Hydrological and Geochemical Implications of Aquifer Depressurization on Expansive Peatland Systems in the Hudson/James Bay Lowlands

by

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Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

The ecological integrity of the peatlands of the Hudson/James Bay Lowlands (HJBL) is vulnerable to hydrological changes associated with climate change and resource development. Dewatering of an open-pit mine (the DeBeers Victor Diamond Mine), located within the interior HJBL, has already shown to increase downward hydraulic gradients and lower seasonal water tables within a peatland dominated watershed located proximal to the mine. Previous studies have yet to fully characterize the implications of increased downward hydraulic gradients and lowered seasonal water tables on surface runoff processes and on the biogeochemistry of the system. In this thesis, peat and stream water samples were analysed for environmental tracers (stable isotopes of water and chloride) and dissolved organic carbon (DOC), and they were used to identify possible alterations in peatland recharge, discharge and the production and transportation of dissolved organic carbon (DOC) to downstream ecosystems. Time series analyses (2008 to 2011) revealed systematic changes in the isotopic composition of hydrogen and oxygen in peat-water within the impacted region of the watershed regardless of non-impacted headwater contributions and variable weather conditions. The changes indicated that increased vertical hydraulic gradients had the greatest control on runoff generation as large storage capacities developed over the winter and summer (through increased deep seepage losses) resulted in a greater infiltration of precipitation and a subsequent reduction in surface flow contributions to stream discharge. Time series analyses of chloride also indicated a reduction in upward flow paths (dispersive and diffusive) and a reduction in the flow of mineral rich waters from the underlying mineral substratum thereby resulting in a change in the biogeochemistry of fen systems. During the summer, DOC analysis indicated that the hydrological alterations associated with aquifer dewatering are also increasing the internal production of DOC within the peatlands through an increase in the aerobic layer (lowered seasonal water tables) and an increase in labile carbon turnover (increased infiltration of event water). And, although greater evaporative isotopic signatures in stream water indicated less surface water contributions to stream channels, incoming precipitation during the fall wet-up period of 2012 was sufficient to reconnect the peatlands thereby providing an avenue for the newly decomposed organic matter to be dissolved and transported into downstream ecosystems. As such, although previous hydrologically based research had noted the role of annual rewetting of the peatlands in minimizing the effects of aquifer depressurization and masking alterations in runoff generation, hydrogeochemical analyses have revealed that such contributions are not isolating the peatlands and downstream ecosystems from the biogeochemical ramifications of an altered hydrological regime. Rather, hydrogeochemical analyses have identified alterations in the magnitude of peatland recharge, a subsequent shift in surface and subsurface contributions to receiving surface waters, a decline in the supply of mineral rich waters to the fens, an increase in peatland decay and subsequent increased surges of DOC to stream networks during the initial stages of aquifer depressurization (first 5 years). Our results demonstrate the usefulness of hydrogeochemical analyses in understanding the hydrological and biogeochemical implications of aquifer depressurization on expansive peatland systems within the HJBL.

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I began this ~~career~~ journey in 2011 with the naïve impression that a Master's would take 2 years to complete… 4.5 years later, I think not about the PhD that could have transpired in that time, but rather the incredible friendships, accomplishments, knowledge, and of course the family (a fiancé, a son, and another on the way) that I have gained from this experience. There are so many people to thank for their role in this academic (and as it turns out, personal) achievement:

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Dedication

This thesis is dedicated to my Dad who taught me that I was capable of doing anything that I set my mind on and to always finish what I start. Miss you Dad!

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1 Introduction

1.1 Background

Nearly 14% of Canada's landmass is covered in wetlands, with the majority of these being peatlands; organic wetlands containing more than 40 cm of peat accumulation (National Wetlands Working Group, 1997). Despite their relatively small terrestrial footprint (~3% of the Earth's surface) (Gorham, 1991), peatland ecosystems are of global importance as they account for an estimated 10% of the world's freshwater resources and about 30% of the world's soil carbon pool (Gorham, 1991; Holden, 2005; Keller et al., 2004). Peatlands can be found in arctic, boreal, temperate and tropical climates; however, 80% are located within the temperate or boreal region of the northern hemisphere (Siegel and Glaser, 2006) where low gradients, impermeable underlying substrates (or topographic convergence) and cool temperatures maintain saturation and promote peat accumulation (Holden et al., 2004). More than a third of the world's peatlands are located within the boreal region of Canada (Riley, 2011).

The Hudson/James Bay Lowlands (HJBL), located mostly in northern Ontario, host one of the largest peatland complexes in the world, blanketing an area of 320 000 km^2 with continuous peat (referred to as "*muskeg*" by the First Nation's People of the region). During the last glaciation, the Laurentide Ice Sheet depressed the Earth's surface in northern Ontario, and upon the retreat of ice, land was inundated with water from the Tyrell Sea. The result was the deposition of an irregular thickness (~0 to 200 m thick) of glacial till and fine-grained marine sediments within a very large and flat basin (Glaser et al., 2004a). As isostatic rebound occurred, wetland ecosystems were initiated atop this re-emerged land. And, through processes of organic matter accumulation, these wetlands moved through the successional stages toward peatland ecosystems due to the cool subarctic climate limiting evapotranspiration (Singer and Chen, 2002), the relative impermeability of the glacio-marine sediments (Dredge and Cowan, 1989) and low basin relief (<1% slope) (Glaser et al., 2004a). Peatland proliferation continues today nearby the coast of James Bay, where rates of isostatic rebound are higher than inland (Glaser et al., 2004a).

Today, the HJBL continues to play an important role in the global carbon cycle (Riley, 2011) through the sequestration of soil carbon during peat accumulation, atmospheric exchanges of carbon dioxide and methane, as well as the transport of dissolved organic carbon to downstream aquatic ecosystems (Blodau, 2002). The rivers draining the HJBL also contribute a substantial amount of freshwater to the saline James Bay (Rouse et al., 1992). The hydrological and geochemical fate of the HJBL is of current concern due to their vulnerability to climate change and anthropogenic influences, and the potential implications to water quantity and quality within the peatlands, and to downstream ecosystems (Déry et al., 2005; Pastor et al., 2003; Rouse et al., 1992).

Climate change is expected to have the greatest influence within arctic and subarctic North America due to the prevalence of freshwater wetlands and permafrost within the region, and their vulnerability to high magnitude changes in temperature and precipitation (Rouse et al., 1997). Warmer temperatures, shallower snowpack, increased evapotranspiration and greater permafrost degradation (associated with climate change) have already shown to alter the delivery of water to, as well as through, northern catchments (including the HJBL), thereby affecting the timing and magnitude of runoff responses (Déry et al., 2005). Declining stream discharge could result in a transition from a surface water-dominated system to a groundwater-dominated system which could have critical implications on carbon cycling, given an associated shift in river transport of organic matter, nutrients and major ions. Currently, there are conflicting theories on whether this shift will result in an increase or a decrease in the transport of organic carbon (Freeman et al., 2001; Frey and McClelland, 2009; Pastor et al., 2003).

In addition to climate change, peatland hydrology can be altered through anthropogenic manipulations in two ways: peat-water drainage through the use of drainage ditches or shallow pumping wells (peat harvest for the horticultural industry, fuel, forestry, agriculture, and livestock grazing) (Ballard et al., 2011; Price et al., 2003); and, aquifer depressurization for the extraction of groundwater via bedrock pumping wells for municipal water supply, open pit mine and quarries, or tunnels (roads, railways, sewage, etc.) (Auterives et al., 2011; Johansen et al., 2011; Kværner and Snilsberg, 2011).

There is a great deal of information on the anticipated effects of climate change within the HJBL (Déry et al., 2005; Gorham, 1991; McLaughlin and Webster, 2014; Philben et al., 2015), as well as the implications of peat-water drainage on northern peatlands (Minkkinen and Laine, 1998; Price, 2003; Strack et al., 2008). Fewer articles have studied the effects of aquifer depressurization on peatlands, particularly within larger northern peatland systems, due to their remoteness and previously scarce infrastructure. Recently, however, current and future resource development pressures within the HJBL (e.g. the DeBeers Victor Diamond Mine) pose a threat to local peatlands, as groundwater extraction is necessary for mineral extraction. A disruption in the natural hydrological regime of a peatland system challenges the ecological integrity of the peatland and downstream ecosystems (Siegel and Glaser, 2006).

1.2 Thesis Overview

To address the concerns of such projects within the HJBL, a Natural Science and Engineering Research Council Collaborative Research and Development Grant (NSERC CRD grant 360525-07) was awarded to Drs. Jonathan Price (University of Waterloo), Brian Branfireun (University of Western Ontario), Vicki Remenda (Queen's University) and Murray Richardson (Carleton University). The aim of the study was to examine peatland drainage systems under the influence of aquifer depressurization for the DeBeers Victor Diamond Mine and to characterize associated changes in peatland structure, hydrological connectivity and the hydrogeochemical implications of the disturbance.

The DeBeers Victor Diamond Mine (henceforth referred to as the Victor Mine, or simply 'the mine') is located 90 km inland from the James Bay coast within the Nayshkootayaow River drainage basin (a tributary of the Attawapiskat River) which comprises a complex assemblage of bogs, fens, and open waters. To extract diamondiferous material from within the underlying fractured limestone bedrock, aquifer depressurization is necessary. The project study site was situated within a small tributary of the Nayshkootayaow River, which at the time of the study (2008-2012), was partially located within the zone of influence of aquifer depressurization.

Depending on topography and peat structure, water typically favours lateral flow through the peatlands of the HJBL (Riley, 2011). However, research conducted under the NSERC CRD objectives, revealed that as a result of aquifer depressurization, vertical hydraulic gradients increase (Whittington and Price, 2013) resulting in lowered seasonal water tables and increased deep seepage losses within peatlands located within the zone of influence of aquifer depressurization (Leclair et al., 2015). The magnitude of these hydrological changes is dependent on peatland type (bogs *vs* fens) (Leclair et al., 2015), the thickness and hydraulic conductivity of the marine sediments located beneath the peatlands (Whittington and Price, 2013), as well as the proximity of limestone bedrock outcrops (bioherms) (Whittington and Price, 2012). Increased downward hydraulic gradients and/or lowered seasonal water tables could alter recharge/discharge relationships and the proportion of surface water/groundwater contributions to stream networks (studied in Chapter 2 of this thesis) (Déry et al., 2005). This could ultimately disrupt the chemical balance of the system by altering the source and transport of solutes and organic matter within and from the peatlands.

Of particular concern are the implications of under-drainage on the production and mobility of dissolved organic carbon (DOC) (organic carbon smaller than 0.45 μm in diameter). DOC is of ecological importance as it affects pH, nutrient availability, metal complexation and light penetration in aquatic environments (Steinberg, 2003). The distribution of DOC within and between a peatland and downstream ecosystems is dependent on the net production of DOC and hydrologic pathways (Moore, 2013). Two major controls on the decomposition of organic material (and ultimately the production of DOC) include the presence of oxygen and solute transport (Baird et al., 2013; Belyea, 1996). Lowered seasonal water tables and increased hydraulic gradients could increase decomposition rates through an increase in the depth of the aerated layer and labile carbon turnover (Hribljan et al., 2014). And, since seasonally lower water tables have also been shown to recover annually within the study area throughout the study period (2008-2011) (Leclair et al., 2015), DOC production and transport may also increase as a result of aquifer depressurization. As the water tables recover into the decomposing layer, they would mobilize the newly decomposed labile material and flush the DOC into downgradient aquatic environments (studied in Chapter 3 of this thesis).

1.3 Thesis Objectives

This thesis forms part of the NSERC CRD project. Previous research has addressed several objectives of the NSERC CRD, including (but not limited to), an assessment of the hydrophysical properties of the peatlands and marine sediments surrounding the mine (e.g. bulk density, porosity, hydraulic conductivity), as well as the hydrological implications of aquifer depressurization on these properties (e.g. peat and marine sediment subsidence, lowered seasonal water tables and increased vertical hydraulic gradients) (Ali, 2013; Whittington and Price, 2013).

The research conducted between 2008 and 2011 by Leclair et al. (2015) was unable to confidently assess the influence of aquifer depressurization on runoff generation (an objective of the NSERC CRD grant that funded this project) due to contributions from non-impacted headwater sections of the watershed and annual variations in weather conditions. As part of this research, stable isotope ratios of oxygen and hydrogen (^{18}O and ^{2}H) in water, as well as major ions, were collected throughout the study period (2008 to 2011). Environmental tracers such as stable isotopes and hydrologically conservative chemical species (e.g. chloride) are useful tools in addressing the evolution of hydrologic flow pathways when hydraulic properties are temporally and spatially variable, which is the case within a complex peatland system undergoing hydrologic manipulations and associated hydrophysical changes over the course of several years.

Stable isotope ratios of oxygen and hydrogen (^{18}O and ^{2}H) in water exhibit systematic spatial and temporal variations along flow pathways due to isotopic fractionation that occurs during phase changes and diffusion (Clark and Fritz, 1997); and, concentrations in hydrologically conservative chemical species decrease with distance along a flow pathway from the zone of solute production (e.g. Dinsmore et al., 2011; Waddington and Roulet, 1997). As such, many studies have successfully utilized stable isotopes and hydrologically conservative chemical species (e.g. chloride) for assessing the source and seasonality of recharge (Levy et al., 2014), runoff generation (Gibson et al., 1993; Orlova and Branfireun, 2014) and the effects of hydrological stressors (Gibson, 2001; Hayashi et al., 2004; Hunt et al., 1998) within peatland systems.

Therefore, the first objective of this thesis was to evaluate the impact of lowered seasonal water tables and increased vertical hydraulic gradients on recharge and discharge relationships through the use of hydrologically conservative chemical species (e.g. chloride) and stable isotopic analyses (Chapter 2: Using stable isotope ratios of oxygen and hydrogen (^{18}O and ^{2}H) in water to track groundwater recharge in a peatland dominated complex under-drained by aquifer depressurization, Hudson/James Bay Lowlands).

The second objective of this thesis meets a currently unevaluated portion of the NSERC CRD, which is to assess the impacts of the hydrological changes on the geochemistry of the system. For reasons stated in Section 1.2, the production and transportation of DOC from peatlands located within the zone of influence are addressed in Chapter 3 (Implications of aquifer depressurization on the production and export of dissolved organic carbon within internal fen-water-tracks of the Hudson/James Bay Lowlands).

1.4 Author's Contributions to the NSERC CRD Project

My involvement in the project began in April 2011; therefore, I was not involved in the instrumentation of the field-site for Chapter 2. I also inherited water samples collected during the 2008, 2009 and 2010 field seasons by other research members of the Muskeg Crew. During the 2011 field season, I helped in the collection of hydrological field data, as well as the collection and preparation (filtration) of all water samples. Although hydrological data generation was generally performed by others, as specified in-text, all interpretations of the hydrological data in Chapter 2 (particularly, those regarding processes involved in the timing and magnitude of peatland recharge and discharge) were made by me. All chemical analyses were performed by the Ecohydrology Laboratory located at the University of Western Ontario (for isotopes) and the Biotron Institute for Experimental Climate Change (for major ions); however, I performed all chemistry related data generation and interpretations for Chapter 2.

During the 2012 field season, I was responsible for the organization and execution of all field research conducted by me and field assistants including, but not limited to, transect locations and piezometer installations, water sample collections, on-site sample filtration and laboratory preparations at the University of Western Ontario. Prepared samples were then analyzed by professional technicians at the Biotron Institute for Experimental Climate Change. I performed all hydrological and chemical data generation and interpretations for Chapter 3.

2 Using stable oxygen and hydrogen isotope ratios to track groundwater recharge in a peatland dominated complex under-drained by aquifer depressurization, Hudson/James Bay Lowlands

2.1 Introduction

Hydrology has a profound influence on peatland ecology (Siegel and Glaser, 2006), so a disruption in a peatland's natural hydrological regime may pose a threat to the peatland and downstream ecosystems. Climate change is expected to alter the hydrology of peatland systems, particularly those located within the arctic and subarctic region of North America, due to the extensive presence of peatlands within the landscape and the susceptibility of permafrost to changes in temperature and precipitation (Rouse et al., 1997). The Hudson/James Bay Lowlands (HJBL), located in Northern Ontario, host one of the largest expansive peatland systems in the world (Siegel and Glaser, 2006). This system is an important freshwater source to the brackish James Bay (Rouse et al., 1992) and plays a significant role in the global carbon cycle (Blodau, 2002). Recent discoveries of mineral deposits (e.g. the DeBeers Victor Diamond Mine) within the HJBL increase concerns within the region, as aquifer dewatering is often necessary for mineral extraction (e.g. open-pit, Victor Mine) which challenge the integrity of local peatlands through hydrological changes. For the Victor Mine, aquifer depressurization has already been shown to increase downward hydraulic gradients and lower seasonal water tables within the peatlands surrounding the Victor Mine (Leclair et al., 2015; Whittington and Price, 2013). With increased downward hydraulic gradients and lowered seasonal water tables, the source and transport of waters and solutes within and between the various peatland types may be altered and peatland deterioration may ensue. In addition, there may be a potential reduction in surface runoff and a subsequent alteration in the chemical balance within and between the peatlands and their receiving surface waters, leading to alterations in aquatic biological communities.

Previous research (conducted between 2008 and 2011) on the implications of aquifer depressurization within the peatlands surrounding the Victor Mine (herein, the mine) was focused within the North Granny Creek drainage basin, a small sub-tributary of the Attawapiskat River. The watershed is primarily composed of a dendritic-like pattern of low-lying channel fens and fen-water-tracks (which eventually converge into 2 defined stream networks) and ovoid-shaped domed-bogs forming the inter-fluvial divides. Beneath the peatlands are varying thicknesses (~0 to 200 m) of glacial till and fine-grained marine sediments (Ali, 2013; Dredge and Cowan, 1989). Aquifer dewatering of the mining pit has resulted in a progressively more extensive cone of depression within the Upper Attawapiskat limestone bedrock which extends below the lower reaches of the North Granny Creek drainage basin (Leclair et al., 2015). Within the zone of influence, downward hydraulic gradients and water table decline were generally found to be stronger within the bogs than the fens (Leclair et al., 2015), within areas underlain by a thinner layer of marine sediments (Whittington and Price, 2013) and within 25 m proximity of bioherms (ancient coral reef limestone outcrops) (McDonald, 1989; Whittington and Price, 2012).

Leclair et al. (2015) also evaluated the hydrological influence of aquifer depressurization at a watershed scale, with an annual (2008-2011) assessment of the water balance of the North Granny Creek watershed and a non-impacted system of similar size (30 km^2) located within the Attawapiskat River drainage basin. Unfortunately, contributions from the upper (non-impacted) reaches of the watershed and inter-annual variations in weather conditions confounded the impacts of aquifer depressurization on runoff generation

(Leclair et al., 2015). In essence, runoff was determined to be greater during ‘wet’ years (i.e. 2009) than in subsequently ‘drier’ years (i.e. 2010 and 2011), resulting in an overall decline in runoff within the North Granny Creek, as well as the non-impacted system. It therefore remains unclear how increased vertical hydraulic gradients and lowered seasonal water tables are affecting recharge and discharge through time.

The geomorphological, hydrological and geochemical complexities of these expansive peatland systems and their covariance with environmental controls (Glaser et al., 2004b; Orlova and Branfireun, 2014; Reeve et al., 1996) make it difficult to study the implications of aquifer depressurization by the use of traditional physical measurements alone, as hydrological measurements provide a mere ‘snapshot’ of its hydrology with little concern to gradient changes between monitoring events (Hunt et al., 1998). Fortunately, hydrologically conservative chemical species (e.g. chloride) and stable oxygen and hydrogen (^{18}O and ^{2}H) isotope ratios of peat and surface water samples collected during the North Granny Creek study can offer insight and support into these hydrological investigations as hydrologically conservative chemical species are less dynamically responsive to short term changes (e.g., a single large rain event and subsequent response in water table back to the surface).

At the mine, pore-water chloride is derived through salt water intrusion or a relic deposit of saline water (e.g. saline water entrapped within the marine sediments underlying the HJBL (Price and Woo, 1988)). Assuming the solute is conservative, it is possible to study the evolution of hydrological flow paths by use of conservative solutes, as their concentrations will become increasingly more dilute with distance from the zone of solute production (e.g. Dinsmore et al., 2011; Waddington and Roulet, 1997). Likewise, stable isotope ratios of oxygen and hydrogen (^{18}O and ^{2}H) in water are ideal environmental tracers for understanding hydrological pathways, as they are part of the water molecule itself, have systematic variations in their abundance in snow, rain, ground and surface waters, and are conservative during mixing (Gibson et al., 1993).

In the absence of continuous hydrological data and rigorous analyses, the use of hydrologically conservative chemical species and stable isotopes as environmental tracers is becoming more widely used to: determine the source and seasonality of recharge within peatlands (Levy et al., 2014); assess runoff generation (Gibson et al., 1993; Orlova and Branfireun, 2014); and to evaluate the effects of hydrological stressors (Gibson, 2001; Hayashi et al., 2004; Hunt et al., 1998). Likewise, the above noted hydrological changes within the peatlands surrounding the mine, should be reflected in alterations in hydrologically conservative chemical species (e.g. chloride) and isotopic analyses, and demonstrate the biogeochemical implications of altered flow processes within peatlands experiencing under-drainage (Hayashi et al., 2004).

The purpose of this paper is therefore to improve our understanding of natural and altered flow processes within peatlands experiencing under-drainage by the use of hydrologically conservative chemical species (i.e., chloride) and isotopic analyses. The specific objectives are (1) to track recharge within the various peatland systems and evaluate how recharge may be altered through increased downward hydraulic gradients and lowered seasonal water tables, and (2) to evaluate a potential shift in the timing and magnitude of runoff generation.

