radiation (Sicart, Pomeroy, Essery, & Bewley, 2006), which is used in the energy-balance snowmelt module.

6) Albedo module: snow albedo is estimated for the duration of winter as well as the melt period. This module also indicated the beginning of melt which is used in the energy-balance snowmelt module.

7) SnobalCRHM: designed for deep alpine snowpacks (Marks, Domingo, Susong, Link, & Garen, 1999), SnowbalCRHM simulates the mass and energy balance of the snowpack to estimate snowmelt by calculating the energy balance of radiation, sensible and latent heat, ground heat flux, advection from rainfall and change in internal energy for two layers of the snowpack (an active top layer and a lower layer).

8) frozenAyers: uses Ayers (1959) infiltration to estimate unfrozen soil infiltration and Zhao & Gray (1999) to estimate frozen soil infiltration and subsequent surface runoff. The soil moisture balance module is linked to both infiltration algorithms. Surface runoff occurs when snowmelt or rainfall exceeds the infiltration rate.

CRHM was initially set up for the 2018 winter season using detailed observations outlined above as one hummock HRU to evaluate snow accumulation and movement of meltwater from hillslopes to the lowlands. The lowlands were not simulated. Parameters were selected from direct observation where possible, and a suitable model performance was obtained (Table 3.2). CRHM was then run for the five previous years of observation to provide simulations for five winters under current climate conditions (T0-P1). Following this, CRHM was used to evaluate the potential impacts of climate change on these five years. Nine future climate scenarios for the WBP were selected from the Intergovernmental Panel on Climate Change report (Meehl et al., 2007) and applied to CRHM in a delta-change approach though systematically changing temperature, precipitation, and both, based on ensemble averages. In CRHM, temperature is an additive change while precipitation is a multiplicative change that only affects days with existing precipitation (a value of 1 represents current precipitation conditions). Model results are grouped based on increases in temperature (T) and precipitation (P) from their baseline temperature (0 °C, and precipitation (a value of 1 indicates no change in precipitation). First, air temperature was increased for 5 years of winter simulation by 2, 4 and 6 °C (T2-P1, T4-P1 and T6-P1, respectively). Second, under current temperature, days with precipitation had volume increases by factors of 1.15, 1.2 and 1.3 (T0-P1.15, T0-P1.2 and T0-P1.3, respectively), based on the IPCC predicted range of increased future precipitation (Meehl et al., 2007). Finally, to assess the impacts of both a warmer and wetter future, precipitation was increased by a factor of 1.2 in addition to the three temperature change scenarios (T2-P1.2, T4-P1.2 and T6-P1.2). CRHM was also used to test the influence of different soil parameters on the partitioning of snowmelt (under current climate conditions only). Soil parameters within the frozenAyers module that were evaluated were soil temperature at the onset of snowmelt in the top 40 cm, the initial soil saturation (VWC/porosity), and soil texture. Values were taken from the observed variability in the six years of data and directly measured soil physical properties.

3.4 RESULTS

3.4.1 Snow accumulation and distribution

The average peak watershed SWE was 89 ± 17 mm (n=278) during the 2017-2018 winter season with small but significant differences among landscape units determined by a simple Wilcoxon rank sum test. The wetland, uplands and hummocks had average peak SWE of 94 ± 13 mm, 90 ± 16 mm and 85 ± 20 mm, respectively (Figure 3.3a), where the hummocks exhibited the most variability as a result of slope position and aspect (Figure 3.3b). Only the hummocks and uplands were statistically similar to one another ($p=0.07$). The smallest snowpack was observed on the crest of the hummocks (71 ± 16 mm), followed by the mid-slope (88 ± 17 mm) and lower-slope (94 ± 26 mm) positions. The crest was significantly different than the mid and lower-slope positions ($p<0.05$) while the mid and lower slopes were similar ($p=0.3$). SWE accumulation and distribution were statistically similar among aspects ($p>0.05$) except between north and east-facing slopes ($p=0.04$). Average SWE on south-, east-, north- and west-facing slopes averaged 91 ± 22 mm,99 ± 17 mm, 84 ± 17 mm and 90 ± 17 mm, respectively.

3.4.2 Snowmelt and melt partitioning

The SWE of all landscape units remained relatively stable until 12 March when SWE began declining during the first melt event which continued to decline to mid-April when the second (and final) melt event depleted the remainder of the snowpack (Figure 3.4). SWE decline was similar among all sites until the first melt event on 12-15 March. Once this melt started in mid-March, the hummock SWE was the most variable (Figure 3.4c) as melt among aspects was considerably different, where the south and west-facing slopes had faster melt rates than the north and east-facing slopes (Figure 3.4d). West-facing slopes had the largest SWE decline of ~25 mm whereas all other sites had a SWE decline ≤ 10 mm. While no snow surveys were conducted between 15-March and 9-April, SWE decline was the largest at South and West aspects during this time period which can be attributed to the warmer temperatures in mid-April as temperatures remained well below zero for the remainder of March. The second melt event began mid-April when air temperatures were above zero during most days. The snowpacks on the south and west-facing slopes lost the majority of SWE by the second melt event in April whereas the remainder of the sites had approximately half of their snowpack left during the second melt event (12-24 April) (Figure 3.4).

To evaluate meltwater partitioning from hillslopes to the wetland area, two runoff collectors were constructed on the north- and south- and facing slopes (Figure 3.1). Snowmelt on the north-facing slope was not measured in March as it was not fully constructed, however, the snowpack stayed deep and experienced minimal melt during the mid-March warm period. Snowmelt began on the south-facing slope on 15 March, with instantaneous discharge peaking the next day at ~190 cm^3/s and total cumulative runoff reaching 10 mm in two days (Figure 3.5a). Following this, runoff occurred from the south-facing slope for the next five days between ~10:30-17:30 each day and in total 15 mm was collected before this hillslope was snow-free on 23 March. In contrast, the north-facing slope did not begin to generate runoff until daily air temperatures were consistently near zero in mid-April; snowmelt began on 12 April with peak instantaneous discharge of 79 cm^3/s on 14 April (Figure 3.5b). The first several days of melt were the most productive, yielding between 2 and 5 mm over the first four days with a shorter duration of daily melt (typically four hours in duration through mid-day). Runoff then declined as the slope became gradually snow-free by 24 April and yielded a total of 23 mm for the entire period. While there were no runoff collectors for upland sites, on-site observations confirm that surface runoff was only generated from hummocks as opposed to low-gradient areas surrounding the lowland.

The average SWE on the south- and north-facing slope prior to melt was 50 mm and 90 mm, respectively, and surface runoff totaled 15 mm and 23 mm, respectively, and their corresponding runoff ratios were 0.31 and 0.26, respectively. Sublimation during the two melt periods was calculated from the eddy-covariance derived latent heat data and amounted to 0.44 mm from 15-21 March (south-facing melt) and 2 mm from 12-24 April (north- facing melt). Infiltration into the frozen ground was calculated as the residual after accounting for runoff and sublimation which was 34 mm and 65 mm for the south- and north-facing slopes, respectively. When all hillslopes feeding the wetland are considered and measurements are scaled, snowmelt surface runoff contributed ~11 mm to the wetland area of SFW.

3.4.3 Model simulation

3.4.3.1 Influence of soil parameters on snowmelt partitioning

To evaluate the role of soil properties and antecedent conditions on runoff generation from the hummocks, parameters within the frozenAyers module, which is derived from the parametric equations of heat and mass transfer in Zhao and Gray (1999), were adjusted based on field and laboratory observations. While the sensitivity of melt partitioning largely reflects parameter assignments in the model equations, it provides guidance on the expected influence of soil conditions on runoff generation from hillslopes based on the ranges of observed values.

Three key parameters that influence frozen soil infiltration are soil texture, the degree of initial soil saturation and ground temperature. Soil texture is important in a reclamation context as different textures are used in various landforms which will eventually be integrated into the closure landscape. The influence of texture on the partitioning of snowmelt is complex as liquid and frozen water influence capillary pressure and permeability. There also exists a relation between texture and soil temperature due to unfrozen water content and latent heat effects. Soil texture was changed within CRHM to silt and clay from the base 2018 winter simulation (sand, 34 mm of runoff). As expected, finer soil texture resulted in greater runoff, with silt loam (37 mm) and clay (41 mm) being 8 and 17 % greater than sand, respectively. It is important to note that degree of soil saturation was not adjusted (Si=0.2), and changes in runoff are from textural differences alone. However, it is expected that finer textured soils would have greater antecedent wetness and therefore greater runoff.

The degree of initial soil saturation (Si), the upper soil volumetric water content at freezeback divided by porosity, influences runoff as soils with greater pore space filled with frozen water restrict infiltration and promotes runoff. In fall 2017, soils froze in late October at a relatively low moisture content in the near surface profile (VWC < 5 % and Si = 0.20). This moisture content was the lowest of any years observed, yet pre-freezeback values as high as 0.5 occurred in some years. As with soil temperature, Si was varied within the range of the observed six-year record for the 2018 winter case. Runoff increased approximately linearly with increasing Si, and results indicate that within the range of observation, pre-freezeback soil moisture can more than triple the expected runoff for a given year (Table 3.3). For every 5 % increase in Si from 0.2, runoff increased ~12 mm with slightly greater increases when dry. It is important to note that 2018 was a high snow year with low initial Si.

Zhao and Gray (1999) use temperature within the top 40 cm at the onset of melt along with a nighttime refreezing parameter to establish the influence of soil thermal status on infiltration. The initial soil temperature (top 40 cm) during 2018 was -2 °C, which was used to run all climate change scenarios in Section 4.3.1. However, between 2013 and 2018 soil temperatures at the onset of melt were as low as -8 °C within the top 40 cm, which reflected cold years with limited snow. The 2018 winter simulation was run with temperatures from -0.5 °C to -8 °C, and results indicate a strong influence of temperature on runoff generation (Table 3.3). Runoff generation was zero at temperatures warmer than -1.5 °C and increased in a negative exponential manner with declining temperatures and at -8 °C, 94 mm of runoff was simulated compared with 34 mm as the base case.

3.4.3.2 Climate Change Scenarios

To test the accuracy of CRHM against observed data, simulations were set up under current climate conditions (T0-P1) for snowpack SWE from 2013-2018 (Figure 3.6a) and surface runoff during the 2017-2018 winter (Figure 3.6b). Surface runoff data only exists for 2017-2018 as the runoff collectors were installed in March 2018. In general, CRHM's simulations overestimated peak snowpack SWE, total surface runoff and underestimated mid-winter melt events (Figure 3.6). Annual peak simulated SWE was higher by an average of ~30 mm over simulated years, but the timing of snow accumulation and the start/end of the snow season is consistent with the observed data (Figure 3.6a) and over all simulated years matched the observed data within 10 days. Simulated surface runoff did not match observed runoff in terms of timing as simulated runoff occurs at the very end of the snow season while observed runoff had two distinct melt periods (Figure 3.6b). While total observed runoff (15 and 23 mm) was less than simulated runoff (34 mm), snowmelt partitioning between infiltration and runoff were similar based on the calculated runoff ratios of 0.3 and 0.2, respectively. Observed and simulated total runoff may differ because by the time the runoff collectors were installed, some of the snowpack had already melted, yielding a smaller total runoff. Additionally, CRHM underestimated the magnitude and duration of mid-winter melt events in all simulated years. For example, CRHM simulated an 8 mm melt event over five days in 2018 compared to observed where the mid-winter melt event caused snowpack SWE to decrease by 83, 45 and 51 mm, respectively for stations 1, 2 and 3 over ~12 days. It is unclear as to why this underestimation occurred, although it may be due to different locations between the SWE sensors and the meteorological data used to drive the model. Note that CRHM was not calibrated but driven by observation data and measured soil parameters (Table 3.1 and 3.2).

Modelled increases in temperature without an alteration in precipitation resulted in an expected reduction in simulated SWE, a shorter snow season and longer and larger mid-winter melt events (Figure 3.7a). Over the simulated years in scenarios T2-P1, T4-P1 and

T6-P1, peak snowpack SWE decreased by an average of 21, 35 and 42 %, respectively (Table 3.4), where deviation from current conditions (T0-P1) was amplified in winters with a higher peak SWE (Figure 3.7a). Duration of the snow season decreased by averages of 10, 37 and 61 days, respectively, some of which were the result of mid-winter melt that eliminated the entire snowpack by January or February in low snow years (i.e., 2013-2014, Figure 3.7a). The presence of mid-winter melt events is an emergent process with warming as its timing moves successively earlier into the winter season. Under these warming scenarios, average mid-winter melt duration increased from 5 days under T0-P1 to 7, 14 and 11 days, respectively for scenarios T2-P1, T4-P1 and T6-P1, respectively. Average snowmelt volume during the mid-winter melt also increased from 12 mm under T0-P1 to 26, 39 and 43 mm, respectively, all of which infiltrated into the frozen ground. Spring snowmelt runoff was small for all warming-only scenarios and decreased from a 5-year average of 11 mm under current conditions to 8, 1 and 0 mm, respectively for each of the two-degree temperature increases (Figure 3.8a) which decrease the average runoff ratio from 0.38 to 0.1, 0.01 and 0, respectively (Table 3.4).

Simulated increases in daily precipitation without changes in temperature increased peak SWE all years by averages of 12, 28 and 36 % in scenarios T0-P1.15, T0-P1.2 and T0-P1.3, respectively compared to initial conditions (Figure 3.7b). Snow season duration is similar to initial conditions and only increased by an average of 7, 8 and 8 days, respectively among precipitation scenarios. Unlike the temperature scenarios, there are no distinct mid-winter melt events among years and snowpack SWE continues to increase until the spring snowmelt event. Surface runoff of spring snowmelt increased from the 11 mm average for current conditions with greater daily precipitation, and averaged 20, 25 and 29 mm, respectively (Figure 3.8b) which kept runoff ratios similar to current conditions (0.2) and averaged 0.17, 0.18 and 0.21, respectively (Table 3.4).

An increase of both temperature and precipitation slightly lessened the effect of climate change on snowpack SWE while surface runoff exhibited the most variability among all climate scenarios. Peak SWE and snow duration under T2-P1.2 are similar to initial conditions but decrease considerably with continued warming with T4-P1.2 and T6-P1.2 (Figure 3.7c). Average peak snowpack SWE and snow season duration both decreased by an average of 0.2, 24 and 32 %, respectively and by an average of 1, 17 and 61 days, respectively in scenarios T2-P1.2, T4-P1.2 and T6-P1.2 (Table 3.4). Mid-winter melt events become increasingly prominent with warming and mid-melt duration increased to 6, 12 and 14 days from 5 days under T0-P1. Mid-winter melt volumes also increased under these conditions to 19, 40 and 44 mm, respectively from 5 mm under T0-P1. Surface runoff from end of season snowmelt increased initially in scenario T2-P1.2 to 20 mm from 11 mm in T0-P1, but then decreased to 10 and 3 mm in scenarios T4-P1.2 and T6-P1.2, respectively (Figure 3.8c). Runoff ratios were maintained at T2-P1.2 and averaged 0.21 but decreased with continued warming to 0.16 and 0.03, respectively indicating that more meltwater infiltrates under warmer scenarios. A summary of the CRHM simulations are in Table 3.4 but detailed results from the climate change simulations can be found in Table A1.

3.5 DISCUSSION

At present, two watersheds in the AOSR inform much of our understanding of hydrology (Ketcheson et al., 2017a; Nicholls et al., 2016; Spennato et al., 2018), carbon dynamics (Clark et al., 2019, 2020) and water quality (Chapter 4; Kessel et al., 2018; Simhayov et al., 2017) for integrated constructed ecosystems. With only two examples over a relatively short time period, it is uncertain as to how representative these systems are and whether early findings can be used to evaluate performance and help guide future design. While ecosystems are built for long-term sustainability, they are dynamic and rapidly changing and it is expected that their hydrological behaviour will evolve in response to changes in vegetation, soil properties, water quality and climate. Examination of winter processes in these constructed ecosystems is scarce (Kelln et al., 2009; Ketcheson & Price, 2016c; Meier & Barbour, 2002), despite several months of temperatures below freezing. In this work we used detailed field observations combined with a purpose-built hydrological model to better understand how ecosystem properties and a changing climate will influence hydrological fluxes.

3.5.1 What controls patterns of snow accumulation and melt on SFW?

The greatest SWE variability arose from differences in slope position on the hummocks (Figure 3.3), where snow accumulated at the bottoms of the hillslopes and decreased towards the crest. The base of hillslope accumulates blowing snow from the relatively flat wetlands and uplands, whereas the crest of the hummocks are exposed to wind erosion as well as enhanced sublimation (Pomeroy & Essery, 1999). SWE distribution among aspects showed no significant difference ($p>0.05$), as wind direction was variable throughout the winter (data not shown). SWE decline was similar between the wetland and upland areas (Figure 3.4a and b) (Ketcheson & Price, 2016c) as both have a relatively flat terrain and receive similar amounts of solar radiation throughout the day. Melt was most variable on the hummocks as slope aspects are exposed to different irradiances and south-facing slopes melted earlier and faster than north-facing slopes (Figure 3.4d).

Vegetation often exerts a strong control on snow accumulation and melt patterns as blowing snow from surrounding landscapes are trapped by shrubs and trees plus snow interception on their canopies (Boon, 2012; Pomeroy et al., 2006, 1998). However, due to the relatively young age of the SFW, topography was the dominant control on snow accumulation and melt as the vegetation is in the early stages of development and has similar height and density across landscape units, despite species differences. This is common among more recently reclaimed ecosystems (Ketcheson & Price, 2016c) which are considerably younger than their natural analogues and require decades for their vegetation to develop fully. As vegetation emerges, increased roughness will lower the influence of wind on scouring and enhance snow trapping via decreased turbulence. However, direct interception may lower snow accumulation due to enhanced sublimation. From a melt perspective, vegetation reduces shortwave radiation to the surface yet enhances long-wave fluxes to the snowpack, thus altering the snowmelt radiative regime (Pomeroy et al., 2009). While aspect plays an important role on how vegetation influences melt (Ellis & Pomeroy, 2007; Ellis, Pomeroy, Essery, & Link, 2011), as the canopy closes melt will be delayed further into spring (Dickerson-Lange et al., 2017). This influence can be observed by comparing SFW with other nearby reclamation landscapes in more advanced stages of regrowth (Figure 3.9). Two mature reclaimed forests, South Bison Hill (~20-year-old mature aspen/white spruce stand) and Jack Pine (~30-year-old jack pine stand) have delayed melt compared with the newly reclaimed Coke Beach site (~10-year young aspen) and the SFW (~6 years). Both sites with tall vegetation had deeper snowpacks that persisted later into April and were less susceptible to early melt. However, the actual snow-free dates were remarkably similar.

3.5.2 The importance of hillslopes on snowmelt runoff generation

The snowmelt period is a critical input of freshwater available to WBP ecosystems, including constructed systems, which rely heavily on this input to replace storage deficits for the upcoming growing season as well as persist in a climate with a long-term water deficit (Devito et al., 2012). Snowmelt also offers an important input of freshwater to dilute the elevated salinity and sodium concentrations that are ubiquitous within the reclaimed landscape (Chapter 4; Kessel et al., 2018). Reconstructed landscapes need to balance between optimizing soil storage for vegetation productivity while providing sufficient water to surrounding landscapes. The hillslope hummocks in the SFW were initially designed to provide groundwater to the adjacent lowlands via recharge and were not expected to produce much surface runoff. The hummocks have no confining soil layer at depth and most of its meltwater infiltrates and percolates downwards. Groundwater flow is the primary delivery of water to the lowlands (Lukenbach et al., 2019), whereas surface runoff is limited to snowmelt periods and potentially extreme rain events.

While total surface snowmelt runoff was similar on both south and north-facing slopes (15 mm and 23 mm, respectively), their timing and rates differed where south-facing slopes melted much earlier and faster (Figure 3.5). The increased air temperature and radiation in mid-March induced melt-generated runoff on the south-facing slope, yet north-facing slope melt was delayed and no runoff occurred until mid-April when conditions remained warm and a rapid decline in snowpacks were observed (Figure 3.5). The SFW hummocks are constructed from Pf sand and tailings sand materials which have average hydraulic conductivities of 1.1 x 10^{-4} m/s and 6.7 x 10^{-4} m/s and an average unfrozen infiltration capacity of 1.8 x 10^{-4} m/s (Table 3.1). These hummocks remain unsaturated all year as water infiltrates and percolates downwards to recharge groundwater at depth (Lukenbach et al., 2019), which can result in low antecedent moisture contents (VWC <5 % on Nov 1, data not shown) prior to freeze-up throughout the upper soil column. However, this varied over the five years of observation. In 2018, these soils maintain a relatively high infiltration capacity when frozen and exceed the maximum melt rate on both the north and south-facing slopes (5.8 x 10^{-7} and 3.7 x 10^{-7} m/s, respectively) therefore partitioning most of the snowmelt water as infiltration. Surface runoff ratios at SFW (~0.3) were much less than those reported by Ketcheson and Price (2016) (0.7-0.9). The hillslopes on Nikanotee fen by comparison, have finer soil textures, considerably higher moisture content overall, and low surface infiltration capacities (Ketcheson & Price, 2016b), resulting in much higher surface runoff during spring snowmelt (Ketcheson & Price, 2016c).

Using CRHM, a well-established platform for simulating winter hydrological processes, the influence of soil properties on the partitioning of meltwater was evaluated. Soil texture alone had only a modest influence, increasing runoff as texture became finer. However, this is complicated by differences in antecedent moisture that would typically accompany finer textures. As antecedent soil wetness increased, melt generated runoff increased as snowmelt infiltration became more restricted (Table 3.3). Progressively finer textures will increase runoff, an important implication for landscape design and an observation which reconciles results presented here and those of Ketcheson and Price (2016a). Surprisingly, soil temperature at the onset of melt had a large influence on limiting infiltration. In 2018, soils were relatively warm (-2 °C) under a deep snowpack, and simulations suggest that if soils were colder, considerably more runoff would have been generated (Table 3.3). However, a negative feedback exists as deeper snowpacks that can generate more runoff insulate soils (enhancing infiltration), whereas thin snowpacks with less potential for runoff generation would have colder soils, enhancing runoff. While texture can be used to directly enhance or reduce melt runoff, a subtle relationship exists that provides for a large range of runoff ratios based on the combination of SWE and soil temperature. From an operational perspective, upland hillslopes can be designed using textural classes to either enhance or limit direct runoff from hillslopes to provision lowland and wetland systems with water. However, model results suggest a large range of variability driven by moisture content and temperature, even within coarse textured soils that were expected to produce little snowmelt runoff. As vegetation on uplands grows, snow accumulation may slightly increase, and melt will be delayed. Furthermore, vegetation will increase soil macro-porosity and infiltration capacity, reducing runoff potential.

3.5.3 What is the influence of climate change on winter processes?

While the WBP is expected to be warmer and wetter with the influence of climate change, an increase in evapotranspiration could lead to an overall drier environment (Ireson et al., 2015; Thompson et al., 2017). CRHM simulations for winter indicate a decrease in snow season length, runoff, peak SWE and an increase in the presence and magnitude of mid-winter melt events. While it suggests that the increase in precipitation by a factor of 1.2 offsets the increase in temperature by 2 °C, further warming clearly decreases the snowpack SWE and shortens the snow season and therefore more of the annual precipitation is delivered as rain instead of snow. Mid-winter melt events are increasingly prevalent under warming scenarios and are expected to become more common in reconstructed systems and may result in the complete loss of the snowpack in drier years when the snowpack is small. Surface runoff is expected to decrease considerably with increased warming and may cease under the warmest scenarios. However, the role of potentially colder runoff-enhancing soils with decreasing snowpacks was not captured in the model. With a smaller snowpack, less meltwater is partitioned as runoff, which may increase soil and groundwater recharge. The influence of climate change may be underestimated by CRHM, as results indicate that simulations overestimate SWE and to a lesser extent runoff compared to observed data. As a result, actual snowpack SWE, runoff and mid-winter melt events with the onset of climate change could be enhanced compared to simulated results in this study. While an increase in precipitation (delivered as rain) may offset the effects of a lengthened growing season and increase in evapotranspiration, it is expected that the water balance of these systems will be impacted over time. The influence of climate change on SFW combined with expected changes in vegetation and soil development provide considerable uncertainty as to how the rate, timing and magnitude of hydrological fluxes within the landscape will change. With the potential for these systems to become increasingly drier with the onset of climate change, it is critical that future design of these constructed systems maximize water retention and storage such as increasing hummock recharge (Ketcheson et al., 2017a; Lukenbach et al., 2019) and maintaining high infiltration capacity-soils to enhance groundwater recharge.

3.6 CONCLUSIONS

The future expansion of bitumen extraction via surface mining projected for the coming decades will result in thousands of hectares of land that will need to be reclaimed. A better understanding of the evolution of runoff pathways and surface-groundwater interactions are required to assess their trajectory and long-term sustainability, which will influence water management and future design of these systems. Results presented here highlight that topography currently controls snow distribution and melt, as vegetation has little influence on winter processes during these early stages. Surface runoff from hillslopes in a large snow year was higher than expected (up to 30 %) and offers a potentially important transport mechanism of water towards the wetland that can help replenish water deficits as well as offer a supply of fresh water to a relatively saline wetland. The remainder of meltwater infiltrated into the unsaturated frozen soils and was partitioned between the rooting zone and deeper percolation. This study provides an important contrast to the other constructed wetland-upland system in the AOSR where the majority of meltwater was transported to the lowland via surface runoff which highlights the influence of construction design and practice on the system's hydrology. The Cold Regions Hydrological Model simulations of snowmelt partitioning with varying soil conditions indicate that antecedent soil saturation (VWC/porosity) and soil temperature had the greatest influence on partitioning snowmelt as surface runoff, while soil texture alone had a moderate affect. Observed and modelled surface runoff results within the AOSR provide evidence that partitioning between surface runoff and infiltration can somewhat be controlled by hillslope construction materials when designed these reclaimed systems. Under various scenarios of a warmer and wetter climate, the Cold Regions Hydrological Model predicts that average annual peak SWE and duration of the snow season could decline by up to 52 % and up to 61 days, respectively while snowmelt runoff ceases completely under the warmest scenarios. This may lead to drier conditions with the onset of climate change as water deficits could increase each year as an increasingly smaller snowpack cannot replenish water stores.

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Data Availability Statement

The data that support the findings of this study are available from the corresponding authors upon reasonable request.

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Tables

Table 3.1. Measured hummock soil properties.

Soil Type	Prescribed Depth	Specific Yield	Porosity	LOI	Bulk Density	K_{sat}	Sand†	Silt†	Clay†
	cm	-	-	%	g/cm^3	m/s	%	%	%
LFH	0 - 15	0.23	0.31	9.3	1.14	1.75E-04	-	-	-
Pf Sand	15 - 55	0.28	0.23	3.5	1.58	1.09E-04	94	6	0
Tailings Sand	> 55	0.29	0.26	1.1	1.54	6.74E-05	92	3	5

†(NorthWind Land Resources Inc., 2012)

Table 3.2. Parameters used to set up CRHM simulations under current and potential climate conditions.