2.2 Study Area

The study area is located within a headwater catchment of the Attawapiskat River, approximately 90 km west of Attawapiskat, Ontario (52°50'N, 83°55'W) (Figure 2-1[a]). Peatlands (primarily bogs and fens) (Riley, 2011) within the study area generally range in thickness up to ~3 m (atop domed bogs). Peatland surface topography exhibit subtle variations, with elevation differences ranging from 0.5 to 2 m between bogs and fens, and an overall topographical slope of ~0.1% (Difebo et al., 2015). The generally flat terrain is also sparsely interspersed with small circular features reaching up to 6 m above surrounding peatland surfaces, including raised permafrost mounds (palsas), and more prominently, ancient coral reef outcrops (bioherms) from the Silurian limestone bedrock of the Upper and Lower Attawapiskat formation (McDonald, 1989; Whittington and Price, 2012).

An assessment of topographical (digital elevation models; DEM) and GeoEye satellite imagery obtained within the main study area, revealed a landscape comprising 64% bog, 30% fen and 6% open water (fen pools, bog pools and stream channels) (Bouffard, 2014). The headwaters of the various drainage networks within the study area are typically arranged in a dendritic-like pattern, made up predominantly of low-lying fen-water-tracks and channel fens with non-existent to intermittently connected channels, during high flow periods (Richardson et al., 2012). Irregularly shaped domed-bogs are often interspersed within the fen-water-track pools, while dense fen-conifer ridges (oriented perpendicular to the flow path) separate the fen-water-track pools (Difebo et al., 2015). Domed-bog lobes and islands are found on the inter-fluvial divides of the low-lying channel fens and fen-water-tracks which tend to be ovoid-shaped, tapering downstream. These bogs often support an irregular network of pools located along the crests, as well as minerotrophic drainage nodes consisting of conspicuous pool networks cascading off the flanks. These features have been referred to as seeps (Sjörs, 1959), internal water tracks (Glaser, 1987) and fen water tracks (Leclair et al., submitted); and herein, as internal fen-water-tracks.

Fen vegetation includes sedges (*Carex spp*) and bilberries (*Vaccinium uliginosum*), which dominate the nutrient-rich channel fens, to *Sphagnum* mosses (particularly *S. magellanicum* and *S. rubellum*) and stunted tamarack (*Larix laricina*), which dominate the nutrient-poor internal fen-water-tracks. The nutrient-poor bogs are typified by Sphagnum (particularly *S. fuscum*), caribou lichen (*Cladonia rangiferina*), and stunted black spruce (*Picea mariana*). Bioherms, palsas and riparian forests are typified by *Sphagnum*, caribou lichen, and more dense black spruce.

The peatlands overlay a varying thickness of glacial till and fine-grained glacio-marine sediments (hereafter referred to as marine sediments, MS) deposited by the Tyrell Sea (0 to 100 m, but more typically ~20 m) which is dominated by clayey silt-sized fractions of quartz, illite, chlinochlore, and calcite, and interspersed with relic shells and layers of coarser-grained materials (Ali, 2013; Dredge and Cowan, 1989).

Located within the subarctic climate region of Ontario, the HJBL experiences long cold winters, short cool summers and moderate precipitation. The two closest meteorological stations with long-term records are the inland community of Lansdowne House (located ~250 km west of the study area), and the coastal community of Moosonee (located ~250 km south-east of the study area). The average yearly temperatures are -1.3^{0}C and -1.1°C, and average yearly precipitation is 700 mm and 682 mm (of which, ~30% and ~28% falls as snow) for Lansdowne House and Moosonee, respectively, based on 1971 to 2000 climate normal (Environment Canada, 2008), which Whittington et al. (2012) showed to be representative of the mine site.

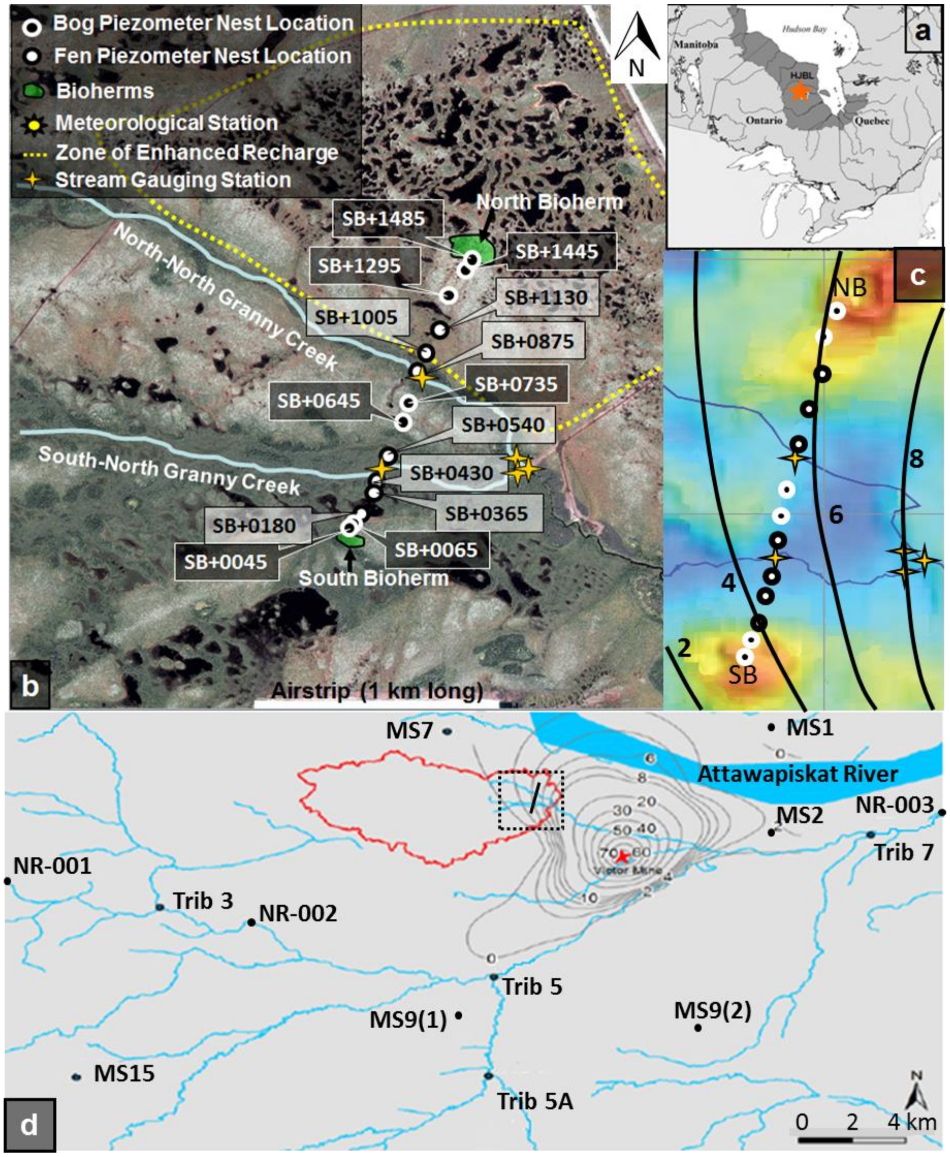

Figure 2-1: Site Map. (a) Location of the Study Area within Ontario (orange star); (b) Nest locations along the Main Research Transect (approximately 1.5 km long), including: North and South Bioherms (which include bedrock monitoring wells), stream gauging stations (yellow stars), piezometer nests (white fill/black outline, bog; black fill/white outline, fen), and the location of the predicted enhanced recharge zone (HCI, 2004). Marine sediment monitoring locations were available at SB+0645, SB+1005, and SB+1295. The meteorological station is outside of the image at the eastern corner of the enhanced recharge zone; (c) Marine sediment thickness map. The gradient for increasing marine sediment thickness is represented by dark red to yellow (i.e. exposed bedrock [0 m] to thin marine sediments [~10 m]) and light blue to dark blue (to a maximum thickness of ~50 m) (map adapted from HCI (2004)). Also included is the relative distance of the Main Research Transect within the upper bedrock aquifer depressurization drawdown cone (m) as of 2011-August (~3 – 6 m) (solid black lines), and; (d) Location of all remote surface water monitoring stations (Tribs, Attawapiskat River and Nayshkootayaow River [NR]), remote bedrock and marine sediment sampling locations (MS), total drainage area for the North Granny Creek (red outline), the Main Research Transect (black line), location of the pit (red star) and the upper bedrock aquifer depressurization drawdown cone (isobars are provided at 2 m intervals from 0-10 m and at 10 m intervals from 10-70 m).

Table 2-1: Name of piezometer nests (Nest) located along the main research transect (SB+), Tributary 5A (Trib 5A+) and MS15 (CB+), landscape unit (Type), landscape sub-unit (Sub-Type) and piezometer depths based on depth below ground surface (mbgs) to centre of 30 cm slotted intakes (Piezometers).

Nest	Type	Sub-Type	Piezometers (mbgs)
SB	Bioherm	Out-cropping	10, 30
SB+0045B	Bog	Domed	0.9, 1.5, 2.4
SB+0065B	Bog	Domed	0.9, 1.5, 2.25
SB+0100B	Bog	Domed	0.9, 1.5, 2.43
SB+0180F	Fen	Interfluvial water-track	0.9, 1.5, 1.9
SB+0365F	Fen	Interfluvial water-track	0.9, 1.5, 1.7
SB+0430F	Fen	Channel	0.9, 1.5
SNGC	Surface Water	Stream	
SB+0540F	Fen	Channel	0.9, 1.5, 1.7
SB+0645B	Bog	Domed	0.9, 1.5
SB+0735B	Bog	Domed	0.9, 1.5, 1.9
NNGC	Surface Water	Stream	
SB+0875F	Fen	Channel	0.9, 1.4
SB+1005F	Fen	Internal water-track	0.9, 1.5, 1.8
SB+1130F	Fen	Internal water-track	0.9, 1.5, 2.4
SB+1295B	Bog	Domed	0.9, 1.5, 2.38
SB+1445B	Bog	Domed	0.9, 1.5, 2.57
SB+1485B	Bog	Domed	0.9, 1.5
NB	Bioherm	Out-cropping	25.5, 58.5, 64.5
Trib 5A	Surface Water	Stream	
5A+0035B	Bog	Domed	0.9, 1.5
5A+0100B	Bog	Domed	0.9, 1.5, 2.14
5A+0210B	Bog	Domed	0.9, 1.5, 2.26
5A+0400F	Fen	Interfluvial water-track	1.5, 2.26
5A+0475F	Fen	Interfluvial water-track	0.9, 1.5
CB	Bioherm	Out-cropping	1.3, 10.85
CB+0025B	Bog	Domed	0.9, 1.5, 2.33
CB+0040B	Bog	Domed	0.9, 1.5, 2.6
CB+0080F	Fen	String	0.9, 1.5, 2.72
CB+0150F	Fen	String	0.9, 1.5, 2.5

2.2.1 Main Study Area (Impacted Site)

The main study area comprised a portion of the North Granny Creek watershed, a sub-tributary of the Attawapiskat River. North Granny Creek (NGC) has two channels: South-North Granny Creek (SNGC) and North-North Granny Creek (NNGC) (Figure 2-1 [b]). The SNGC and the NNGC have drainage areas of 21 km^2 and 9 km^2, respectively (Richardson et al., 2012) and are minimally (if at all) incised into the underlying mineral sediments. The thickness of these underlying fine-grained sediments varies across the main research transect (Figure 2-1 [c]), from 0 to ~10 m near the North and South Bioherms and up to ~50 m near the mid-section of the transect (HCI, 2004). Hydraulic conductivity of the marine sediments averaged ~10 mm day^{-1} (Whittington and Price, 2013). Peat thickness along the transect ranges from 1.4 to 2.57 m, with hydraulic conductivities averaging 340 mm day^{-1}, generally decreasing with depth from 698, 230 and 58 mm day^{-1} at depths of 0.9 mbgs, 1.5 mbgs and just above the peat/marine sediment interface, respectively (Whittington and Price, 2013). The main research transect crosses the lower reaches of the NGC watershed, spanning 1.5 km from the South to North Bioherms (SB; NB) [nomenclature: (Whittington and Price, 2012)] and is located within ~3 km of the DeBeers Victor Mine (Figure 2-1 [d]).

Peat piezometer nests were located within the various peatland types along the main research transect and named according to their relative distance (in meters) from the bedrock monitoring wells located beside the South Bioherm (see Figure 2-1 [b] and Table 2-1). The North Bioherm also includes a nest of bedrock monitoring wells. SB+0645, SB+1005 and SB+1295 also include piezometers in the marine sediments. Surface water monitoring stations were located within each of the stream channels along the main research transect (SNGC-MID and NNGC-MID), as well as downstream of the main research transect at their confluence (SNGC-LOW, NNGC-LOW and NGC-CON) (Figure 2-1 [b]).

2.2.2 Aquifer Depressurization

The DeBeers Victor Mine commenced dewatering of the pit in January 2007 at ~8,200-18,000 m^3 day^{-1}, increasing to ~85,000 m^3 day^{-1} by February, 2010, and averaged ~80,000 m^3 day^{-1} throughout 2011 (Itasca Dever Inc, 2011). Pit dewatering has resulted in a progressively more extensive cone of depression within the Upper Attawapiskat bedrock, which extends below the lower reaches of the NGC drainage basin. It was estimated that the cone of depression encompassed 6.7, 11.2 and 14.5% of the North Granny Creek watershed during 2009, 2010 and 2011, respectively (Leclair et al., 2015).

The shape of the drawdown cone is irregular (Figure 2-1 [d]) due to the presence of sub-cropping and out-cropping bioherms where fine-grained sediments are thin or not present (e.g. Figure 2-1 [c]). Drainage through these underlying marine sediments is dependent on their hydraulic conductivity and thickness (Whittington and Price, 2013). Due to the presence of cropping and sub-cropping bioherms and thin sediments near the northern end of the transect (SB+1005 to the North Bioherm) there is a zone of enhanced recharge (HCI, 2004; Whittington and Price, 2012; Whittington and Price, 2013) (Figure 2-1 [b]). By the end of the study period (August-2011), the main research transect was situated over the zone of bedrock depressurization (South Bioherm ~3 m; North Bioherm ~6 m) (Figure 2-1[c]).

After 5 years of aquifer depressurization, vertical hydraulic gradients (negative indicating downward) increased from near-zero values (-0.001) to -0.45 near the bioherms (SB+0045 to SB+0100; SB+1445 and SB+1485), -0.03 within the middle the main research transect (SB+0180 to SB+0875) and -0.20 within the enhanced recharge zone (SB+1005 to SB+1295) (Whittington and Price, 2013). These gradients have also resulted in an increased groundwater recharge flux from an estimated pre-mining flux of 0.007 to 0.07 mm day^{-1} (HCI, 2004) to approximately 0.26 and 2.1 mm day^{-1} in non-bioherm and near-bioherm areas, respectively.

2.2.3 Remote Sites (Non-impacted Sites)

All remote monitoring sites were outside the zone of influence of mine dewatering but within ~25 km of the main research area (Figure 2-1 [d]). A total of 8 remote stream sampling locations were used in this study; one at the Attawapiskat River, three along the Nayshkootayaow River (NR) and four along tributaries of the Nayshkootayaow River (Tribs). The Attawapiskat River, Nayshkootayaow River, and Trib 7 are incised into the bedrock; Trib 3, Trib 5 and Trib 5A are partially incised into the marine sediments. The reader is directed to Richardson et al. (2012) for a more in depth comparison of the various catchments, including: geomorphology, scale and runoff generation.

A total of 6 remote marine sediments and bedrock monitoring stations (MS) and 2 remote transects consisting of a series of peat piezometer nests (MS15 and Trib 5A) were used in this study (Figure 2-1 [d]). The control site MS15, consists of a 150 m transect of piezometer nests beginning by a bioherm (CB), through bog (CB+0025B and CB+0040B) and terminating within a string fen (CB+0080F and CB+0150F) (Table 2-1). The transect represents a non-impacted bioherm and an adjacent bog similar to those located at either end of the main research transect (SB+0045 and SB+0065; SB+1295 to SB+1485). The control site Trib 5A, consists of a transect (~475 m long) spanning from a stream channel (Trib 5A) through bog (5A+0035B to 5A+0210B) and terminating within a low-lying fen-water-track (5A+0400F and 5A+0475F) (Table 2-1). The Trib 5A watershed has equivalent drainage area to the North Granny Creek (30 km^2) (Richardson et al., 2012) and represents the bog nests located within the middle of the main research transect (SB+0645 and SB+0735) and the fen nests located within the southern portion of the main research transect (SB+0180 to SB+0430).

2.2.4 Review of Previous Hydrological Assessments Pertinent to the Results of this Study

Frost table depth, thickness and thaw were found to be dependent on landscape type (Leclair et al., submitted). Fens developed a thicker frost layer (~25 cm) than the bogs (~15 cm), and frost layers within fen and bog hollows thawed earlier (mid-May) than bog hummocks (remained into June).

Annual oscillations in water tables followed the typical pattern of declining through the summer, recovering in the fall, declining over the winter and recovering through the spring; this pattern was more pronounced within the impacted sites (Figure 2-2 [b]). While fall water tables generally recovered at both impacted and non-impacted sites, spring recharge did not necessarily recover completely along the main research transect (Figure 2-2 [b]) (Leclair et al., 2015) as a consequence of enhanced winter drawdown. As a result, from 2008 to 2011, seasonal water tables declined 0.52 and 0.35 m within the bogs and fens, respectively, and 0.71 and 0.31 m within areas impacted by bioherms and not impacted by bioherms (i.e. impacted areas are within 25 m proximity to bioherms), respectively (Whittington and Price, 2013).

Runoff also followed the seasonal patterns of peatland water tables (Figure 2-2 [c]) noted above (i.e. runoff decreased/increased with decreasing/increasing water tables). Overall, the North Granny Creek watershed experienced a decline in seasonal runoff over the course of the study period (Leclair et al., 2015). This was not exclusively attributable to aquifer depressurization, as the reference sites also experienced the same trend due to seasonal weather; runoff was determined to be greater during ‘wet’ years (i.e. 2009) than in subsequently ‘drier’ years (i.e. 2010 and 2011). However, individual runoff events were found to be greater within the non-impacted catchments, as compared to the impacted catchment, which was determined to be an indication of less water being detained in storage. In essence, dewatering influence was reflected more in the increased downward hydraulic gradients within the zone of influence rather than a decline in runoff.

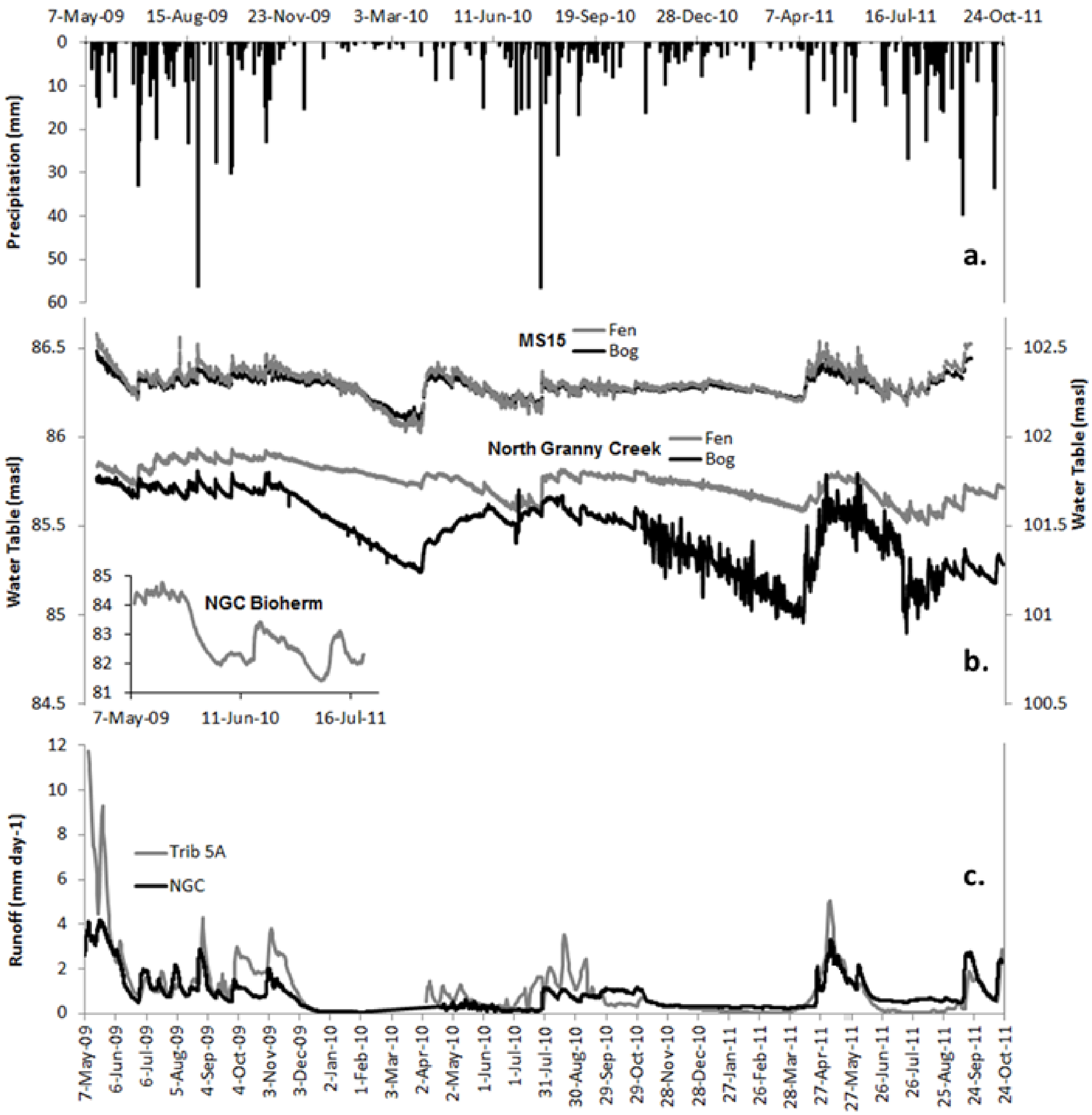

Figure 2-2: Daily precipitation events (a); continuous long-term water tables of bogs, fens and bioherm (inset) located within the North Granny Creek (NGC) and MS15 catchments (b); and, continuous long-term runoff of the NGC and Trib5A catchments (c). Note that the MS15 bioherm is not shown on this figure, but would resemble a straight line on the same Y-axis as the NGC bioherm (b. inset) (Whittington and Price, 2012). Merged Figures 3 and 4 from Leclair et al. (2015).

2.3 Methods

2.3.1 Hydrological Instrumentation and Monitoring

All bedrock (3 m long screening) and marine sediment (30 cm long screening) monitoring locations were instrumented by DeBeers' as part of the mine's compliance monitoring program. They were installed at various depths (Table 2-1), sand packed and sealed with bentonite. Each peatland observation point along the main research transect, MS15 and Trib 5A was instrumented with one fully slotted monitoring well (~1 m in length) and a nest of piezometers perforated with 30 cm slotted intakes. Peat depth permitting, each piezometer nest consisted of three piezometers (located in a line, within ~50 cm of one another); two having their slotted intakes centered at 0.9 m and 1.5 m (relative to ground surface) (where possible) and the third located just above the marine sediment-peat interface (Table 2-1). Monitoring wells and sample collection locations were also included at Trib 5A, each stream channel along the main research transect (SNGC-MID; NNGC-MID), as well as at the confluence of the SNGC and NNGC stream channels (SNGC-LOW; NNGC-LOW; NGC-CON) (Figure 2-1 [b]). Peatland wells and piezometers were constructed from 2.5 cm inside diameter PVC pipes and wrapped with 250 μm Nitex mesh. Each pipe was instrumented by hand-auguring a pilot hole to the required depth prior to pipe insertion.

Manual hydraulic head measurements were made on a regular basis (< weekly) during the ice-free seasons of 2009 to 2011. Continuous water table measurements were obtained from select groundwater monitoring wells using pressure transducers set to record at hourly intervals throughout the study period. The recordings were corrected for barometric pressure with a transducer located at the mine site (<4 km away) and adjusted according to manual measurement comparisons.

2.3.2 Water Sampling and Chemical Analyses

Water samples were collected from piezometers at various times throughout the spring and summer of 2008, 2009, 2010, and 2011, and once during the fall of each year. The tops of the piezometers were loosely covered between monitoring events to prevent contamination via precipitation and/or airborne particulate matter. Prior to porewater sampling, piezometers were purged dry (when possible) via low-flow peristaltic pumps and allowed to recover overnight. Peat-water was extracted with low-flow peristaltic pumps; stream-water samples were obtained from monitoring stations at the time of discharge measurements via standard grab sampling procedures; snow and surface water samples were taken from locations adjacent to piezometer nests, and rain samples were extracted from a bulk rain gauge within 24 hours of secession of storm events.

A 10% HCl bath and a thorough de-ionized water rinse were used to clean Teflon tubing and sampling bottles between each sampling event; to prevent piezometer cross-contamination, tubing and sample bottles were rinsed thoroughly with de-ionized water between each piezometer purge and sample extraction, and each sample bottle was rinsed with piezometer peat-water (or stream-water) from the intended source prior to sample collection.

Samples were collected in clean 250 ml Nalgene bottles, filtered through 0.45 μm membrane filters, and then decanted for storage prior to analysis of stable isotope ratios of oxygen and hydrogen (^{18}O and ^{2}H) and chloride. Samples were stored at room temperature with zero headspace for isotopes and frozen for preservation of chloride, prior to being transported to the University of Western Ontario for analysis. Chloride samples were analyzed by the Biotron Institute for Experimental Climate Change (Western University) using a Dionex ICS-1600 ion chromatography system, with a detection limit of 0.01 mg L^{-1}. Water samples were analyzed for stable isotope ratios of oxygen and hydrogen using Cavity Ring Down Spectroscopy (Picarro L2120-i) at the Ecohydrology Laboratory (Western University); ratios were expressed in δ notation as deviations per mil (‰) from the Vienna Standard Mean Ocean Water (V-SMOW, precision: $\delta^{18}O \pm 0.1‰$, $\delta^{2}H \pm 0.5‰$).

2.4 Results

2.4.1 Meteorological Conditions throughout the Study Period

The average monthly temperatures and precipitation recorded at the on-site weather station were generally within the range of values recorded at Lansdowne House and Moosonee each year of the study. The average temperature recorded on-site during the 2008 and 2009 field seasons (1-May to 31-August) were slightly cooler than the long-term averages and during the 2010 and 2011 field seasons were within the normal range (Table 2-2). In addition to having a shallow snowpack and early melt (February/March) in 2010, precipitation was less than normal during the 2010 field season, as well as during the 2011 field season. Total seasonal precipitation for 2008 was normal and was much higher than normal for 2009 (Table 2-2).

Table 2-2: Meteorological variables recorded on-site, Lansdowne House (LH), and Moosonee (M) from (1-May to 31-Aug). LH and M are based on the 30-year (1071-2000) Canadian Climate Normals from Environment Canada.

	2008	2009	2010	2011	LH	M
Average Air Temperature (°C)	11.9	10.4	13.0	12.5	13.4	12.0
Precipitation (mm)	298	380	276	284	333	294
Snow Depth (cm)	43	75	0	39	56	35
Snow Date	05-Apr	09-Apr	-	12-Apr	31-Mar	31-Mar

2.4.2 Isotopic Analyses and Chloride Concentrations

All precipitation samples obtained throughout the entire study period were used to determine the local meteoric water line (LMWL) ($\delta^2H = 7.5x + 4.6$) (Figure 2-3 [a]). The seasonal range in isotopic composition of precipitation was large and became more depleted with time; early spring rains comprised the highest ratio of heavy isotopes, while snow samples comprised the most depleted. The local evaporation lines (LEL) of individual data groups (e.g., bogs, fens, bedrock) ranged in slope from 4.4 (bedrock) to 5.8 (bogs), but generally intersected the LMWL at approximately the same value (~-15.2 for $\delta^{18}O$ and ~-109.7 for δ^2H) (Figure 2-3 [b-d]). There were no discernable differences in isotopic composition with depth or season of sample extractions for bog, fen, marine sediment and bedrock samples (deciphered data not shown). There were however, differences in isotopic composition with year of sample extraction; this is addressed in Section 2.4.2.1.

Stream sample collection typically did not commence until snowmelt was well under way each year; only 2 samples were obtained prior to May of each year. These samples were obtained from Trib 5A (2009) and NGC (2011) on 12-April and had isotopic compositions of -26.13 ($\delta^{18}O$) and -190 (δ^2H), and -21.1 ($\delta^{18}O$) and -156.3 (δ^2H), respectively (not shown on the figure). The majority of stream samples collected during early spring (May of each year) (located within the dashed ellipse in Figure 2-3 [e]) tracked relatively well along the LMWL but below the LEL of all stream samples. There were no discernable isotopic differences between samples obtained from the Remote Tributaries and the North Granny Creek. Differentiation in the isotopic composition of stream samples occurred later in the season (June) (dotted ellipse in Figure 2-3 [e]), and the much more enriched summer stream samples (within the dotted ellipse in Figure 2-3 [d]) were obtained from the North Granny Creek during 2010 and 2011. All other samples collected during the summer, fall and winter generally plotted on or between the LEL and LMWL with no discernable differences between the Remote Tributaries and North Granny Creek samples.

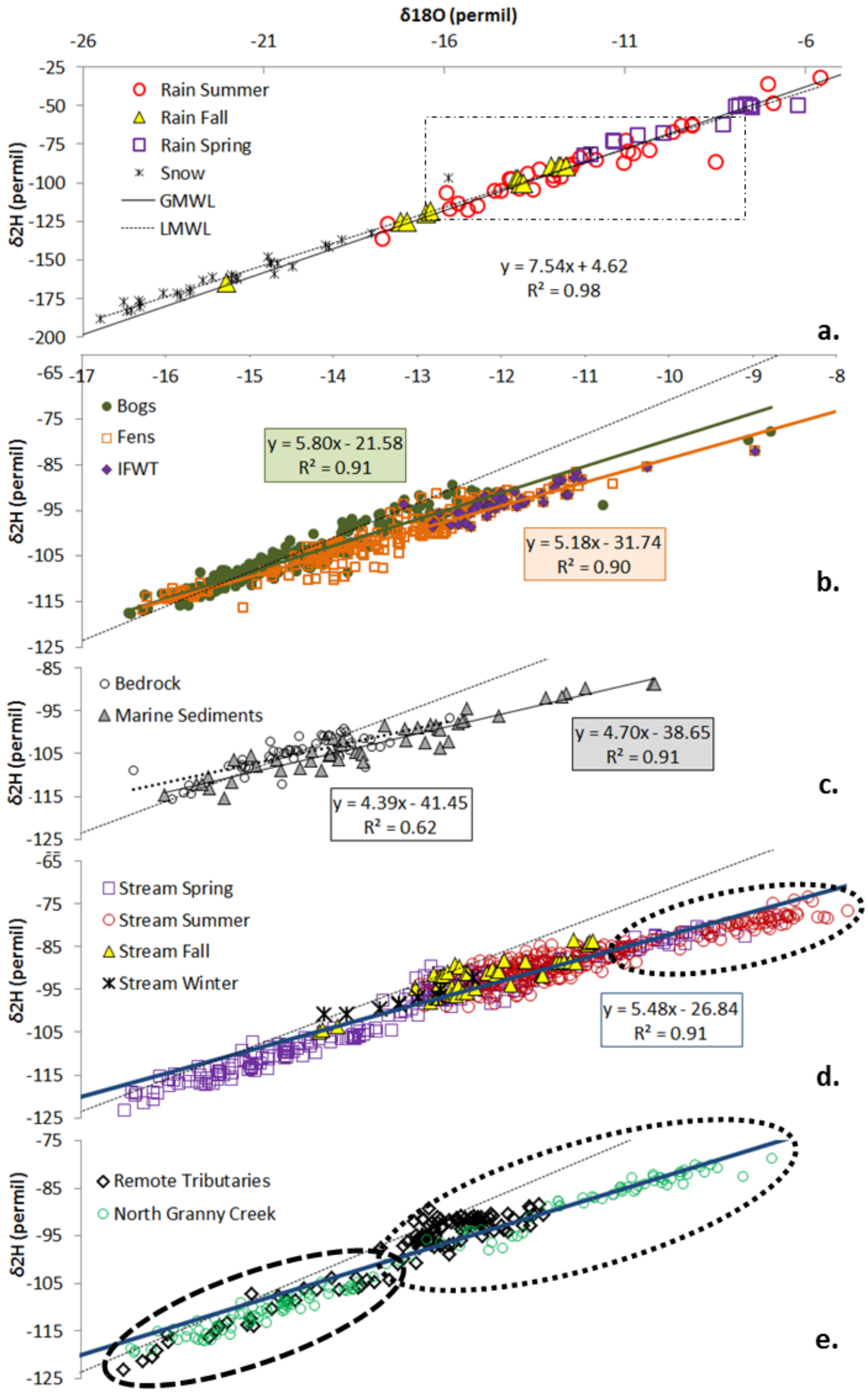

Figure 2-3: Plots of stable isotope ratios of hydrogen and oxygen (δ^2H versus δ^{18}O, ‰ VSMOW) for all water samples obtained over the entire study period (May-2008 to October-2011) from all rain gauging stations, piezometer nests and stream gauging stations located along the main research transect and remote sites. Plots are separated into seasonal precipitation (a); bog, fen and internal fen-water-track (b); bedrock and marine sediments (c); seasonal stream samples (d); and, spring stream samples from all Remote Tributaries and the North Granny Creek (e). The dotted ellipse in (d) encompasses stream samples obtained from the North Granny Creek during late-spring (June) and summer (July and August) 2010 and 2011. The dashed and dotted ellipses in (e) encompass all stream samples obtained during May and June of each year, respectively. The global meteoric water line (GMWL) and local meteoric water line (LMWL) (based on all rain and snow samples) are displayed as solid and dashed lines, respectively. Evaporation lines for each data set (based on linear relationships) are also displayed and colour coded.

Box-and-whisker plots were created for the isotopic composition and chloride concentrations of all available water samples obtained from each piezometer nest (each piezometer depth interval inclusive in each plot, with the exception of interface piezometers for chloride), stream sample location and precipitation sampling events (broken into seasons of sample collection for isotopes; rain and snow for chloride), and is displayed in Figure 2-4 (a. and b., respectively). Chloride concentrations tended to increase slightly with depth, particularly at nests located within the middle of the main research transect and at Trib 5A (deciphered data not shown).

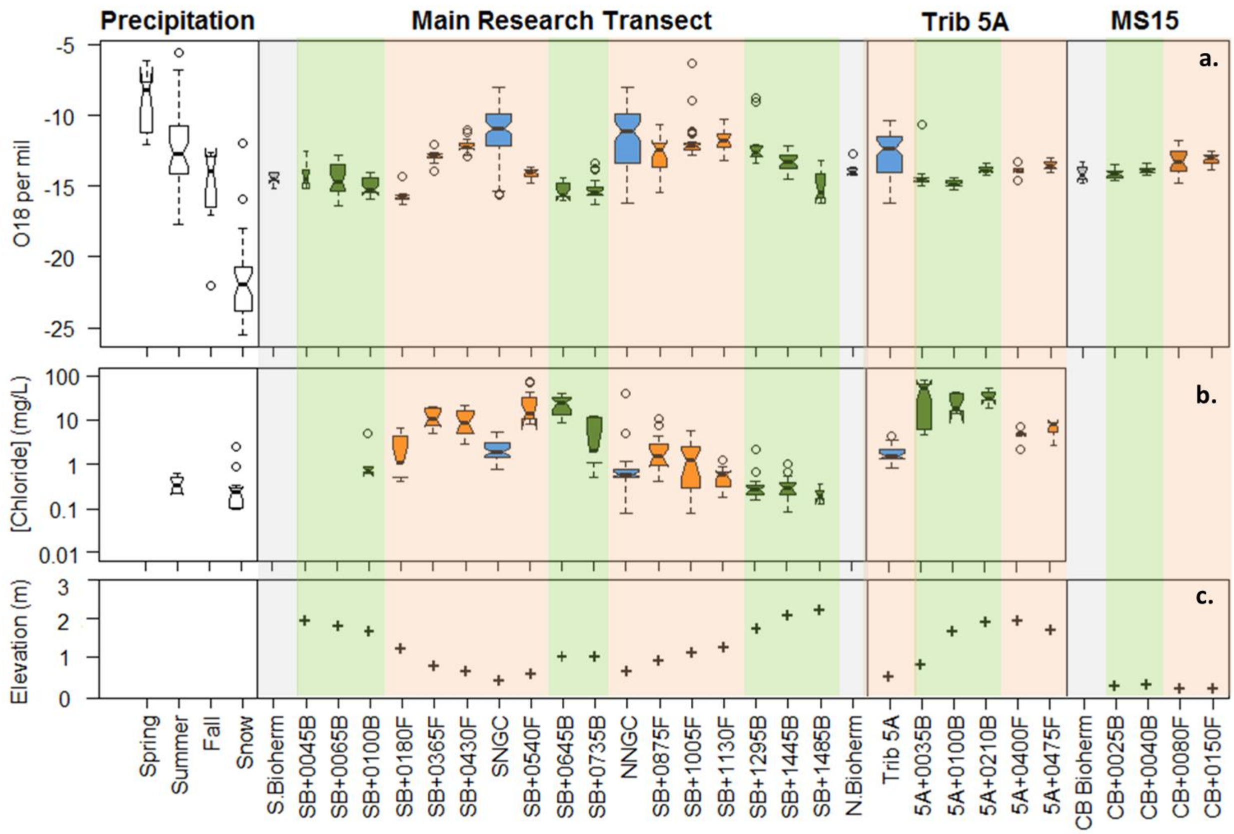

Figure 2-4: Box plots of stable isotope ratios of $\delta^{18}O$ (‰ VSMOW) (a) and chloride concentrations (b) for all available precipitation (white), stream water (blue) and bioherm groundwater (grey) samples, as well as for all peat-water samples obtained from piezometer nests (bog [green]; fen [orange]) located along the main research transect, Trib 5A and MS15. Stable isotope ratios of $\delta^{2}H$ are not shown but followed similar trends. Peat-water samples include all piezometer depths where available (0.9, 1.5 mbgs and interface for isotopes; 0.9 and 1.5 mbgs for chloride). There were insufficient data for analysis of chloride within the interface piezometers and from all nests located at MS15. Width of box plots represent sample size (e.g. top Figure: n=5 at SB+0045 to n=70 at NNGC). Box end points (whiskers) are the 5 and 95% percentiles and outliers are shown as open dots. Where notches do not overlap, the median of the data are significantly different, with 95% confidence (R Development Core Team, 2009). Bottom graph (c) represents the ground elevation (m) relative to an arbitrary local datum at each piezometer nest.

The isotopic signatures of all peat and groundwater samples were similar to the isotopic signatures of summer and/or fall rain (based on medians) (Figure 2-4 [a]); however, the SNGC and NNGC stream channels were significantly more enriched in $\delta^{18}O$. In general, the isotopic composition of peat-water became increasingly more enriched from bog to fen within Trib 5A, MS15, downslope of the North Bioherm toward the NNGC, as well as from the central domed bog (SB+0735) toward the SNGC. Conversely, the isotopic composition of peat-water became more depleted with distance downslope of the

South Bioherm (from SB+0045 to SB+0180) and was much more enriched with proximity to the SNGC. Based on interquartile ranges, the isotopic composition of peat-water samples at the remote sites were generally tight and evenly distributed; conversely, data obtained from the main research transect were more variable and right skewed toward more enriched isotopes (particularly within the bogs).

All precipitation and stream samples contained very low concentrations of chloride (Figure 2-4 [b]). All peat-water obtained from piezometer nests located at the southern and northern ends of the main research transect (i.e. SB+0100 and SB+0180; NNGC to SB+1485, respectively) were also dilute in chloride. Peat-water samples obtained from within the middle of the main research transect (bogs and fens inclusive) (SB+0365 to SB+645) displayed much higher concentrations. These areas generally coincide with lower topographical locations (Figure 2-4 [c]) and greater marine sediment thickness (Figure 2-1 [c]). All bog samples obtained from Trib 5A displayed high chloride concentrations, significantly higher than the adjacent fen. There were insufficient data to display box-and-whisker plots for a few piezometer nests along the main research transect and for all piezometer nests at MS15.

2.4.2.1 Temporal (Yearly) Changes in Isotopes and Chloride Concentrations

To assess the temporal (yearly) changes in isotopic composition and chloride concentration at individual piezometer locations, a simple linear regression was performed to determine the slope of the relationship between isotopic composition, or chloride concentration, and time. A positive slope means enrichment with time (or increase in concentration) (e.g. slope for SB+0065B 1.5mbgs is +0.0021 which is an enrichment of $\delta^{18}O$). Details of the calculations and an example are provided in Appendix A. Each data point in Figure 2-5 (a. and b.) represents a regression at the indicated location. For example, the $\delta^{18}O$ trend through time at SB+0065B displays a positive slope at all depths (blue diamond 0.9 mbgs, 0.0026; red square 1.5 mbgs, 0.0021 [from Appendix A, noted above]); and green triangle interface, 0.0014), meaning that all piezometers located at SB+0065B are showing a trend of enrichment through time. Conversely, the 0.9 mbgs piezometer at SB+1485B (the last point along the main research transect) displays a negative slope (-0.0005) indicating isotopic depletion through time.

The majority of water samples obtained from piezometers located along the main research transect displayed isotopic enrichment through time, particularly within the bogs (Figure 2-5 [a]). Isotopic enrichment was greatest atop the domed bogs (with the exception of SB+1485B which was located directly adjacent the North Bioherm) and became less apparent with distance downslope towards the fens (Figure 2-5 [a. and b., respectively]). Enrichment tended to be greatest within the 0.9 mbgs zone of the bogs, while the 0.9 mbgs zone of the internal fen-water-track (SB+1005F and SB+1130F) displayed the greatest isotopic depletion. Conversely, the majority of peat-water extracted from the remote sites displayed either no change or isotopic depletion with time. There were no discernable trends between the remote bogs and fens. Caution must be used in interpreting these data as sample numbers were low, thus many of the slopes were from regression relationships that were not significant (Table A-1 and A-2).

Chloride concentrations declined through time along the main research transect, particularly within the 1.5 mbgs zone of the fens (Figure 2-5 [b]). Data points for SB+0540F are not shown on the Figure due to having considerably larger regression slopes compared to all other data (0.03 for 0.9 cmbgs; 0.09 for 1.5 cmbgs); see Appendix A. There were only 2 nests with sufficient interface piezometer data for regression of chloride (Trib5A+0210B and SB+1130B); both were located in bogs and displayed little to no change in chloride concentration with time. SB+0635B was the only bog nest to display a noticeable decline in chloride concentrations with time. As noted above, this nest contained higher chloride concentrations (Figure 2-4 [b]) than other bog nests along the transect, and was located within an area underlain by a thicker layer of marine sediments (Figure 2-1 [c]). Chloride concentrations generally increased with time within Trib 5A, with little distinction between bogs and fens. There were insufficient data for regression of chloride concentrations at MS15.