Parameter	Units	Assigned	Module
HRU area	km^2	3.31E+4 & 1.6E+4	Shared
Aspect	N/E/S/W	S & N	
Elevation	m	318	
Slope	°	8	
Latitude	°	57	
Vegetation height	m	1	
Si (initial soil saturation)	mm^3 mm^{-3}	0.2	
Max available water holding capacity	mm	375	
Max values for soil recharge zone	mm	60	
Albedo_bare	-	0.17	Albedo
Albedo_snow	-	0.85	
Groundcover	-	1 (bare ground)	frozenAyers
Soil Texture	-	1 (coarse/medium coarse)	
Soil Temperature	K	271.1	
Coefficient	-	2.82	
Rain_soil_snow	-	1	SnobalCRHM
T_g / G_flux		0	
Climate Chg Temp	°C	0, 2, 4, 6	obs
Climate Change Ppt factor	-	1, 1.15, 1.2, 1.3	

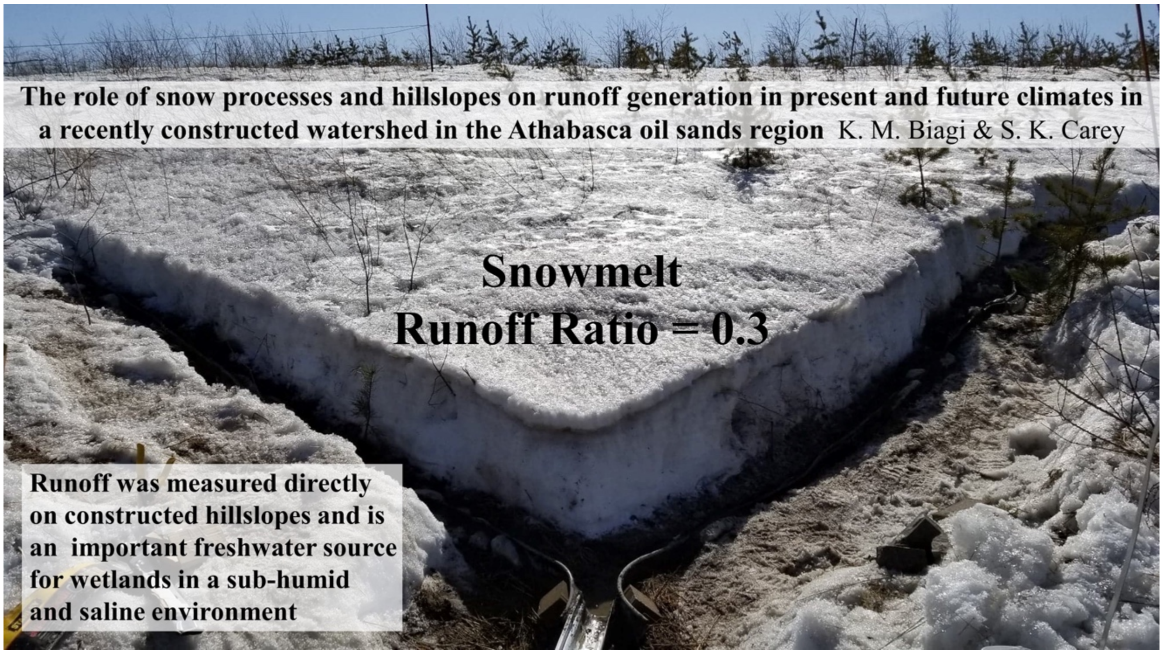
Table 3.3. CRHM simulation results from the frozenAyers module. Variables tested were initial soil saturation (Si), soil temperature (Tsoil) and soil texture.

Si	**Runoff**	**Tsoil**	**Runoff**	**Soil Texture**	**Runoff**
-	mm	°C	mm	-	mm
0.2	34	0	0	Sand	34
0.25	47	-1	0	Silt loam	37
0.3	60	-2	34	Clay	41
0.35	72	-4	69		
0.4	84	-6	85		
0.45	95	-8	94		
0.5	105				

Table 3.4. Summary of CRHM simulation results for projected climate change scenarios. Positive and negative signs indicate if values increased or decreased, respectively from current conditions. Note that values were averaged over the five simulated years (2013-2018).

Climate Scenario	Change in peak snowpack SWE	Change in snow season	Mid-winter melt duration	Mid-winter melt	Runoff	Runoff Ratio
2013-2018	%	days	days	mm	mm	-
T0-P1	0	0	5	12	11	0.38
T2-P1	-21	-10	7	26	8	0.1
T4-P1	-35	-37	14	39	1	0.01
T6-P1	-42	-61	11	43	0	-
T0-P1.15	+12	+7	0	0	20	0.17
T0-P1.2	+28	+8	0	0	25	0.18
T0-P1.3	+36	+8	0	0	29	0.21
T2-P1.2	-0.2	-1	6	19	20	0.21
T4-P1.2	-24	-17	12	40	10	0.16
T6-P1.2	-32	-16	14	44	3	0.03

Figures



Graphical Abstract. Runoff was measured directly on constructed hillslopes and is an important freshwater source for wetlands in a sub-humid and saline environment.

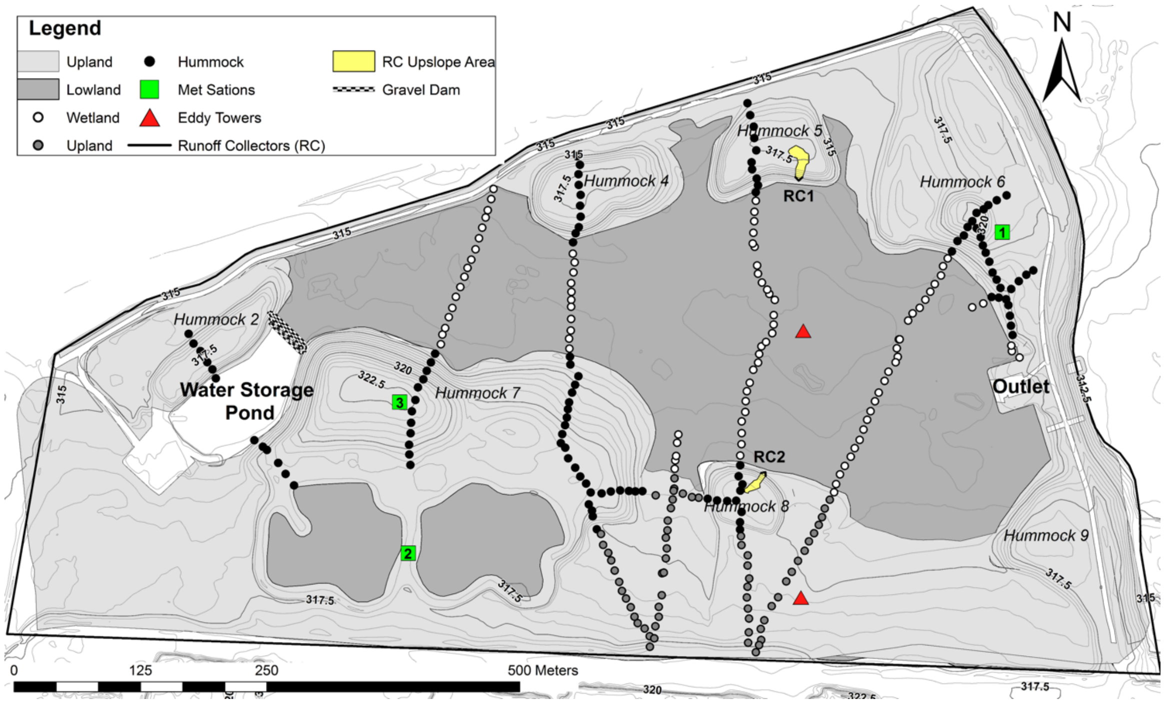


Figure 3.1. Instrumentation map of Sandhill Fen Watershed. Dark grey areas represent the lowland (wetland) area and lighter grey represents the margins and upland areas. Circles represent snow survey points which are coloured based on landscape type. The “V” shape at the base of the runoff collectors indicate the physical constructed boundary and thinner lines represent the calculated upslope contributing areas.

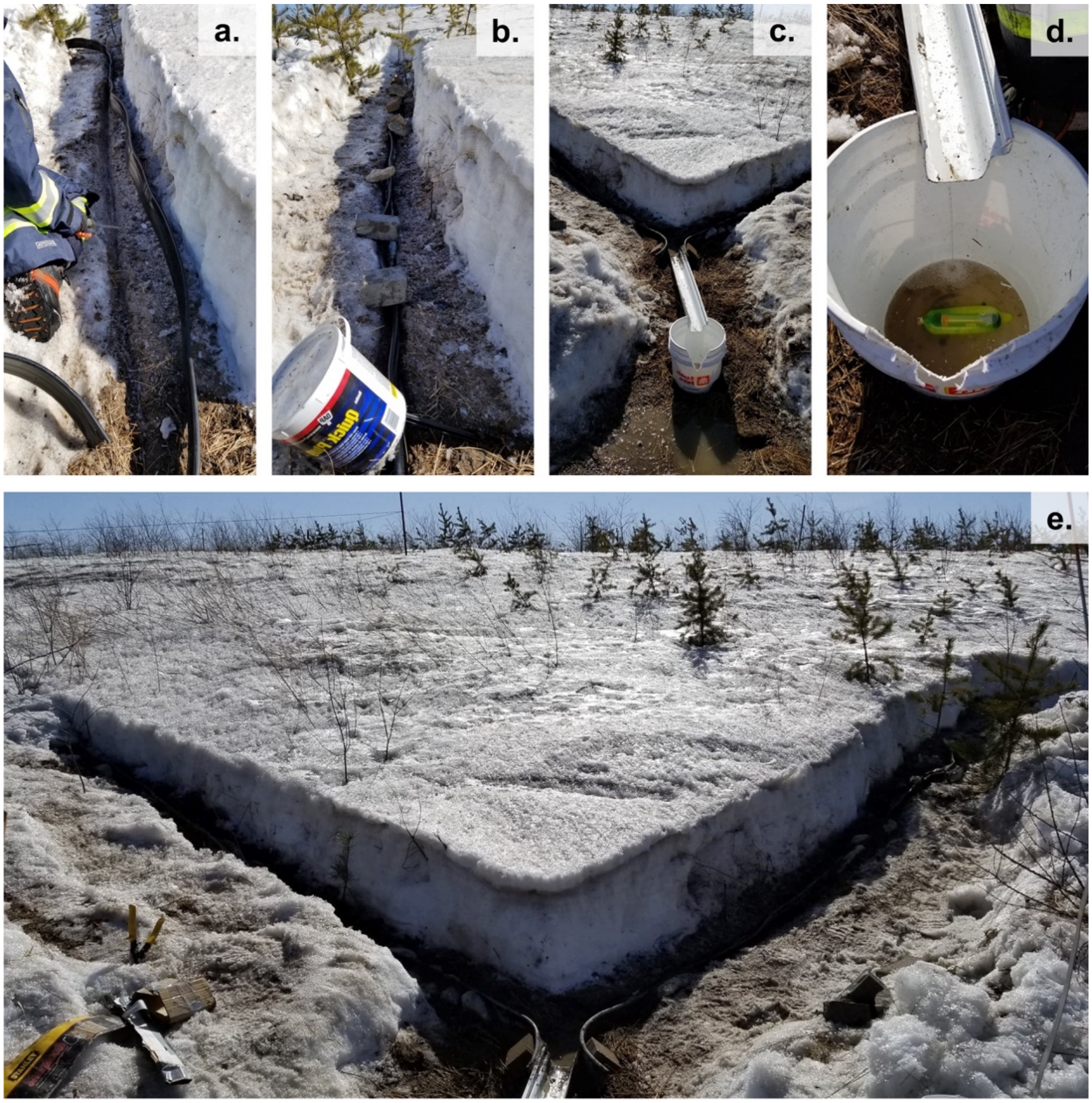


Figure 3.2. Runoff collector construction phases. a) Two 10-15 cm trenches were dug into the frozen ground, b) hydraulic cement was used to seal the garden edging into the trenches, c) trenches tapered at the bottom of the slope to form a "V" where an eavestrough was used to funnel water towards the bucket with a v-notch, d) a pressure transducer was used to continuously measure water height and discharge, e) completed runoff collector where each arm of the "V" was ~4 m.

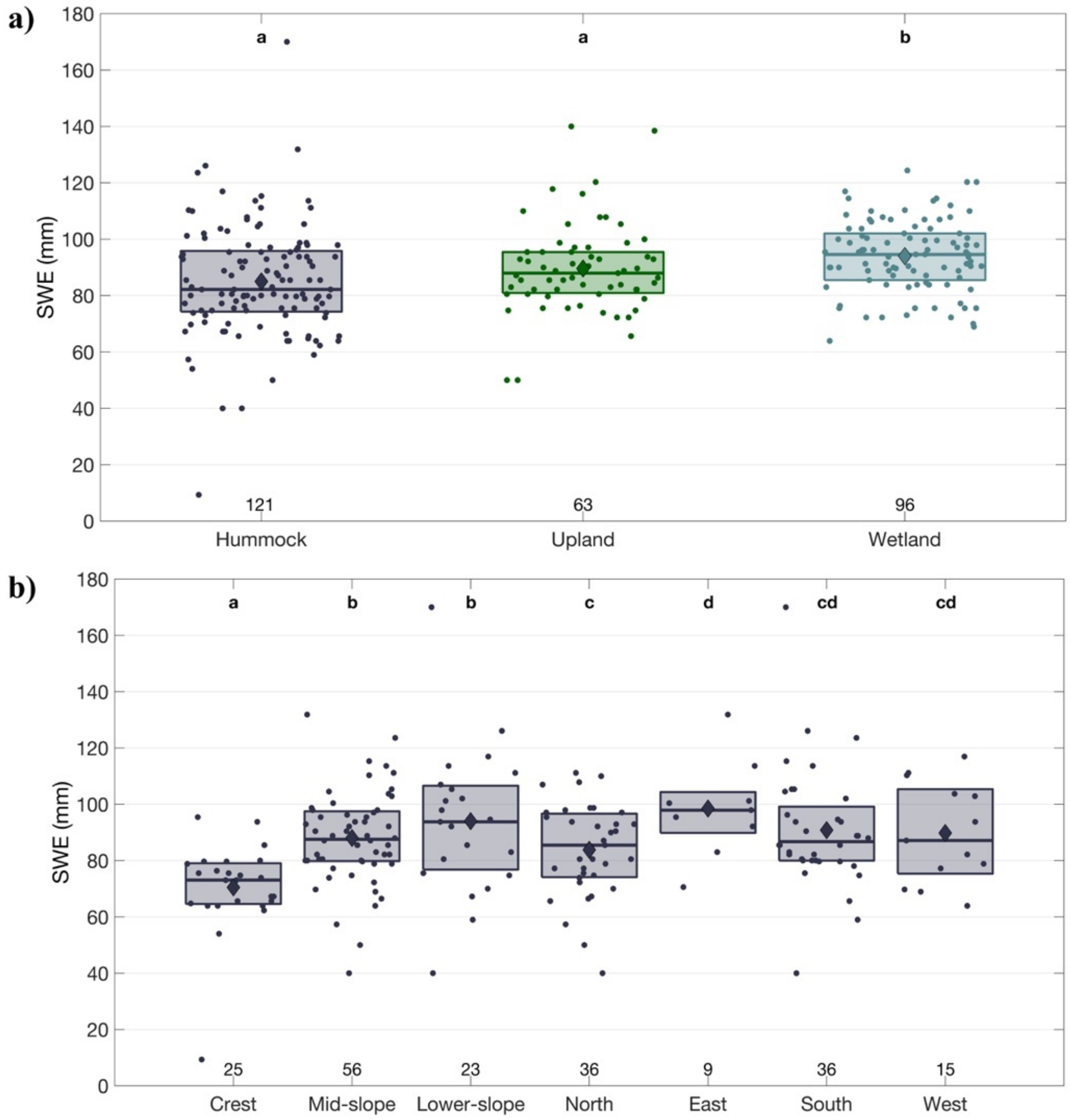


Figure 3.3. Boxplots of snowpack SWE prior to melt in a) landscape units and b) hummock slope position and aspect. Circular points are data points from the snow surveys. Outer border of the boxplots represents the 25th and 75th percentiles and mid box lines represent the group median. The diamond points represent the group mean and numbers above the x-axis indicate sample size. Means sharing a letter are not significantly different (Wilcoxon rank sum test).

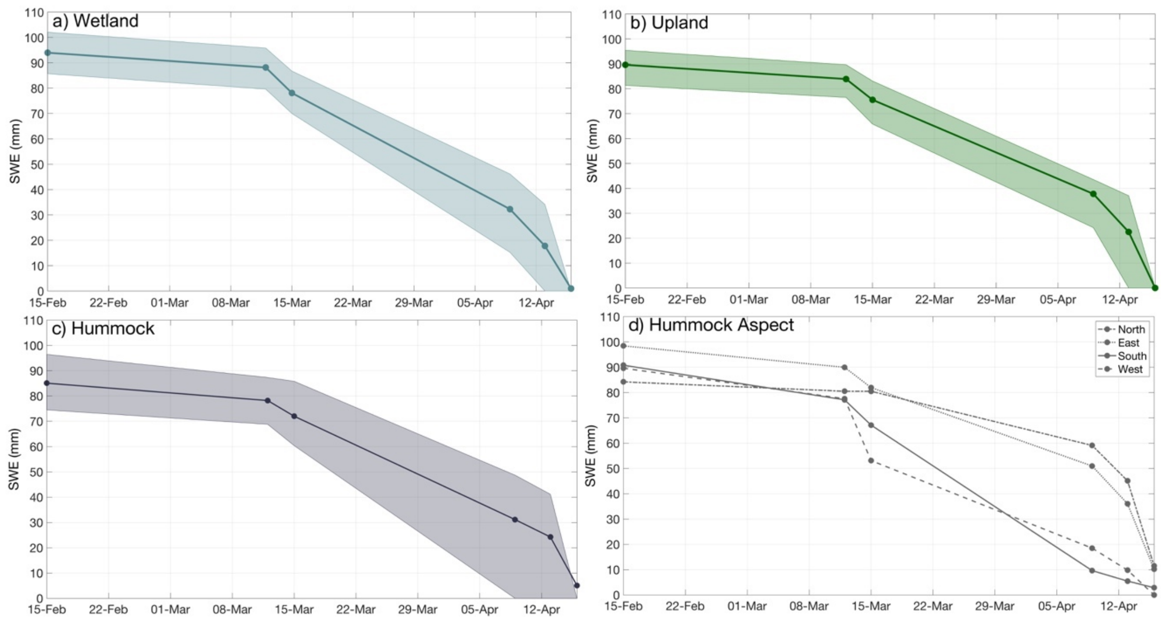


Figure 3.4. Snowpack SWE of each landscape unit from before and during melt for the a) wetland, b) upland, c) hummocks and d) hummock aspects. The data points represent the mean SWE and the top and bottom of the shaded areas represent the 75th and 25th percentiles, respectively.

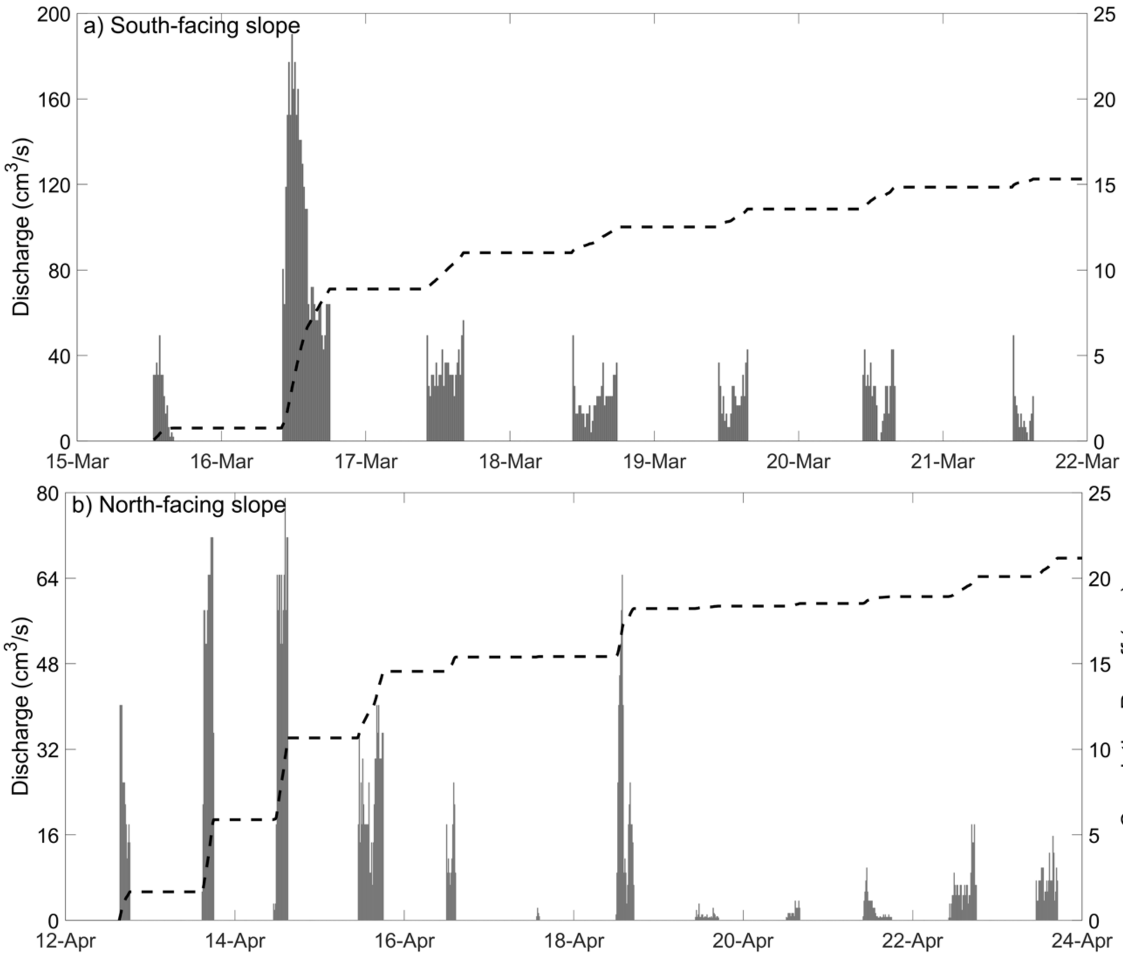


Figure 3.5. Snowmelt runoff collector discharge (cm^3/s) and cumulative runoff (mm) at 15-minute intervals for two distinct melt periods of a) South-facing runoff collector from 15-22 March 2018 and b) North-facing collector during 12-24 April 2018. Note: 1) figure scales are different and 2) the entire snowpack had melted by the end of each melt period shown.

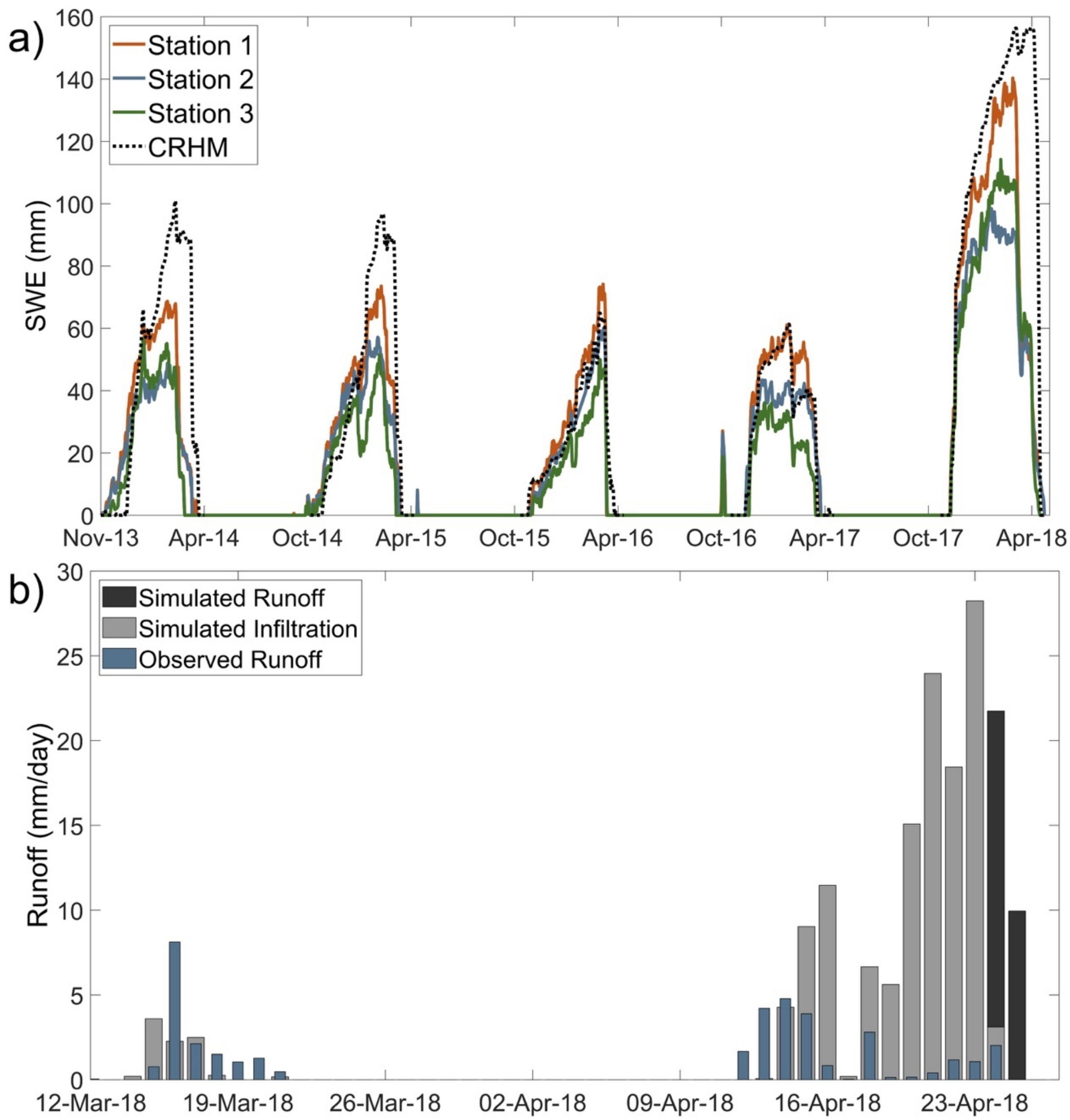


Figure 3.6. Observed and CRHM simulated a) hourly snowpack SWE from 2013-2018 and b) daily snowmelt surface runoff for the 2017-2018 winter season.

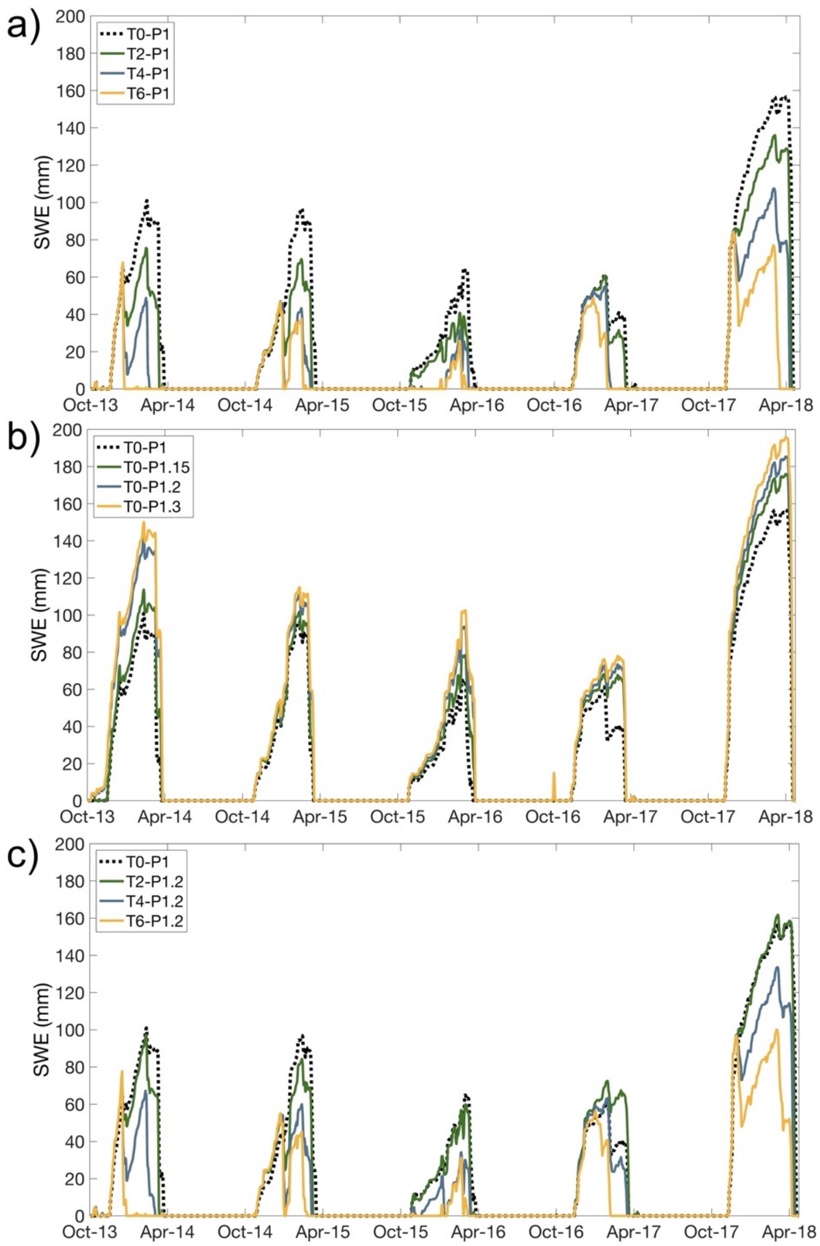


Figure 3.7. Simulated SWE under climate scenarios of a) increased temperature (T2-P1, T4-P1, T6-P1), b) increased precipitation (T0-P1.5, T0-P1.2, T0-P1.3) and c) increased temperature and precipitation (T2-P1.2, T4-P1.2, T6-P1.2). The black dotted line is the simulated SWE under current climate conditions (T0-P1).

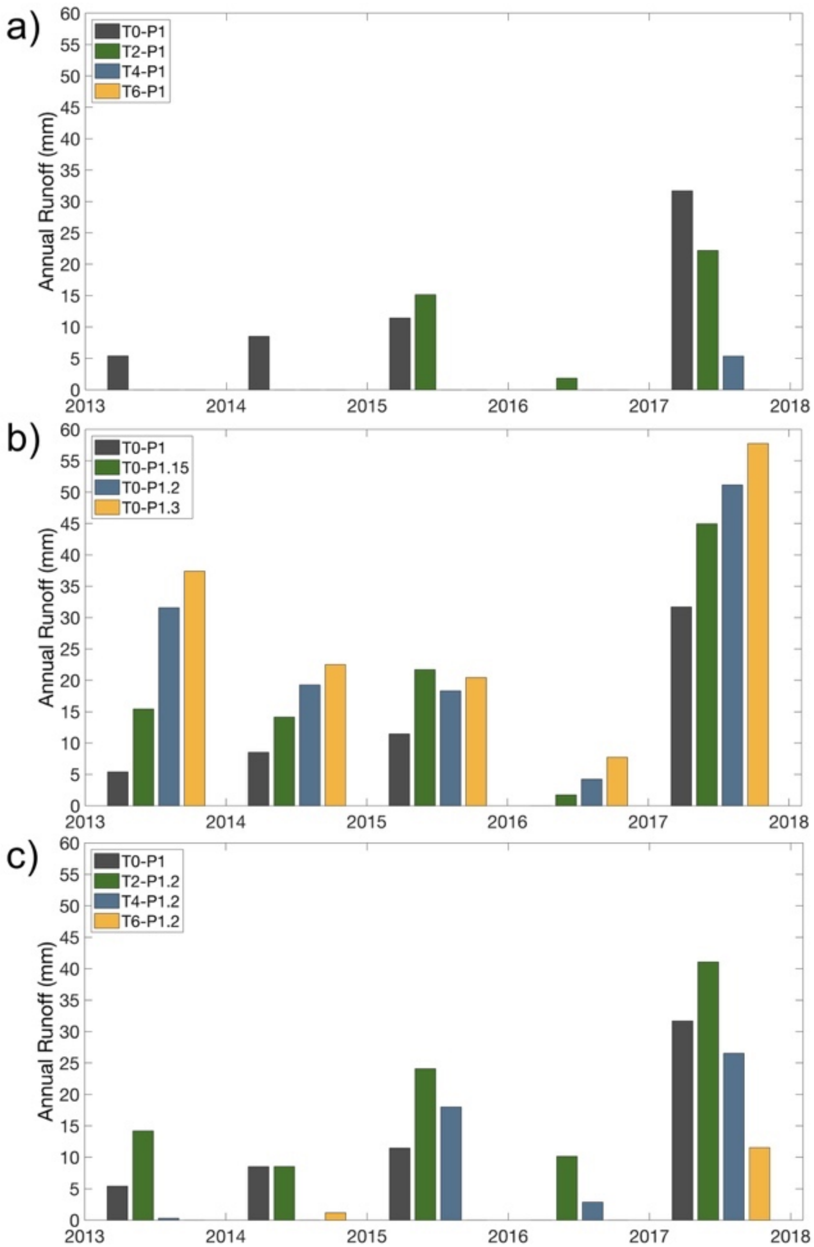


Figure 3.8. Simulated snowmelt runoff under climate scenarios of a) increased temperature (T2-P1, T4-P1, T6-P1), b) increased precipitation (T0-P1.5, T0-P1.2, T0-P1.3) and c) increased temperature and precipitation (T2-P1.2, T4-P1.2, T6-P1.2). The dark grey bar represents the simulated runoff under current climate conditions (T0-P1).

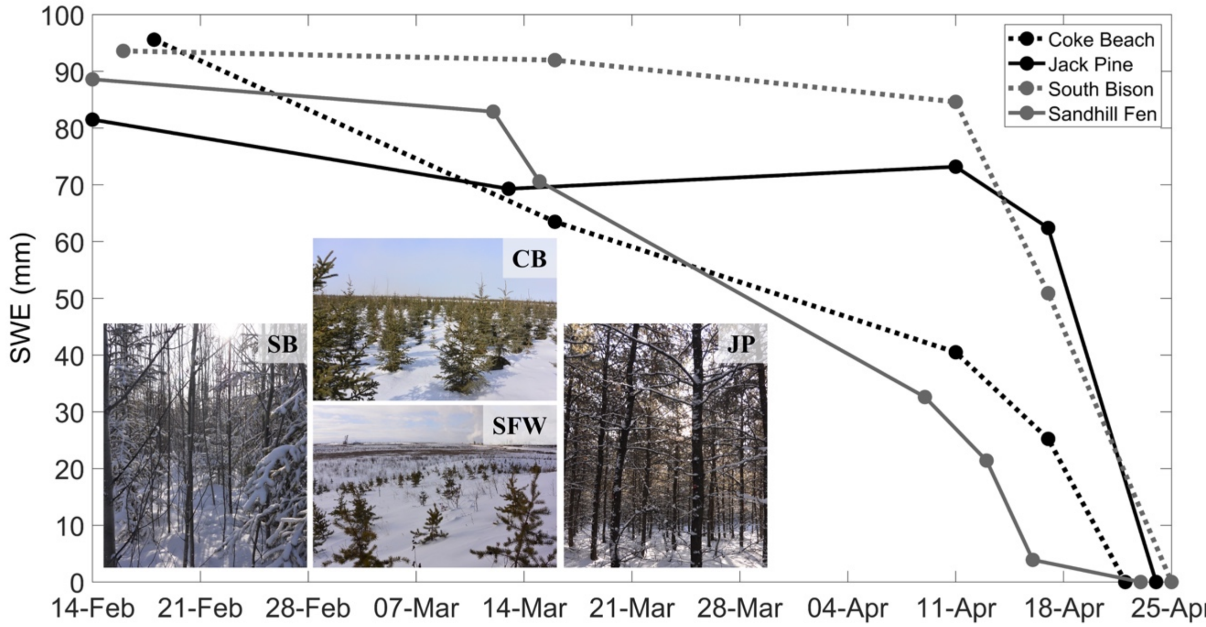


Figure 3.9. Average SWE of older constructed systems on the Syncrude Canada Ltd. property. Data points represent daily average SWE data from snow surveys conducted during the 2017-2018 winter. Inset: photographs to indicate maturity differences among sites for South Bison (SB), Coke Beach (CB), Jake Pine (JP) and Sandhill Fen Watershed (SFW).

CHAPTER 4

INCREASES IN SALINITY FOLLOWING A SHIFT IN HYDROLOGIC REGIME IN A CONSTRUCTED WETLAND WATERSHED IN A POST-MINING OIL SANDS LANDSCAPE

This chapter examines early hydrological and related salinity changes at the study site following a shift from heavy management to minimal management (first three years). Drivers behind the increase in salinity are explained and justified.

It is published as Biagi, K.M., Oswald, C.J., Nicholls, E.M., and Carey, S.K., 2019, Increases in salinity following a shift in hydrologic regime in a constructed wetland watershed in a post-mining oil sands landscape: Science of the Total Environment, v. 653, p. 1445–1457, doi:10.1016/j.scitotenv.2018.10.341. This chapter has been printed under licence from Elsevier (Appendix C).

ABSTRACT

Bitumen extraction via surface mining in the Athabasca Oil Sands Region results in permanent alteration of boreal forests and wetlands. As part of their legal requirements, oil companies must reclaim disturbed landscapes into functioning ecosystems. Despite considerable work establishing upland forests, only two pilot wetland-peatland systems integrated within a watershed have been constructed to date. Peatland reclamation is challenging as it requires complete reconstruction with few guidelines or previous work in this region. Furthermore, the variable sub-humid climate and salinity of tailings materials present additional challenges. In 2012, Syncrude Canada Ltd. constructed a 52-ha pilot upland-wetland system, the Sandhill Fen Watershed, which was designed with a pump and underdrain system to provide freshwater and enhance drainage to limit salinization from underlying soft tailings materials that have elevated electrical conductivity (EC) and Na^+. The objective of this research is to evaluate the hydrochemical response of a constructed wetland to variations in hydrology and water management with respect to water sources, flow pathways and major chemical transformations in the three years following commissioning. Results suggest that active water management practices in 2013 kept EC relatively low, with most wetland sites <1000 µS/cm with Na^+ concentrations <250 mg/L. With limited management in 2014 and 2015, the EC increased in the wetland to >1000 µS/cm in 2014 and >2000 µS/cm in 2015. The most notable change was the emergence of several Na^+ enriched zones in the margins. Here, Na^+ concentrations were two to three times higher than other sites. Stable isotopes of water support that the Na^+ enriched areas arise from underlying process-affected water in the tailings, providing evidence of its upward transport and seepage under a natural hydrologic regime. In future years, salinity is expected to evolve in its flow pathways and diffusion, yet the timeline and extent of these changes are uncertain.

4.1 INTRODUCTION

Bitumen extraction via open-pit surface mining has resulted in over 800 km^2 of disturbance of Boreal forests and wetlands in the Athabasca oil sands region (AOSR) (Government of Alberta, 2015). During the mining processes, entire landscape surfaces are removed and excavated up to 75 m deep to access the bitumen-rich McMurray Formation (Government of Alberta, 2013), permanently altering these systems (Price et al., 2010; Rooney & Bayley, 2011; Rooney et al., 2012; Trites & Bayley, 2009b). In addition to this physical disturbance, high levels of salts are ubiquitous in the post-mining landscape from the tailings material used to backfill open pits and placed directly on the landscape. Due to the large volume of tailings, a treatment process was adopted in the oil sands to decrease the total volume of Mature Fine Tailings. The product of this treatment, composite tailings, is a saline non-segregating slurry that is composed of tailings sand, fine tailings and oil sands process water (OSPW) (Chalaturnyk, Scott, & Özüm, 2002; Matthews, Shaw, MacKinnon, & Cuddy, 2002). The highly saline nature of this waste material is a result of: 1) naturally occurring saline sediments and aquifers, and 2) chemicals used during bitumen recovery. Naturally occurring marine shale sediments and aquifers are disturbed during extraction of the McMurray Formation, which contains high concentrations of soluble salts, particularly Na^+ and Cl^- (Kessler et al., 2010; Leung et al., 2003; Lord & Isaac, 1989). Bitumen recovery involves the use of chemicals to extract bitumen from the sand sediments that contribute soluble salts to tailings such as, caustic hot water (NaOH) used to extract bitumen from the sand sediments, and gypsum ($CaSO_4 \bullet 2H_2O$) used to initiate the formation of composite tailings (Chalaturnyk et al., 2002; Matthews et al., 2002). The resultant OSPW from these processes is characterized by elevated Na^+ concentrations as a result of cation exchange between Na^+ on the marine clays and free Ca^{+2} ions in solution (MacKinnon, Matthews, Shaw, & Cuddy, 2001). Much of the concern with these waste materials surround the negative ecological effects such as vegetation stress associated with elevated

Na^+ concentrations as these substrates will underlie many reclaimed systems and have the potential to be transported to the surface (MacKinnon & Boerger, 1986).

The current disturbance footprint will increase as the total minable area in the AOSR is six-fold that of the area currently affected by mine activities (~4800 km^2) (Government of Alberta, 2015). To mitigate this large-scale landscape disturbance, the Environmental Protection and Enhancement Act implemented by the Government of Alberta requires industries to reclaim altered landscapes back to pre-disturbance functioning capability as part of their closure plans (OSWWG, 2008). Forest and wetland ecosystems must be reconstructed, as open pits are backfilled with waste material and are capped with salvaged soil material from the pre-mining landscape. Although previous research efforts in this region have largely focused on the construction of forested-upland systems (Carey, 2008; Elshorbagy et al., 2005; Huang, Barbour, et al., 2015; Huang, Hilderman, et al., 2015; Kessler et al., 2010; Lilles, Purdy, Chang, & Macdonald, 2010; Sorenson, Quideau, MacKenzie, Landhäusser, & Oh, 2011; Strilesky et al., 2017), efforts have recently shifted to wetland construction (Daly et al., 2012; Hartsock, House, & Vitt, 2016; Kessel et al., 2018; Ketcheson et al., 2016b; Nicholls et al., 2016; Vitt et al., 2016a; Wytrykush et al., 2012), as wetlands covered >50 % of the pre-disturbance Western Boreal Plains landscape, the majority of which were fen peatlands (>90 %) (Rooney et al., 2012; Vitt et al., 1996). The current challenge is to construct peatland systems atop soft tailings that are resilient to the highly disturbed and saline post-mining landscape, with the capability to mimic the functioning capacity of natural peatlands within the Western Boreal Plains.

4.1.1 Reclamation challenges

Peatlands are identified as keystone ecosystems within this region due to the vital ecosystem functions they provide, such as water storage and transmission, peat formation, carbon sequestration and storage, nutrient transport, vegetation growth, habitat, improved water quality and biodiversity (Devito et al., 2012; Ferone & Devito, 2004; Rooney et al., 2012; Vitt, 2006). Constructing peatlands to mimic these important ecological and hydrological functions presents many challenges, as peatland reclamation involves the design and building of complex ecosystems with no benchmarks or previous knowledge of such methods in this region. The initiation of natural peatland formation requires decades to centuries, and succession requires complex interactions between physical, biological and chemical factors to facilitate long-term peatland development which naturally takes thousands of years (Clymo, 1984; Nwaishi, Petrone, Price, & Andersen, 2015). The properties of peat soils used in reclamation have been considerably altered as they are drained and removed from natural sites, stockpiled and transported for placement (Faubert & Carey, 2014). It is currently unknown if disturbed peat can perform the vital ecohydrological functions at the same capacity as natural peatlands due to the disruption of the organic soil profile (Nwaishi, Petrone, Price, & Andersen, 2015).

4.1.2 Peatland hydrology and salt movement

Constructed peatlands must establish a hydrological regime capable of supporting various ecosystem functions that are critical to their long-term success. The hydrology of natural landscapes in the Western Boreal Plains is largely controlled by the sub-humid climate and geology (Devito et al., 2012; Ferone & Devito, 2004), and peatlands play a key role in storing and transmitting water to adjacent landscape units during seasonal and decadal wet and dry cycles (Devito, Creed, Gan, et al., 2005; Smerdon et al., 2005). During wet cycles, peatlands are recharged and hydrologically connected to forest uplands, whereas during dry periods they can provide water to adjacent forests through capillary action or root uptake to maintain vegetation growth (Devito et al., 2012; Ferone & Devito, 2004). Peatlands also have complex feedback mechanisms that enable them to maintain wetness in a variety of climate conditions, suggesting potential resilience to climate change (Waddington et al., 2015).

Another critical design requirement of constructed peatlands is to minimize salinization from underlying waste materials, as elevated salinity may restrict vegetation growth and succession required for long-term peatland development (Nwaishi, Petrone, Price, & Andersen, 2015). Electrical conductivity (EC) above 4000 μS/cm can have detrimental effects on plants (Renault, Lait, Zwiazek, & MacKinnon, 1998; Richards, 1954) and exposure to OSPW (typically > 4000 μS/cm) has shown a significant reduction in plant taxa and species richness (Rezanezhad, Andersen, et al., 2012; Rooney & Bayley, 2011; Trites & Bayley, 2009b) as well as changes to phytoplankton communities in wetlands (Leung et al., 2003). Salts at depth from the OSPW can be transported to surface in reconstructed peatlands via: 1) advection, as fen peatlands receive input from deep groundwater systems (Vitt, Bayley, & Jin, 1995) and, 2) diffusion from the strong concentration gradient between the highly saline OSPW at depth and the relatively fresh water at the surface (Bailey, 2001; Kessler et al., 2010; Merrill, Doering, Power, & Sandoval, 1983). Evapotranspiration also has the potential to enhance diffusion of salts to the surface (Merrill *et al.*, 1983; Moran *et al.*, 1990; Rezanezhad *et al.*, 2012), as it is the largest annual loss of water in the Western Boreal Plains (Devito, Creed, Gan, et al., 2005). Evapo-concentration is also an important process contributing to salt accumulation in constructed peatlands with surface or near-surface water (Biagi, 2015; Simhayov et al., 2017).

Understanding the mechanisms that govern solute transport throughout constructed peatlands and the long-term evolution of salinity is fundamental to the success of these systems. Kessel *et al.* (2018) reported the distribution and movement of Na^+ in the other nearby constructed fen in the AOSR, yet the two peatlands have starkly different designs and likely trajectories of chemical evolution (Ketcheson et al., 2016b). Considering the novelty of newly constructed ecosystems and the importance of limiting salinization in post-oil sands mining environments, the objectives of this research are to: 1) understand the chemical behavior of a constructed wetland in the three years following commissioning, 2) evaluate the role of water management on wetland chemistry, and 3) provide information to guide future wetland creation in the AOSR.

4.2 SANDHILL FEN WATERSHED (SFW)

4.2.1 Location

The SFW is located in the northwest corner of a formerly mined area (1977-1999) called East-In-Pit and is part of Syncrude Canada Ltd.'s Base Mine (57°02'N, 111°35'W) located approximately 40 km north of Fort McMurray, Alberta. This region is situated in the Western Boreal Plains ecoregion of western Canada which is characterized by a cold sub-humid climate. Thirty-year climate normal data (1981-2010) for Fort McMurray indicate average January and July mean daily temperatures of -17.4 °C and 17.1 °C, respectively with an annual mean temperature of 1.0 °C (Environment Canada, 2016). Mean annual potential evapotranspiration (607 mm) exceeds mean annual precipitation (419 mm), with 316 mm of precipitation falling predominantly as rain between May and September (Environment Canada, 2020; Wytrykush et al., 2012).

4.2.2 Sandhill Fen Watershed construction

The SFW is one of two constructed peatlands that currently exist in the AOSR, both of which vary considerably in design (Daly et al., 2012; Wytrykush et al., 2012). East-In Pit was filled with 35 m of inter-bedded composite tailings and tailings sand layers followed by a 10 m structural cap of tailings sand. The SFW was constructed on top of these materials and is approximately 52 ha in area, which includes a 35 ha upland area and 17 ha fen wetland (Figure 4.1). SFW is comprised of several landscape units including upland hills (referred to as hummocks), vegetated swales, perched fens, woody berms, a freshwater storage pond, an underdrain system and a lowland fen wetland (Wytrykush et al., 2012). Seven hummocks, varying in elevation from three to eight meters above the lowland, were constructed of mechanically placed tailings sand to create upland recharge areas that could supply water to the lowland area (Nicholls et al., 2016). The hummocks were capped with 0.1 – 0.5 m of Pleistocene fluvial sand and compacted by a bulldozer. The non-hummock areas of the upland were capped with 0.5 m of peat-mineral mix soils while the lowlands stratigraphy comprises of 0.5 m of clay (soil base) topped with 0.5 m of peat material to provide a surface organic soil. Peat placed in the lowlands was harvested in the fall of 2010 from the upper 0.6 m of a poor fen north of Syncrude's Base Mine and was placed with a bulldozer in January 2011. SFW was vegetated with species native to the area, and seeds were harvested from natural sites nearby in the summer of 2011 and spread throughout the watershed in November of 2011 (Nicholls et al., 2016; Vitt et al., 2016a; Wytrykush et al., 2012). Three boardwalks were installed in the wetland and are named one to three starting in the west near the water storage pond (Figure 4.1). Near-surface well and surface samples are labelled based on the location within SFW: water storage pond (WSP), boardwalk one (B1), boardwalk two (B2), boardwalk three (B3), margins/transition area (TR) and uplands (UP). B1, B2 and B3 collectively are the wetland sites. The B prefix is followed by the well/sample number which increased sequentially based on the order of installation.

The SFW system was designed to provide water to the lowland area from the uplands while limiting salinization through a fresh water supply and usage of underdrains (Nicholls et al., 2016; Wytrykush et al., 2012). In addition to precipitation and groundwater, water can be supplied from a nearby reservoir to the clay-lined water storage pond. Once in the water storage pond, water gradually flows through a gravel dam into the lowland area near boardwalk one. This water flows east towards the outlet where surface water flows through a V-notch weir and into the sump, where both surface and underdrain water are collected. The underdrain system consists of perforated 20.32 cm high-density polyethylene pipes covered in a fine-screen geotextile cloth and lie one to three meters below the lowland area of the SFW. Underdrains were installed to further reduce the potential for salinization of the SFW and, when open, provide an overall downward hydraulic gradient to limit the upward movement of salts from the composite tailings layer, as well as transport any OSPW that has migrated upwards from depth directly to the sump

(Wytrykush et al., 2012). When on, water in the sump, which includes a combination of surface and underdrain water, is collectively pumped out of the SFW and is referred to as the outlet. Although water supply and drainage can be managed at the SFW, it was not intended for long-term use but rather a means to provide freshwater to minimize salinity and to encourage vegetation survival as the system develops.

4.3 METHODS

4.3.1 Inflow and outflow measurements

The lateral inflow via pumping in the water storage pond was measured using a Model AT868 AquaTrans Ultrasonic Flow Transmitter as well as a pressure transducer (Rosemount Inc., Chanhassen, MN, USA) to measure level in the water storage pond. The water level of surface outflow through the V-notch weir was continuously measured with a transducer when pumps were on. Further details on the pump system and other instrumentation are found in Nicholls *et al.* (2016).

4.3.2 Meteorological measurements

Each of the three meteorological towers (Figure 4.1) was instrumented with a tipping bucket rain gauge (Model CS700, Campbell Scientific Inc., Logan UT, USA (CSI)) to measure rainfall, a CSI *CS725* to measure snow water equivalent (SWE) as well as a CSI SR50A sonic ranger to monitor snow depth. All measurements were recorded on an hourly basis with CSI CR1000 data loggers. In addition, three eddy covariance towers were instrumented in 2013 to measure micrometeorological parameters, one of which operates year-round (Nicholls *et al.*, 2016).

4.3.3 Hydrometric measurements

Near-surface water table fluctuations were measured using PVC slotted wells installed to depths between 0.48 – 1.02 m throughout SFW and instrumented with Solinst Junior Edge Levelogger pressure transducers. Eleven near-surface wells were installed in 2013, and 22 additional wells with transducers were installed in 2014 (Figure 4.1). Transducers measured water level and temperature every 15 minutes from May to October in 2013, 2014 and 2015. Manual measurements of water level were made weekly using a Solinst LTC water level tape for quality assurance and control. Levels were corrected each year for barometric pressure using a Solinst Barologger. Wells were named according to their position within SFW, where the suffix B represents wells along boardwalks one to three, TR represents wells in the margins (transition) and UP represents wells in the uplands.

4.3.4 Electrical conductivity

Continuous and discrete EC measurements of surface and pore water were taken from 2013 to 2015. In 2013, Solinst LTC (level, temperature, conductivity) Levelogger Junior pressure transducers were installed in five of the wells along B3 and took measurements every 15 minutes from May to October. In 2014, a total of ten LTC transducers were installed down the major near-surface flow pathways (Figure 4.1). Discrete measurements of EC were taken weekly using a YSI Professional Plus Multiparameter instrument at over 50 surface and pore water sampling sites. All EC values, both discrete and continuous, were corrected to 25 °C (Hayashi, 2004).

4.3.5 Major ions

In 2013, surface water samples were taken at four sites weekly and 13 pore water samples from near-surface slotted wells were taken monthly and analyzed for major ions from May through October. In 2014 and 2015, the sampling program was expanded to include 17 surface and 23 pore water samples were taken bi-weekly from May to October and analyzed for major ions and stable hydrogen and oxygen isotopes. Wells were purged prior to sample collection and samples were collected manually using a Model 428 BioBailer and were poured into 1-L amber bottles, both of which were environmentalized for each sample. Grab samples of surface water samples were also collected using 1-L amber bottles. Pore water samples were filtered using 47-mm, 0.45-μm nitro-cellulose filters, while surface water samples were filtered through GF/F 25-mm, 0.7-μm filters (for particulate organic carbon analysis), both using vacuum suction. Filtered samples were refrigerated in 60-mL, translucent HDPE bottles. Water samples were analyzed at the Biogeochemistry Laboratory, University of Waterloo, for major ions including Na^{+}, K^{+}, Mg^{+2}, Ca^{+2}, Cl^{-} and SO_4^{-2} using a Dionex AS40 Automated Sampler, and alkalinity using a Bran & Luebbe AutoAnalyzer3 (Folio Instruments) with all results reported in mg/L. HCO_3^{-} concentrations were calculated by multiplying the alkalinity values by a conversion factor of 1.22 (Csuros, 1994). Minimum detection limits for Na^{+}, K^{+}, Mg^{+2}, Ca^{+2}, Cl^{-}, SO_4^{-2} and alkalinity were 0.13, 0.1, 0.3, 0.67, 0.02, 0.1 and 5 in mg/L, respectively. A correction curve was built with standards from Thermofisher and five percent of samples are re-run as duplicates for analytical QA/QC procedures. The sodium adsorption ratio (SAR) was calculated for each sample (Equation 1) to infer the relative activity of Na^{+} to Ca^{+2} and

Mg^{+2}, which indicates the degree of cation exchange and the potential for soils to become saline-sodic, where cation concentrations are in mmol/L (Robbins, 1984).

$$SAR = [Na^{+}] / (\sqrt{[Ca^{+2}]} + [Mg^{+2}]) \qquad (1)$$

4.3.6 Stable isotopes

Stable hydrogen and oxygen isotope samples were collected simultaneously with the pore and surface water samples described above. Samples were filtered and stored in 20-mL polyethylene scintillation vials at room temperature. Stable isotope ratios of hydrogen and oxygen were determined using a Los Gatos Research DTL-100 Water Isotope Analyzer at the University of Toronto. Five standards of known isotope composition, with δ^2H ranging from -154‰ to -4‰, purchased from Los Gatos Research were used for calibration, in addition to periodic checks using the international standards, VSMOW2 and SLAP2. During analytical runs, samples were interweaved with standards at a ratio of 3:1. Results of δ^2H and δ^{18}O (‰) were plotted against a Local Meteoric Water Line (LMWL) developed for Syncrude Canada Ltd.'s Mildred Lake Base Mine (Baer, Barbour, & Gibson, 2016) and source water isotope values were taken from Baer (2014). A Local Evaporation Line (LEL) was established for the SFW by plotting a regression line through the δ^2H - δ^{18}O relationship of surface waters within the SFW. The slope of the regression line is equivalent to the slope of the LEL. Statistical analysis of results was completed using the R language for statistical computing (R Core Team, 2018).

4.4 RESULTS

4.4.1 Climate

Mean daily air temperatures at SFW in 2013, 2014 and 2015 were 1.9 °C, 1.4 °C and 3.6 °C, respectively; higher than the 1.0 ± 1.3 °C 30-year climate normal for the Fort McMurray Airport (Environment Canada, 2020). Total annual rainfall was greatest in 2013 (367 mm), followed by 2014 (308 mm) and 2015 (252 mm), with the latter years falling below the 30-year climate normal of 316 mm (Environment Canada, 2020). High intensity and short duration convective storms dominate rainfall events in this region, most occurring in the summer months. In 2013, 334 mm fell as rain between May and September, whereas 270 mm and 236 mm fell in 2014 and 2015, respectively.

Cumulative evapotranspiration (ET) over the growing season (May-Sept) in the lowland, measured via the eddy covariance technique, were similar in 2013 and 2014 (358 mm and 350 mm, respectively) (Nicholls et al., 2016) and slightly less in 2015 (295 mm).