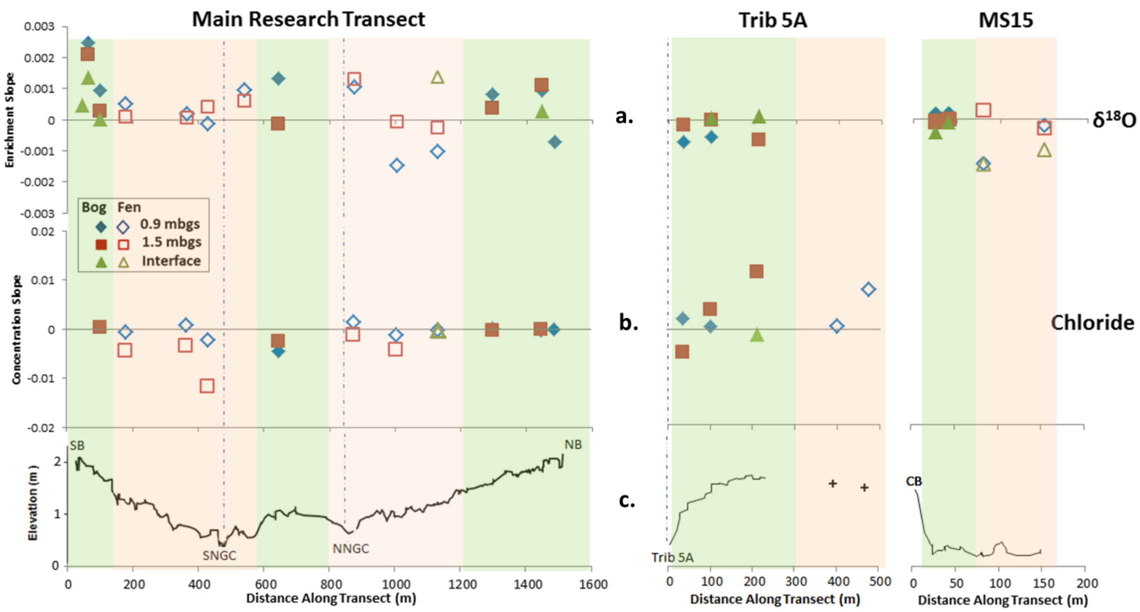

Figure 2-5: Linear regression analysis was performed on the isotopic analyses (a) and chloride concentrations (b) of all peat-waters against time of sample extraction for each piezometer located along the main research (left), Trib 5A (middle) and MS15 (right) transects. The slope of each regression line is displayed on the figure. Open (fen) and closed (bog) points are coded according to depth of sample extraction (0.9 mbgs, 1.5 mbgs and interface piezometers; blue diamond, red square and green triangle, respectively) and displayed with relative distance of sample extraction along each transect. Positive value indicates isotopic enrichment and/or concentration in chloride over time; negative value indicates isotopic depletion and/or decline in chloride concentration with time. Stable isotope ratios of hydrogen (δ^2H) is not shown but followed similar trends; see Appendix A. Chloride data points for SB+0540F are not shown due to considerably larger regression slopes to all other chloride data (0.03 for 0.9 cmbgs; 0.09 for 1.5 cmbgs); see Appendix A. Bottom graph (c) represents the ground elevation (m) along each transect. The ground elevation survey at Trib 5A was incomplete; therefore, the 2 crosses represent ground elevation at piezometer nests 5A+0400F and 5A+0475F.

2.5 Discussion

2.5.1 Differentiation and Connectivity between Landscape Units

Owing to the proximity of the study site to Hudson Bay and its location within the subarctic, the Local Meteoric Water Line (LMWL) plotted close to the Global Meteoric Water Line (GMWL), and within the lower range of global values (Clark and Fritz, 1997) (Figure 2-3 [a]). The isotopic composition of precipitation was seasonally dependent, with early spring and summer rains comprising the highest ratio of heavy isotopes and snow comprising the most depleted. This is due to varying condensation temperatures, degree of evaporation from falling precipitation, and seasonal shift in source area of precipitation (Dansgaard, 1964).

Fens tended to display significantly more enriched isotopic signatures than adjacent bogs (i.e. Trib 5A; MS15; SB to SNGC; SNGC to NNGC; NNGC to NB) (Figure 2-4 [a]). Open bodies of water (more frequently present within fen systems) undergo direct evaporation which results in the fractionation of isotopes by preferentially evaporating isotopically light particles and resulting in the isotopic enrichment of water left behind (Faure and Mensing, 2005). As such, despite being ~100 m apart, the more isotopically enriched nature of SB+0540F (located within a channel fen), as compared to SB+0645B (domed bog draining into the channel fen) (Figure 2-4 [a]), supports the assessment that the channel fen receives a larger portion of its waters from further upstream (Leclair et al., submitted) where more evaporation can occur due to its large proportion of open water. This is also true for the low-lying fen-water-track and channel fen nests located at SB+0365F and SB+0430F, respectively. Higher chloride concentrations measured at these nests (Figure 2-4 [b]) and upward fluxes of water determined for several years of the study period (0-2 mm day^{-1}) (Whittington and Price, 2013) also indicate the importance of groundwater contributions within the low-lying fen-water-tracks (Leclair et al., 2015). While groundwater contributions and stream contributions from the upper non-impacted region of the watershed help to maintain fen water levels during aquifer depressurization (Leclair et al., 2015; Leclair et al., submitted), they are not isolating the fens from hydrogeochemical effects of increased downward vertical hydraulic gradients. This is apparent by the isotopic enrichment of waters obtained from these fen systems with time (discussed further in Section 2.5.2.2) and a decline in chloride concentrations (Figure 2-5), discussed below.

Leclair et al. (submitted) indicated that internal fen-water-tracks are in large part, hydrologically connected to their harbouring bogs; however, the isotopically enriched signatures of the fen nests (SB+1005F and SB+1130F), compared to the adjacent bog nest (SB+1295B) (Figure 2-4 [a]), indicate that while contributed bog waters may have had time for evaporation within the fen pools, the fen likely also receives large contributions from the pool network located within the upper reaches of the internal fen-water-track. Leclair et al. (submitted) found that increased vertical hydraulic gradients and lowered seasonal water tables are resulting in a decline in connectivity between the bogs and internal fen-water-tracks (and likely from the headwater ponds of the internal fen-water-track)(Whittington and Price, 2013). This assessment is supported as the bog nests (SB+1295B and SB+1445B) witnessed isotopic enrichment with time (with the exception of SB+1485B), while the internal fen-water-track nests (SB+1005F and SB+1130F) witnessed isotopic depletion (Figure 2-5 [a]) (this is discussed further in Section 2.5.2.2). In addition, while higher chloride concentrations within the internal fen-water-tracks (Figure 2-4 [b]) indicate the minerotrophic nature of these features (Glaser et al., 2004a; Reeve et al., 1996; Sjörs, 1959), vertical hydraulic gradients have reversed as a result of aquifer depressurization (Leclair et al., submitted), and is resulting in a decline in chloride levels (Figure 2-5 [b]); this is discussed further below.

High chloride concentrations noted within the bog nest of SB+0645B, as well as Trib 5A (Figure 2-4 [b]), may be an indication of the upward movement of chloride within a marginal area (where upward gradients may exist), or the upward diffusion of chloride (where downward gradients are negligible) from the marine sediments since bog proliferation began. For example, outside the zone of influence, some bog nests have been found to have negligible or upward gradients (Leclair et al., 2015) thus allowing for the upward diffusion or dispersion of chloride, respectively (Price and Woo, 1988) and ultimately, the presence of quantifiable levels of chloride (Figure 2-4 [b]). Within the zone of influence however, downward hydraulic gradients are increasing as a result of aquifer depressurization (Leclair et al., 2015).

And, although SB+0645B is somewhat hydrologically protected from the effects of aquifer depressurization, with having a thicker layer of marine sediments beneath it (Whittington and Price, 2013), similar to the fens, the marine sediments are not isolating the bog from the biogeochemical effects of increased leaching of dilute precipitation, as indicated by a decline in chloride levels (Figure 2-5 [b]).

A decline in chloride concentration (as shown in Figure 2-5 [b]) is reflective of less upward movement (dispersive or diffusive) of mineral rich waters, due to reversed hydraulic gradients (downward instead of upward) and a subsequent increase in recharge and leaching. This decline occurred within all fens along the main research transect (particularly those displaying the highest concentrations of chloride; e.g. SB+0540F), regardless of water contributions from the upper reaches and protection from thicker layering of marine sediments (Leclair et al., 2015; Whittington and Price, 2013). Less upward movement of mineral rich waters can be particularly detrimental to the health of the fen systems, as fen vegetation relies on these chemical contributions to buffer acidity (Mitsch and Gosselink, 2000).

Conversely, chloride concentrations tended to increase within Trib 5A over the course of the study (Figure 2-5 [b]). The reason for this is not entirely clear, particularly within the bogs; however, given the high levels of chloride within these nests (Figure 2-4 [b]) it could be a result of (variably weather related) decreasing downward (within bogs) and increasing upward (within fens and marginal bog areas) vertical hydraulic gradients found within non-impacted systems throughout the study period (Leclair et al., submitted); this would allow for more diffusive and dispersive transport of chloride from the marine sediments, respectively, and a subsequent increase in chloride levels. And, although this would be occurring within the impacted study site as well, the greater hydraulic gradients changes found within the impacted region (1 order of magnitude increase (Leclair et al., submitted)) is outweighing any weather or climactic-related variability.

2.5.2 Conceptual model of Peatland Recharge

2.5.2.1 Non-impacted

A conceptual model, Figure 2-6 illustrates the nature of recharge patterns without (top: non-impacted) and with (bottom: impacted) enhanced recharge caused by mine dewatering**.** Figure 2-6 (non-impacted) illustrates the balance between inputs and outputs within a natural system from a typical spring to the following fall. By the end of the first spring (Spring-1), water tables are high. As the first summer approaches, temperatures increase and evapotranspiration dominates the water budget, resulting in a decline in the water table (Leclair et al., 2015). In the fall, evapotranspiration declines and water tables recover (Ulanowski, 2014)(Figure 3-2). As the season progresses and temperatures drop below zero, a frost layer is formed and snow begins to accumulate. Over the course of the winter, water tables decline (Figure 2-2 [b]; Figure 2-6) as baseflow continues toward the stream channel (Figure 2-2 [c]). By the following spring (Spring-2), temperatures rise, snow ripens and begins to melt. The majority of snowmelt occurs while the frost table is still present (Leclair et al., submitted; Whittington et al., 2012). Since frozen saturated peat is considered to be relatively impermeable (Price and Fitzgibbon, 1987), the majority of snowmelt is unable to infiltrate; some is detained in depressions and some runs off through the highly conductive acrotelm (Bowling et al., 2003) (represented as a black arrow in Figure 2-6). The isotopically depleted stream samples obtained on 12-April-2009 and 2011 confirm that snowmelt makes up a large proportion of the streamflow during the onset of the spring freshet. As the remaining snow continues to melt and the frost layer begins to thaw, these waters mix (represented as a horizontal solid dark blue line in Figure 2-6) and some runs off. This is indicated by the isotopically depleted stream samples displayed within the dashed ellipse in Figure 2-3 [e], which were obtained during May of each year. The samples plotted below the LMWL and LEL, indicating that stream water was primarily composed of a mixture of snowmelt and peat-water. As the frost layer continues to thaw, its waters (made up of late-summer and fall precipitation from the previous year), and any residual snowmelt that has not run off, becomes available for recharging the hydrologically depleted system (represented as a diagonal solid dark blue line Figure 2-6). This is particularly evident during the spring of 2010 when a shallow snowpack and early snowmelt (February/March) left little to no snow by the beginning of April (Whittington et al., 2012). Despite a lack of snow and minimal rain inputs, water tables began to recover by the beginning of April 2010 (Figure 2-2 [a. and b., respectively]). Within the non-impacted system (MS15), these waters were sufficient to fill the deficit that had formed, thereby leaving little room for any further incoming precipitation and as such, spring rains runoff (Figure 2-2 [c])(represented as a solid blue horizontal arrows in Figure 2-6). This is reflected in the isotopic composition of Remote Tributary stream water samples located within the dotted ellipse in Figure 2-3 [e] which show that during late spring (June), the vast majority of isotopic signatures plotted between the LMWL and LEL, indicating a mixture of groundwater and rain (Turner et al., 2010).

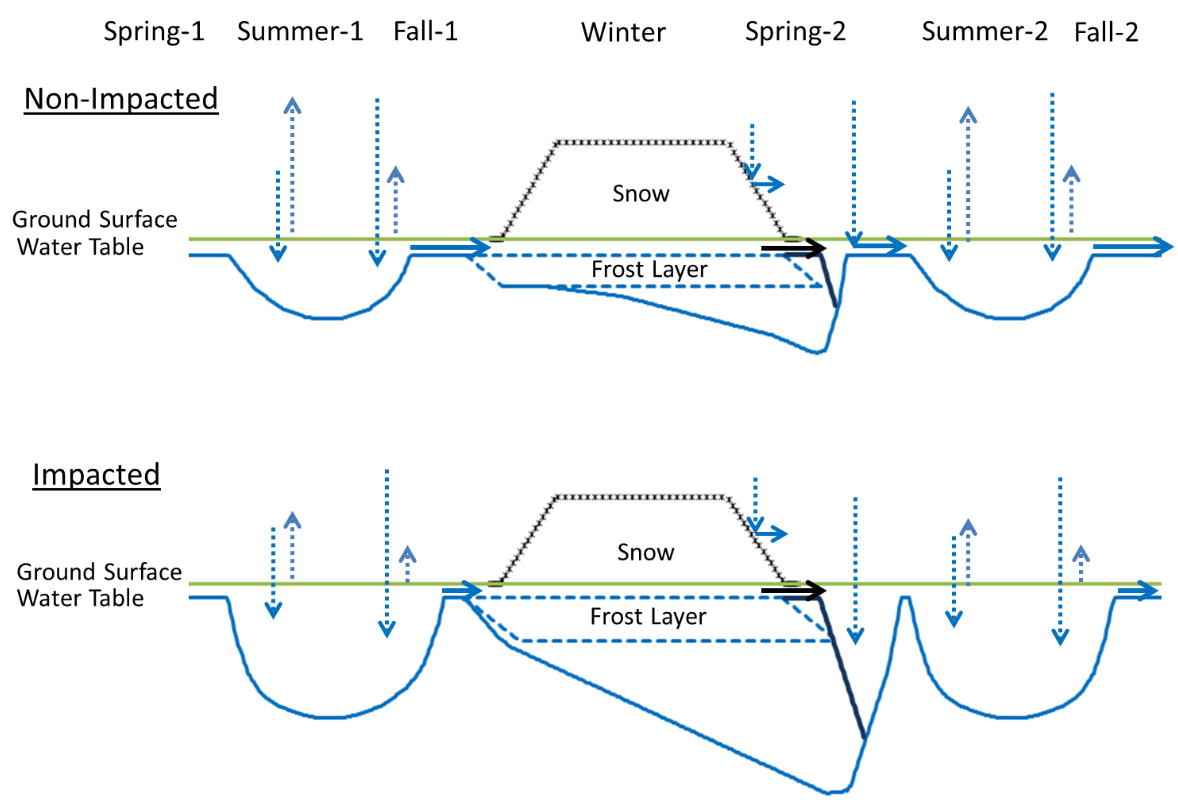

Figure 2-6: Conceptual diagram of the seasonal oscillations in water tables (solid light blue line), frost layer proliferation and thaw (dashed blue line), incoming precipitation, evapotranspiration and runoff within a natural peatland system located within the HJBL (non-impacted; top) and a system being under-drained through bedrock aquifer depressurization (impacted; bottom). Blue arrows represent incoming rain (down), evapotranspiration (up) and runoff (right). Their length represents relative magnitude. Black arrows represent snowmelt runoff. Solid dark blue line represents snowmelt and frost thaw ponded atop a thawing frost table (horizontal) and the infiltration of this water upon complete frost thaw (diagonal).

2.5.2.2 *Impacted*

As a result of aquifer depressurization, the peatlands located within proximity of the Victor Mine are experiencing various hydrological effects, such as a decline in seasonal water tables and an increase in deep seepage losses (Leclair et al., 2015; Whittington and Price, 2013). These responses have been shown to be more pronounced within proximity of bioherms (Whittington and Price, 2012), within areas underlain by a relatively thin layer of marine sediments (Whittington and Price, 2013) and are peatland specific (bogs > fens) (Leclair et al., submitted). Hydrologically based research conducted between 2008 and 2011 by Leclair et al. (2015) was unclear on the effects of these hydrological changes on runoff responses due to annual variability in weather conditions and contributions from the non-impacted upper reaches of the watershed; a closer look at the geochemistry demonstrates a shift in hydrological conditions, particularly the source of recharge and subsequent discharge. The isotopic composition of the peat-water surrounding Victor Mine is changing over time (Figure 2-5 [a]). With the exception of the internal fen-water-tracks, waters are becoming more enriched with heavier isotopes with time, particularly within the bogs, as a result of a greater infiltration of isotopically enriched spring rains; this is explained below.

Within the impacted system, the conceptual model (Figure 2-6 [impacted]) illustrates peatlands experience synchronous seasonal oscillations in water tables and runoff; however, water tables decline to a greater extent within each season and runoff response is less (see Figure 2-2). By the end of spring (Spring-1), water tables are high (at or near the surface) (see Figure 2-6 [impacted]), but as summer approaches, a combination of increased temperatures, evapotranspiration and deep seepage losses result in a greater decline in summer water tables (Leclair et al., 2015). Temperatures and evapotranspiration decline through the fall and incoming precipitation is sufficient for water tables to recover (see Figure 2-2; Figure 3-2). Below freezing temperatures form a frost layer and snow accumulates. Aquifer depressurization continues over the course of the winter resulting in a greater decline in water levels, particularly within impacted bogs (Figure 2-2 [b])(Leclair et al., 2015). As in the non-impacted system, snow melts while the frost table is still in place and the majority of snowmelt runs off (Figure 2-6; black arrow) resulting in isotopically depleted stream water samples similar to the isotopic composition of snow (i.e. the 2 stream samples obtained on 12-April-2009 and 2011). And, similar to the non-impacted system, as the frost layer begins to thaw, its waters mix with any residual snowmelt (Figure 2-6; horizontal dark blue line) and some runs off. As such, stream samples obtained from the North Granny Creek during May of each year were isotopically similar to the stream samples obtained from the Remote Tributaries (within the dashed ellipse in Figure 2-3 [e]). Differentiation between the isotopic compositions of stream water samples obtained within the Remote Tributaries and the North Granny Creek occurred later within the season (June); as shown within the dotted ellipse in Figure 2-3 [e]. As in the non-impacted sites, water tables begin to recover upon the completion of frost thaw; however, the infiltration of waters stored within the frost layer and ponded snowmelt (Figure 2-6; diagonal dark blue line) is insufficient to fill the deficit from the winter drawdown, particularly within the bogs (Figure 2-2 [b])(Leclair et al., 2015). This is particularly evident in Figure 2-2 where water levels within the impacted bog displayed a partial recovery in water table prior to a subsequent recovery and subsequent runoff response in April, 2010. Since there remains a deficit within the profile, incoming spring precipitation is available for infiltration (as opposed to exiting as runoff) and recharges the hydrologically depleted system (see Spring-2; Figure 2-6). Since the majority of incoming precipitation is infiltrating, stream samples obtained from the North Granny Creek in June of each year (within the dotted ellipse in Figure 2-3 [e]) are more enriched in heavy isotopes and plot well along the LEL, indicating less runoff of precipitation and a greater proportion of ground and peat-water contributions than the non-impacted sites. And, since the infiltrating spring rains are significantly more enriched in heavier isotopes (Figure 2-3 [a]; Figure 2-4 [a]), isotopic enrichment of the groundwater occurs over time (Figure 2-5 [a]).

Isotopic enrichment of the peatlands occurred more within the bogs than in the fens, and was progressively less pronounced with distance downslope of the bogs and toward the low-lying fen-water-tracks and channel fens (Figure 2-5 [a]). Bogs are ombrogenous features that rely solely on atmospheric sources (snowmelt in the spring, rain in the summer and fall) to recharge low water tables. Downward recharge of event waters may be greatest at the crest of domed bogs, while lateral or upward flow-paths may occur with distance downslope (toward bog margins) (Reeve et al., 2000). As such, bog peat-water contributions from further upslope (along the crest/top of domed bog) may increase the winter storage deficit along the crest because this water contributes to winter storage downslope (along the flanks and marginal area), thereby leaving more room for spring rains along the crest and less room with distance downslope, and thus, resulting in a progressive decline in isotopic enrichment downslope of the domed bogs (Figure 2-5 [a]). Since fens are minerotrophic features that rely on both atmospheric and groundwater contributions from upstream and adjacent bogs (as noted in Section 2.5.1) their winter storage deficit was less than in the bogs (Figure 2-2 [b]), resulting in less infiltration of isotopically enriched spring rain, and ultimately less isotopic enrichment (Figure 2-5 [a]).