4.4.2 Hydrological management

In 2013, the SFW was commissioned. To saturate the lowland and test the engineered infrastructure, there was near-continuous pumping over the 17 ha lowland totaling ~809 mm (1.38 x 10^5 m^3), which was more than twice the annual average precipitation. Total outflow in 2013 over the lowland was ~883 mm (1.5 x 10^5 m^3), which was primarily conveyed through the subsurface as the open underdrains contributed over 90 % to the total discharge (Nicholls et al., 2016). Although pumps were highly active in 2013, there was no direct hydrological management strategy following the wetting of the lowland as the utility and impact of the pumps was being assessed. In 2014 and 2015, the outflow pump was turned on for either short export experiments or to minimize high water tables in the wetland, as there is no surface outflow and only limited groundwater drainage occurs when the pumps are off. There was no target water level that triggered the operation of the pumps, yet it was recognized that the center of the wetland (B2 and B3) had excessively high water tables compared to natural wetlands. In 2014, the only inflow was on 19-20 May, when 14 mm (2470 m^3) of water was pumped into the water storage pond, while in 2015 there was no freshwater input. The valve connecting the underdrains to the sump was closed in May 2014, restricting outflows to surface discharge in 2014 and 2015. In 2014, several short surface drainage events occurred, lasting one to four hours which totaled 18 mm (3095 m^3). In 2015, one 56-hour drainage event of 54 mm (9334 m^3) occurred. It is important to note that surface outflow from SFW can only occur at this time via artificial pump management.

4.4.3 Hydrological dynamics

In 2013, the water table (WT) across the lowland responded primarily to pumping and large rainfall events, which at times were coincident (Figure 4.2a). Inflow pumping through June and July caused a rise in WT at all lowland sites, with more immediate responses closest to the water storage pond in the west (B1-W2). Conversely, the WT near the outlet in the east (B3-W1 and B3-W4) responded rapidly to outflow pumping events, with less response at B1 (Figure 4.2a) (Nicholls *et al.*, 2016). Limited pumping in July suggests that without management, WT fluctuated in response to rainfall and evapotranspiration. However, by the end of July inflow and outflow pumping resulted in large and rapid WT response throughout the SFW lowlands.

With reduced inflow and outflow pumping in 2014 and 2015, the WT was above or near the surface for most of the year and was largely influenced by rainfall events and evaporative losses which resulted in gradual responses in WT, suggesting that active pumps resulted in a large and more variable WT response as observed in 2013 (Figure 4.2b-c). In 2014, small inflow events (14 mm) in mid-May only resulted in a small response in wells close to the water storage pond (B1-W2) and outflow events throughout the summer (18 mm) were short and responses were small in wells close to the outlet (B3-W1, B3-W4). Responses to rainfall and evaporation were prominent as the WT rose across the lowland in May after 70 mm of rain over five days, followed by steady declines from early June through late August (Nicholls et al., 2016). Several small rainfall events coupled with decreased evapotranspiration in early Fall caused the WT to remain relatively stable until a large precipitation event late September increased water levels (Figure 4.2b). WT patterns were similar in 2015, where the single outflow event of 54 mm at the beginning of June lowered the WT at all west lowland wells close to the outlet (B2-W7, B3-W1 and B3-W4), yet had a limited influence on overall levels throughout the course of the season (Figure 4.2c) (Spennato et al., 2018). The most prevalent pattern in 2015 was the increased WT decline in August during a protracted period of limited precipitation. It is noteworthy that WT decline at B1 in the west lowland was more rapid than the down-gradient sites. A large precipitation event in late August combined with lower evapotranspiration increased water tables at the end of the growing season.

Vertical hydraulic gradients beneath the wetland were evaluated using several nests of deep piezometers throughout the watershed (Longval and Mendoza, 2014 – data not shown). During the initial saturation of the wetland in 2013, vertical hydraulic gradients were downward at many locations throughout the lowland and highly responsive to pumping. In 2014 and 2015 vertical gradients were negligible throughout the lowland, suggesting little vertical recharge or discharge of water.

4.4.4 Hydrochemical dynamics

4.4.4.1 Electrical conductivity (EC)

EC patterns were spatially and temporally dynamic as they were variable across sampling sites and changed throughout the season and annually from 2013 to 2015. In 2013, EC in the lowlands remained relatively low where sites averaged 792 ± 616 µS/cm (Figure 4.3; Table 4.1), with slightly higher values in the margins and uplands. The freshwater supplied to the water storage pond is ~500 µS/cm (Wytrykush et al., 2012), and following the general water flow from west to east, values in the lowland generally increased from B1 through B3 (Figure 4.3a-c) towards the outlet where EC averaged 2567 ± 984 µS/cm in 2013. EC was highest and most variable in the upland and margin areas (1587 ± 1437 µS/cm), with values that ranged from 400 – 4600 µS/cm in 2013.

In 2014, EC increased across all sites in SFW to averages 1481 ± 546 µS/cm, 1718 ± 546 µS/cm and 2593 ± 971 µS/cm in the water storage pond, lowlands and margins/uplands respectively (Table 4.1). However, average values decreased at the outlet from 3160 µS/cm prior to closing the underdrains to 1460 µS/cm in May 2014 once the underdrains were closed. EC increased seasonally at boardwalk one (B1) (Figure 4.3a) where WTs exhibited the greatest decline throughout the summer, whereas EC remained relatively consistent in the mid-fen (B2 and B3) (Figure 4.3b,c) where WTs were consistently above the ground surface. The uplands had no clear EC pattern as values across all sites were variable throughout the season (Figure 4.3d).

In 2015, there was a continued increase in the average EC at the water storage pond, sites B1 and B2 (Figure 4.3a and b), yet values declined slightly at B3 (Figure 4.3c) and remained similar at the margins/uplands on average (Figure 4.3d, Table 4.1). Seasonal patterns in 2015 were relatively weak, yet a small increase was observed at B1 and B2 where EC approached 4000 μS/cm. Areas with persistent standing water throughout the season (B3) had lower and more consistent EC values than observed in 2014 and typically did not exceed 2000 μS/cm (Figure 4.3c). The margins and uplands remained highly variable with respect to EC, and exhibited a slight decline throughout the 2015 season. A simple Wilcoxon rank sum test to assess differences betewen years for pooled data suggests that all years were significnatly different than each other for all four groupings, with increases at B1 and B2 for each of the three years and a decline in EC at B3 in 2015 ($p<0.05$). For the margins/uplands, EC in 2015 as a pooled data set suggest a small decline in 2015 from 2014 at $p=0.05$.

4.4.4.2 Major ions

Major ion concentrations followed similar patterns to EC, as concentrations were relatively low in 2013 and increased throughout 2014 and 2015 (Figure 4.4). While there was an overall increase in all major ions, the results will focus on Na^+ and Ca^{+2} because of important cation exchange processes and the deleterious impact of excessive sodium on vegetation. In 2013, Ca^{+2} exceeded Na^+ concentrations at most sites and were relatively low (Figure 4.4; Table 4.1). Na^+ and Ca^{+2} concentrations were similar in the water storage pond and averaged 46 ± 49 mg/L and 57 ± 25 mg/L, respectively and had an average SAR of 1.1 ± 0.08 (Table 4.1). In the lowlands, average Na^{+} concentrations (56 ± 52 mg/L) were lower than Ca^{+2} (105 ± 108 mg/L) (Figure 4.4) and the average SAR was 1.1 ± 0.7. This pattern was preserved with higher concentrations in the margins and uplands where Na^{+} and Ca^{+2} concentrations averaged 121 ± 60 mg/L and 402 ± 200 mg/L, respectively and had an average SAR of 1.2 ± 0.3. The outlet, 90 % of which was underdrain water, exhibited the highest average ion concentrations across the SFW, as Na^{+} concentrations (355 ± 255 mg/L) were considerably higher than Ca^{+2} (99 ± 34 mg/L) (Figure 4.4 and Table 4.1) and the average SAR was 5.2 ± 4.6.

Although major ion concentrations increased site-wide in 2014 and 2015, the most noticeable change was the increase of Na^{+} concentrations relative to other major ions (Table 4.1). The observed increase was more prevalent in areas termed "Na^{+} enriched zones" that appeared along the border between the base of the southern hummocks and the lowland (Hummocks #2, 7 and 8; Figure 4.5), where Na^{+} was more than double that of other sites. At these locations, average Na^{+} concentrations were 431 ± 159 mg/L in 2014 and 491 ± 120 mg/L in 2015, yet values reached as high as 886 mg/L in 2014. Ca^{+2} concentrations at these enriched zones were considerably lower than Na^{+} in both 2014 (256 ± 134 mg/L) and 2015 (261 ± 66 mg/L). The SARs in these Na^{+} enriched zones were also much higher than any other site in SFW and averaged 6.6 ± 2 in 2014 and 6.2 ± 1.5 in 2015. Average Na^{+} and Ca^{+2} concentrations were elevated in the water storage pond in 2014 (157 ± 76 mg/L and 93 ± 22 mg/L respectively), which continued to increase in 2015 (356 ± 172 mg/L and 132 ± 44 mg/L, respectively). The SAR in the water storage pond increased to an average of 2.2 ± 2 in 2014 and 5.9 ± 2 in 2015. Excluding the Na^+ enriched zones, Na^+ and Ca^{+2} concentrations remained higher in the margins/uplands (153 ± 119 mg/L and 446 ± 161 mg/L, respectively) than the lowlands (122 ± 60 mg/L and 195 ± 123 mg/L, respectively) in 2014. In 2015, Na^+ continued to rise slightly across the lowlands and more notably across the margins and uplands, whereas Ca^{+2} remained level (Table 4.1). The average SAR remained similar in 2014 and 2015 where values were 2.2 ± 1.1 and 1.9 ± 1.5, respectively in the lowlands and were 1.8 ± 1.2 and 2.4 ± 1.7, respectively in the margins/uplands. Na^+ concentrations decreased at the outlet in 2014 and 2015 and averaged 246 ± 175 mg/L and 90 ± 47 mg/L respectively whereas Ca^{+2} remained relatively constant throughout 2014 and 2015 and averaged 125 ± 30 mg/L and 99 ± 31 mg/L, respectively. The largest decline in SAR was observed in the outlet, as the average decreased from 5.3 ± 4.4 in 2014 to 2.0 ± 0.8 in 2015.

4.4.4.3 Stable isotopes of water

Stable isotopes of water from SFW exhibit three distinct groupings which include groundwater, surface water, and water from "Na^+ enriched zones" identified above (Figure 4.6a-c). Near-surface groundwater is a combination of precipitation and water from the input reservoir (Mildred Lake) and falls generally along the LMWL for the site developed by Baer *et al.*, (2016). Samples from standing surface waters plot along a well-developed local evaporation line (LEL) and have isotopically enriched ^{2}H and ^{18}O signatures that result from open-water evaporation. Oil sands process water (OSPW) has a distinct isotopic signature, between the LMWL and LEL, that arises from high humidity near-equilibrium isotopic enrichment of ^{2}H and ^{18}O in cooling towers in the bitumen extraction process, followed by a second open-water evaporation loss in ponds exposed to the atmosphere (Baer et al., 2016). The enriched zones identified via elevated Na^+ (B1-W2, B1-W3, TR-S1 and TR-S2) were isotopically similar to OSPW (Figure 4.6d). A Wilcoxon rank sum test indicates no difference between the enriched zones and OSPW ($p=0.41$), and surface water and OSPW ($p=0.12$), yet there was a significant difference between the enriched zones and the groundwater ($p<0.05$) and enriched zones and surface water samples ($p<0.05$).

4.5 DISCUSSION

Current mine closure plans are required to include strategies for both upland forest and wetland (including peatland) reclamation. This poses extensive challenges considering the disturbance footprint will be in the thousands of square kilometres and there are no benchmark or industry-accepted construction techniques and methods, making integrated watershed construction experimental. Building these systems is complicated by several material challenges. Soils must be salvaged from nearby ecosystems prior to mining, stockpiled and placed in reclaimed systems. Saline tailings must be placed in previous mine pits and in other areas of the landscape, and then salvaged materials placed atop the tailings. These constructed ecosystems must be designed to host of ecological and hydrological functions while being resilient to potential salinization from underlying materials and natural disturbances such as wildfire and climate change (Ketcheson et al., 2016b).

To date, two contrasting upland-fen ecosystems have been built, whose design strategies are outlined in Wytrykush *et al*. (2012) and Daly *et al.* (2012) and sodium dynamics in Kessel *et al*. (2018). The SFW is the larger of these two systems and was designed with infrastructure including pumps and underdrains that allow for hydrological management. Considering that infrastructure at SFW is unrealistic to implement at larger scales, the variable levels of management in this study provide guidance to the importance of active management during the early stages of reclamation and offers insight into the future sustainability of reclamation on soft tailings. In particular, the salinization of the lowland, particularly from Na^+, is a concern as it can negatively affect vegetation success (Rezanezhad, Andersen, et al., 2012; Rooney & Bayley, 2011; Trites & Bayley, 2009b).

4.5.1 What influences major ions and electrical conductivity?

In 2013, the combination of freshwater supply, open underdrains and frequent discharge at the outlet kept salinity levels reduced across the lowlands. While there were several heavy rainfall events, the relative volume of water added from the rainfall events (~100 mm over the season) was relatively small compared to the volume of water added and removed from pumping (~800 mm), which likely overshadowed atmospheric influences on WT and hydrochemistry. Frequent pumping resulted in large variations in the water table, EC and ion concentrations in response to both inflow and outflow. Throughout 2013, sites near the water storage pond (near B1) received freshwater continuously through the leaky gravel dam and WT remained high. Sites closer to the outlet (near B3) exhibited WT that rapidly declined during outflow events as the underdrains were particularly effective at rapidly draining large volumes (~90 %) of water to the outlet sump. Outlet water had the highest Na^+ concentrations and SAR in 2013 due to the active removal of both surface and subsurface waters that contained OSPW. Most sites across SFW exhibited low EC, Na^+ concentrations and SAR values which indicates the limited influence of

OSPW, most of which was discharged through the underdrains, and the addition of fresh water from the water storage pond. While sampling was more limited in 2013, early evidence of an enriched zone of upwelling OSPW was present in late 2013 near boardwalk one (B1) when the underdrains were inactive. When the underdrains were active, OSPW was effectively moved to the drains and hydraulic gradients were downward, keeping the wetland water relatively fresh. Overall, WT dynamics and hydrochemistry were primarily controlled by the pumps and underdrains in 2013, while the influence of atmospheric fluxes was difficult to evaluate because of the large influence of the pumps (Nicholls *et al*., 2016).

The SFW was only lightly managed the underdrains were closed in May 2014, except for a short inflow event in 2014 and several small drainage events in 2014 and 2015 to decrease high water tables in the wetland as well as to capture hydrochemical changes during drainage. As a result, the changes in hydrochemistry were largely controlled by vertical atmospheric fluxes as rainfall events increased the WTs, which gradually declined throughout the season in response to evapotranspiration (Figure 4.2b). As surface waters became stagnant without any inflow or drainage through the outlet, EC and ion concentrations increased although the differences between 2014 and 2015 varied throughout the wetland and upland/margins. Despite the overall increase in ion concentrations in the lowlands, most sites had relatively low SAR values, indicating little influence of Na^+ sources at these sites and suggesting evapo-concentration during dry periods and dissolution of salts in the unsaturated zone as a mechanism for increased salinity during rainfall. Whereas rainfall itself acts to dilute the system, residual salts can complicate the influence of precipitation on salinity. A notable change under the less managed hydrological regime in 2014 and 2015 was elevated Na^+ concentrations at certain sites along the margin between the southern hummocks (hummocks #2, 7, and 8) and the lowlands, where OSPW is presumed to be discharging from depth in the near-surface. These areas exhibited Na^+ concentrations and SAR values more than two-fold of those observed at all other sites across SFW, indicating they are more active areas of cation exchange between Ca^{+2} and Na^+ as a result from the input of Na^+-rich waters from OSPW. Soils are considered saline-sodic when EC and SAR are >4000 μS/cm and 13, respectively (Richards, 1954). Reported Na^+ concentrations in composite tailings and OSPW range between 690 – 1100 mg/L while Ca^{+2} is <100 mg/L (Renault *et al.*, 1998; Renault et al., 1999; Leung *et al.*, 2003; Zubot, 2010; Zubot *et al.*, 2012). Stable isotopes further support the emergence of OSPW in these Na^+-rich areas, as OSPW has a distinct isotopic signature as a result of the industrial refining process (Baer et al., 2016). In contrast, stable isotopes of surface water in the wetland plot along the local evaporation line, and near-surface groundwater plot along the LMWL and have more negative δ^2H and δ^{18}O signatures. Surface water samples show significant similarities to OSPW due to the overlap of the OSPW isotopic signatures which plot in between the LMWL and LEL, close to evaporative-enriched surface water samples. While there was precipitation water input throughout the season, there was insufficient freshwater provided to keep ion concentrations at the 2013 levels, particularly with the contribution of oil sands process water from depth after closing the underdrains.

The overall hydrogeological flow pattern transports OSPW in deeper groundwater moving from the south to northeast in the 10-m sand cap which emerges along the margins of hummocks 2, 7 and 8 via a combination of advection and diffusion (BGC Engineering Inc., 2015). These hummocks (2, 7 and 8) provide a preferential pathway for the transport of OSPW to the surface, as they are constructed of tailings sand material that has a high horizontal and vertical saturated hydraulic conductivity (K_h=10^{-4} m/s, K_v=10^{-5} m/s) and are the only areas without a clay layer present. As the tailings sand in the hummocks is two to three orders of magnitude greater than the clay layer ($K_{h,v}$=10^{-7} m/s), the deep groundwater that transports OSPW can emerge at the base of the hummocks between the tailings sand material in the hummock and the clay-underlain areas. OSPW has not been observed at the base of the northern hummocks in SFW but may appear in the future as the system continues to develop.

There are a number of factors that may influence the salinity of SFW that were not assessed in this study. (1) When the SFW was initially saturated in 2013, solubility reactions would result in a site-wide increase in water salinity, yet the rate of these reactions is uncertain. The rapid pumping did not allow for an observation salinity changes associated with this initial wetting, yet we assume by 2014 a relative measure of equilibrium was achieved. (2) There were differences in weather among the three years, which although in 2013 had a reduced influence on salinity patterns, may exert an important role over longer-term as the Western Boreal Plain undergoes considerable long-term cycles in climate that strongly influence the presence of water on the landscape (Devito *et al.,* 2012). (3) The vegetation within SFW continues to change rapidly (Vitt *et al.,* 2016), and although their influence on system chemistry it has been documented in natural systems (W.-L. L. Chee

& Vitt, 1989; Vitt & Chee, 1990a), the influence of vegetation assemblages on salinity at SFW is unknown.

4.5.2 Incorporating hydrochemistry to evaluate success in constructed peatlands

Defining metrics to evaluate success and to provide insight into the long-term trajectory of constructed wetlands in the AOSR is challenging due to their uniqueness. Currently, much of the framework for assessing success of reclaimed wetlands is based on indicator species for open-water marshes and freshwater peatlands which may be inappropriate for constructed peatlands due to their elevated salinity and ion concentrations that many native freshwater peatland species cannot tolerate (Purdy, Macdonald, & Lieffers, 2005; Rooney & Bayley, 2011; Trites & Bayley, 2009b). Natural saline peatlands in the Western Boral Plains have also been suggested to act as analogues and benchmarks as a measure of ecological success in reclaimed peatlands, as they have adapted to high salinities with salt-tolerant vegetation communities (Purdy et al., 2005; Rooney & Bayley, 2011; Timoney, 2001; Trites & Bayley, 2009b), and can still form organic matter and peat (Trites & Bayley, 2009a). However, the hydrochemistry of constructed peatlands differs considerably as reported values of EC and Na^+ concentrations are orders of magnitude lower in freshwater peatlands (<70 µS/cm and <10 mg/L, respectively) (W.-L. L. Chee & Vitt, 1989; Vitt & Chee, 1990a), and higher in saline peatlands (up to 27000 µS/cm and 13000 mg/L, respectively) (Trites & Bayley, 2009b; Wells & Price, 2015a). Constructed peatlands are also highly dynamic systems and have undergone considerable change within the first five years after commission (Biagi, 2015; Kessel et al., 2018; Ketcheson & Price, 2016a; Nicholls et al., 2016; Simhayov et al., 2017; Vitt et al., 2016a) and may potentially develop into unique ecosystems in the Western Boreal Plains. As such, a functional-based approach to evaluating ecosystem success may be more appropriate in order to monitor and understand how constructed wetlands develop (Ketcheson et al., 2016b; Nwaishi, Petrone, Price, & Andersen, 2015).

Hydrology plays an important role in solute transport and distribution which can influence many ecosystem functions. Elevated salinity and Na^+ concentrations are a primary water quality concern in the AOSR, and development and persistence of saline-sodic conditions in constructed peatlands can cause long-term effects on ecosystem health and success, as elevated Na^+ concentrations support different vegetation and phytoplankton communities that do not promote peat formation at the same rate of natural peatlands (Leung et al., 2003; Rooney & Bayley, 2011; Trites & Bayley, 2009b, 2009a). Results presented here highlight the potential development of saline-sodic conditions in constructed peatlands, as zones of elevated Na^+ concentrations developed within a matter of weeks following the cessation of hydrological management. The SAR could be an effective hydrochemical indicator of water quality in the AOSR, as it infers the relative activity of Na^+ to Ca^{+2} and Mg^{+2} and can indicate the degree of cation exchange in reclamation material (Robbins, 1984). A combination of EC and SAR data can be used as indicators of areas with the potential to become saline-sodic and highlight areas where OSPW is emerging from depth. Incorporating hydrochemical indicators into the overall assessment of ecosystem performance in constructed wetlands can provide a holistic evaluation of ecological functionality and insight on the long-term trajectory of these systems.

4.6 CONCLUSIONS

Mining activities in the Athabasca oil sands region may continue for decades, which will continue to produce significant quantities of soft tailings that will underlie reclaimed landscapes as a method of containment and storage. The ubiquitous salinity of this waste material presents many challenges in constructing self-sustaining wetlands that provide the necessary hydrological and ecological functions required for wetland success. Key results from three years of hydrochemical monitoring from the pilot Sandhill Fen Watershed are that hydrological management has a strong influence on the hydrology and major ion chemistry of the system and that salinity increases in some areas when pumping of freshwater through the system is reduced. With a reduction in pump activity in 2014 and 2015, vertical fluxes controlled the hydrology, as there was no natural inflow or outflow. The water table in these years gradually declined throughout the season with increases in response to rainfall. Ion concentrations and electrical conductivity varied within the year, yet the mechanisms for this change based on major ion composition (SAR) was principally evapo-concentration and dissolution of salts in the unsaturated zone when water tables rose. Additionally, several Na^+ enriched zones appeared at the interface between the base of the hummocks and the wetland, where oil sands process water from depth was emerging at the surface. These zones had markedly higher Na^+ concentrations than the rest of SFW and matched the isotope signature of oil sands process water. Considering that the Sandhill Fen Watershed is newly constructed, it is expected that its chemistry will change over decadal time scales as transport process redistribute ions within the landscape in response to the predominant hydrological processes. In addition, Sandhill Fen Watershed in time will be integrated with the broader reclaimed landscape, providing a surface outlet that will provide the ability for surface waters to flush during high flow events. Future work will continue to monitor the long-term evolution of the Sandhill Fen Watershed to provide guidance and set expectations for integrated watershed reclamation programs in the Athabasca oil sands region.

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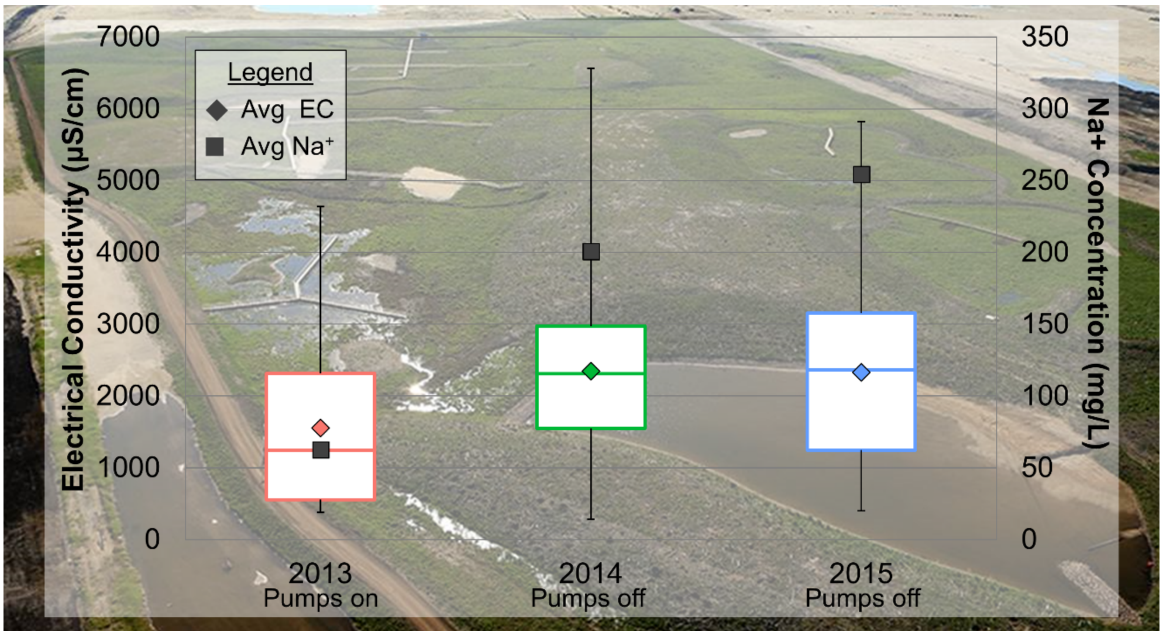
Tables

Table 4.1. Average major ion concentrations and electrical conductivity (EC) values with standard deviations from 2013-2015.

Site	Year	Na^{+} (mg/L)	K^{+} (mg/L)	Mg^{+2} (mg/L)	Ca^{+2} (mg/L)	Cl^{-} (mg/L)	SO_4^{-2} (mg/L)	Alkalinity (mg/L)	HCO_3^{-} (mg/L)	EC (µS/cm)	SAR
WSP	2013	46 ± 49	2.2 ± 1.4	15 ± 7	57 ± 25	39 ± 27	83 ± 81	68 ± 38	83 ± 47	~500[1]	1.1 ± 0.8
	2014	157 ± 76	3.1 ± 0.9	27 ± 8	93 ± 22	115 ± 51	357 ± 138	111 ± 19	136 ± 23	1481 ± 546	2.2 ± 2
	2015	356 ± 172	3.6 ± 0.9	42 ± 16	132 ± 44	120 ± 89	347 ± 240	104 ± 28	127 ± 34	2076 ± 560	5.9 ± 1.9
Lowlands	2013	56 ± 52	2.9 ± 2.9	29 ± 35	105 ± 109	41 ± 28	218 ± 352	115 ± 74	141 ± 90	792 ± 616	1.1 ± 0.7
	2014	122 ± 60	3.2 ± 2	51 ± 27	195 ± 123	76 ± 31	574 ± 434	176 ± 78	214 ± 95	1718 ± 546	2.2 ± 1.1
	2015	130 ± 109	2.3. ± 2	52 ± 30	204 ± 125	102 ± 90	517 ± 336	196 ± 90	380 ± 132	1785 ± 929	1.9 ± 1.5
Margins & Uplands	2013	121 ± 60	3.2 ± 0.9	94 ± 52	402 ± 200	103 ± 31	1123 ± 672	216 ± 87	263 ± 107	1587 ± 1437	1.2 ± 0.3
	2014	153 ± 119	5.4 ± 3.3	89 ± 40	446 ± 161	145 ± 157	1170 ± 488	301 ± 129	367 ± 158	2593 ± 971	1.8 ± 1.2
	2015	249 ± 165	4 ± 2.1	93 ± 42	420 ± 135	235 ± 185	996 ± 386	312 ± 108	370 ± 145	2632 ± 1279	2.4 ± 1.7
Na^{+} Enriched Zones	2013	-	-	-	-	-	-	-	-	-	-
	2014	431 ± 159	6.8 ± 4	54 ± 19	256 ± 134	303 ± 127	898 ± 414	351 ± 97	428 ± 119	2759 ± 695	6.6 ± 2.0
	2015	491 ± 120	7.2 ± 6	66 ± 16	261 ± 66	294 ± 148	960 ± 415	371 ± 57	452 ± 70	3034 ± 628	6.2 ± 1.5
Outlet	2013	273 ± 223	7.0 ± 3.5	25 ± 9	98 ± 35	173 ± 137	343 ± 208	160 ± 133	336 ± 251	2567 ± 984	5.2 ± 4.6
	2014	246 ± 175	5.4 ± 3	33 ± 8	125 ± 30	166 ± 85	412 ± 120	282 ± 88	368 ± 155	1548 ± 141	5.3 ± 4.4
	2015	90 ± 47	3.1 ± 1.04	24 ± 10	99 ± 31	83 ± 43	227 ± 95	190 ± 73	232 ± 89	931 ± 103	2.0 ± 0.8

[1]Values obtained from Wytrykush *et al.*, 2012.

Figures



Graphical Abstract. Wetland electrical conductivity and Na^+ concentrations from 2013 – 2015.

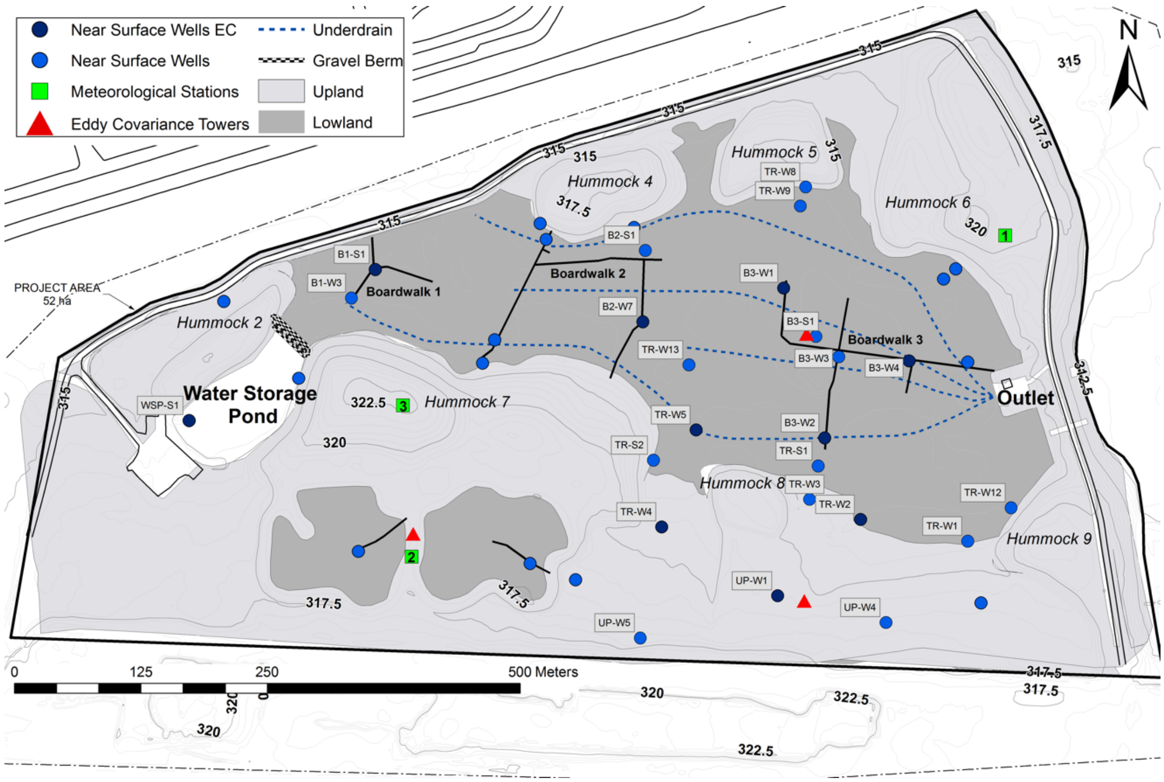


Figure 4.1. Instrumentation map of Sandhill Fen Watershed. Dark grey areas represent the lowland (wetland) area and lighter grey represents the margins and upland areas. Well locations with a black dot represent locations with continuous electrical conductivity (EC) measurements in addition to water level and temperature.

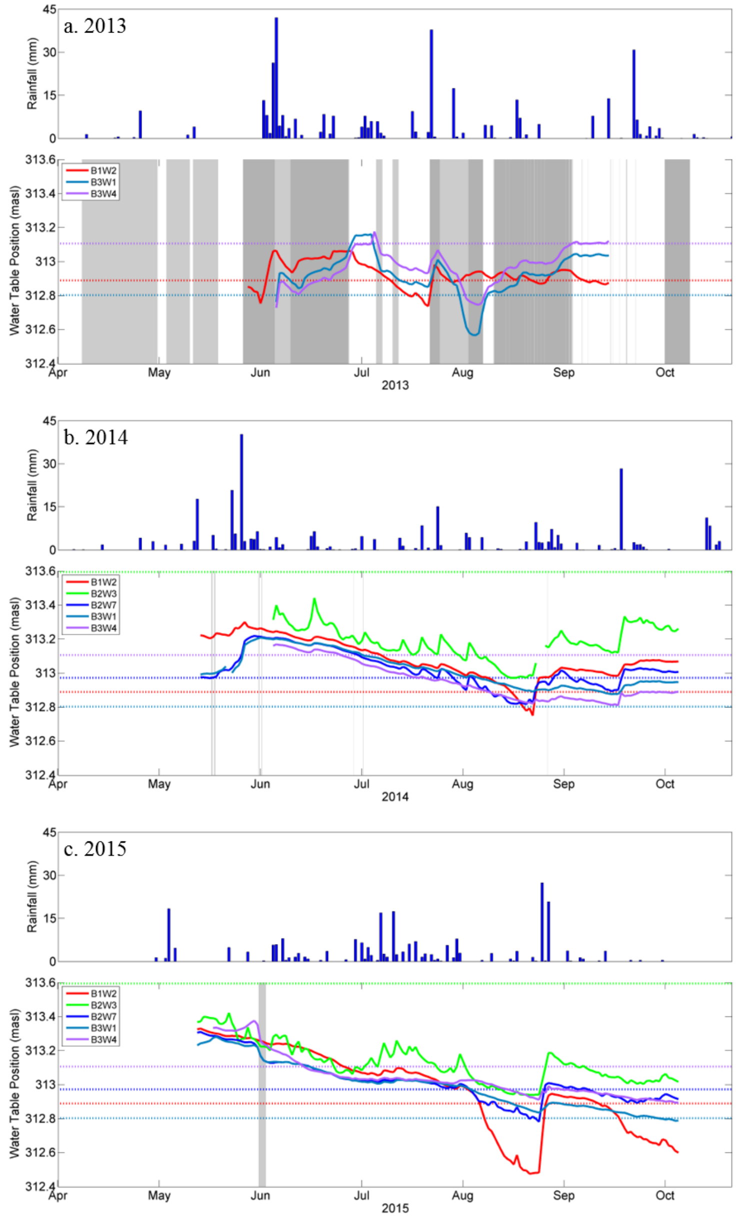


Figure 4.2. Water table position (masl) along the lowland transect from April to October in a) 2013, b) 2014 and c) 2015 (tick marks indicate the start of each month). Dashed lines represent the ground surface of each site and the dark and light grey boxes depict when the inflow and outflow pumps were on, respectively. The suffix B represents the well position on boardwalks one to three (B1, B2, B3), followed by the well number (W1, W2 etc.).

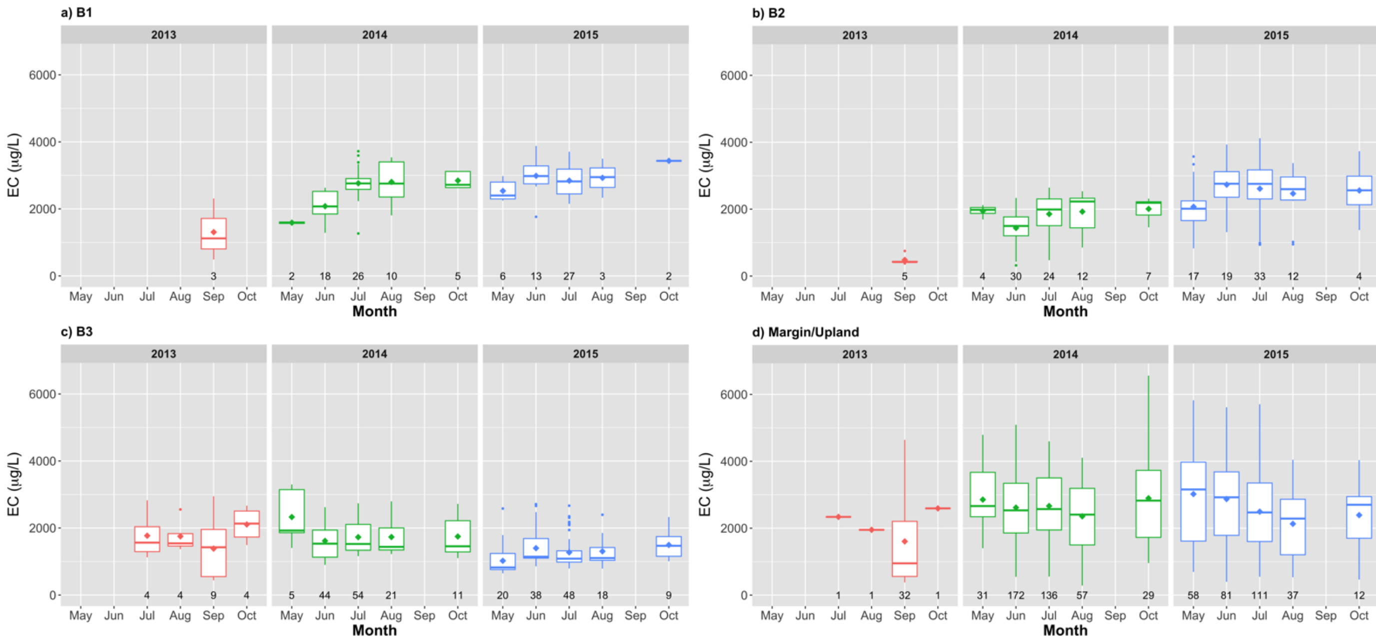


Figure 4.3. Monthly electrical conductivity (EC) from 2013 – 2015 at boardwalks a) B1, b) B2, c) B3 and d) margins/uplands. The horizontal lines, diamond points and circular points represent the median, mean and outliers respectively. The numbers below the box plots represent the sample size.

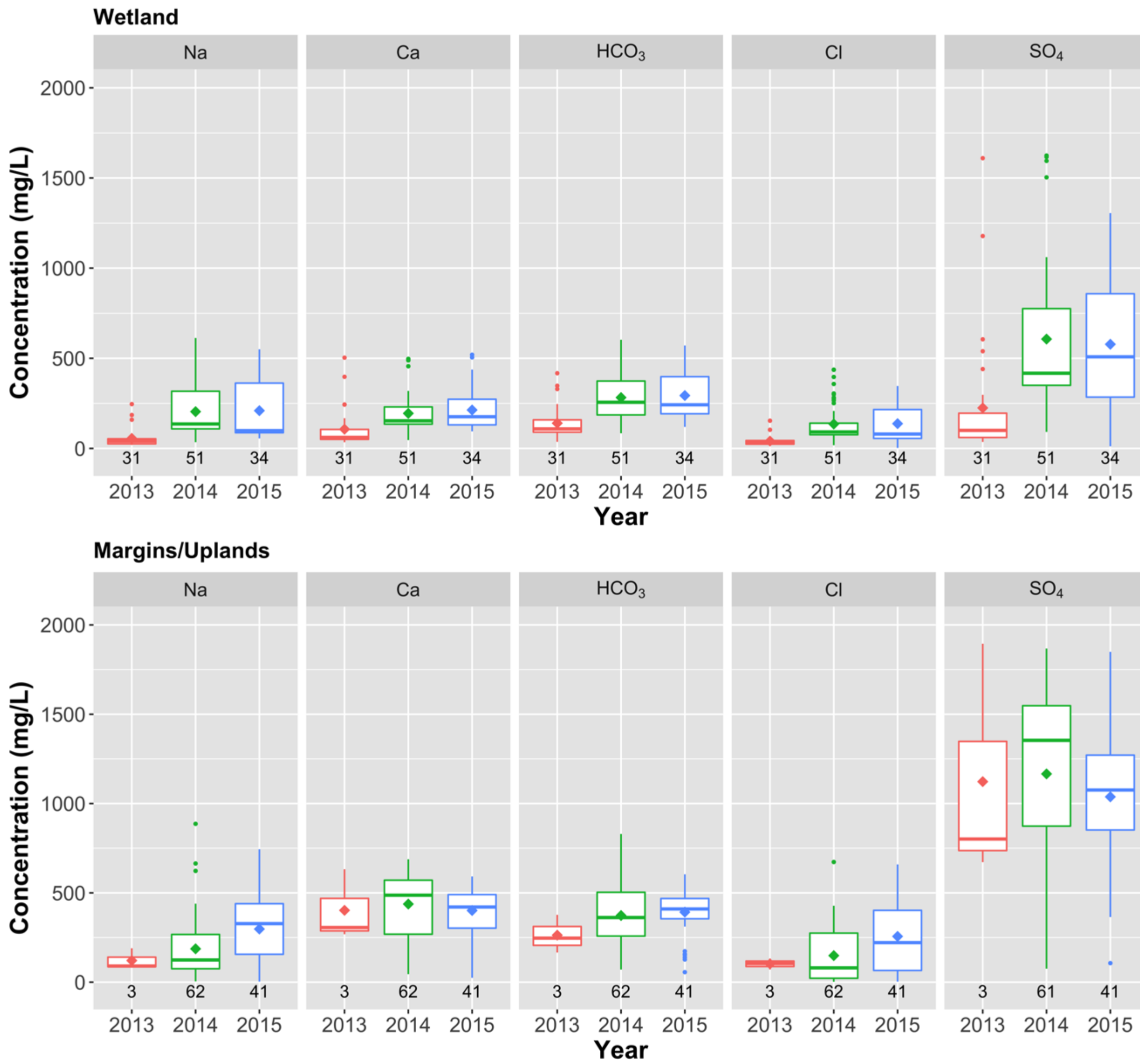


Figure 4.4. Ion concentrations in the wetland and margins/uplands from 2013 – 2015. The horizontal lines, diamond points and circular points represent the median, mean and outliers respectively.

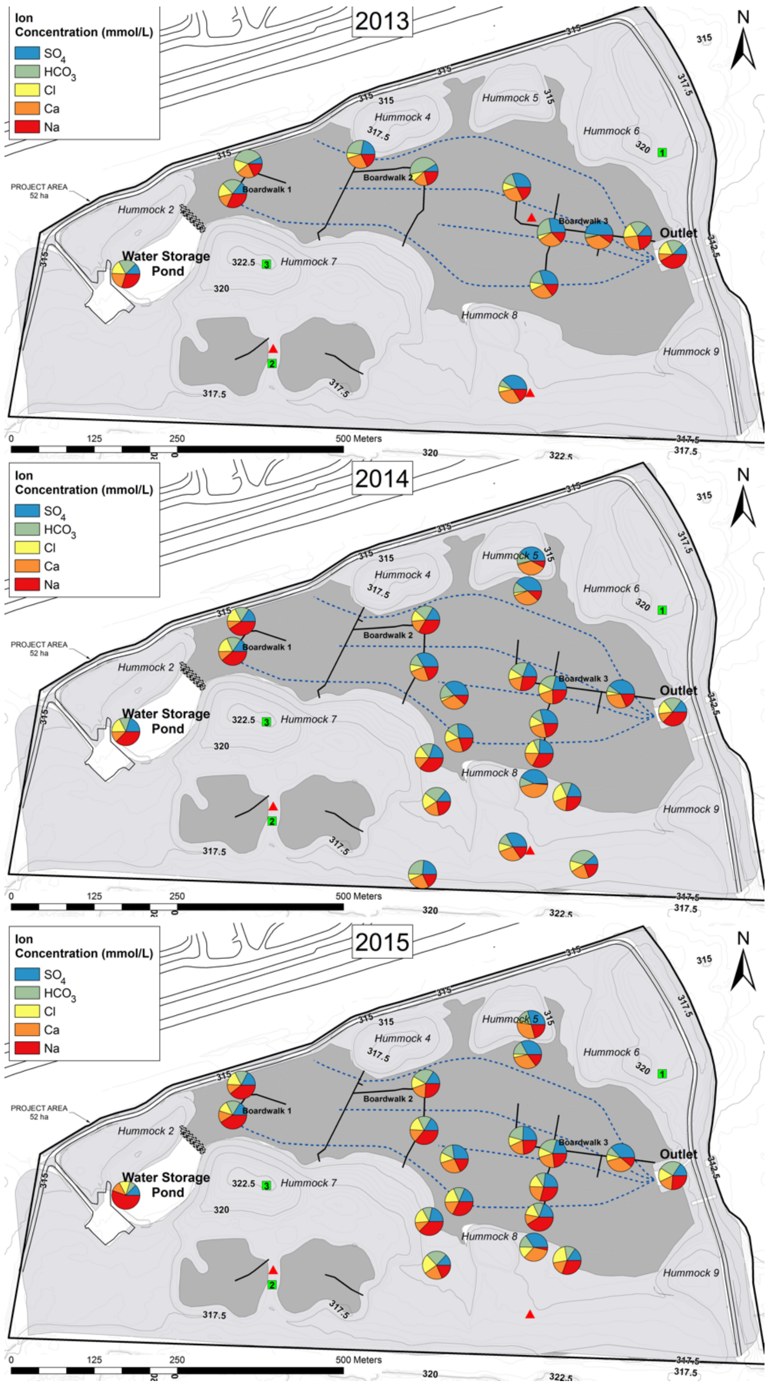


Figure 4.5. Average major ion concentrations at each water sampling site in 2013, 2014 and 2015. Note that concentrations are in mmol/L in order to compare the relative abundance of ions.

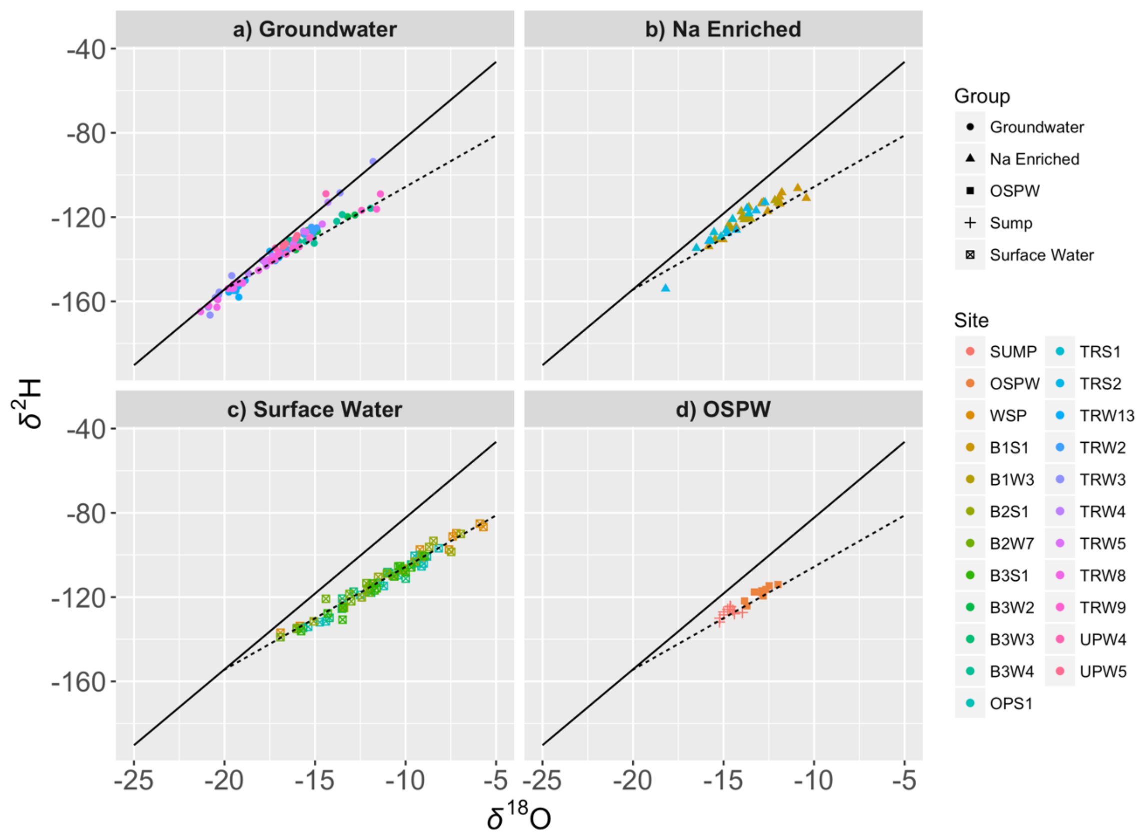


Figure 4.6. Average stable isotope concentrations (δ^{18}O and δ^2H) of a) groundwater, b) Na^+ enriched zones, c) surface water and d) oil sands process water (OSPW) from 2014 and 2015.

CHAPTER 5

HYDROCHEMICAL DEVELOPMENT IN A CONSTRUCTED PEATLAND IN A POST-MINING LANDSCAPE SIX YEARS AFTER CONSTRUCTION: SHORT-TERM AND LEGACY EFFECTS OF ELEVATED SALINITY

ABSTRACT

Reclamation of forest and wetlands is legally required in the Athabasca oil sands region following bitumen extraction via surface mining which leaves large open pits that are backfilled with saline tailings waste. This makes reclamation challenging, as constructed ecosystems need to be capable of sustaining itself in a highly saline and disturbed landscape within a dry and variable climate. To date, two upland-wetland systems have been constructed in the AOSR, and research has focused on the immediate hydrological, biogeochemical and soil physical changes following construction. The objective of this work was to understand the development of salinity and hydrochemical patterns in a constructed peatland watershed following a heavily managed hydrologic regime to provide insight on its potential for long-term success as a peat-forming wetland. Six years of hydrochemical data (2013 – 2018), including water quality parameters, major ion concentrations and stable isotopes of water, are presented from the Sandhill Fen Watershed (SFW), a 52-ha upland-wetland catchment that was built with a pump system to provide freshwater and enhance drainage to limit salinization from the highly saline soft tailings the system is built upon. In general, electrical conductivity (EC) increased throughout SFW over the study period as a result of 1) reduction of inflow and outflow, 2)

changes in water table positions and 3) increased mixing of site-wide waters. Inflow and outflow dominated the water balance in 2013 which kept salinity relatively low. Following 2013, vertical fluxes became dominant and the wetland and margins observed year-over-year increases in EC by 1585 and 2313 μS/cm in the wetland and margins, respectively from 2013 to 2018. The uplands were the only region to observe a decrease in EC, by 1747 μS/cm over the same time frame. Spatially, EC was variable throughout SFW and gradients of salinity developed throughout the wetland and across landscape units. In the wetland, EC was highest in the west wetland which decreased towards the east wetlands but when considering site-wide EC, it was highest in the margins and lowest in the uplands. With a decrease in outflow following 2013, water table position increased across SFW which were significantly negatively and positively correlated with EC in the wetlands and margins, respectively (p-val < 0.05). In addition to these salinity patterns, the chemical composition of SFW waters shifted over the study period from largely Ca-dominant in 2013 (>90 % of samples) to Na-dominant in 2018 (>70 % of samples) and became increasingly similar to the chemical composition of the underlying tailings waste (Na-Cl dominant). Stable isotopes of water support this claim, as they indicate a decreased influence of evaporation and increased mixing across SFW over time. Based on its current conditions, SFW cannot support freshwater peat-forming vegetation and is most chemically similar to brackish marshes and saline fens. Shifts in design strategies are required for constructed peatland watersheds to increase their chance of success in this highly saline and disturbed post-mining landscape.

5.1 INTRODUCTION

The current disturbance footprint from bitumen extraction via surface mining in the Athabasca oil sands region (AOSR) is approximately 900 km^2, and has to potential to increase up to 4800 km^2 if all current leases are utilized (Government of Alberta, 2020). During the surface mining processes, entire landscape surfaces are removed and excavated up to 75 m deep to access the bitumen-rich McMurray Formation (Government of Alberta, 2020), leaving large open pits in the post-mining landscape. To mitigate this disturbance, the Environmental Protection and Enhancement Act legally requires oil companies to reclaim these landscapes back to pre-disturbance functioning capability (Province of Alberta, 2020). As part of the reclamation strategy, these pits are backfilled with tailings waste material, referred to as composite tailings, which are a saline non-segregating slurry containing tailings sand, fine tailings and oil sands process water (OSPW). Composite tailings have elevated salinity from naturally-occurring saline sediments disturbed during mining (contributes Na^+ and Cl^-) (Kessler et al., 2010; Leung et al., 2003; Lord & Isaac, 1989) and from chemicals (NaOH and CaSO4) used during bitumen recovery (contributes Na^+, Ca^{+2} and $SO4^{-2}$) (Chalaturnyk et al., 2002; Matthews et al., 2002). Since these composite tailings are used to backfill open pits as the first step of the reclamation process, there is high potential for salinization of constructed ecosystems in the AOSR.

Wetlands are an important component of the reclaimed landscape as they comprise of approximately half of the pre-disturbance Western Boreal Plains landscape, with >90 % considered peatlands (Rooney et al., 2012; Vitt et al., 1996). The natural stratification of the undisturbed organic soil profile enables peatlands to perform many important ecosystem functions and feedbacks (Nwaishi, Petrone, Price, & Andersen, 2015; Waddington et al., 2015), including water regulating functions that are critical for adjacent ecosystems as the AOSR is characterized by a sub-humid climate with long-term water deficits (Devito, Creed, Gan, et al., 2005; Devito et al., 2012; Smerdon et al., 2005, 2007). Constructing peatlands comes with an array of challenges (Ketcheson et al., 2016a), as their critical ecosystem functions and feedbacks typically form over millennia (Clymo, 1983; Nwaishi, Petrone, Price, & Andersen, 2015) and there are no established methods for their creation following surface mining. The potential for ecosystem salinization adds additional complexity and barrier towards long-term success of constructed peatlands in the AOSR.

While water quality may not directly influence hydrological function, it has an important influence on the microbial (Leung et al., 2003) and vegetation communities present (Lilles et al., 2010; Trites & Bayley, 2009b), which in turn affect peat accumulation, hydrologic properties and long-term peatland development (Trites & Bayley, 2009a). The movement of water throughout this landscape is inextricably linked to the transport of solutes in reclaimed landscapes in the AOSR. Salt ingress into reclaimed systems is of concern for their long-term success as elevated salinity can negatively impact vegetation growth and peat accumulation (Glaeser, Vitt, & Ebbs, 2016; Lilles et al., 2010; Pouliot, Rochefort, & Graf, 2012; Rooney & Bayley, 2011; Trites & Bayley, 2009a, 2009b; Vitt et al., 2020). Despite evidence of salt ingress in reclaimed forested upland systems, successful vegetation growth has been sustained as salt accumulation is maintained at depths beyond the rooting zone or is limited to low slope positions (Kelln et al., 2008; Kessler et al., 2010).