Peat-water obtained from the piezometers nests located within the internal fen-water-track (SB+1005F and SB+1130F) showed a depletion in heavy isotopes through time (Figure 2-5 [a]). This is likely a result of their location near the enhanced recharge zone (Figure 2-1 [b]). The headwater ponds of the internal fen-water-track, which feed the subsequent ponds that SB+1005F and SB+1130F are situated, are located atop the domed bog, within the enhanced recharge zone and within close proximity to the North Bioherm (Figure 2-1 [b]). Since downward hydraulic gradients were greater within the enhanced recharge zone (due to the presence of several out-cropping and sub-cropping bioherms and the thin layer of marine sediments) and within proximity of the North Bioherm, water tables were seasonally lower within the area, thereby providing more storage capacity within these ponds (Whittington and Price, 2012; Whittington and Price, 2013) and less downgradient contributions to the internal fen-water-track pools (as noted in Section 2.5.1). Less seasonal contributions from headwater ponds, as well as from the bogs (as noted in Section 2.5.1), resulted in lower water tables prior to freeze-up (unpublished data). Therefore, upon the spring freshet, a greater proportion of (isotopically depleted) snowmelt would have been withheld in storage within the internal fen-water-tracks pools, as well as headwater ponds, which would eventually be transported downslope within the internal fen-water-track, explaining its isotopic depletion; this transport can occur within a matter of days or months depending on water levels and location along the water track (Leclair et al., submitted).

The isotopic composition of most samples obtained within Trib 5A and MS15 were relatively consistent or showing minor depletion through time (Figure 2-5 [a]). The 2 points displaying greater isotopic depletion noted within MS15 were primarily at the peat/mineral sediment interface, with no differentiation between bog and fen. The cause of this depletion is uncertain, although it may be a signal of variable weather conditions (potentially on a decadal time scale) being preserved within this layer, since recharge rates are very low within the area (Leclair et al., 2015). Isotopic depletion may also be an indication of climate change, but this has not been extensively reviewed.

While the above noted processes (weather and/or climate driven) would be occurring within the impacted study site as well, the exacerbated effects of aquifer depressurization (i.e. greater contributions of isotopically enriched spring rains; decreased diffusive and dispersive groundwater contributions of chloride) are outweighing the possible effects of variable weather conditions and climate change. This is evidenced by not only the quantity of points displaying isotopic enrichment and chloride depletion, but the discernable trends between bogs and fens and with distance along the transect (Figure 2-5). Conversely, within the remote sites, regression slope analyses were more random with respect to differences between bogs and fens and with distance along the transects.

2.5.3 Peatland Discharge and the Effects of Aquifer Depressurization

Although Leclair et al. (2015) noted lowered runoff responses within the North Granny Creek (as compared to Trib5A), they were unsure whether it was a result of aquifer depressurization or differences in watershed characteristics. Indeed, an increase in the infiltration of spring rain, as noted above, was shown to reduce the availability of precipitation for runoff and result in a shift in the proportion of groundwater/surface water contributions to the stream channel during the spring (Figure 2-3 [e]). And, a greater infiltration of event waters was not limited to the spring and can help to explain the lowered runoff responses displayed within the North Granny Creek, as compared to Trib5A throughout the remainder of each year. During the summer and fall, water tables also declined to a greater extent within the North Granny Creek watershed, than in the undisturbed system on account of aquifer depressurization (Figure 2-2 [b])(Leclair et al., 2015). Therefore, more storage space was available for incoming summer and fall precipitation. As a larger proportion of precipitation infiltrated the peatlands, less precipitation was available for runoff to the stream channels (Figure 2-2 [c]). As such, during 2010 and 2011, the isotopic composition of late spring and summer stream water within the North Granny Creek watershed was more enriched and followed the LEL more closely (i.e. the samples located within the dashed circle in Figure 2-3 [d]). These values indicate less addition of quick-flow during precipitation events and a greater proportion of base-flow with an evaporative signature. And, although 2010 and 2011 were noticeably drier years, which would naturally result in greater infiltration and less quick-flow contributions, these enriched values were not consistent within the Trib 5A catchment (or any other remote stream channels) (deciphered data not shown), indicating aquifer depressurization to be the primary explanation.

2.6 Conclusion

Peatlands of the North Granny Creek Watershed located within the zone of influence of aquifer depressurization are experiencing hydrological (Leclair et al., 2015; Whittington and Price, 2012; Whittington and Price, 2013) and hydrogeochemical alterations (Figure 2-5) which have implications on not only the peatlands themselves, but to their downstream ecosystems.

For the peatlands, the isotopic alterations of peat-water obtained from the peatlands surrounding the Victor Mine indicate that drier antecedent moisture conditions (lowered seasonal water tables) are allowing for a greater infiltration of precipitation. A decline in seasonal water tables could increase desiccation through oxygenation and higher decomposition rates (Whittington and Price, 2006), and an increase in infiltration of event waters could exacerbate these effects by flushing decay inhibiting chemical constituents (i.e. recalcitrant carbon and methane), as well as increase the supply of electron acceptors necessary for decay (Baird et al., 2013; Morris and Waddington, 2011; Straková et al., 2011). An increase in infiltrating rainwater, and less upward movement of mineral groundwater (as shown by a decline in fen chloride concentrations), may also initiate a loss in fen biodiversity through the development of a precipitation lens (which would be deficient in solutes) and the establishment of precipitation-dependent species-poor vegetation (Schot et al., 2004).

For receiving surface waters, previous research has emphasized the importance of antecedent moisture conditions on the partitioning of precipitation between recharge and discharge to stream networks (Branfireun and Roulet, 1998). With increased storage capacities, a greater proportion of precipitation is becoming available for infiltration which leaves less precipitation available for runoff and, ultimately a reduction in stream discharge. With less water available for runoff, water quality may be altered as base-flows become more important at sustaining stream flow than surface runoff. In particular, modifications in the surface and subsurface flow pathways of a peatland to a receiving surface water, could impact the chemical balance of the stream channel, including the production and transport of methylmercury (Branfireun and Roulet, 1998).

Although Leclair (2015) found that non-impacted headwaters were helping to sustain water levels within down-gradient fen systems and stream networks, our findings show that these contributions are not isolating the fens from potential hydrogeochemcial effects of increased downward hydraulic gradients and an increase in recharge. And, in a climate change scenario, where lowered seasonal water tables would reflect higher seasonal temperatures, the headwaters would be impacted regardless; resulting in an increase in storage capacity, greater infiltration of event waters and a decline in surface runoff within the upper reaches as well.

2.7 Acknowledgements

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3 Implications of Aquifer Depressurization on the Production and Export of Dissolved Organic Carbon within Internal Fen-Water-Tracks of the Hudson/James Bay Lowlands

3.1 Introduction

While only covering ~3% of the Earth's surface (Gorham, 1991) peatland ecosystems make up an estimated 10% of the world's freshwater resources, 30% of the global terrestrial soil carbon (C) reserves (Ballard et al., 2011) and are responsible for almost 10% of the global methane (CH_4) flux (Keller et al., 2004). Unfortunately, with the uncertainty of climate change and anthropogenic influences, the function of peatlands as a net carbon sink in the future remains uncertain. The Hudson/James Bay Lowlands (HJBL) host one of the largest peatland expanses in the world (Siegel and Glaser, 2006), is a substantial contributor of freshwater to the brackish James Bay (Rouse et al., 1992), plays a significant role in the global carbon cycle (Blodau, 2002) and has recently been identified as host to mineral deposits (e.g. the DeBeers Victor Diamond Mine). As with the Victor Diamond Mine (located ~90 km inland of Attawapiskat, Ontario), aquifer dewatering is necessary for mineral extraction, which may pose potential implications to surrounding peatland systems which encounter increased recharge.

It has been well documented that water table drawdown in a peatland through drainage ditches or climate change can cause changes to the hydrology and geochemistry of a system (e.g. Blodau and Siems, 2012; Keller et al., 2004; Whittington and Price, 2006); however, few articles have assessed the effects of aquifer depressurization on overlying wetland systems (Auterives et al., 2011; Johansen et al., 2011; Kværner and Snilsberg, 2011; Whittington and Price, 2012; Whittington and Price, 2013), particularly the geochemical implications. Current and future resource development pressures throughout the HJBL, highlights the importance of improving scientific understanding of biogeochemical processes within this landscape and the possible implications of aquifer depressurization. The implications of aquifer depressurization on surface water quality and fisheries are of particular concern to northern communities that rely significantly on the land for sustenance.

Dissolved organic carbon (DOC) in surface waters plays a significant role in the ecology of peatlands and their downstream aquatic ecosystems as it affects acidity, nutrient availability, metal complexation and light penetration in aquatic environments (Steinberg, 2003). Operationally, DOC is defined as any organic carbon that passes through a 0.45 μm diameter filter, and is composed of root exudates, microbial by-products, or leached material from live or dead organic matter (Webster and McLaughlin, 2010). The transport of DOC within a peatland complex is dependent on the net production of DOC and hydrologic pathways (Moore, 2013). It is well-known that temperature, plant litter quality and solute transport have a fundamental control on the decomposition rates of organic material (and ultimately, the production of DOC) (Baird et al., 2013; Belyea, 1996), as well as the presence of oxygen. Aerobic decomposition of organic material occurs at a rate ~50 times faster than anaerobic decomposition (Siegel and Glaser, 2006); as a result, the greatest rate of DOC production occurs within the zone of water table fluctuations (i.e. within the acrotelm, where oxic conditions exist) (Belyea, 1996). Likewise, lateral water movement in peatlands dominates within the acrotelm as hydraulic conductivity is greatest near the surface and decreases rapidly with depth (Price et al., 2003). Shorter groundwater residence times within the acrotelm, therefore, increases the production and export of DOC (Beer and Blodau, 2007; Morris and Waddington, 2011) as the turnover of fresh water (low in DOC) (i.e. precipitation) enhances the release and transport of DOC from the peat matrix while DOC accumulation in pore-water constrains it (Blodau et al., 2004). Consequently, the zone for the highest level of production and transport of DOC has been shown to be just below the vegetation layer and within the upper 50 cm of a peat profile (Blodau et al., 2004; Moore et al., 2003).

The water table (in relation to the ground surface) plays a significant role in both the decomposition and export of organic material within peatland systems. Altering the natural water level of a system (e.g., by aquifer dewatering) can lead to a change in DOC production and transport (Clark et al., 2009; Hribljan et al., 2014; Pastor et al., 2003) which could have large implications on aquatic biodiversity (Limpens et al., 2008), and ultimately fisheries. Recent research proximal to the Victor Diamond Mine (hereafter, 'the mine') have found lowered seasonal water tables, increased downward hydraulic gradients (Whittington and Price, 2013) and increased infiltration of precipitation (Chapter 2, this thesis) on account of aquifer depressurization. According to the above noted processes, lower water tables, increased hydraulic gradients and increased infiltration of event waters could increase decomposition through aeration and labile carbon turnover (Hribljan et al., 2014); however, lowered water tables could promote DOC accumulation, as surface runoff would be reduced (Kværner and Snilsberg, 2011; Pastor et al., 2003). In addition, under persistently lower water levels, studies have shown a decrease in pore-water DOC concentrations as the newly decomposed organic matter would be retained in the solid phase, due to the associated reduction in DOC solubility (i.e. soil water acidification). For the newly decomposed organic matter to become soluble, and subsequently cause a rise in DOC concentration, water levels must recover (Blodau et al., 2004; Clark et al., 2012; Pastor et al., 2003).

Despite lower seasonal water tables within the peatlands surrounding the mine, annual precipitation periodically recharged the hydrologically depleted system during the study period (2008-2011) (Whittington and Price, 2013) (see also Chapter 2). Of particular interest is the fall wet-up period which has been overlooked in detail during earlier studies. This period may be important for DOC export, as DOC concentrations are often greatest following periods of warm temperatures and dry conditions, when water levels recover and the accumulated decomposed organic material is flushed from the system (Eimers et al., 2008; Hinton et al., 1997; Limpens et al., 2008). The lack of a dedicated fall sampling regime may therefore explain the enigmatic hydrological and geochemical sources to creeks and rivers of the Attawapiskat River basin (Orlova and Branfireun, 2014). For example, Orlova and Branfireun (2014) could not explain the significant spatial and temporal variability in stream water DOC concentrations through end-member mixing models; however, their defined end-members were assumed to be temporally invariant, without accounting for the potential flushing (and subsequent pulse) of DOC during the fall wet-up period.

In natural peatlands, carbon cycling (and therefore, the hydrological and geochemical source of DOC to receiving surface waters) is influenced by peatland type and successional stage (Webster and McLaughlin, 2010). In peatland complexes such as the HJBL, bogs typically drain to fens, thus typically export DOC to fens, which subsequently convey it from the peatland to downstream ecosystems (Pastor et al., 2003). As such, bogs are influenced only by ombrogenous waters derived from precipitation (rain, snow and dew), are typically highly acidic (pH <4), and contain low amounts of base cations (e.g. Calcium [Ca^{2+}] < 2 mg L^{-1}). Fens are dominated by geogenous ground or surface waters (stagnant, flowing or flood water), are typically less acidic than bogs (pH 3.8-8.5), and depending on the underlying mineral substrate and the proportion of groundwater contributions, tend to be base rich (e.g. Ca^{2+} from 2 - 60 mg L^{-1}) (National Wetlands Working Group, 1997; Reddy and DeLaune, 2008; Wieder and Vitt, 2006). The water table in a natural fen system is typically located at or near the ground surface, whereas water tables within bog systems may seasonally fluctuate by up to 50 cm below ground surface (Clymo, 1984).

Located within the interior HJBL, the mine is surrounded by a complex assemblage of bogs, fens and open waters. As Sjörs (1959) indicated the overall direction of drainage across the various peatland types within the HJBL is easily seen from the air. Satellite imagery around the mine indicates watershed boundaries which encompass a multitude of these peatland features (Difebo et al., 2015) which are expected to be hydrologically and geochemically connected across various flow pathways. Of interest, and ubiquitously situated along the flanks of all large domed bogs (>20 km^2) within the HJBL (Glaser, 1987), are series of internal fen-water-tracks which have been noted by several authors (Glaser et al., 2004a; Reeve et al., 1996; Sjörs, 1959), but rarely investigated in depth.

As implied by their name, these features are similar in appearance to large fen-water-tracks (typically ≥1 km^2) although typically at a much smaller scale (<0.1 km^2) which function as conduits for runoff across large peatlands (Price and Maloney, 1994). Internal fen-water-tracks originate atop large domed bogs where large, erratic ponds begin to converge into a series of conspicuous elongated pool networks perpendicular to the slope extending down the flanks of the bogs, becoming gradually narrower toward the bog margin (Sjörs, 1959). Despite being located atop the otherwise ombrogenous domed bogs, these features have been shown to have higher pH and mineral constituents than the bogs they drain (e.g. fen: pH 5-6.7, [Ca^{2+}] of 2.5 to 30.8 mg L^{-1}; bog: pH<4, [Ca^{2+}] of <2 mg L^{-1}, respectively) (Glaser et al., 2004b). As such, they are able to support minerotophic indicator species such as tamarack and adapted sedges (National Wetlands Working Group, 1997). It is possible then, that internal fen-water-tracks may be large producers and contributors of DOC to stream networks since they are not only located atop domed bogs and connect to source areas with relatively large water table fluctuations, but they also contain a comparatively larger concentration of vascular plants which are a ready source of labile carbon (Strack et al., 2006).

The objectives of this paper are therefore three-fold: (1) to better understand water and DOC flow within and between internal fen-water-tracks and the bogs in which they are situated; (2) to determine the significance of internal fen-water-tracks in the transport of water and DOC to receiving surface waters, particularly during the fall wet-up period; and (3) to determine the possible influence of aquifer depressurization on the production and transport of DOC to receiving surface waters.

3.2 Study Site

The site is located ~90 km inland of the coastal community of Attawapiskat, Ontario within the Hudson/James Bay Lowlands (HJBL) (Figure 3-1[a]), and is approximately 3 km northwest of the DeBeers Victor Diamond Mine (52°50'N, 83°55'W). Aquifer depressurization for the Victor Mine is mostly from the Upper and Lower Attawapiskat formation, which are approximately 20 m and 80 m thick, respectively, separated by a thin (2 to 3 m) blue-green claystone layer. In places, bedrock exposures (bioherms) connected to the Upper Attawapiskat formation, reach up to six meters above the otherwise flat peatland surface as isolated circular features (Whittington and Price, 2012). Deglaciation (Laurentide Ice Sheet) and seawater inundation (Tyrell Sea) gave rise to a varying thickness (0 to 100 m, but more typically ~20 m) of glacial till and fine-grained glacio-marine sediments which are dominated by clayey silt-sized fractions of quartz, illite, chlinochlore and calcite, and interspersed with relic shells and layers of coarser-grained materials (Ali, 2013; Dredge and Cowan, 1989). The low permeability of the fine-grained sediments and its low relief, coupled with the cool temperatures of Northern Ontario, has resulted in extensive peatland development. Peatlands within the study area range in thickness up to ~3 m and comprise a complex series of bogs, fens and open channels (National Wetlands Working Group, 1997). Of particular importance to this study is the connectivity between large domed bogs (>20 km^2) and their receiving stream channels via seeps (Sjörs, 1959), internal water tracks (Glaser, 1987) and fen-water-tracks (Leclair et al., 2015); herein they are referred to as internal fen-water-tracks (FWT).

This study area comprises four internal fen-water-tracks (denoted: FWT1-4) and a small stream channel (North-North Granny Creek [NNGC] to which the internal fen-water-tracks drain [Figure 3-1 (b)]). The internal fen-water-tracks are located with increasing distance from the mine (with FWT1 being nearer the mine and FWT4 being further), but are all within approximately 3 to 5 km of the pit. Situated parallel to the aquifer depressurization drawdown contours (Figure 3-1 [b]), greater distance from the mine should represent lesser disturbance; however, geomorphological differences between each internal fen-water-track may influence each differently. For example, FWT3 is located in an area underlain by a thick layer of marine sediments (Figure 3-1 [c]) while FWT2 and FWT1 are located within close proximity to an outcropping bioherm. These 2 factors may increase the influence of aquifer depressurization based on the hydrological implications of marine sediment thickness and bioherms (Whittington, 2013; Whittington and Price, 2012). Likewise, during high-flow periods, a distinct surface water rivulet (to which an internal fen-water-track ultimately channels its waters) was visible at FWT1 and FWT4, whereas a more diffuse discharge zone was apparent at FWT2 and FWT3. So, comparisons between internal fen-water-tracks (for the purposes of identifying aquifer depressurization influence) was limited to FWT1 and FWT4 on account of their relative distances from the mine, similar shapes (albeit FWT1 is larger), equivalent pond convergences and comparable thicknesses of underlying marine sediment (Figure 3-1 [c]).

The upper reaches of the NNGC watershed predominantly comprises a network of channel fens and fen-water-tracks which converge just upstream of the NNGC-UP stream gauging station (Figure 3-1 [d]); the stream then flows through a narrow channel fen confined between two large domed bogs (the northern one harbours the four internal fen-water-tracks). NNGC is minimally (if at all) incised into the underlying fine-grained glacio-marine sediments.

The surface of the large domed bog is isolated from regional surface and groundwater inflows and is therefore nutrient-poor and typified by *Sphagnum* mosses (particularly *S. fuscum*), caribou lichen (*Cladonia rangiferina*) and black spruce (*Picea mariana*). *Sphagnum spp.* also dominate the nutrient poor internal fen-water-tracks, but the presence of adapted sedges (*Carex spp*) and tamarack (*Larix laricina*) indicate that their waters have a higher dissolved mineral content than the adjacent bogs.

The study site is located within the subarctic climate region of northern Ontario, and as such, winters are long and cold whereas summers are short and cool. The inland community of Lansdowne House (located ~250 km west of the study site) and the coastal community of Moosonee (located ~250 km south-east of the study site) are the two closest meteorological stations with long-term meteorological records. The two communities experience average yearly temperatures of -1.3^{0}C and -1.1°C and average yearly precipitation of 700 mm and 682 mm, respectively (of which, ~30% and ~28% falls as snow) (based on 1971-2000 climate normals) (Environment Canada, 2008); Whittington et al. (2012) showed that these values are also representative of the mine site.