Elevated salinity and Na^{+} concentrations are a greater concern in reclaimed lowland and peatland ecosystems due to the higher WT positions associated with wetlands. Saturated soils can promote upward diffusion from the strong concentration gradient between the composite tailings at depth and the relatively fresh water at the surface (Bailey, 2001; Kessler et al., 2010; Merrill et al., 1983). Evapotranspiration can draw water and associated solutes towards the surface (Merrill et al., 1983; Moran et al., 1990; Rezanezhad, Andersen, et al., 2012) where evapo-concentration can increase salinity (Chapter 4; Simhayov et al., 2017). There is also potential for advective transport throughout the peatlands and the broader reclamation landscape from areas influenced by deeper saline groundwater systems (Kessel et al., 2018; Vessey, Lindsay, & Barbour, 2019). While peat structure can buffer the effects of saline waters via ion attenuation from the polarity and presence of closed and dead-end pores (Hoag & Price, 1995; Rezanezhad et al., 2016a), peat used in reclamation quickly reaches its buffering capacity due to the elevated ion concentrations and degraded peat material (Chapter 2; Rezanezhad, Price, & Craig, 2012; Vessey et al., 2019). Additionally, cation exchange can release adsorbed Na^{+} back into solution as higher valent ions, such as Ca^{+2} and Mg^{+2}, have a higher affinity to soil particles (Carroll, 1959). Evidence of a combination of these mechanisms has already been observed in reclaimed peatlands as salinities and ion concentrations have increased considerably since construction (Chapter 4; Kessel et al., 2018; Simhayov et al., 2017; Vessey et al., 2019) that are considerably higher than undisturbed peatlands in the AOSR (W.-L. Chee & Vitt, 1989; Trites & Bayley, 2009b; Vitt & Chee, 1990b). As a result, shifts in the reclaimed vegetation communities have already been observed (Borkenhagen & Cooper, 2019; Vitt et al., 2020, 2016b) as the native peatland vegetation originally planted cannot persist in saline waters.

Currently, only two constructed peatland watersheds exist within the AOSR (Daly et al., 2012; Wytrykush et al., 2012) and with the likelihood of salinization, it is critical to understand the most effective design strategies that can effectively flush salts while limiting their impact on the ecosystem and promote peatland development. The objectives of this research are to use the first six years of hydrochemical data from a recently constructed peatland to: 1) examine changes in inorganic solute distribution, 2) understand the influence of the dominant hydrological fluxes on the concentration and distribution of inorganic solutes and 3) use stable isotopes of water to identify major source waters and their interactions. This work advances upon that presented in Chapter 4 which assesses the early hydrochemical patterns as a result of management practices while this study assessed the longer-term patterns since active management ceased.

5.2 SITE DESCRIPTION

The Sandhill Fen Watershed (SFW) was constructed in the northwest corner of a formerly mined area (1977-1999) called East-In-Pit and is part of Syncrude Canada Ltd.'s Base Mine (57°02'N, 111°35'W) located approximately 40 km north of Fort McMurray, Alberta. This region is situated in the WBP ecoregion of western Canada which is characterized by a cold and sub-humid climate where mean annual potential evapotranspiration (607 mm) exceeds mean annual precipitation (456 mm), with 342 mm of precipitation falling predominantly as rain between May and September (Environment

Canada, 2020; Wytrykush et al., 2012). Thirty-year climate normal data (1981-2010) for Fort McMurray indicates that average annual daily temperature is 0.7 ± 1.2 °C where lowest and warmest temperatures typically occur in January averaging -18.8 ± 5.0 °C and July averaging 16.8 ± 1.1 °C, respectively (Environment Canada, 2020).

The SFW was built atop 35 m of composite tailings (containing OSPW) followed by a 10 m tailings sand structural cap. The SFW comprises of a 17 ha fen wetland, 35 ha upland including 20 ha of upland hillslopes that vary in elevation from 3-8 m above the wetland (referred to as hummocks) (Figure 5.1) (Wytrykush et al., 2012). There is a total of seven hummocks which were constructed of mechanically-placed tailings sand and were designed to create upland recharge areas that could supply water to the lowland area. Excluding the hummocks, 0.5 m of clay was laid on top of the tailings sand cap to maintain water table (WT) position and to attenuate upward migration of salts from the CT layer, followed by 0.5 m of peat mineral mix and 0.5 m of peat in the uplands and wetland, respectively. Peat placed in the lowlands was harvested and salvaged in the fall 2010 from the upper 0.6 m of a poor fen north of Syncrude's Base Mine and was placed with a bulldozer in January 2011. Inflow and outflow can be managed via a pump system that was installed during construction, intended to maintain wetness and mitigate the elevated salinity. In addition to precipitation and groundwater, water can be supplied to the clay-lined water storage pond (referred to as the inlet) from a nearby undisturbed lake (Mildred Lake) which gradually flows through a gravel dam into the lowland area of SFW. It should be noted that outflow of surface and near-surface water through a V-notch weir can only occur when the outlet pump is activated. Details on the pump system can be found in Chapter 4 and Nicholls *et al*., 2016.

5.3 METHODS

5.3.1 Field measurements

A network ~30 near-surface PVC slotted wells were installed during 2013 – 2014 throughout SFW (Figure 5.1). Ten Solinst LTC (level, temperature, conductivity) Levelogger Junior pressure transducers were deployed in these near-surface wells down the major near-surface flow pathways which took measurements every 15 minutes from May to October each year from 2014 – 2018 (Figure 5.1). These measurements were only made in five wells surrounding Boardwalk 3 in 2013 before the network of wells was expanded in 2014. Manual measurements of water level were made monthly at minimum for quality assurance and control. Solinst level measurements were corrected for barometric pressure using a Solinst Barologger. To increase the spatial representation of water quality parameters, discrete measurements of temperature, electrical conductivity and pH were taken monthly at minimum using a YSI Professional Plus Multiparameter or YSI Pro DSS instrument at over 50 surface and well sampling sites (Figure 5.1). All EC values, both discrete and continuous, were corrected to 25 °C (Hayashi, 2004). The YSI instruments were calibrated before each use.

Wells sampled for chemical analysis varied slightly each year as sampling strategy changed over the years (Table B1). Sampling protocol for major ions and stable hydrogen and oxygen isotope samples is outlined in Chapter 4 unless stated otherwise. Sampling frequency decreased from bi-weekly during 2013 – 2015 to monthly from 2016 onwards. Samples measured during pumping events from each boardwalk are indicated on Figure 5.1. Duplicate samples were taken every ten samples and both field and laboratory blanks were included in each sampling campaign for quality assurance and quality control. Wells were named according to their position within SFW, where the suffix B represents wells along boardwalks one to three, TR represents wells in the margins (transition) and UP represents wells in the uplands (Figure 5.1).

5.3.2 Laboratory sample preparation

Water samples were treated and filtered with the same laboratory protocols outlined in Chapter 4. Filtered samples were analyzed for major ions (Na^+, K^+, Mg^{+2}, Ca^{+2}, Cl^- and SO_4^{-2}) using ion chromatography with Dionex ICS 6000, IonPac AS18 and CS12A analytical columns at the Biogeochemistry Laboratory, University of Waterloo. The minimum detection limit was 0.2 mg/L for Na^+, K^+ and Mg^{+2} and was 0.1 mg/L for Ca^{+2}, Cl^- and S-Sulfate, N-Nitrate and N-Nitrite. As part of the analytical quality assurance and control procedures, 5% of samples were re-run as duplicates. HCO_3^- concentrations were calculated by multiplying the alkalinity values by a conversion factor of 1.22 (Csuros, 1994). Stable isotope ratios of hydrogen and oxygen were determined using a Los Gatos Research DTL-100 Water Isotope Analyzer at the University of Toronto. Five standards of known isotope composition, with δ^2H ranging from -154‰ to -4‰, purchased from Los Gatos Research were used for calibration, in addition to periodic checks using the international standards, VSMOW2 and SLAP2. During analytical runs, samples were interweaved with standards at a ratio of 3:1.

5.3.3 Data analysis

Ion concentrations were converted from mg/L to mmol/L or meq/L to compare the relative abundance of ions in Piper diagrams, as the molecular weight for each ion is incorporated into the converted values. Stable isotope of water results (δ^2H and $\delta^{18}O$) were plotted against a local meteoric water line (LMWL) that was developed for Syncrude Canada Ltd.'s Mildred Lake Base Mine along with mine source waters (Baer, 2014; Baer et al., 2016). A local evaporation line (LEL) was developed in Chapter 4 and is used in this study. The line-conditioned excess (LC excess) was calculated for all isotope samples (Equation 1) to infer the influence of evaporation (Landwehr & Coplen, 2006).

$$\text{LC excess} = \delta^2H - a \times \delta^{18}O - b \quad (1)$$

Variables a and b represent the slope and intercept of the local meteoric water line (LMWL) and concentration of δ^2H and $\delta^{18}O$ are in per mil (‰). Increasingly negative LC excess values indicate a greater evaporative signal.

5.4 RESULTS

5.4.1 Salinity

5.4.1.1 Annual patterns

In general, EC has increased across the SFW and the occurrence of wet and dry years (Chapter 2) had little influence on the observed year-over-year increase (Figure 5.2). Overall, average wetland EC increased by ~1600 µS/cm from 2013 to 2018 (~530 µS/cm every two years) (Table B2). The three boardwalk areas within the wetland had distinct EC patterns, where the highest average increase in EC was observed at B1 and B2 of ~2500 and 2400 µS/cm, respectively from 2013 to 2018 with some stabilization after 2016 (Figure 5.2a & b). In contrast, average EC at B3 was similar over the study period where EC only increased by ~500 µS/cm from 2013 to 2018 (Figure 5.2c). The average margin EC increased by ~2300 µS/cm from 2013 to 2018 and appears to have stabilized since 2016 (Figure 5.2d). Annual EC maximums were considerably higher in the margins than the wetlands and reached 8675 µS/cm in 2018. Unlike the wetlands where a year-over-year increase was typically observed, the highest average EC in the margins was observed during wet years (2016 and 2018) of 4272 and 3922 µS/cm, respectively, whereas dry years (2014, 2015 and 2017) were considerably lower averaging 2765, 2911 and 3531 µS/cm, respectively (Figure 5.2; Table B2). The upland areas were the only region that exhibited a decrease in annual EC across study years, where values decreased from 3870 µS/cm in 2013 to 2123 µS/cm in 2018, however the sample size was much smaller (Figure 5.2e).

While there was an overall pattern of increasing salinity, EC was highly variable across all regions in the SFW, as some areas remained relatively fresh (<1000 µS/cm) while other areas exceeded 8500 µS/cm (Figure 5.3). Through the first six years, a salinity gradient developed along two main transects: 1) across the boardwalk down the centre of the wetland, and 2) from the south upland swale towards the north margins of SFW (Figure 5.3). In 2013, there was little gradient across the wetland as all boardwalk areas had similar average salinities (~1000 µS/cm) while average EC was highest in the uplands (3870 µS/cm) and decreased northward to the south margins (2391 µS/cm), the wetland (1580 µS/cm) and to the northern margins close to hummocks 4 and 5 (670 µS/cm) (Figure 5.3a).

The salinity across the wetland transect increased in the following years where EC was highest at B1 and decreased towards B3. ECs in 2015 and 2018 at B1, B2 and B3 averaged 2897 and 3803 μS/cm, 2445 and 3309 μS/cm and 1323 and 2242 μS/cm, respectively (Figure 5.3b, c) where maximum EC at each boardwalk reached 7426, 5498 and 3745 μS/cm, respectively in 2018. Following 2013, EC patterns in the transect from the southern upland swale to the northern margins shifted, where EC was lowest in the uplands, averaging 1295 and 993 μS/cm in 2015 and 2018, respectively and highest in the southern margins averaging 3984 and 5230 μS/cm in 2015 and 2018, respectively. The average wetland and northern margin EC were intermediate and averaged 1563 and 2710 μS/cm in the wetland and 2029 and 2007 μS/cm in the northern margins for 2015 and 2018, respectively (Figure 5.3).

5.4.1.2 Seasonal patterns

Compared to the year-over-year changes, drivers of seasonal salinity changes were subtle as the influence of precipitation and evapo-concentration sometimes offset one another in addition to any pump activity (Figure 5.4). Wetland and margin EC had opposing responses to changes in WT from precipitation (average EC of 45 μS/cm) and evaporation (Figure 5.4). Considering all wetland sites and years, Pearson correlation results indicate a significant negative correlation exists between wetland WT position and EC (p-val < 0.05) where increases and decreases in wetland WTs were followed by a decrease (dilution of ions) and increase (concentration of ion) in EC, respectively (Figure 5.4). This pattern was more evident in dry years (2014, 2015 and 2017) where WTs show a continuous decline, with minimal influences from precipitation, and an increase in EC over the season at all boardwalks (Table B3). In wet years, it was more difficult to identify this pattern when multiple heavy rain events and pumping events obscure evapo-concentration signals. Seasonal increases in EC became greater as the years progressed, as average EC in the wetland from May to October increased by 402, 368 and 865 μS/cm in 2013, 2015 and 2018, respectively (Table B3).

In contrast, margin EC was positively correlated with WT position where increasing WT position increased EC (ion mobilization) and a decrease in WT lowered EC (p-val < 0.05). Following a rise in WT from rain events in both wet and dry years, a clear increase in EC was observed in the margin sites which decreased only when the WT declined (Figure 5.4). Seasonal EC changes were subtle in the margin compared to the wetland where average EC either decreased or showed minimal change from May to October across study years (Figure 5.4). Overall, seasonal changes in margin EC were not consistent throughout the study period as average margin EC increased by 260 μS/cm in 2013, decreased by ~1000 μS/cm in 2015 and increased by 230 μS/cm in 2018. Upland WT position and EC was variable over the study period and were not significantly correlated (p-val > 0.05). The upland seasonal trends were variable where average seasonal EC decreased by ~1700 μS/cm in 2015 (dry year) and increased by ~1500 μS/cm in 2018 (wet year) (Table B3).

5.4.1.2 Influence of the outflow pump on wetland salinity

In early years, the combination of inflow and outflow maintained lower salinities in the wetland. In 2013, the outflow pump was highly active (Chapter 2), and resultant wetland EC was relatively low (Figure 5.4). Following 2013, the outflow pump was only active for pre-planned drainage events in 2015, 2017 and 2018 of 55, 102 and 210 mm, respectively (Chapter 2). While management decreased in 2015, outflow pumping still decreased the salinity at all boardwalks by ~1000 μS/cm, however these values returned back to pre-pumping salinities within two weeks following the pumping event (~2800 μS/cm) (Table 5.1). During the 2017 and 2018 outflow pump events, EC remained similar to pre-pumping values or increased post-pumping (Table 5.1) despite the events being considerably longer (122 and 276 hours, respectively) than the 2015 outflow pumping event (60 hours) (Chapter 2). A 100-year rainfall event (92 mm in 24 hours) occurred shortly after the 2018 pumping event, resulting in the considerably lower EC two weeks following pumping (Table 5.1).

5.4.2 Chemical constituents

Ion concentrations emulated EC patterns with values increasing site-wide year-over-year (Table B2). Of particular importance is the rising Na^+ concentrations compared to other ions. Average Na^+ concentrations were relatively low in 2013, whereas Ca^{+2} typically exceeded Na^+ in early years. Over the study period (2013 to 2018), average annual Na^+ increased by 293, 382 and 161 mg/L in the wetland, margins and uplands, respectively whereas Ca^{+2} increased considerably less over the same time period in the wetland (156 mg/L) and decreased by 35 and 516 mg/L in the margins and uplands, respectively. Average Na^+ and Ca^{+2} concentrations in 2013 were 56, 87 and 190 mg/L and 107, 287 and 632 mg/L in the wetland, margins and uplands, respectively. Na^+ increased considerably by 2018 where average concentrations were 349, 469 and 350 mg/L in the wetland, margins and uplands, respectively, where some sites reached up to 886, 1090 and 570 mg/L, respectively. Comparatively, average Ca^{+2} only moderately increased in the wetland to 263 mg/L from 2013 to 2018 and decreased in the margins and uplands to an average of 252 and 116 mg/L, respectively.

To evaluate the spatial variability in ion concentrations, Piper diagrams were used to identify water facies across the SFW, and show a shift in the dominant chemical constituents from 2013 to 2018 (Figure 5.5). In 2013, Ca-dominant facies comprised >90 % of all samples, mostly as Ca-SO_4 (47 %) and Ca-HCO_3 (41 %), whereas 6 % of samples were Na-dominant (Figure 5.5). This chemical composition matched that of input waters (inflow and rain) which were Ca-HCO_3 dominant. Following 2013, the chemical composition of site-wide waters began to shift where 69 % of samples were Ca-dominant and 31 % were Na-dominant in 2015, and by 2018, 73 %, were Na-dominant and only 27 % of samples were Ca-dominant (Figure 5.5). Additionally, 32 % of samples in 2018 (compared to 2 % and 0 % in 2015 and 2013, respectively) are Na-Cl dominant which is similar to the chemical composition of OSPW. While this shift towards Na-dominant waters is evident across all sites, this pattern varied spatially, where the margins and B1 experienced the greatest shift from Ca- to Na-dominant waters followed by B2, B3 and the uplands (Figure 5.5). These high Na^+ areas corresponded to the highest salinity areas in SFW (Figure 5.4) whereas more Ca-dominant areas correspond with lower EC regions within SFW (Figure 5.4). Compared to other undisturbed sites in the region, much of the SFW waters have a similar chemical composition to that of natural saline fens which are characterized by elevated Na and Cl concentrations (Figure 5.5) (Hartsock et al., 2021). Comparatively, SFW are least chemically similar to brackish marshes, which are dominant in Na and HCO_3^-, as well as extreme rich fens which are chemically similar to precipitation (Figure 5.5).

5.4.3 Stable isotopes of water

Stable isotopes of water indicate that the variability in both δ^2H and $\delta^{18}O$ decreased year-over-year as the SFW becomes more mixed (Figure 5.6). Average δ^2H and $\delta^{18}O$ and their respective standard deviations decreased from -129.2 ± 16.9 ‰ and -15.1 ± 3.2 ‰, respectively in 2014 to -134.6 ± 9.6 ‰ and -16.3 ± 1.5 ‰, respectively in 2018. In 2014 and 2015, samples taken early or late in the growing season plot closely along the LMWL whereas samples taken during June to August, plot along the LEL, suggesting an evaporation signal (Figure 5.6a, b). Although there were few samples in 2016, values plot lower along the LEL, which continued into 2017 and 2018 where most samples plot on the lower portion of the LEL between the isotopic signatures of SFW groundwater and OSPW regardless of sample collection time (Figure 5.6c-e).

To further evaluate source water, mixing, and the influence of evaporation, Na^+ concentrations were plotted against the LC-excess (Figure 5.7). In the wetland, many samples in 2014 and 2015 were isotopically enriched via evaporation as evident from the highly negative LC-excess values (Figure 7a). As the years progressed, the LC-excess values increased and samples shifted upwards along the mixing line between SFW input waters and OSPW. While margin waters had limited evaporative signatures (Figure 7b), samples progressed upwards along the mixing line each year. In general, wetland and margin samples from early years (2014, 2015) plot lower along the mixing line and are closer to the signature for SFW input waters which have migrated towards the chemical signature of OSPW over time.

5.5 DISCUSSION

The SFW was designed as a pilot project to test how peatlands could be constructed on top of composite tailings in the AOSR, and to evaluate how the designed watershed develops in terms of hydrology, water chemistry and vegetation. The ubiquitous salinity in AOSR reclaimed landscapes poses a considerable challenge for reclamation success as excess salinity negatively affects peatland vegetation development and ultimately peat accumulation, overall ecosystem function (Chapter 2) and feedback mechanisms (Waddington et al., 2015). As a result, reclaimed systems are expected to have higher salinity than their natural analogues until these salts are flushed from the system over time (Ketcheson et al., 2016b; Nwaishi, Petrone, Price, & Andersen, 2015). Since salt accumulation in these constructed systems is inevitable, it is critical that these reclaimed peatlands are designed to limit accumulation.

5.4.1 Why is salinity increasing?

Three main mechanisms can explain the observed year-over-year increase in salinity and distribution of ions throughout SFW: 1) a reduction in inflow and outflow, 2) shifts in WT position and 3) increased mixing of site-wide waters with deeper saline OSPW.

5.4.1.1 Reduction of inflow and outflow

The major hydrological change observed during the first six years was the shift in the dominant fluxes related to management (Chapter 2; Chapter 4; Nicholls et al., 2016).

In 2013, inflow and outflow were more than double the precipitation and evapotranspiration (both >800 mm), which kept salinity concentrations relatively low (500 – 1000 µS/cm) as relatively fresh water was supplied to the water storage pond and saline water was removed via outlet pumping (Figure 5.2 & 5.3). EC at the margins was considerably higher than the lowland as they were not directly affected by the pumping (Figure 5.2 & 5.4). Following 2013, the management regime shifted to pump out water only when WTs became high and flooded the boardwalks, and was typically infrequent (once or twice a year after 2014). In 2015, outflow pumping was effective at temporarily lowering wetland EC, which decreased by ~1000 µS/cm before increasing back to pre-pumping EC after two weeks (Table 5.1). In contrast, the 2017 and 2018 pumping events had minimal influence in lowering EC and at some sites resulted in an increase despite greater volume of water removal. Due to the year-over-year accumulation of ions, single-event outflow pumping (in 2017 and 2018) was likely of insufficient duration to flush salts and may have led to mobilization of water into the wetland from the saturated and highly saline margins. It is unlikely that without a consistent outflow, the SFW will be unable to effectively flush salts and lower overall EC.

5.4.1.2 Water table position

Following 2013, WT position primarily responded to precipitation inputs and evaporation where WT position was negatively correlated with EC (p-val < 0.05). During dry years in the wetland (i.e. 2015), evapo-concentration was evident where EC (and consequently ion concentrations) increased throughout the growing season in response to WT decline (Figure 5.3). The only decrease in EC during dry years was during pumping as dilution from precipitation was minimal (Figure 5.4). Additionally, two pumping events were required in 2018 to mitigate flooding which also contributed to lower EC (Figure 5.4). With the combined effects of increased rainfall and pumping, the effects of evapo-concentration were suppressed as ions were diluted from freshwater input (rain and inflow) and flushing via outflow.

In contrast, margin WT position was positively correlated to EC (p-val < 0.05) where ions were mobilized during wet conditions. While this relationship occurred in both wet and dry years, EC was generally higher in wet years as a result of increased ion mobilization associated with higher WTs (Figure 5.4). Most of the margin area borders the base of the hummocks and the south swale, which has been identified as the region where upwelling and lateral advection of saline OSPW occurs as the clay liner does not extend under the hummocks (Chapter 4; Vessey et al., 2019). During wet conditions, an increase in the WT directs water from the hummocks to the wetland (Spennato et al., 2018) and transports saline OSPW water into the wetland. While increases in EC were evident at the margins following an increase WT position, increases in EC were highest in wet years (i.e. 2018). Additionally, the spatial gradient of high salinity in the margins that decreases towards the wetland (Figure 5.3) mirrors deeper groundwater salinity patterns reported for the tailings sand cap that underlies the entire SFW where EC is lowest underneath the wetland (1000 – 2000 μS/cm) and increases towards the margin areas (2000 – 4000 μS/cm) (Twerdy, 2019). The uplands were the only region within SFW that had EC decline with time, suggesting that there was some flushing of salts via recharge, which were transported downgradient to the margins and lowlands.

5.4.1.3 Increased mixing of waters

The shift in chemical and isotopic composition of waters suggests increased mixing across SFW. Piper diagrams indicate that over time many sites, primarily margin and B1 sites that border hummocks, shifted towards the chemical composition of OSPW (Na-Cl dominant). In 2013, >90 % of samples were Ca^{+2} dominant as many of the construction materials used in SFW were high in Ca^{+2} (Chapter 4) and while Ca^{+2} was still the dominant ion in 2015, a shift towards increasingly Na^{+} dominant, particularly in the margins, emerged (Figure 5.5). By 2018, most of the site-wide samples (>70 %) were Na^{+} dominant, indicating considerable presence of OSPW.

This shift in chemical composition is an indication of increased mixing and is supported by the decreased variance in $\delta^{2}H$ and $\delta^{18}O$ with time, implying that surface and near-surface SFW water is becoming increasingly similar (Figure 5.6). Additionally, the evaporative signal (isotopic enrichment) declined over time as fewer samples plot along the LEL. This matches the observed decline in evapotranspiration since 2016 directly measured at the SFW lowland (Chapter 2). To confirm that the rise in Na^{+} concentrations was not solely from evapo-concentration, LC-excess and Na^{+} values for SFW waters were plotted against a mixing line for Mildred Lake water (used to flood SFW in 2013) and OSPW (Figure 5.7). In early years (2014 and 2015), it appears that much of the wetland was influenced by evapo-concentration as indicated by the highly negative LC-excess values. With time, samples shifted away from this evaporative signal and moved upwards on the mixing line towards the signature for OSPW, suggesting increased mixing of SFW waters with OSPW.

Similar mixing was observed in the margins yet there was little signature of evaporated water in the groundwater samples taken (Figure 5.7).

The margin and B1 areas (highest EC and Na^+ regions) are at the base of the hummocks (hillslopes) and are areas where saline groundwater and OSPW emerges at the surface (Chapter 2; Chapter 4; Vessey et al., 2019). The clay liner that extends throughout the upland swale, margins and wetland which was intended to limit the upward flux of OSPW, is absent from the hummocks (constructed of tailings sand) which provides windows for OSPW to be transported to the surface and spread to other regions of SFW (see Figure 2.9 in Chapter 2). In general, solutes originated from the OSPW at depth are likely mobilized into the 10 m tailings sand cap via diffusion due to the strong concentration gradient between OSPW (>4000 μS/cm) and the overlying groundwater which flows northeast (ranges from 1000 – 4000 μS/cm) (Twerdy, 2019). Advection is likely the primary mechanism of solute transfer of saline groundwater into SFW at the base of the hummocks and then mixing throughout the system which diffusion through the clay and peat soil layers is a slow process and was likely a minimal contributor to the overall salinity. While these soil materials have the ability to remove solutes from water via adsorption, their capacity has been reached as Na^+ concentrations are high and continue to rise throughout SFW (Vessey et al., 2019).

5.4.2 What is the fate of SFW?

Elevated salinity will be a lasting legacy in SFW and across AOSR due to the sheer volume of highly saline tailings waste that is incorporated back into the reclaimed landscape. While reclaimed forested systems have exhibited resilience to elevated salinity and Na^+ due to the lower WTs (Lilles et al., 2010), wetlands are more susceptible to salinization as near-surface saturation enhances ion mobilization to the surface (Kelln et al., 2008; Merrill et al., 1983). Considering the total volume of OSPW underneath the SFW (2.25×10^7 m^3) and the average Na^+ concentration of OSPW (1005 mg/L), there is approximately 2.25×10^{10} g of Na^+ within the footprint of the SFW at the time of commission. Based on the current volume and chemistry of water that has been flushed from the SFW via outflow from 2013 to 2018 (2.2×10^5 m^3), only 0.2 % of Na^+ from the underlying OSPW has been flushed. The capacity for SFW to remove salts within the wetland is limited by the adsorption capacity of the soils, the lack of flushing capability via outflow and discharge of OSPW at the base of the hummocks in wet conditions. While peat has the capacity to immobilize salts due to the high adsorption capacity and dead-end and closed pores (Hoag & Price, 1995; Rezanezhad, Price, et al., 2012; Simhayov et al., 2017), this ability is limited in reclaimed peat soils due to the relatively small volume of peat (0.5 m), degraded properties and the abundance of ions present in the system (Vessey et al., 2019). This, in addition to minimal flushing via outflow, ions will only continue to accumulate in SFW waters as there are no mechanisms of ion removal to mitigate the elevated ion concentrations.

Evidence has materialized that the SFW wetland is developing marsh-like characteristics based on the current WT position (Chapter 2), shift in vegetation (Vitt et al., 2020, 2016b), degraded peat properties (Chapter 2) and continual increases in salinity (Figure 5.4) (Chapter 4). In general, vegetation communities and productivity decline with increasing salinity and sodium concentrations (Phillips, Petrone, Wells, & Price, 2016;

Trites & Bayley, 2009b, 2009a) and while some species can tolerate saline and sodic waters (Nwaishi, Petrone, Price, & Andersen, 2015; Rezanezhad et al., 2016b; Vitt et al., 2020; Wells & Price, 2015b), many peat-forming species, such as mosses, cannot. Increases in WT position and soil moisture have been linked to increases in ion mobilization via both diffusion and advection (Hoag & Price, 1995; Kelln et al., 2008; Kessler et al., 2010), which is a key characteristic of peatlands where the ideal WT position is close to the surface. The salinities and Na^+ concentrations reported in SFW have long exceeded that of natural freshwater peatlands in the region (W.-L. Chee & Vitt, 1989; Hartsock et al., 2021; Vitt & Chee, 1990b) and are most similar to that of brackish marshes and saline fens (Hartsock et al., 2021; Wells & Price, 2015b) that naturally occur in Alberta.

Climate change is an additional barrier for the success of reclaimed peatlands as the potential ecosystem response to warmer and wetter climate (Meehl et al., 2007) could lead to an overall decline in peatland WT positions and a subsequently drier environment from an increase in evapotranspiration (Ireson et al., 2015; Thompson et al., 2017). While natural peatlands have water-conserving feedback mechanisms that could potentially mitigate wetter or drier conditions (Waddington et al., 2015), these mechanisms are currently absent in constructed peatlands (Chapter 2) which put them at much greater risk for continued salinization over time. While an increase in precipitation delivered as rain could increase dilution in the wetland, higher WTs in the margins would increase mobilization of ions from the saline OSPW to the surface and subsequently spread throughout SFW. Comparatively, drier conditions could continue to enhance ion concentrations in constructed peatlands as WT decline will amplify via further evapo-concentration and reduced snowpacks will greatly reduce the annual freshwater input during spring snowmelt (Chapter 3). The absence of these feedback mechanisms, in addition to the limited flushing capability, put SFW at risk for enhanced saline conditions regardless of wetter or drier conditions with the onset of climate change.

5.4.3 Recommendations

Elevated salinity is an unavoidable and long-lasting barrier towards constructed peatland success in the AOSR and designing landscapes capable of flushing these salts without accumulating excessive amounts over time will be the key to their long-term success. While using a deeper peat/soil column during construction would increase the adsorption capacity and removal of ions, this is only a temporary solution as soil adsorption capacities are quickly reached due to the elevated concentrations (Vessey et al., 2019). Following the recommendations in Chapter 2, a general shift in the design strategy for constructed peatlands could help the overall success of these systems which include: 1) increase the flushing capacity of the SFW via enhanced recharge and a naturally-flowing outlet, 2) adjust the current soil design to extend the clay liner beneath the entire peatland-watershed and increase the depth of the peat column and 3) shift design goals to better match the outcome in a disturbed landscape. First, enhancing system recharge via constructing larger hummock structures with high infiltration soils will increase dilution and limit the upward movement of saline water (Chapter 3; Kelln et al., 2008; Lukenbach et al., 2019), while a constructed outlet that is consistently flowing will help remove solutes from the system and limit ion accumulation in the wetland. Secondly, extending the clay liner beneath the hummocks will help limit the formation of Na^+ enriched areas where

OSPW from depth is emerging at the surface. This will also help direct recharge water directly to the margins and wetland and limit recharge water lost to deep percolation. Additionally, a deeper peat column in the fen area (greater than the current 0.5 m) could encourage more attenuation of ions from the saline water. The Nikanotee Fen has both of these components and as a result have shown slightly lower salinities and influence of OSPW (Kessel et al., 2018; Ketcheson et al., 2017b). Lastly, incorporating perched or upgradient peatland systems within the reclaimed landscape, which was part of the original SFW design, may reduce potential for salinization as the upgradient areas in SFW have lower salinities as ions have been flushed from them over time (Figure 5.3). By constructing peatlands upgradient of a marsh system, excess water and salts can run off and collect in the marsh where vegetation species tolerant to saline waters and inundated conditions can thrive (see Figure 2.9 in Chapter 2). With a mechanism for freshwater supply via precipitation and recharge but slightly isolated from the down-gradient saline waters, peat-forming vegetation species (not necessarily freshwater species) could thrive. That said, there is unlikely any ecosystem design that will completely eliminate the salinity in peatlands as the near-surface WT required for peatland development promotes the upward movement of salts to the surface. Avoiding salinization all together in constructed peatlands may not be possible and as such, designing constructed peatlands similar to naturally occurring saline peatlands with salt-tolerant species capable of accumulating peat (Trites & Bayley, 2009a; Volik et al., 2017; Wells & Price, 2015b) may offer the best chance for success.

5.6 CONCLUSIONS

Elevated salinity may be unavoidable in constructed peatland watersheds due to ubiquitous salinity in the materials used to build these ecosystems as well as from the composite tailings and oil sands process water (OSPW) that will underlie all reclaimed systems in the post-mining landscape. As a result, these systems will have legacy effects of this salinity for decades so designing and constructing peatland-watersheds capable of effectively flushing saline waters while minimizing accumulation will be critical. This research aimed to understand the development of salinity patterns in the Sandhill Fen Watershed (SFW), a constructed peatland watershed, following a heavily managed hydrologic regime to provide insight on its potential for long-term success as a peat-forming wetland. In general, salinity increased year-over-year throughout SFW from 2013 to 2018, particularly in the wetland and margin areas. The uplands were the only region that observed a decrease in salinity over the study period. Salinity was variable throughout SFW where gradients of salinity developed across the wetland and landscape units. In general, salinity was highest in the west wetland (close to the inlet) and decreased towards the east wetland (outlet). Across landscape units, salinity was lowest in the southern uplands, highest in the margins and intermediate throughout the wetland.

Three main mechanisms can explain these salinity patterns: 1) reduction of inflow and outflow, 2) WT position and 3) increased mixing of waters. Firstly, heavy artificial management of inflow and outflow in 2013 maintained relatively low salinity levels. Following that year, hydrologic management largely ceased and the dominant hydrological fluxes shifted to precipitation and evapotranspiration. As a result, ions accumulated throughout the wetland as there was little outflow and flushing of saline water declined.

Secondly, with reduced outflow, WTs increased throughout the wetland and margins which was amplified by the presence of wet years. In the wetland, there was a significant negative correlation with WT depth and salinity, where an increase in WT diluted solutes and resulted in a decrease in salinity. The margins observed an opposing pattern, where a significant positive correlation existed between WT depth and salinity, despite the presence of wet or dry years, indicating an increase in ion mobilization with higher WTs. Lastly, the chemical and isotopic signature of site waters indicate increased mixing of SFW waters with saline groundwater at depth as waters have a decreased evaporation signal and are becoming increasing similar to the chemical composition of OSPW (characterized by high salinity and Na^+ concentrations). Based on its current conditions, SFW cannot support peat-forming vegetation and is most similar to brackish marshes and saline fens. By altering the design of constructed peatland watersheds to include a naturally-flowing outlet, continuous clay liner and an upgradient peatland could help maintain lower salinities in the wetland area. Additionally, shifting designs strategies from reconstructing naturally-occurring freshwater peatlands to naturally-occurring saline peatlands might offer another avenue to increase the success of these systems in a highly disturbed and saline landscape.

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Tables

Table 5.1. Electrical conductivities (µS/cm) of before, during and after pump events in 2015, 2017 and 2018 at boardwalks 1, 2, 3 and the outlet. The * indicates where data is missing.

Site	Year	Sample Time	EC µS/cm
B1	**2015**	Pre-event	2199
		Mid-event	2476
		Post-event	1222
		>1 week post	2825
	2017	Pre-event	2361
		Mid-event	2679
		Post-event	*
		>1 week post	4526
	2018	Pre-event	2800
		Mid-event	2475
		Post-event	2700
		>1 week post	1900
B2	**2015**	Pre-event	1629
		Mid-event	991
		Post-event	718
		>1 week post	1314
	2017	Pre-event	2849
		Mid-event	1669
		Post-event	*
		>1 week post	2857
	2018	Pre-event	1700
		Mid-event	2625
		Post-event	3800
		>1 week post	1200

Site	Year	Sample Time	EC µS/cm
B3	**2015**	Pre-event	1914
		Mid-event	1098
		Post-event	674
		>1 week post	1147
	2017	Pre-event	1620
		Mid-event	1714
		Post-event	1660
		>1 week post	2121
	2018	Pre-event	1800
		Mid-event	1800
		Post-event	1900
		>1 week post	1300
Outlet	**2015**	Pre-event	1307
		Mid-event	978
		Post-event	1027
		>1 week post	1146
	2017	Pre-event	*
		Mid-event	1745
		Post-event	1740
		>1 week post	*
	2018	Pre-event	1700
		Mid-event	2217
		Post-event	1400
		>1 week post	*

Figures

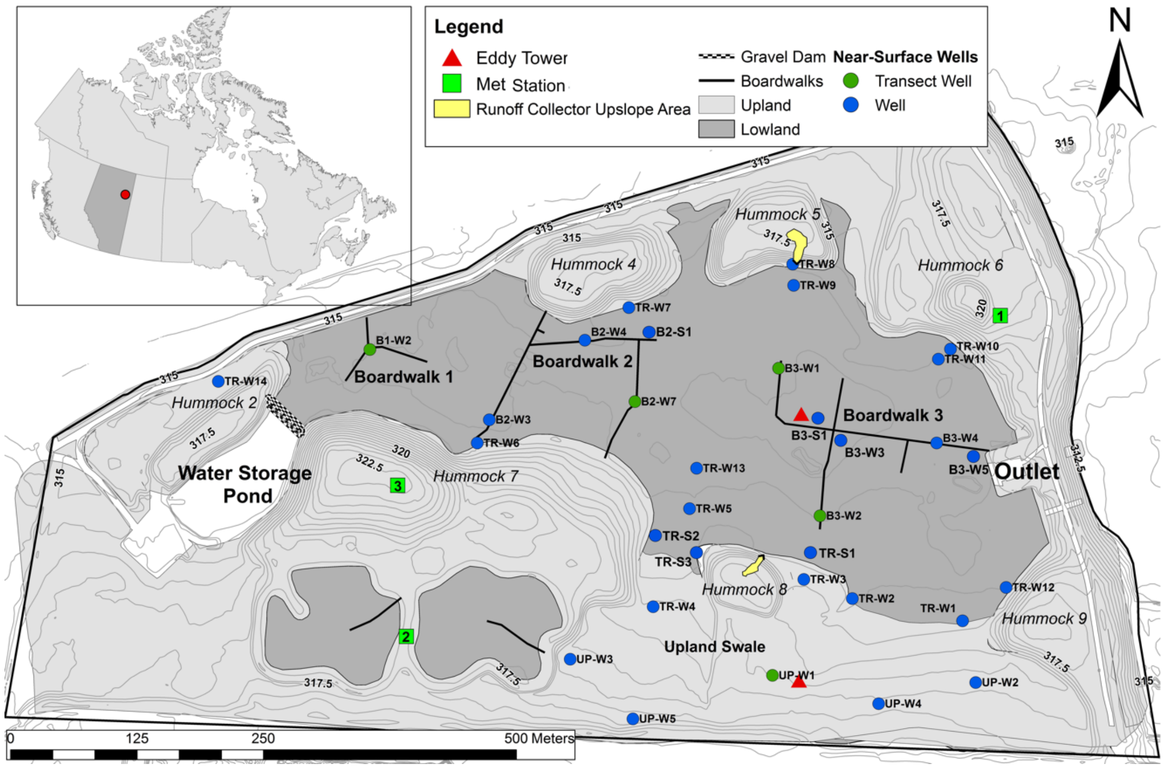

Figure 5.1. Instrumentation map of SFW. Dark and light areas represent the wetland and uplands, respectively. The border between the wetland and uplands represents the margins.

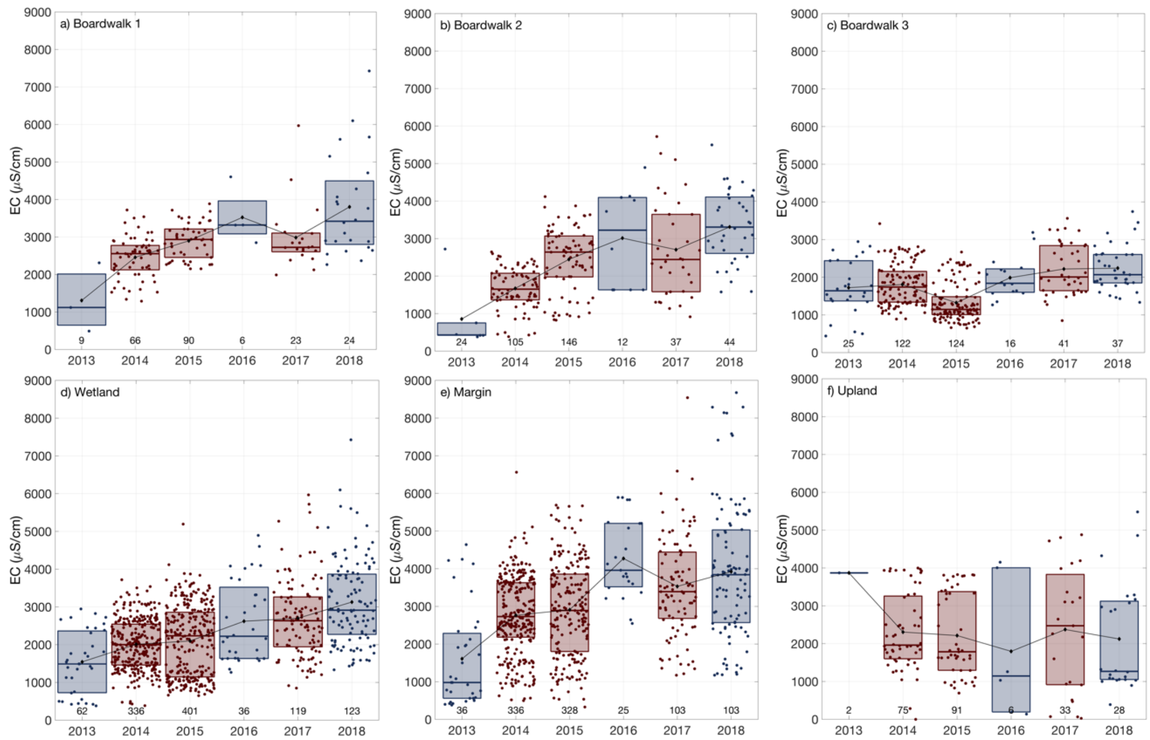

Figure 5.2. Boxplots of annual electrical conductivity (µS/cm) at boardwalks 1, 2, 3, wetland, margins and upland swale from 2013 – 2018. The red and blue colouring represent the dry and wet years, respectively outline in Chapter 2. The wetland boxplot includes all boardwalk sites plus any additional sites classified in the wetland region. The upper and lower limits of the boxplots represent the 75th and 25th percentiles, respectively. The bolded line and black diamond represent the median and mean, respectively. Numbers above the x-axis labels represent the sample size for each year.

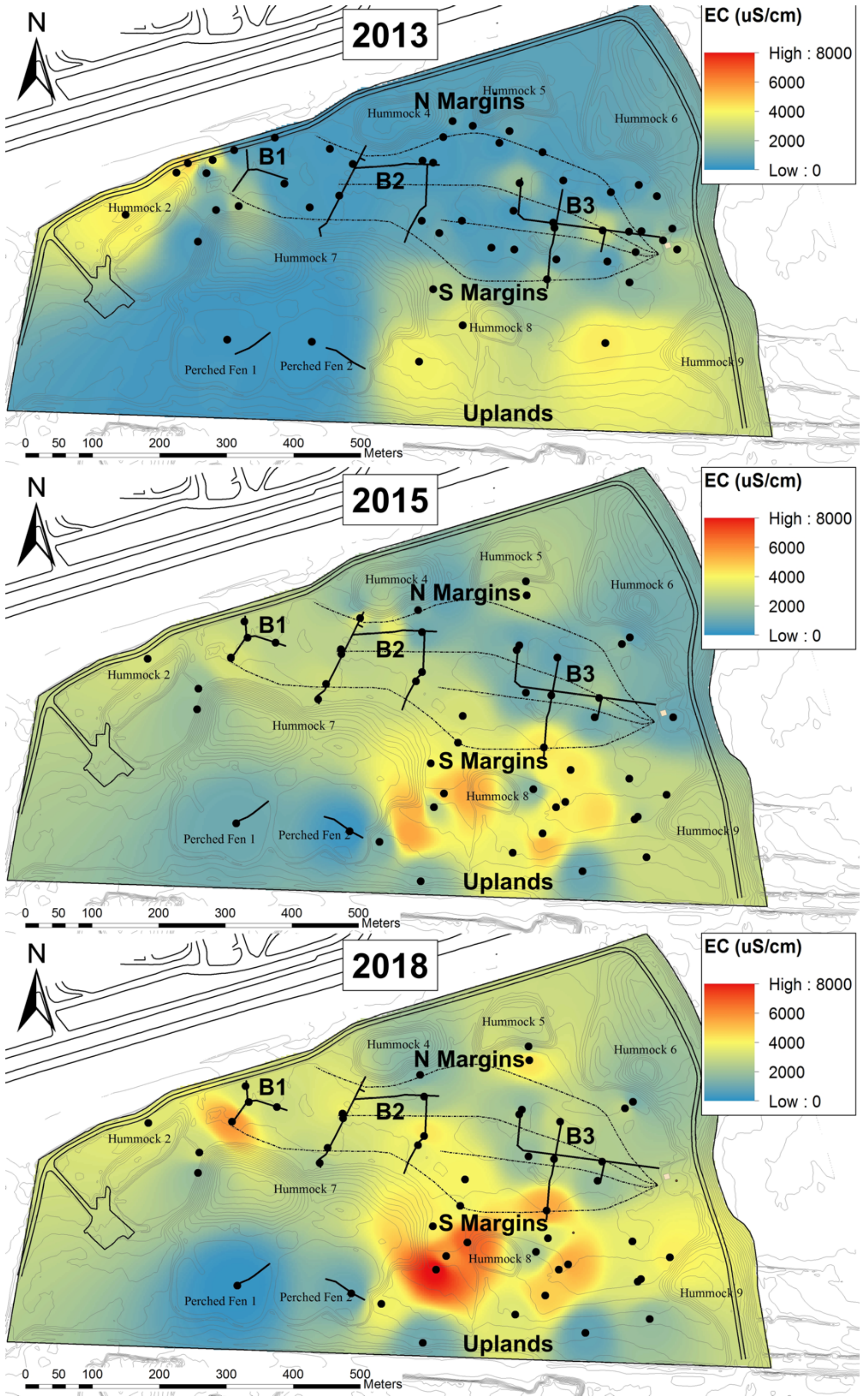

Figure 5.3. Average annual electrical conductivity (μS/cm) in 2013, 2015 and 2018. Black dots represent sampling locations and areas in between have been interpolated in arcGIS.

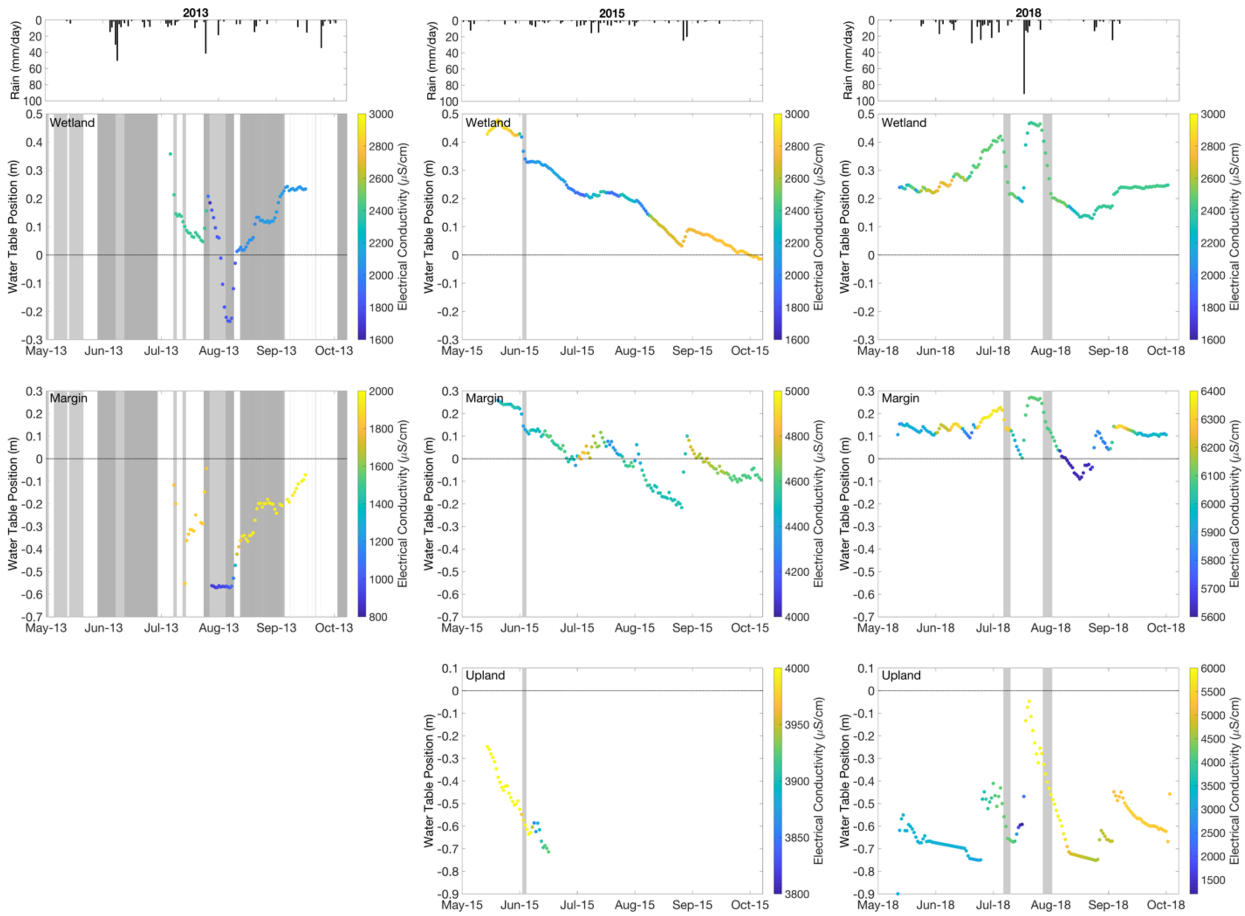

Figure 5.4. Seasonal WT position (m) and electrical conductivity (μS/cm) from May to Oct in 2013, 2015 and 2018 for the wetland, margins and upland swale. Electrical conductivity is represented by the colour of the data points and is scaled specifically to the site. The black dotted line represents the ground surface. The dark and light grey areas represent the duration of inflow and outflow pump activity, respectively. Note the scale change for each figure. The sites used to represent the wetland, margins and uplands were B3W1, B3W2 and UPW1, respectively.

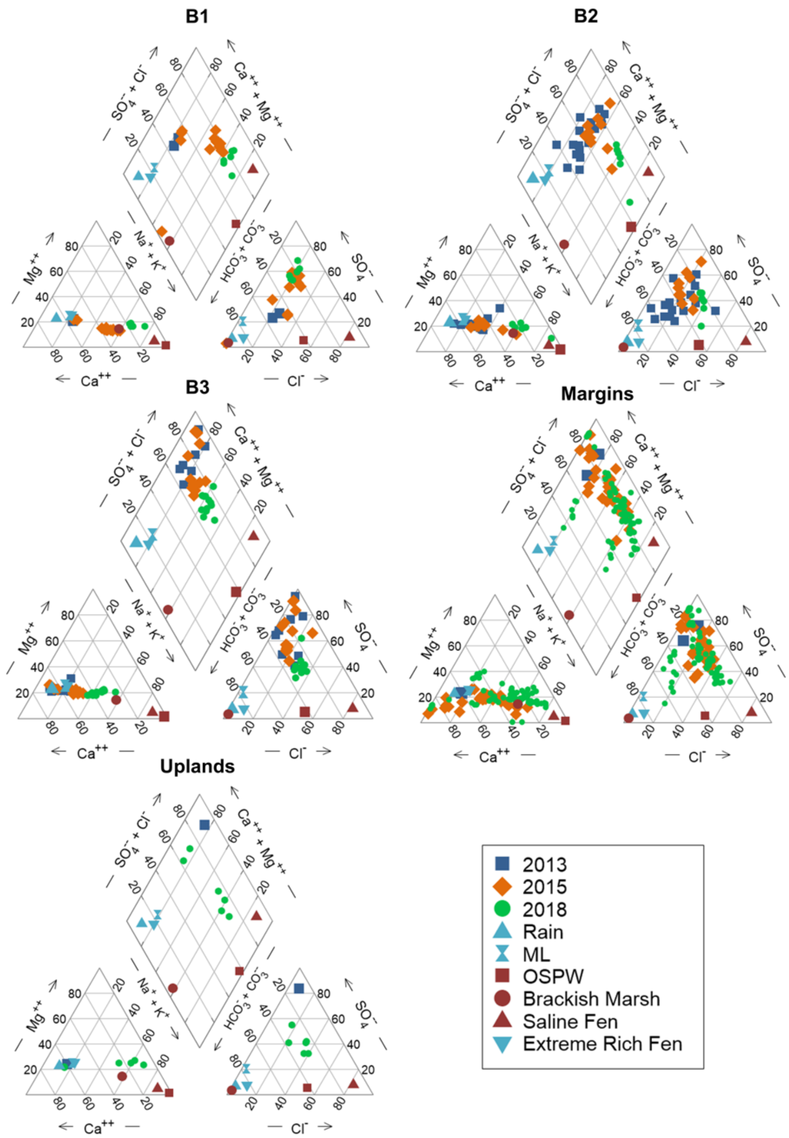

Figure 5.5. Piper diagrams depicting the chemical composition (% meq/kg) of SFW waters in 2013, 2015 and 2018 at boardwalks 1, 2, 3, margins and upland swale. 2013, 2015 and 2018 are represented by blue squares, orange diamonds and green circles, respectively. Light blue and red squares represent the chemical composition for input waters and oil sands process water (OSPW), respectively. The Brackish Marsh, Saline Fen and Extreme Rich Fen data was taken from Hartsock et al., 2021.

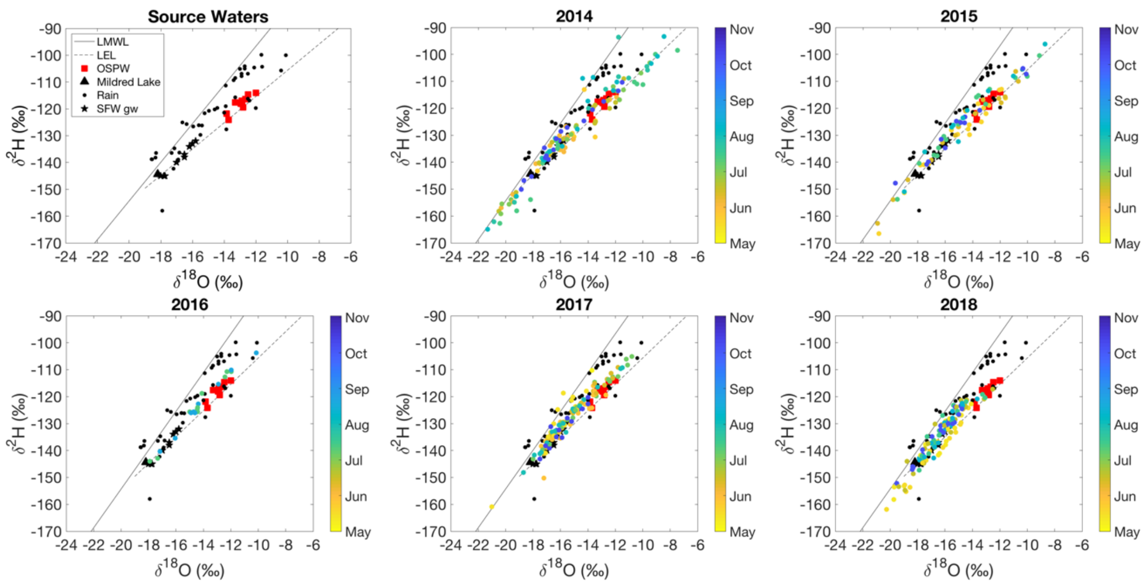

Figure 5.6. Stable isotopes of water for SFW waters across all landscape units for 2014 to 2018. Colours of the dots represent seasonal time of sampling (May to October). Black stars, green squares and red squares represent the isotopic signature for SFW groundwater (gw), precipitation and oil sands process water (OSPW), respectively. The LMWL is taken from Baer et al., 2016 and the SFW gw points are from Twerdy, 2019.