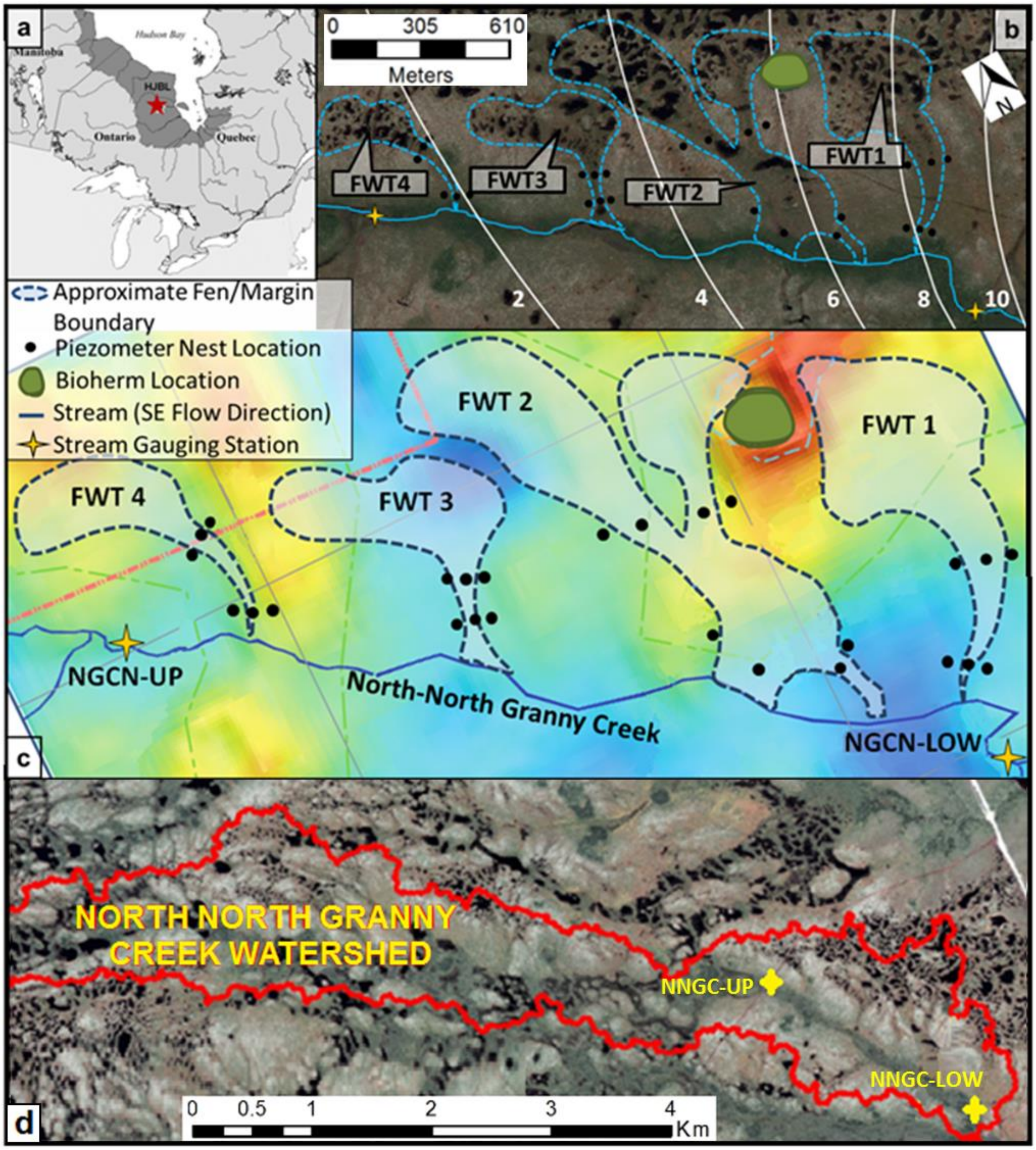

Figure 3-1: Site Map displaying a) the location of the Study Site in Ontario (red star); b) IKONOS imagery of the study site, overlain with: internal fen-water-track location and approximate fen boundary (blue dashed lines), piezometer nest locations (black dots), location of stream gauging stations (yellow stars) along the North-North Granny Creek (solid blue line), bioherm location (green circle) and upper bedrock aquifer depressurization drawdown contour intervals (white lines), in meters (2012-April); c) the study site superimposed on the marine sediment thickness map. The yellow to dark red gradient represents thin (<10 m) marine sediments to exposed bedrock; light blue to dark blue gradient represents increasing thickness of marine sediments (maximum ~50 m); and, (d) IKONOS imagery, total drainage area of the North-North Granny Creek watershed (red outline) and location of stream gauging stations along the NNGC (yellow stars).

3.3 Methods

3.3.1 Hydrological Instrumentation and Monitoring

Two transects were placed across all four internal fen-water-tracks; one transect was situated near the outlet (south transect) and another up gradient (north transect). North transects were situated approximately half way down the internal fen-water-tracks where flow-paths were more organized (as compared to the upper reaches with widespread ponded areas). Each transect included one observation point within the centre of the internal fen-water-track (e.g. FEN-N) and two were placed on either side, within the bogs (e.g. BOG-NW and BOG-NE). FWT2, the largest of the three, had four observation points along each transect: four along the north to capture both branches of the internal fen-water-track, as well as four along the south where the flow path splits around a palsa (discontinuous permafrost) located within the riparian area of the stream channel. Fen nests were located along the edge of ponds and in front of visible flow pathways (ridges between ponds were avoided as groundwater mounds within ridges of water tracks have been shown to impede flow (Price and Maloney, 1994)). Attempts were made to locate bog nests within exclusively bog areas (seen as white on the IKONOS imagery due to the abundance of lichen) and not within the margin areas and within the watershed boundaries. Watershed delineation was done with the use of Digital Elevation Model (DEM) data derived from Light Detection and Ranging (LiDAR, 5 m resolution), courtesy of Bouffard and Richardson (personal communication).

Each observation point was instrumented with one fully slotted monitoring well (~ 1 m in length) and a nest of piezometers with 25 cm perforated intakes; nests consisted of three piezometers with slotted intakes stepping down in depth from 0 to 75 centimeters below ground surface (cmbgs) (i.e. 0-25; 25-50; and 50-75 cmbgs). Within each nest, wells and piezometers were located in a straight line, perpendicular to the horizontal water flow direction (i.e. fens nests were installed in line, east-west; bogs were generally north-south) and within ~50 cm of one another. Monitoring wells were also included at each of the stream gauging stations (NNGC-UP and NNGC-LOW).

Wells and piezometers were constructed from 2.54 cm inside diameter PVC pipes closed at the bottom. For the piezometers, 250 μm nylon mesh was wrapped around the slotted intakes. All peat wells and piezometers had sufficient total length to reach ~1 m in depth, for stability from the firmer catotelm peat. All pipes were purged several times to flush water and sediments that had mixed during installation. Pipe top elevations were surveyed using a dual frequency survey-grade GPS in real time kinematic survey mode (Topcon GMS-2) and set to a vertical and horizontal accuracy of 0.003 m and 0.005 m, respectively.

Stream discharge, water tables and hydraulic head measurements were taken manually at all stream gauging, well and piezometer locations on a weekly basis and/or after any major rainfall event. Stream water levels are hereafter referred to as “stage” (masl); and, peat-water measurements are hereafter referred to as “water table” (relative to a datum [masl]) and “water level” (below ground surface [mbgs]), even when it was above the surface (i.e. ponded). Wells located at the southern fen nests of FWT1 and FWT4 were instrumented with Schlumberger Diver water pressure transducers set to record at one-hour intervals during the fall. Wells located at the stream gauging stations were also instrumented with pressure transducers throughout the study period. These recordings were later corrected for barometric pressure from a barometric pressure transducer located at the mine site, as well as from manual measurements taken throughout the study period. Manual discharge measurements were obtained at each of the gauging stations at the time of hydraulic head surveys using a YSI Flowtracker Acoustic Doppler Velocimeter. Stage-discharge rating curves were generated to convert hourly stage measurements to continuous (hourly) discharge measurements based on the corrected pressure transducer stage measurements. Spring measurements were omitted on account of ice damming effects on stage-discharge curves (Sui et al., 2007), as well as the inability to properly capture overbank flow on account of equipment limitations on measuring turbulent flow through vegetation (sedges). Converted discharge measurements approximately > 0.17 $m^3 s^{-1}$ made for May (when overbank flow was present) were therefore uncertain and are only displayed for qualitative purposes.

Hydraulic conductivity (*K*) was determined with bail tests (*K*-test) (Hvorslev, 1951). Hydraulic conductivity was measured at least once for all piezometers with perpetually saturated slotted intakes; for piezometers found to be straddling the water table, *K* was measured within two days of every sampling event (*L* was determined based on the water level within the slotted area). In cases where the recovery was too fast for manual measurements, recovery was monitored with a pressure transducer set to record at 0.5 second intervals (Fetter, 1994). Results from multiple *K*-tests were averaged for use in this study. The transmissivity of the upper 75 cm of peat was determined based on changes in the water table,

$$\boldsymbol{T = bK} \qquad \textbf{[3-1]}$$

where, *T* is transmissivity (L^2/t), *b* is saturated thickness of the screens (*L*), and *K* is saturated horizontal hydraulic conductivity (*L*/*t*). For a multilayer aquifer (i.e. 0-25, 25-50 and 50-75 cm), the total transmissivity was determined as the sum of the transmissivity of each layer,

$$\boldsymbol{T_{Total} = \sum_{i=1}^{n} T_i} \qquad \textbf{[3-2]}$$

We were unable to measure flow through the rivulets on account of equipment limitations on measuring turbulent flow through vegetation (sedges) and flumes were not constructed at the time of this study; therefore, surface flows are unavailable for this study. A flume was constructed following this study (in 2014); the applicability of the results will be addressed in Section 3.5.2.

Discharge from the upper 75 cm of the fens was calculated to determine relative internal fen-water-track contributions to the stream channel using Darcy's Law:

$$\boldsymbol{Q = T_{Total} \frac{dh}{dl} W} \qquad \textbf{[3-3]}$$

where *Q* is discharge ($m^3 s^{-1}$), T_{Total} is transmissivity of the upper 75 cm of the peat profile ($m^2 s^{-1}$), *dh/dl* is the hydraulic gradient (unitless), and *W* is the width of the saturated flow (m). Hydraulic gradients were determined by head difference (*dh*) and distance between (*dl*) fen wells and the stream channel. Width of contributing area was determined by saturated thickness (m) and width of flow (m). Width of flow was determined by the width of the internal fen-water-track at the southern transect.

Bulk incident rainfall was collected at the base of FWT2 in a standard rain gauge with a 12.7 cm diameter collecting orifice; the collected rain was measured after the cessation of every rain event. These data were used to data check meteorological station data (which measured rainfall via tipping bucket set to record every 10 minutes). Field data agreed well with MET data, and since the tipping bucket was of better quality and within three kilometers of all internal fen-water-tracks, apportioning of the on-site data was considered unnecessary.

3.3.2 Water Sampling and Chemical Analyses

Three campaign style sampling events, whereby all piezometers and stream locations were sampled on the same day, were performed and intended to capture high water tables following the spring freshet (and summer antecedent conditions) (Spring; 10-June-2012), low seasonal water tables (and fall antecedent conditions) (Summer; 09-August-2012) and a final measurement near the end of the fall wet-up period and prior to freeze (Fall; 20-October-2012). Each sampling campaign was an involved multi-day process: Day 1- heads measurements were obtained and then the piezometers were purged to allow a day for slow recovering piezometers (details on the purging process are below); Day 2- stream discharge was measured and piezometers and stream channels were sampled; and, Day 3- samples were filtered (mine site laboratory) and *K*-tests were performed. To capture the "flushing" of DOC throughout the fall wet-up period, an additional 7 sampling events were performed. An attempt was made to obtain these samples directly after (or during) all rainfall events. So, heads measurements, piezometer purging, water sampling, *K*-testing, stream gauging and stream water sampling were all completed over the course of one day (water filtering was performed the following day). Since such large sampling events were not possible to perform within a single day, fall wet-up sampling events (from 1-September-2012 to 20-October-2012) were limited to the south nests of FWT1 and FWT4. These nests were chosen on account of their similarities discussed above, and their relative distances from the mine (near and far).

Prior to porewater sampling, piezometers were purged dry (when possible) via teflon tubing affixed to a low-flow peristaltic pump and then allowed to recover overnight. Piezometer tops were loosely covered after purging and between all monitoring events to prevent contamination via precipitation and/or airborne particulate matter. Peristaltic pumps were also used for sample collection. Teflon tubing was washed with a 10% HCl solution and thoroughly rinsed with de-ionized water between each purging and sampling event. Teflon tubing was also rinsed and flushed thoroughly with de-ionized water between each piezometer purge, as well as between each sample extraction. Disposable nitrile gloves were worn and changed as necessary to purge piezometers, collect all samples, filter all samples and handle sample bottles while preparing for shipping to the laboratory.

Dedicated sample collection bottles (125 mL high-density polyethylene water bottles) were thoroughly washed and rinsed with 10% HCl solution and de-ionized water between each sampling event, and then "environmentalized" three times with piezometer peat-water (or stream-water) prior to sample collection. Stream water samples were collected with standard grab sampling methods by submerging the sample bottles and pointing them upstream, with care being taken to avoid large organic matter from being captured in the water samples. In-field chemical parameters (temperature, pH and electrical conductivity) were measured with an Oakton PCSTestr 35 waterproof pH/conductivity/TDS/salinity handheld tester (calibrated prior to every sampling event). Samples were stored in coolers with ice packs during sampling events and again when couriered to the laboratory, under Chain of Custody procedures. As part of the quality assurance/quality control (QA/QC), multiple sample duplicates, field blanks, Teflon tubing line blanks and filter blanks were collected during each sampling/filtering event.

On three occasions, rain was collected for ionic and DOC analyses via rain sampler consisting of a large Teflon funnel attached to a 500 mL high-density polyethylene water bottle (both washed and rinsed thoroughly with 10% HCl solution and de-ionized water prior to each sampling event). Collection was performed at the beginning of the rainfall events when the concentrations of solutes are considered to be highest due to washout of particulates and aerosols from the lower atmosphere (Orlova and Branfireun, 2014). Rainwater was filtered in-field through nylon syringe micro-filter (0.45 μm) directly into laboratory-supplied vials. Pore-water and stream-water samples were filtered in the DeBeers environment laboratory located on-site through 0.45 μm filters, and then decanted into laboratory-supplied sample bottles for storage prior to analysis of DOC and major ions. Samples were frozen for preservation of DOC and ions, prior to being transported to the University of Western Ontario for analysis. Samples were analyzed for DOC and ions by the Biotron Institute for Experimental Climate Change. Calcium and chloride were analyzed using Dionex ICS-3000 and Dionex ICS-1600 ion chromatography systems, respectively, with detection limits of 0.01 mg L^{-1}. DOC was analyzed using an OI Analytical 76 Aurora 1030W TOC analyzer using heated persulfate oxidation, with a detection limit of 0.2 mg L^{-1}. All filter, field, and line blanks contained unquantifiable concentrations of ions and DOC, and sample duplicates were within acceptable range of one another (<20% relative difference between samples) (Canadian Council of Ministers of the Environment, 2011).

3.4 Results

3.4.1 Meteorological Conditions throughout the Study Period

Local seasonal precipitation recorded throughout the study period was generally within the range of values of Lansdowne House and Moosonee, and local average monthly temperatures recorded on-site were within 3°C of the 2 meteorological stations (Table 3-1) (Environment Canada, 2008); the use of long term (30-year climate normal; 1971-2000) climate data from these two stations are therefore reasonable for on-site comparisons (the averages of these stations are used hereafter). Average temperatures recorded on-site during spring (May and June), summer (July and August) and fall (September and October) compared well with normal temperatures (Table 3-1). During March 2012 however, a period of abnormally high temperatures resulted in an early snowmelt, leaving no snowpack by the beginning of this study. The study site received slightly lower than normal precipitation during the spring; however, the majority occurred during a major rainfall event in mid-May (+197% of normal May precipitation). The importance of this will be discussed later. Precipitation was less than normal during the summer and slightly higher than normal during the fall.

3.4.2 Peatland Hydrological Responses throughout the Study Period

All observation points experienced a decline in water levels throughout the spring and into summer, with all points reaching their minimum measured depths by 15-August (daily averages of bogs and fens displayed in Figure 3-2). Subsequently, after 71 mm of rain fell on-site within 3 days (5 to 7-September), water levels measured at nearly all observation points on 8-September (with the exception of FWT2-BOG-SW, FWT2-FEN-NW, -NE and -SW nests) had recovered to within 3 cm of (and several exceeding) their respective spring measurements. Moreover, despite FWT2-BOG-NE reaching the lowest observed water level of all observation points, water levels at this nest had recovered to nearly 8 cm above its respective springtime measurement by 8-September, resulting in ponded water. After an additional 48 mm of rain fell on-site (5 to 10-October), all other FWT2 nests that had not recovered by 8-September had done so by 11-October (to within 6 cm of, or exceeding springtime measurements). The use of the term *"fall wet-up period"* is therefore used hereafter, to describe this (i.e., from 5-September to 10-October).

Table 3-1: Meteorological variables for the study period (1-May to 31-October-2012) obtained from DeBeers, Lansdowne House (LH) and Moosonee (M) weather stations. The climate normal values are an average of Lansdowne House and Moosonee based on the 30 year (1971-2000) Canadian Climate Normals from Environment Canada. Snow depth is by the end of March. An "*" indicates maximum, rather than average temperature.

		March	Spring	Summer	Fall
Average Air Temperature (°C)	DeBeers	14.2*	9.9	15.1	5.4
	LH (2012)	22.4*	11.5	16.6	6.0
	M (2012)	24.3*	11.7	16.2	8.2
	Normal	-4.4*	9.7	15.7	6.0
Precipitation (mm)	DeBeers	-	137	93	146
	LH (2012)	-	231	109	112
	M (2012)	-	112	125	146
	Normal	-	120	194	135
Snow Depth (cm)	DeBeers	0	-	-	-
	LH (2012)	17	-	-	-
	M (2012)	0	-	-	-
	Normal	46	-	-	-

3.4.2.1 Bogs vs Fens

Mean bog water levels declined and recovered at a faster rate than the fens, based on daily mean water levels of all nests, resulting in a greater water level decline within the bogs, as compared to the fens (Figure 3-2). Bog water levels also recovered to initial spring values earlier than the fens during the *fall wet-up period*. Water level variability around these mean values (standard deviation) was greatest within the bogs, particularly during the summer. For example, on 15-August-2012, the standard deviations for mean water levels were ±15.6 and ±7.3 cmbgs within the bogs and fens, respectively.

Bog water levels (relative to ground surface) and bog water table elevation (relative to a datum) were very similar (Figure 3-2). Essentially, for every drop (or recovery) in water level, there was a corresponding drop (or recover) in water table elevation, reflecting a rigid matrix within the bogs. Fluctuations in fen water table elevation did not correspond to fluctuations in water table depth throughout the study period. Fen water table elevation declined over 3 cm more than water levels did relative to the ground surface, reflecting greater subsidence within the fens.

Vertical hydraulic gradients were calculated based on water table and piezometric head differences (*dh*) of the deepest piezometers (50-75 cmbgs), and distance between water table and centre of the slotted intake (*dl*). Average vertical hydraulic gradients were downwards (negative) within the bogs and fens; however, gradients were generally stronger within the bogs (Figure 3-3 [a]). Average downward hydraulic gradients generally decreased from spring to summer within the fens (with the exception of a temporary increase after the early August rainfall event) and recovered during the fall. Average downward hydraulic gradients were relatively consistent within the bogs throughout the study period, with the exception of a positive flow reversal on 15-August.

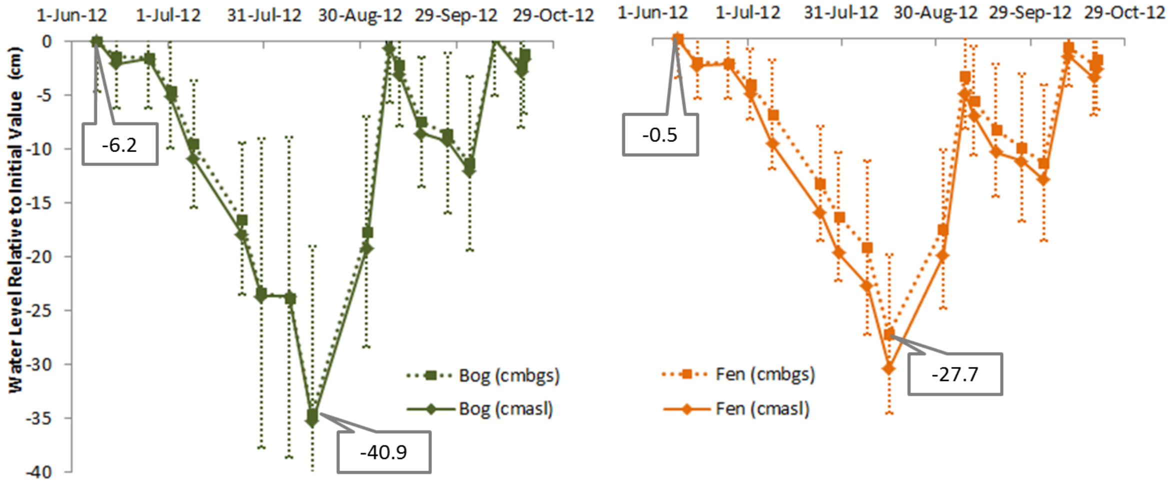

Figure 3-2: Average bog (green) and fen (orange) daily water table depth (cmbgs) and elevation (cmasl) fluctuations relative to initial values. Initial average values relative to ground surface were -6.2 and -0.5 cmbgs for the bogs and fens, respectively. Average depths measured on 15-August were -40.9 and -27.7 cmbgs for the bogs and fens, respectively.