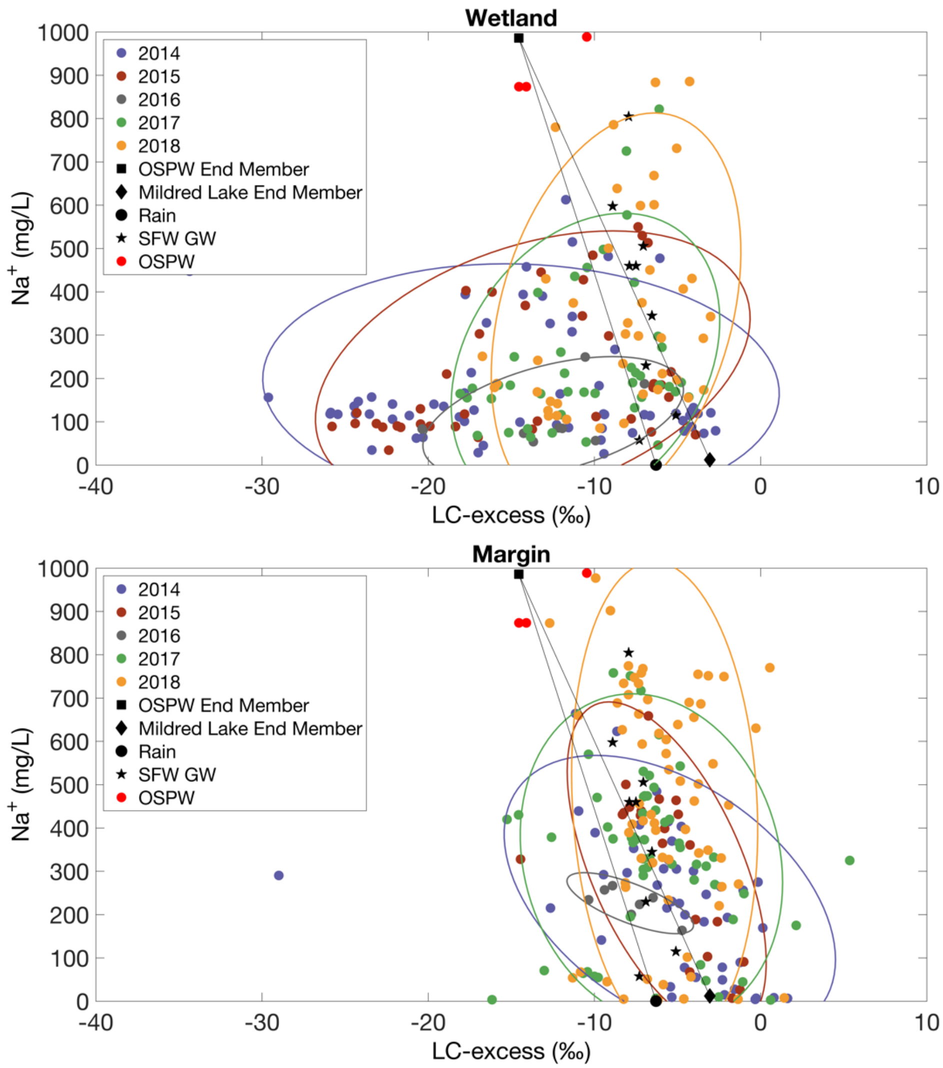

Figure 5.7. Mixing analysis of wetland, margin and upland waters. Coloured dots represent surface and near-surface waters where the colours separate the samples by year (2014 to 2018). The ellipses represent two standard deviations from the mean of the samples. The SFW groundwater (gw) samples are taken from Twerdy, 2019.

CHAPTER 6

CONCLUSIONS

6.1 Summary and conclusions

The goal of this thesis was to evaluate the development and trajectory of the hydrological and hydrochemical functioning of a constructed peatland watershed built atop soft tailings in a post-mining landscape. While the SFW has shown to be capable of providing adequate wetness to maintain vegetation growth, several barriers including elevated salinity, increased mixing with OSPW, limited outflow and uncertain future conditions in a changing climate have decreased the likelihood that the SFW will develop into a peat-forming wetland in the long term. Primarily, the high and unconstrained water tables, degraded peat material and elevated salinity has prevented peat-forming vegetation to grow and has prevented feedback mechanisms, typically found in undisturbed peatlands (Waddington et al., 2015), to develop. Maintaining a near-surface water table in the SFW wetland proved to be challenging as the use of inflow and outflow pumps moved an exceptionally high volume of water through the wetland (greater than twice that of natural fluxes) and resulted in variable water tables that fluctuated above and below the ground surface but maintained lower salinities due to the influx of freshwater and flushing of saline waters. Without the use of the pumps (except for pre-planned drainage events), water balance fluxes were of typical volumes and primarily vertical, however, water tables throughout the wetland increased and resided above the ground surface for much of the growing season at many lowland sites. This was supported by precipitation input via annual snowmelt and heavy rain events that moved water downgradient from the uplands and margins towards the wetland where water accumulated. Fully saturated soils also inhibit processes that encourage the stratification in the peat column which is essential for the development of feedback mechanisms that are currently absent at SFW. Without inflow, precipitation, particularly snowmelt, is an important input of water to replace storage deficits from the previous growing season as much of the precipitation delivered as rain is balanced by evapotranspiration.

Winter processes, including snow accumulation, redistribution and melt at SFW, were largely influenced by topography. Snow accumulation was controlled by slope position where it was highest at the base of hillslopes and decreased towards the crest while snowmelt was controlled by slope aspect where west and south-facing slopes melted the fastest. While the role of the relatively young vegetation on winter processes was minimal, its influence is expected to increase as it develops over time. Runoff ratios of snowmelt from hillslopes was low (30 %) which differed considerably than those previously reported for the other reclaimed peatland watershed (70 – 90 %) highlighting the influence of different soil materials used during construction. Climate change is an additional threat to reclaimed wetlands as the projected increase in temperature and precipitation the AOSR (Meehl et al., 2007) may actually lead to overall drier conditions with the potential for an increase in evapotranspiration (Ireson et al., 2015; Thompson et al., 2017). Under various climate change scenarios of a warmer and wetter climate, simulation results indicate that the influence of winter processes will decrease potentially putting reclaimed systems at greater risk. Modelling results support this assertion, as simulations under a warmer and wetter climate indicate smaller snowpacks, more mid-winter melt events and decrease in snow season length meaning more precipitation will be delivered as rain instead of snow. While an increase in rain would add more water to the system, it may be utilized by evapotranspiration and act to decrease groundwater recharge and storage.

Salinity was relatively low in 2013, as the SFW was in early stages of its development and highly active inflow and outflow pumps increased the exchange of saline water for freshwater. Following 2013, salinity increased year-over-year as outflow was limited and the only freshwater input was from precipitation. While precipitation offers a source of freshwater to SFW, it enhanced the already high WTs and salinity in some regions in SFW. The fluctuation of the WT from evaporation and precipitation affected salinity differently in the wetlands and margins. Water table position and salinity was negatively correlated in the wetland where ion concentrations increased as WT decreased (evapo-concentration) but was positively correlated in the margins where ion concentrations increased as WTs increased (mobilization). High WTs in the margins mobilize the highly saline OSPW from depth where it emerged at the surface surrounding the base of the hummocks which then spreads throughout SFW when areas are more hydraulically connected during wet conditions. This was supported by stable isotopes of water that confirmed increased site-wide mixing over time between SFW waters and OSPW. Combining the effects of high WTs and elevated salinity, much of the peat is fully saturated and peat-forming vegetation cannot survive in inundated conditions. As a result, invasive species, such as Typha, make up much of the species composition within the wetland (Vitt et al., 2020, 2016a).

6.2 Recommendations

Culminating the results presented in this thesis, several design changes for future peatland construction are provided to increase the long-term success of these constructed systems especially throughout prolonged climatic dry periods which did not occur over the course of this study. Recommendations include changes to the soil configuration throughout SFW as well as a shift in general design strategy away from replicating freshwater peatlands.

Continued use of high infiltration soils in construction will be important for water availability and salinity management as recharge is the sole input of freshwater to these reclaimed systems. Model results indicated that the combination of soil texture and antecedent moisture content strongly influenced the infiltration capacity of reclamation soils and thus partitioning between runoff and infiltration. Soil temperature also exerted a strong control on infiltration where colder soils increased runoff production. With the potential for smaller snowpacks in a warmer climate, maintaining high infiltration and unsaturated soils will encourage recharge despite future scenarios in a changing climate. Incorporating similar or larger hummock structures into future reclamation projects will help ensure recharge as the hillslopes remain unsaturated and encourage percolation of precipitation input to groundwater stores. Currently, the clay liner is omitted underneath the hummocks which has developed into an area where OSPW is mobilized to the surface and spreads throughout SFW. A fully lined system would decrease the transportation of OSPW to the surface as the only mechanism of its transport would be via diffusion through the clay liner, which is a more gradual process. This could decrease the salt accumulation within the wetland and margin areas and keep OSPW isolated with deeper groundwater beneath the system. Although not a permanent solution, increasing the peat column depth in the wetland could help attenuate salinity via adsorption and limit the salt interaction with wetland vegetation.

From a design perspective, the incorporation of a naturally-flowing outlet instead of an artificially managed pump system is ideal for constructed wetlands to mitigate the high water tables and elevated salinity. This will enable natural drainage from the wetland to maintain lower water tables while offering a flushing mechanism for saline water that has accumulated over time. A shift in the design goal for constructed wetlands is needed as the original target of replicating undisturbed freshwater peatlands in the region is unlikely to be successful in the long-term. Building a combination of peat-forming and non-peat-forming wetlands where the peatlands are upgradient and a marsh or swamp is downgradient should be considered to limit interaction of peat-forming species with saline groundwater while the non-peat forming wetlands could accumulate excess water and salts downgradient could increase the success of these systems. Part of the original design of SFW was to incorporate perched peatlands, which exist naturally in the WBP, to try and isolate these systems from interacting with saline groundwater instead of constructing the wetland to be the lowest elevation within the watershed where all water accumulates. Lastly, incorporating salt tolerant vegetation capable of accumulating organic matter is likely more realistic than attempting to create ideal conditions for freshwater peatland vegetation as the ingress of saline water is inevitable.

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APPENDICES

APPENDIX A

Supplementary Material Chapter 3

Table A1 All simulation results from the Cold Regions Hydrological Model under current and various potential future climate scenarios.

Climate Scenario			Simulation Results											
Scenario	Temperature Increase	Precipitation increase	Year	Snow Season Start	Snow Season End	Snow Season Duration	Cumulative SWE	Peak SWE	Annual Runoff	Melt Duration	Melt Volume	Mid-winter melt	Mid-winter melt SWE
	°C	mm/day		-	-	days	mm	mm	mm	days	mm	days	mm
			2013-2014	9-Dec-13	21-Apr-14	133	176	102	5	15	89	4	9
			2014-2015	22-Nov-14	9-Apr-15	138	148	97	9	24	90	2	4
T0-P1	0	0	2015-2016	15-Nov-15	16-Apr-16	153	131	65	11	29	65	5	7
			2016-2017	27-Nov-16	1-Apr-17	125	103	61	0	18	41	8	30
			2017-2018	19-Nov-17	26-Apr-18	158	192	157	34	18	157	5	8
			2013-2014	16-Dec-13	9-Apr-14	114	171	77	0	38	75	11	33
			2014-2015	21-Nov-14	31-Mar-15	130	138	70	0	25	70	5	27
T2-P1	2	0	2015-2016	15-Nov-15	30-Mar-16	136	118	41	15	12	39	5	18
			2016-2017	27-Nov-16	30-Mar-17	123	95	61	2	16	32	8	37
			2017-2018	19-Nov-17	21-Apr-18	153	180	136	22	13	129	7†	17†
			2013-2014	16-Dec-13	18-Mar-14	92	164	68	0	40	49	11	59
			2014-2015	21-Nov-14	27-Mar-15	126	134	48	0	21	43	10	47
T4-P1	4	0	2015-2016	23-Jan-16	30-Mar-16	67	106	32	0	12	26	5	22
			2016-2017	27-Nov-16	24-Feb-17	89	82	55	0	13	55	16	5
			2017-2018	19-Nov-17	17-Apr-18	149	173	108	5	10	79	26†	61†
			2013-2014	17-Dec-13	18-Jan-14	32	156	69	0	4	68	4	68
			2014-2015	21-Nov-14	13-Mar-15	112	130	48	0	7	38	8	47
T6-P1	6	0	2015-2016	23-Jan-16	14-Mar-16	51	91	26	0	5	26	5	26
			2016-2017	27-Nov-16	16-Feb-17	81	71	49	0	5	30	16	26
			2017-2018	19-Nov-17	24-Mar-18	125	187	85	0	15	77	21	50
			2013-2014	9-Nov-13	22-Apr-14	164	193	114	15	42	114	-	-
			2014-2015	22-Nov-14	10-Apr-15	139	162	102	14	34	102	-	-
T0-P1.15	0	0.1	2015-2016	15-Nov-15	17-Apr-16	154	143	78	22	23	78	-	-
			2016-2017	27-Nov-16	2-Apr-17	126	113	68	2	19	68	-	-
			2017-2018	19-Nov-17	25-Apr-18	157	211	176	45	17	176	-	-
			2013-2014	9-Nov-13	24-Apr-14	166	201	142	32	44	142	-	-
			2014-2015	21-Nov-14	11-Apr-15	141	169	111	19	35	111	-	-
T0-P1.2	0	0.15	2015-2016	15-Nov-15	18-Apr-16	155	149	94	18	24	94	-	-
			2016-2017	27-Nov-16	3-Apr-17	127	118	74	4	20	73	-	-
			2017-2018	19-Nov-17	26-Apr-18	158	220	186	51	18	185	-	-
			2013-2014	9-Nov-13	24-Apr-14	166	209	151	38	44	150	-	-
			2014-2015	21-Nov-14	11-Apr-15	141	176	115	23	35	115	-	-
T0-P1.3	0	0.2	2015-2016	15-Nov-15	18-Apr-16	155	155	103	20	24	102	-	-
			2016-2017	26-Nov-16	3-Apr-17	128	123	78	8	20	78	-	-
			2017-2018	19-Nov-17	26-Apr-18	158	229	196	58	18	196	-	-

			2013-2014	9-Nov-13	10-Apr-14	152	195	98	14	38	97	4	27
			2014-2015	21-Nov-14	1-Apr-15	131	158	85	9	26	84	6	30
T2-P-1.2	2	0.15	2015-2016	15-Nov-15	1-Apr-16	138	135	60	24	13	60	4	8
			2016-2017	27-Nov-16	2-Apr-17	126	108	73	10	19	67	7	17
			2017-2018	19-Nov-17	22-Apr-18	154	180	162	41	13	159	10	12
			2013-2014	17-Dec-13	1-Apr-14	105	187	78	0.3	40	67	11	57
			2014-2015	21-Nov-14	29-Mar-15	128	153	60	0	23	60	7	48
T4-P1.2	4	0.15	2015-2016	8-Dec-15	31-Mar-16	114	122	34	18	13	30	6	20
			2016-2017	27-Nov-16	29-Mar-17	122	94	63	3	15	32	11	26
			2017-2018	19-Nov-17	19-Apr-18	151	197	134	27	11	114	25†	47†
			2013-2014	17-Dec-13	18-Jan-14	32	178	79	0	10	78	N/A	N/A
			2014-2015	22-Nov-14	13-Mar-15	111	148	56	1	13	45	11	59
T6-P1.2	6	0.15	2015-2016	23-Jan-16	14-Mar-16	51	103	31	0	5	31	9	6
			2016-2017	27-Nov-16	16-Feb-17	81	81	57	0	7	41	9	8
			2017-2018	19-Nov-17	24-Mar-18	125	187	101	12	7	52	28†	102†

†Sum of two mid-winter melt events

APPENDIX B

Supplementary Material Chapter 5

Table B1. List of sampled wells from 2013 – 2018 at SFW. Sampled wells are indicated by the grey shading and wells that were instrumented with an LTC (level, temperature, conductivity) Levelogger are indicated by a dot.

Type	Site	2013	2014	2015	2016	2017	2018
Wetland	B1-W1						
Wetland	B1-W2	•	•	•	•	•	•
Wetland	B1-W3						
Wetland	B1-W4						
Wetland	B2-S1						
Wetland	B2-W3			•			
Wetland	B2-W4						
Wetland	B2-W6						
Wetland	B2-W7		•	•	•	•	•
Wetland	B3-W1	•	•	•	•	•	•
Margin	B3-W2	•	•	•			
Wetland	B3-W3						
Wetland	B3-W4	•	•	•	•	•	•
Wetland	B3-W5						
Outlet	OP-S1						
Margin	TR-S1						
Margin	TR-S2						
Margin	TR-S3						
Margin	TR-W1					•	
Margin	TR-W10						
Margin	TR-W11						•
Margin	TR-W12					•	
Wetland	TR-W13			•		•	
Margin	TR-W14					•	•
Margin	TR-W2		•				•

Margin	TR-W3						
Margin	TR-W4		•				
Margin	TR-W5		•	•			•
Margin	TR-W6					•	•
Margin	TR-W7					•	•
Margin	TR-W8						
Wetland	TR-W9						•
Upland	UP-W1		•	•		•	•
Upland	UP-W2						
Upland	UP-W3						
Upland	UP-W4						
Upland	UP-W5						
Inlet	WSP		•	•			

Table B2. Annual mean ($\overline{x}$) and standard deviation (σ) of SFW water chemistry. The wetland category includes all boardwalks plus sites off of the boardwalks.

Site	Year	Na^{+} (mg/L)		K^{+} (mg/L)		Mg^{+2} (mg/L)		Ca^{+2} (mg/L)		Cl^{-} (mg/L)		SO_4^{-2} (mg/L)		HCO_3^{-} (mg/L)		EC (µS/cm)		pH	
		$\overline{x}$	σ	$\overline{x}$	σ	$\overline{x}$	σ	$\overline{x}$	σ	$\overline{x}$	σ	$\overline{x}$	σ	$\overline{x}$	σ	$\overline{x}$	σ	$\overline{x}$	σ
B1	2013	49	55	2	1	14	6	61	27	46	29	95	102	136	59	1307	924	-	-
	2014	394	122	5	2	44	10	180	52	272	106	654	143	429	113	2457	552	7.6	0.5
	2015	432	81	4	2	59	14	239	62	236	104	748	329	446	76	2897	476	7.2	0.4
	2016	305	6	0	0	49	2	213	32	220	10	710	5	656	200	3522	754	-	-
	2017	563	177	5	3	76	22	293	66	276	131	1124	395	902	117	2984	874	7.3	0.5
	2018	759	116	7	2	100	8	152	36	465	47	1889	406	874	197	3802	1353	7.3	0.4
B2	2013	56	52	3	4	22	25	71	33	38	15	103	58	132	87	855	924	7.8	-
	2014	146	63	3	2	53	21	188	77	91	27	557	289	260	99	1672	542	7.2	0.9
	2015	196	151	2	1	48	24	167	73	149	109	412	232	289	143	2445	848	7.1	0.5
	2016	149	85	1	1	31	14	108	43	136	89	314	158	363	107	3012	1299	6.7	0.7
	2017	317	126	5	1	57	10	184	25	302	176	492	72	826	215	2704	1285	7.1	0.6
	2018	554	149	6	2	82	29	128	57	670	200	854	240	732	238	3309	959	7.0	0.4
B3	2013	62	53	4	1	54	50	217	166	43	46	582	554	160	113	1730	730	-	-
	2014	113	50	3	2	51	29	206	139	71	27	599	492	198	74	1809	544	6.9	1.0
	2015	85	17	3	2	54	34	229	148	71	59	587	383	205	68	1323	501	6.7	0.5
	2016	70	17	2	1	20	5	82	18	70	23	147	33	386	117	1988	544	6.3	0.6
	2017	154	38	6	1	40	17	154	64	158	48	312	134	577	220	2219	675	6.8	0.8
	2018	198	58	10	3	47	11	140	20	259	67	451	152	386	102	2242	596	6.7	0.5
Wetland	2013	56	51	3	3	29	34	107	107	41	28	225	348	140	88	1544	827	7.8	-
	2014	176	136	4	2	60	33	258	163	113	101	770	477	297	145	2010	654	7.1	0.9
	2015	197	152	3	2	65	35	267	149	137	103	694	391	313	133	2098	934	6.9	0.5
	2016	149	103	1	1	30	14	118	59	126	79	326	236	431	165	2621	1077	6.5	0.6
	2017	236	173	5	2	75	44	292	177	193	133	692	432	686	244	2699	1035	6.9	0.7
	2018	349	232	7	3	87	33	263	174	379	217	1137	620	547	257	3129	1133	6.8	0.6
Margins	2013	87	5	3	0	65	6	287	27	88	24	737	91	271	149	1609	1316	-	-
	2014	243	187	7	4	87	37	444	150	222	178	1205	452	387	150	2765	1032	7.3	0.9
	2015	344	206	6	5	85	41	385	140	295	197	1020	422	397	137	2911	1248	7.1	0.4
	2016	256	76	3	1	48	14	198	53	193	109	635	199	625	129	4272	1051	6.6	0.1
	2017	339	198	5	4	68	31	303	131	267	192	825	345	815	219	3531	1344	6.9	0.5
	2018	469	281	8	5	92	38	252	158	551	462	1310	773	616	216	3922	1769	6.8	0.4
Uplands	2013	190	-	4	-	153	-	632	-	132	-	1895	-	246	-	3870	-	-	-
	2014	115	54	4	2	78	36	288	157	97	74	629	534	412	134	2349	989	7.6	1.3

	2015	-	-	-	-	-	-	-	-	-	-	-	-	-	-	2214	1088	-	-
	2016	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1797	1824	-	-
	2017	269	56	4	4	142	31	523	147	398	89	1187	303	676	340	2722	1468	7.1	0.8
	2018	351	256	7	3	83	50	116	66	482	341	713	484	651	407	2123	1425	7.0	0.5

Table B3. Seasonal chemistry data at Boardwalks 1, 2, 3 (B1, B2, B3), wetland, margins and uplands in SFW. The wetland category includes all boardwalks plus sites off of the boardwalks.

Site	Year	Month	Count	mean EC µS/cm	std EC µS/cm	max EC µS/cm	min EC µS/cm	Year	Month	Count	mean EC µS/cm	std EC µS/cm	max EC µS/cm	min EC µS/cm
B1	2013	Sep	3	1307	924	2310	490	2016	Jul	5	3522	754	4603	2848
	2014	May	3	1607	47	1636	1553	2017	May	9	2697	582	3723	1988
		Jun	19	1970	408	2622	1288		Jun	4	2647	195	2844	2383
		Jul	27	2711	350	3719	2145		Jul	4	2705	135	2844	2520
		Aug	11	2794	527	3539	2177		Aug	4	3426	737	4526	2953
		Oct	6	2809	244	3125	2630		Oct	1	5968	-	-	-
	2015	May	8	2498	332	2930	2251	2018	May	8	3270	1340	5664	2264
		Jun	24	3123	372	3884	2737		Jun	4	4612	1888	7426	3383
		Jul	36	2847	490	3715	2152		Jul	4	3845	1532	6099	2724
		Aug	16	2859	503	3498	2279		Aug	4	3344	910	4709	2880
		Oct	6	3437	5	3440	3433		Oct	4	4473	767	5604	3940
B2	2013	Aug	1	2720	0	2720	2720	2016	Jul	8	3117	1345	4894	1633
		Sep	5	482	151	750	380		Aug	4	2854	1410	4128	1633
	2014	May	6	1937	163	2125	1671	2017	May	12	2536	1441	5105	915
		Jun	36	1366	416	1962	385		Jun	7	2360	998	3949	1291
		Jul	40	1796	587	2644	475		Jul	9	2380	874	3749	1303
		Aug	15	1901	634	2535	854		Aug	6	3056	838	4207	2232
		Oct	8	1939	364	2312	1458		Oct	3	5495	318	5720	5270
	2015	May	26	2088	755	3573	836	2018	May	15	2840	894	4374	1576
		Jun	40	2618	795	3837	1283		Jun	8	3736	1184	5498	1845
		Jul	48	2666	882	4118	939		Jul	7	3313	802	4287	2346
		Aug	20	2299	879	3206	951		Aug	7	3286	949	4510	2176
		Oct	12	2056	784	2739	824		Oct	7	3839	804	4601	2701
B3	2013	July	7	1770	753	2825	1127	2016	Jul	8	2150	671	3189	1351
		Aug	8	1823	523	2724	1371		Aug	8	1826	351	2404	1267
		Sep	9	1384	908	2946	440							
		Oct	5	2172	498	2668	1491							
	2014	May	5	2269	658	3424	1860	2017	May	14	2069	855	3563	849

		Jun	37	1736	540	2811	889		Jun	7	2184	584	2951	1610
		Jul	48	1808	528	2748	1161		Jul	7	2020	509	2830	1602
		Aug	21	1819	532	2793	1223		Aug	7	2109	346	2864	1837
		Oct	11	1832	593	2718	1103		Oct	6	2972	244	3258	2630
	2015	May	22	924	262	1459	650	2018	May	12	1830	449	2946	1334
		Jun	29	1409	524	2474	858		Jun	6	2546	755	3454	1832
		Jul	39	1350	502	2669	805		Jul	6	2469	760	3745	1774
		Aug	21	1390	513	2396	790		Aug	7	2213	352	2601	1808
		Oct	13	1545	443	2335	1008		Oct	6	2571	326	3174	2297
Wetland	2013	Jul	16	1770	753	2825	1127	2016	Jul	22	2846	1095	4894	1351
		Aug	13	1823	523	2724	1371		Aug	14	2291	1000	4128	1267
		Sep	17	1105	835	2946	380							
		Oct	9	2172	498	2668	1491							
	2014	May	15	2029	473	3424	1553	2017	May	41	2510	1052	5105	849
		Jun	103	1735	578	3117	385		Jun	21	2487	779	3958	1291
		Jul	133	2118	655	3719	475		Jul	23	2411	705	3749	1303
		Aug	55	2171	687	3539	854		Aug	20	2849	829	4526	1837
		Oct	30	2225	654	3324	1103		Oct	14	4019	1272	5968	2630
	2015	May	63	1708	863	3573	650	2018	May	41	2706	1086	5664	1334
		Jun	103	2288	993	5194	858		Jun	21	3502	1348	7426	1832
		Jul	137	2202	927	4118	805		Jul	20	3244	1085	6099	1774
		Aug	62	1982	877	3498	790		Aug	21	3050	994	5105	1808
		Oct	36	2077	841	3440	824		Oct	20	3571	955	5604	2297
Margin	2013	Jul	1	2338	-	-	-	2016	Jul	13	4558	1119	5891	2646
		Aug	2	1962	16	1973	1951		Aug	12	3961	922	5204	2547
		Sep	31	1531	1377	4640	380							
		Oct	1	2593	-	-	-							
	2014	May	23	3048	946	4808	1401	2017	May	38	3391	1412	8541	1173
		Jun	122	2718	971	5110	550		Jun	21	3445	1136	5375	1359
		Jul	122	2795	928	4580	955		Jul	18	3612	1284	5567	1466
		Aug	42	2438	1165	4110	499		Aug	14	3476	1328	5497	1596
		Oct	26	3127	1426	6562	626		Oct	12	4091	1664	6592	1847
	2015	May	69	3319	1315	5669	776	2018	May	37	3708	1590	7538	1175
		Jun	96	3062	1272	5687	931		Jun	17	4211	1922	8291	1342
		Jul	95	2778	1189	5657	567		Jul	17	4039	1811	8289	1248
		Aug	43	2441	996	4046	535		Aug	17	3974	1714	8131	1440

		Oct	25	2300	1147	4036	328		Oct	15	3933	2174	8675	1198
Upland	2013	Sep	1	3870	-	-	-	2016	July	3	1869	2049	4153	194
									Aug	3	1725	2025	4006	138
	2014	May	4	3162	636	3984	2502	2017	May	4	2611	1708	4878	471
		Jun	21	2234	875	3800	1247		Jun	5	2612	1363	4192	983
		Jul	28	2595	934	3967	1443		Jul	4	3025	1661	4804	509
		Aug	14	2132	1153	3998	284		Aug	9	1364	1560	3102	27
		Oct	9	1873	1097	3938	1045		Oct	1	72	-	-	-
	2015	May	63	1708	863	3573	650	2018	May	8	1831	1040	3288	896
		Jun	103	2288	993	5194	858		Jun	5	1564	954	3258	1032
		Jul	137	2202	927	4118	805		Jul	5	1453	1201	3170	389
		Aug	62	1982	877	3498	790		Aug	5	3158	2207	5482	1090
		Oct	36	2077	841	3440	824		Oct	5	3300	1698	4858	1050

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