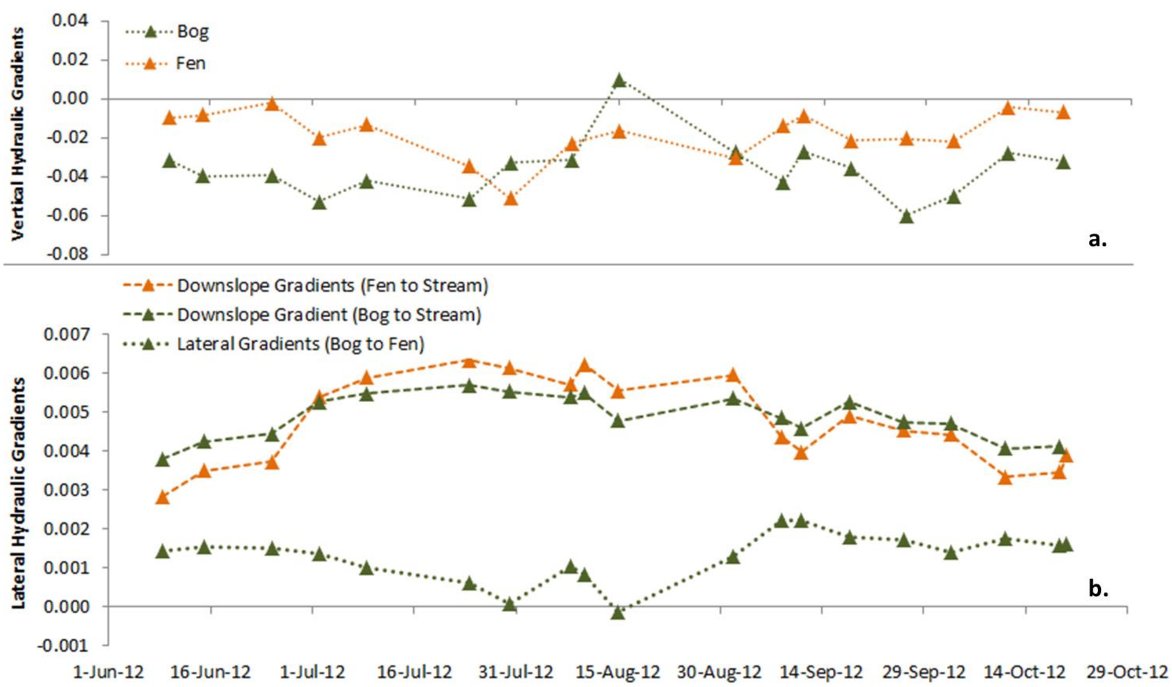

Figure 3-3: Average vertical hydraulic gradients (a) of the bogs (green) and fens (orange). Calculations were based on water table and piezeometric head differences (*dh*) from the deepest piezometers (50-75 cm), and distance between water table and centre of slotted intakes (*dl*). And, average lateral hydraulic gradients (b) between the bogs and fens (green dotted line) and downslope toward the stream channel within the bogs (green dashed line) and fens (orange dashed line).

Average lateral hydraulic gradients were positive (flow from bogs to fens) during the spring, declined over the summer (with a temporary increase after the early August rainfall event) and resulted in a minor flow reversal (from fen to bog) by 15-August (Figure 3-3 [b]). Upon the onset of the *fall wet-up period*, lateral hydraulic gradients increased to values higher than initial spring measurements and subsequently declined slightly through the fall to near spring measurements. Average lateral hydraulic gradients (downslope toward the stream) increased through the spring and into summer and subsequently declined through the fall, particularly within the fens.

Transmissivity (T) within the upper 75 cm of the bogs and fens decreased logarithmically by 3 to 4 orders of magnitude with depth (Figure 3-4 [a., inset]). Fens had higher transmissivity than the bogs at the same water level and decreased slightly less quickly with water level decline, based on the slope of the logarithmic relationships. Therefore, bogs witnessed a greater decline in transmissivity during the summer and a quicker recovery when wetter conditions returned resulting in transmissivity values most comparable between the bogs and fens upon the onset of the *fall-wet-up period* (Figure 3-4 [b]).

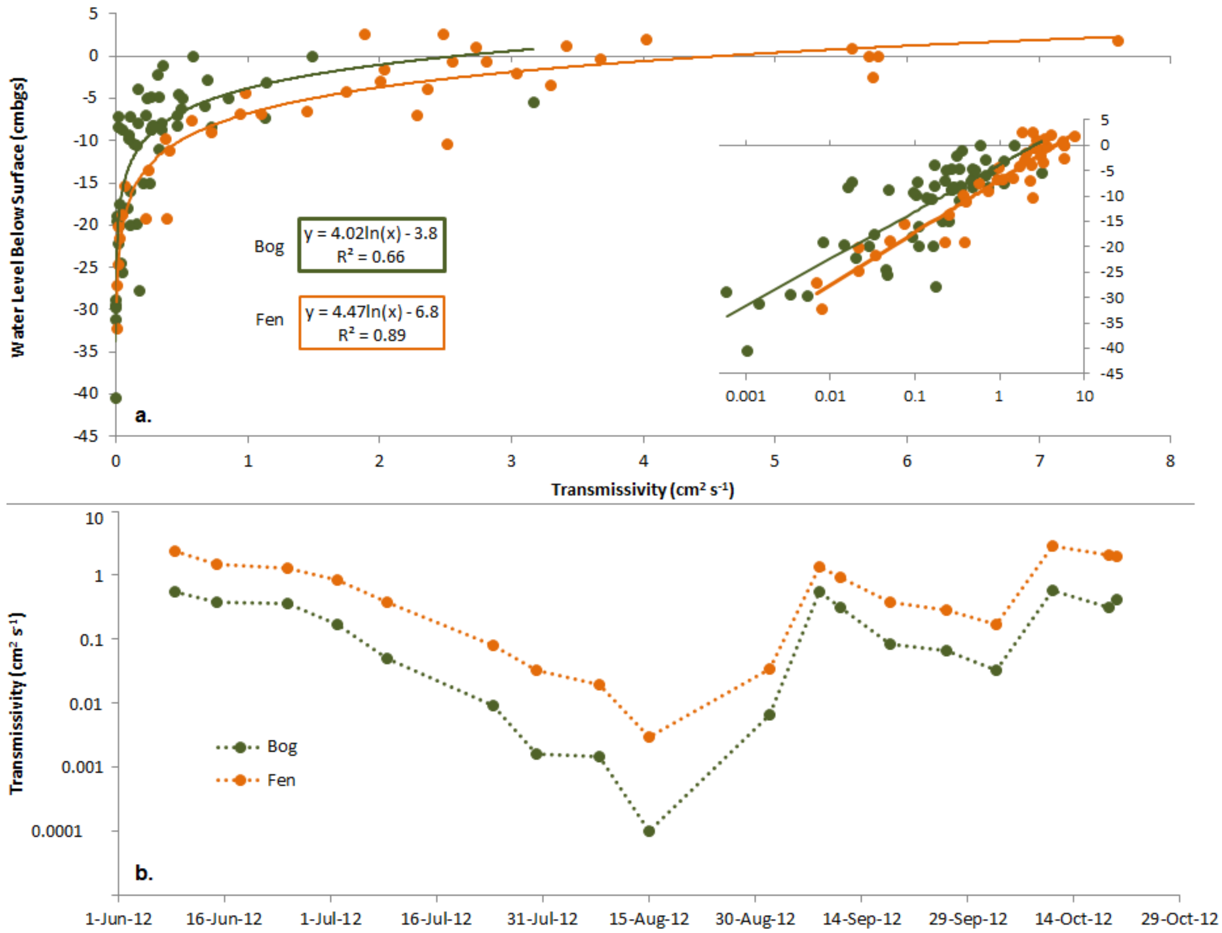

Figure 3-4: Transmissivity of the upper 75 cm of all bog (green) and fen (orange) piezometer nests, under different water-levels (a); inset displays transmissivity on a logarithmic scale with same axes as top figure. Water flow above the ground surface was not measured. Time series of average bog (green) and fen (orange) transmissivity (determined by substituting average bog and fen daily water levels into the equation of the logarithmic relationships) (b).

3.4.2.2 FWT1 (more impacted) vs FWT4 (less impacted)

As noted in Section 3.2, FWT1 was located with closer proximity to the mining pit (representing greater disturbance) and FWT4 was located with greater distance from the mining pit (representing lesser disturbance). As such, the hydrological response of the southern fen nests (located near the outlet to the stream) of both internal fen-water-tracks are reported to determine possible impacts of aquifer depressurization. The range in water level within the southern fen nest of FWT1 was slightly greater than in FWT4, but followed a similar pattern (Figure 3-5). In both sites, water table elevation (cmasl) declined more than water table depth (cmbgs) during the summer, reflecting the amount of subsidence at each site. At the deepest water table position (15-August), the difference between water table elevation and water table depth was about 2.6 cm and 4.8 cm in FWT 1 and FWT 4, respectively, reflecting the greater subsidence at the latter (less impacted) location. Vertical hydraulic gradients were comparable between the southern fen nests of FWT1 and FWT4 during the spring and fall; however, declined much further within FWT1 by 24 and 30-July (Figure 3-6).

Transmissivity (T) within the upper 75 cm of the southern fen nests of FWT1 and FWT4 decreased logarithmically (Figure 3-7) by 1 to 2 orders of magnitude with depth (FWT1: 0.6 to 0.002 $cm^2\ s^{-1}$; FWT4: 0.3 to 0.02 $cm^2\ s^{-1}$). Transmissivity declined at a faster rate within FWT1, based on the logarithmic relationships (FWT1: y = 3.68ln[T] + 2.16; FWT4: y = 6.57ln[T] + 8.73) (R^2 values were 0.95 and 0.93, respectively). Substituting water level measurements into these logarithmic relationships, it is apparent that transmissivity was greater within the southern fen nest of FWT1 during the spring, but declined much lower than FWT4 during the summer (Figure 3-7).

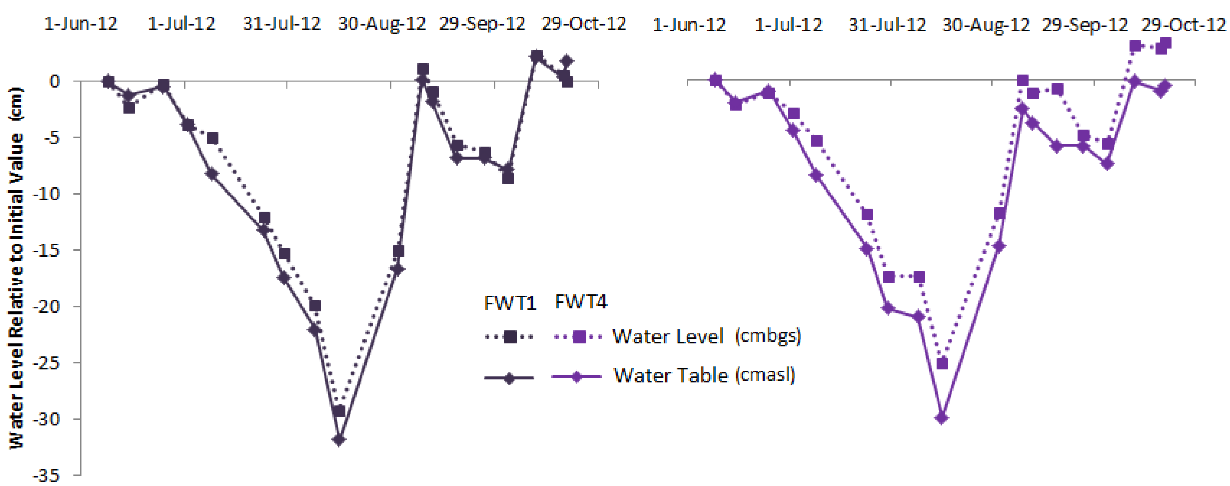

Figure 3-5: Time series of daily water table depth (cmbgs) and elevation (cmasl) fluctuations relative to initial values within the southern fen nests of the relatively more impacted FWT1 (left) and the relatively less impacted FWT4 (right). Initial values relative to ground surface were -0.4 and -2 cmbgs for FWT1 and FWT4, respectively. Depths measured on 15-August were -29.6 and -28.9 cmbgs for FWT1 and FWT4, respectively.

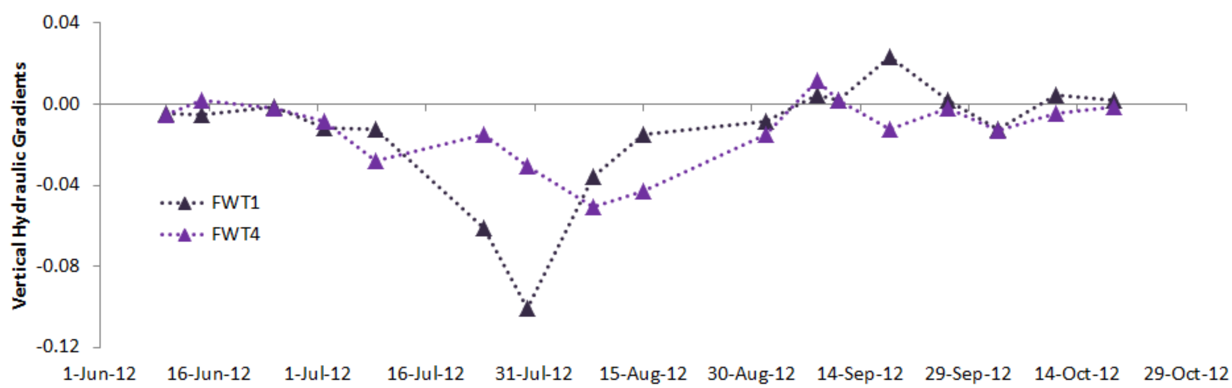

Figure 3-6: Time series of vertical hydraulic gradients for the southern fen nests of the relatively more impacted FWT1 (dark purple) and the relatively less impacted FWT4 (light purple), based on piezeometric head (50-75 cm piezometers) and water table differences. Negative values represent downward gradients.

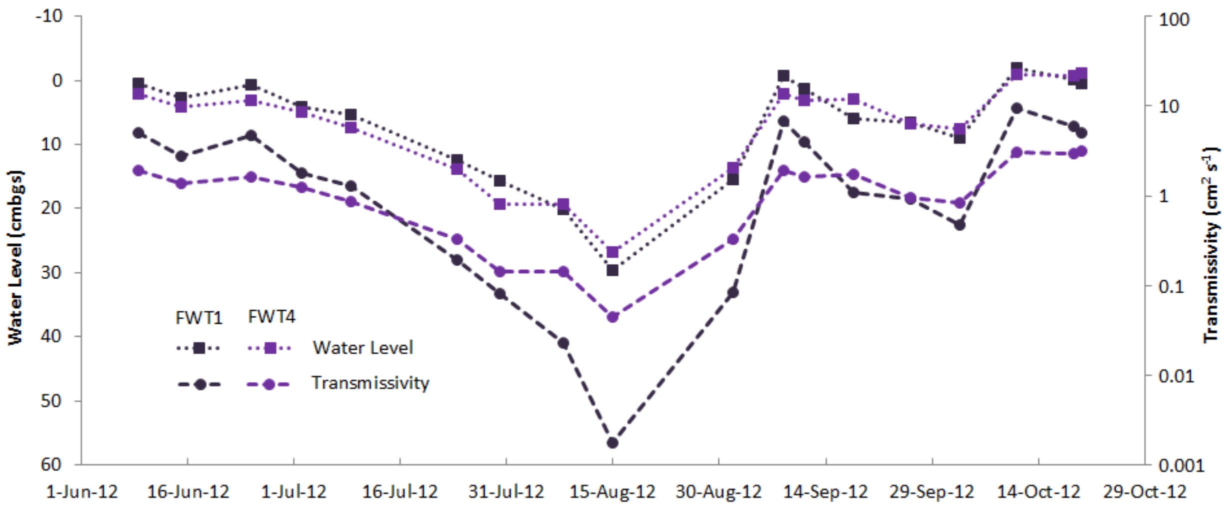

Figure 3-7: Time series of daily fen water level (cmbgs) fluctuations (top) and transmissivity (bottom) within the southern fen nests of the relatively more impacted FWT1 (dark purple) and the relatively less impacted FWT4 (light purple). Transmissivity was determined by substituting daily fen water levels into the equation of the logarithmic relationships of water level *versus* transmissivity for each of the southern fen nests of FWT1 and FWT4.

3.4.3 Peatland Geochemistry and DOC Concentrations

Box-and-whisker plots were derived with the use of R software for computational statistics; the boxes represent the interquartile range (25-75% of the observed data) with the median resting at 50%, the whiskers mark the upper 95% and lower 5% of the observed data (outliers are displayed as open circles) and the notches represent the 95% confidence interval around the median. Accordingly, if the notches between two box plots do not overlap, there is strong evidence (at $p = 0.05$) that their medians differ (R Core Team, 2014).

3.4.3.1 Bogs vs Fens

All bog and fen peat-waters were generally acidic (on average pH4.8±0.6 and pH5.4±0.5, respectively) but the fens had significantly higher pH levels than the bogs at all equivalent depths (Figure 3-8 [a]). pH increased significantly between each depth interval within the bogs and below 50 cm within the fens. All bog and fen nests had similarly low levels of chloride [Cl^-] (<1 mg L^{-1}) (with the exception of a few nests with Cl^- concentrations between 1 to 2 mg L^{-1}), with no noticeable pattern with depth (Figure 3-8 [b]). Calcium [Ca^{2+}] concentrations were more variable than Cl^- concentrations, ranging from 1.3 to 29.1 mg L^{-1} (Figure 3-8 [c]). Ca^{2+} concentrations increased significantly with depth in the bogs and were slightly higher at equivalent depth within the fens. Ca^{2+} concentrations only increased significantly below 50 cm within the fens, relative to the shallower depths.

DOC concentrations responded differently within the bog and fen nests with depth and through time (Figure 3-9). Within the bogs, DOC concentration increased significantly between spring and summer within the upper 50 cm of the peat profile, but not between summer and fall. DOC concentration increased with depth in the bogs (within season); however, this increase weakened with time. DOC concentrations increased between spring and summer within the entire profile of the fens, but this increase was not significant. DOC levels increased significantly between spring and fall within the entire profile of the fens. There were no differences in DOC concentration with depth in the fens, no matter the season (Figure 3-9 [b]).

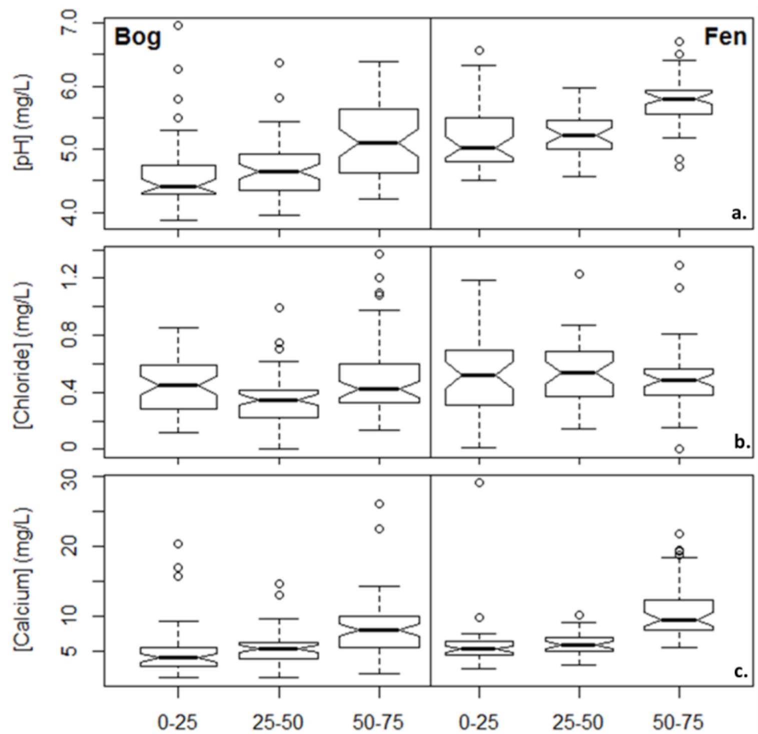

Figure 3-8: Box plots of pH (a), chloride (b) and calcium (c) concentrations of all bog and fen at the various depth intervals. All bog plots have n values between 52 and 61; all fen plots have n values between 40 and 44. Where notches do not overlap (e.g. Bog pH; 0-25, 25-50 and 50-75), the median of the data are significantly different, with 95% confidence (i.e. at $p = 0.05$).

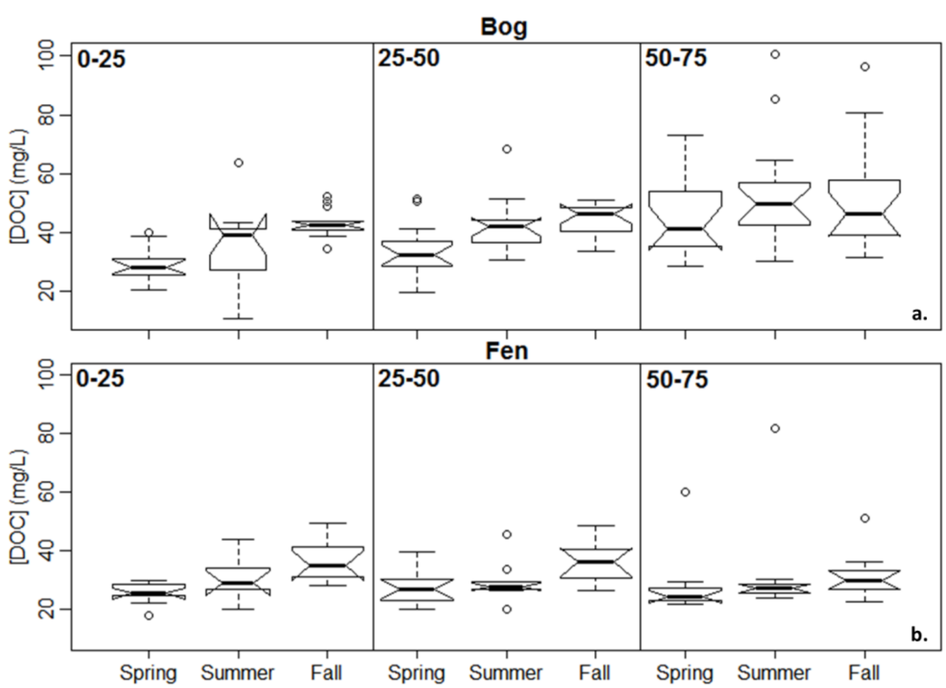

Figure 3-9: Box plots of DOC concentrations within the various depths of all bogs (a) and fens (b) for all sample periods. Bog and fen plots derived from samples obtained from the 0-25 cmbgs zone during the summer have n values of 11 and 8, respectively; all other bog and fen plots have n values of 17 and 10, respectively. Where notches do not overlap (e.g. Bog 25-50 Spring *vs* Summer), the medians of the data are significantly different, with 95% confidence (R Development Core Team, 2009).

3.4.3.2 *FWT1-FEN-S (more impacted) vs FWT4-FEN-S (less impacted)*

The southern fen nests of FWT1 (more impacted) and FWT4 (less impacted) were monitored for DOC concentrations with greater frequency than all other nests during the *fall wet-up period* and the results are displayed in Figure 3-10. DOC concentrations were similar between the southern fen nests of FWT1 and FWT4 during the spring sampling event, albeit FWT4 was slightly more variable with depth. Similar to average fen measurements, DOC concentrations within the less impacted internal fen-water-track (FWT4) did not increase between the spring and summer sampling events; whereas, more similar to average bogs, DOC concentrations within the more impacted internal fen-water-track (FWT1) increased above 25 cm between the spring and summer sampling events. DOC concentrations increased upon the onset of the *fall wet-up-period* within the upper 50 cm of both internal fen-water-tracks. And, similar to average fen measurements, DOC concentrations obtained during the major fall sampling event were greater than summer values; however, concentrations were still more variable with depth in the more impacted internal fen-water-track (FWT1) as compared to the less impacted internal fen-water-track (FWT4) which had similar concentrations with depth, similar to average fen measurements (Figure 3-10 and Figure 3-9, respectively).

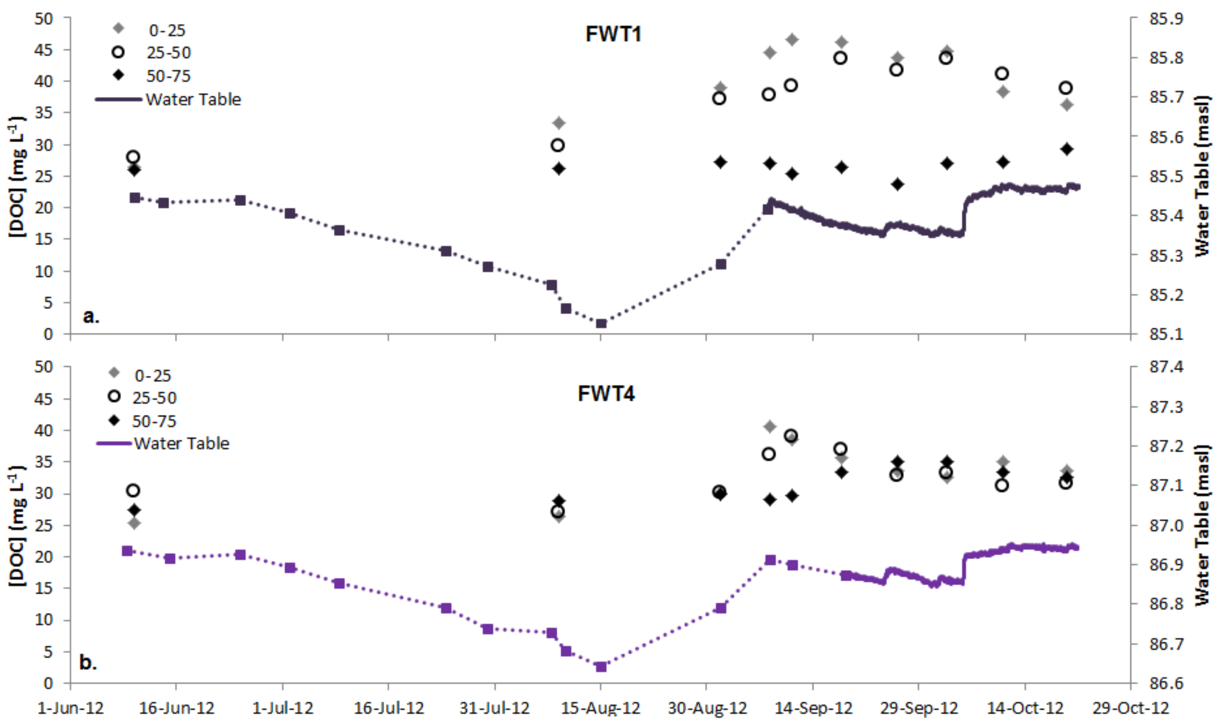

Figure 3-10: Time series of water table elevations (purple) and DOC concentrations (black and grey) at the southern fen nests of FWT1 (a) and FWT4 (b) (relatively high *versus* relatively low impact, respectively).

3.4.4 *Streamflow Measurements throughout the Study Period*

As noted in Section 3.3.1, discharge values above ~0.17 $m^3 s^{-1}$ are merely included to display general stream response and are not to be considered accurate, due to the presence of overbank flow (not captured in the stage/discharge rating curve). Also, aside from 2 spot measurements (24-April-2012 and 10-June-2012), continuous NNGC-UP stage measurements did not commence until 28-June-2012; therefore, UP and LOW comparisons are only available for 24-April-2012, 10-June-2012 and from 28-June-2012 through to the end of the study period.

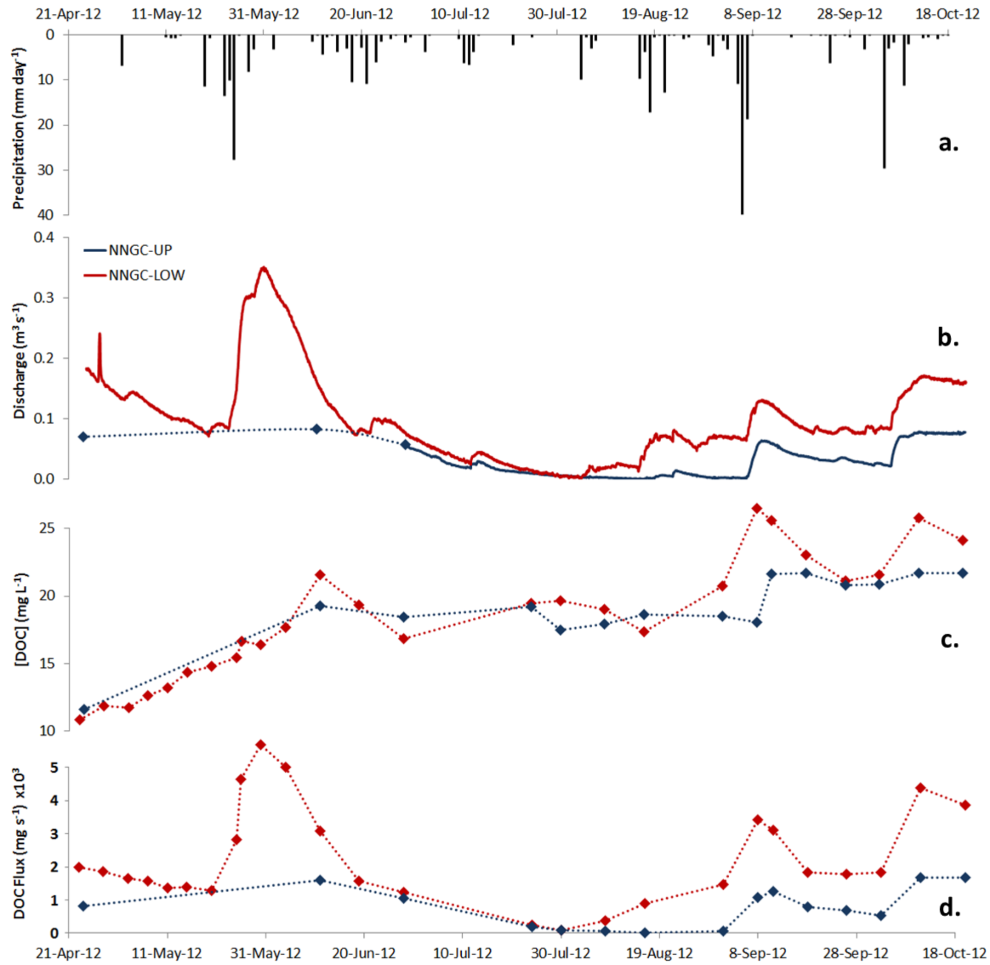

Figure 3-11: Time series graph of precipitation (a), stream discharge (b), DOC concentration (c) and DOC flux (d).

Stream discharge (NNGC-LOW) was in decline through April and into the beginning of May (Figure 3-11 [b]). With the exception of the response to the major rainfall event of mid-May (Section 3.4.1), discharge continued to decline through the spring and into August, with responses limited to rainfall events exceeding 5 mm. After the mid-May rainfall event, stream discharge declined at a faster rate at the LOW station until both stations were showing similar stream discharge values by late-July. By the beginning of August however, a series of small rainfall events (exceeding 5 mm) resulted in an increase in stream discharge at the LOW station only. The UP station did not witness any quantifiable change in discharge until after the 18-August rain event, which exceeded 17 mm. After this event, small (<5 mm) rain events were able to sustain water levels at the LOW station while the UP station declined to near pre-event values. Between 5-September and 7-September, nearly 71 mm of rain fell on-site, and within one day of the commencement of this event, both stations experienced a response in discharge. Overall, stage continued to climb through September and into October. During this time, stream discharge responded relatively evenly across both stations (although the LOW station had higher values and showed flashier responses to events less than 5 mm). Ultimately, by the end of the study period, discharge at the UP station had increased to levels comparable to those measured at the end of June; while discharge at the LOW station had surpassed late-June discharge measurements by nearly 0.1 $m^3 s^{-1}$. The *"fall wet-up period"* seen in water table (e.g. Figure 3-2) is similarly reflected in increased streamflow (Figure 3-11 [b]).

3.4.5 Stream DOC Concentrations and Fluxes

Stream DOC concentrations increased at both the UP and LOW stations throughout the study period, with the lowest concentrations obtained in the spring (24-April), and the highest obtained upon the onset of the *fall wet-up period* (1-September and 11-September at the LOW and UP stations, respectively) (Figure 3-11 [c]). The rate of increase was not consistent throughout the study period; DOC concentration increased most rapidly (and consistently) during the spring, remained reasonably steady throughout the summer, and increased through the fall. DOC concentrations were most variable during the fall, particularly at the LOW station, and corresponded to peaks in stream discharge. Springtime DOC concentrations on the other hand, were relatively unresponsive to changes in stream discharge. The LOW station experienced more variable DOC concentrations throughout the study period with the greatest variations (and deviation from the UP station) occurring during the *fall wet-up period*.

The flux of DOC within the stream channel generally mimicked stream discharge at both the UP and LOW stations (Figure 3-11 [d]). With the exception of a large pulse of DOC during the major rain event of mid-May, stream DOC flux was in decline throughout the spring and into the summer. The flux of DOC began to increase at the LOW station by the beginning of August, as stream discharge increased. The flux of DOC remained low at the UP station until the start of the *fall wet-up period*. Similar to stream discharge and DOC concentrations, the flux of DOC within the stream channel generally increased throughout the *fall wet-up period*. The use of the term *"fall flush"* is also therefore reasonable.

3.5 Discussion

The study period presented here began in June 2012; over 5 years after pit dewatering commenced (January 2007). The cone of influence of aquifer depressurization was therefore extending below all FWTs at the start of this study period, increasingly so nearer to the mine. Dewatering impact may therefore be reflected in the hydrology and geochemistry of the system, as presented below. Since information on pre-disturbance conditions is unavailable, site and treatments are confounded (Davies and Gray, 2015) and pseudoreplication should be considered (Hurlbert, 1984). Care is therefore needed to avoid over-interpretation of the results concerning hydrological and geochemical pathways of non-impacted internal fen-water-tracks within the HJBL. To address this, inference of the results are not only made with the use of a large data set (4 FWTs), but with inference to impact itself and reference to other literature within the region.

Average vertical hydraulic gradients were downward within the bogs and fens. Downward hydraulic gradients are typical within undisturbed bog systems (Siegel and Glaser, 2006), but not necessarily within fens, which are generally known as groundwater discharge zones (Siegel and Glaser, 1987). Leclair et al. (submitted) found vertical hydraulic gradients within the bogs and fens of FWT1 increased between 2010 and 2011 from -0.07 to -0.09 (bog) and -0.01 to -0.04 (fen); an order of magnitude greater than those measured at a non-impacted reference site (from 0.003 to 0.000 [bog] and -0.009 to -0.002 [fen] in 2010 and 2011, respectively). During the spring and summer of 2012, vertical hydraulic gradients were downward and of similar magnitude as gradients measured by Leclair et al. (submitted) (Figure 3-3 [a] and Figure 3-6) (average bogs: -0.03 and average fens: -0.02), reflecting the influence of deep aquifer depressurization.

As a result of increased vertical hydraulic gradients, Leclair et al. (submitted) found that mid-summer water levels at FWT1 (bogs: 44 cmbgs and fens: 24 cmbgs) had dropped 20 and 17 cm further below the ground surface than non-impacted bogs and fens, respectively. During this study in the summer of 2012, water levels similar to those reported by Leclair et al. (submitted) were found within the bogs (-40.9 cmbgs) and fens (-27.7 cmbgs) (Figure 3-2), similarly reflecting the impact of deep aquifer depressurization.

In addition to displaying greater water table decline (-40.9 *vs* -27.7 cmbgs), the bogs displayed greater water table variability than the fens (based on standard deviation) (Figure 3-2) due in part to bog water contributions helping to regulate water levels within the fens (Figure 3-3 [b]) (Quinton and Roulet, 1998) and a more rigid bog peat matrix. Fluctuations in fen water table elevation (relative to a datum; masl) deviated from fluctuations in water table depth (below ground surface) throughout the study period (Figure 3-2 and Figure 3-5) due to peat surface oscillation (Whittington and Price, 2006). As water is removed from the compressible peat, pore spaces collapse and the surface elevation declines (Price and Schlotzhauer, 1999). Since sedge peat is more compressible than *Sphagnum* mosses (due to higher macroporosity), the fens experienced notable surface oscillation, while the bogs remained relatively rigid (Figure 3-2). This contributes to the greater apparent water level stability within the fens.

Lower water levels within the bogs compared to the fens (Figure 3-2) was in part a consequence of lower transmissivity, as hydraulic conductivity declines considerably with depth (Price et al., 2003). Transmissivity declined less quickly with water table decline within the fens (Figure 3-4 [a]), partly due to peat subsidence; although hydraulic conductivity declines with depth, peat subsidence allows for water flow to continue within the relatively permeable upper layers, thereby offsetting a decline in saturated thickness (Whittington and Price, 2006).

Despite downward hydraulic gradients within the fens, and greater water storage capacities within the bogs due to their lower seasonal water table, horizontal gradients indicate that the bogs continue to supply water to the fens during wetter periods of the growing season (spring and fall), and the fens continue to function as water conveyors (Figure 3-3 [b]).

3.5.1 Hydrologic and Geochemical Connectivity within the Landscape

3.5.1.1 Peatland Acidity and Major Ion Concentrations

Within the upper 25 cm of the bogs and fens, average pH (4.8±0.6) measurements were comparable to the lumped values (bog and fen peat pore-waters: pH 4.0-5.0) displayed by Ulanowski (2014) within an internal fen-water-track and its harbouring bog located approximately 25 km south of Victor Mine (i.e. outside the zone of mine influence). An internal fen-water-track studied within the Albany River watershed of the HJBL displayed similar ranges in near-surface pH (between 5 and 6.7) (Glaser et al., 2004b) as the fens in the NNGC watershed (between 4.5 and 6.6); however, bogs of the Albany River watershed were slightly more acidic (Albany River: <4.2; NNGC $Bog_{Average}$ 4.6±0.6). Bog and fen chloride levels were dilute within all 3 studies; <2 mg L^{-1} within the Attawapiskat River watershed [NNGC and Ulanowski (2014)] and <4 mg L^{-1} in the Albany River watershed (Glaser et al., 2004b). Ulanowski (2014) found Ca^{2+} levels increased with depth within the bog/internal fen-water-track transect (<5 mg L^{-1} at the surface, to 100 mg L^{-1} near the mineral contact). Ca^{2+} concentrations ranged from 1.3 to 26.1 mg L^{-1} and 2.5 to 29.1 mg L^{-1} within the upper 75 cm of the NNGC bogs and fens, respectively.

Glaser et al. (2004b) found lower bog Ca^{2+} concentrations (on average, <2 mg L^{-1}) but similar Ca^{2+} concentrations within the fens (between 2.5 and 30.8 mg L^{-1}).

The significantly higher pH and calcium concentrations noted within the fens (Figure 3-8 [a. and b., respectively]), indicate the higher base-element concentration of the internal fen-water-tracks compared to their harbouring bog. The presence of Ca^{2+} within the bogs (>2 mg L^{-1}) may reflect ion exchange during decomposition or upward diffusion from the calcium-rich mineral sediments (i.e. dissolution of calcium carbonates and calcite). Price and Woo (1988) showed diffusion of ions from the marine sediments of coastal HJBL marshes was responsible for their presence in shallow peat. Further inland, where bogs have developed, diffusion would only be important if downward gradients, and thus freshwater recharge, are negligible. Indeed, this was what Leclair et al. (2015) showed at a bog located outside the zone of aquifer depressurization. In the current study, the bogs displayed downward hydraulic gradients for the majority of the study period (Figure 3-3 [a]), but the ~5 years of leaching has evidently not yet removed Ca^{2+} from the profile (Figure 3-8 [c]).

3.5.1.2 Peatland DOC Concentrations

Average bog and fen DOC concentrations (42.9±13.2 and 32.3±8.7, respectively) were comparable to average bog and fen DOC concentrations collected within the bogs and internal fen-water-tracks of the Albany River basin (42.9 and 31.3, respectively) (Glaser et al., 2004b). Lumped values within the upper 1 m of the bog and internal fen-water-track of Ulanowski (2014) (31.0±3.6) were similar to the average fen values found within this study. DOC concentrations were more variable within the bogs with depth and time (Figure 3-9 [a]). Since decomposition rates are strongly related to water table fluctuations (Belyea, 1996), the greater range in bog DOC concentrations compared to the fens (i.e. larger interquartile ranges within the bogs; Figure 3-9 [a]) is likely a result of the higher variability in water levels in the bogs (Figure 3-2).

DOC concentrations increased with depth within the bogs (Figure 3-9 [a]). DOC concentrations did not vary with depth in the fens, no matter the season (Figure 3-9 [b]). Dispersive mixing increases with higher flow velocities and longer transport distances (Reeve et al., 2001). Since the fens have higher transmissivity (Figure 3-4) and lower downward vertical hydraulic gradients (albeit affected through aquifer depressurization) (Figure 3-3 [a]), fen DOC concentrations reflect more mixing between layers, thereby displaying similar DOC concentrations with depth. DOC concentrations within the bogs on the other hand, are more stratified with depth because of lower flow velocities and greater downward hydraulic gradients. This stratification diminishes with time as bog DOC concentrations increase significantly within the 0-25 and 25-50 cm layers during the summer and fall, while the 50-75 cm layer remains relatively unchanged.

Bog water levels dropped further below the ground surface over the course of the summer (Figure 3-2) and the lateral transmission of bog water to the fens decreased (Figure 3-3 [b] and Figure 3-4 [b]). Lower water levels promote aerobic decomposition (Wieder and Vitt, 2006), but newly decomposed organic matter may remain in the solid phase until fresh water becomes available (Blodau et al., 2004). There were several small rainfall events during the summer of 2012 (Figure 3-11 [a]), which likely contributed to dissolution of freshly decomposed organic matter (i.e. DOC). However, transmissivity remained very low [~0.0001 cm^2 s^{-1}] during the summer, releasing little water and DOC to the fens at this time. DOC released into the water column but not exported, resulted in a significant increase in DOC concentrations within the bogs during the summer (Figure 3-9 [a]). DOC concentrations within the 50-75 cm zone changed very little with time, compared to shallower locations because of persistent anoxic conditions and low freshwater turnover rates. DOC concentrations did not change considerably by the fall sampling event (Figure 3-9 [a]), which is likely a result of more balanced DOC production and export, since connectivity with the fens improved (Figure 3-4 [b] and Figure 3-3 [b]).

Although fen water levels also dropped below the ground surface over the course of the summer, water levels were nearer the surface than in the bogs (minimum average water level was -27.7 cmbgs) (Figure 3-2) thereby reducing the opportunity for aerobic decomposition and DOC production (Wieder and Vitt, 2006). Furthermore, transmissivity remained 2 orders of magnitude higher than in the bogs (Figure 3-4 [b]), causing some flushing; consequently DOC concentration was lower than in bogs and did not increase significantly during the summer (Figure 3-9 [b]). As fen water levels recovered during the fall (Figure 3-2) and connectivity with the bogs was re-established (Figure 3-3 [b]), dissolution of decomposed organic materials and flushing of DOC from the bogs into the fens resulted in an increase in DOC concentrations within the fens (Figure 3-9 [b]).

3.5.2 Stream Channel Hydrology and Geochemistry

3.5.2.1 Peatland Contributions to Streamflow

North-North Granny Creek basin appears to alternate between 2 hydrological phases: the first occurs when water levels are sufficiently high that depression storage capacities are met and the system drains as a single source area (spring and fall), aided by the high transmissivity of the upper layers of saturated peat; the second phase occurs when the loss of water through seepage and evaporation dominate the water budget and the system becomes disconnected (summer) (cf. Quinton and Roulet, 1998). Richardson et al. (2012) suggests the variable source area concept applies in this landscape. As shown on Figure 3-1, the upper reaches of the watershed consists of a high proportion of low-lying fen-water-tracks, while the lower reaches are primarily made up of 2 large domed bogs hosting a series of internal fen-water-tracks, including FWT1 to FWT 4.

During the spring and fall, water levels were high within the bogs and internal fen-water-tracks (Figure 3-2). As such, transmissivity was high (Figure 3-4 [b]), the bogs were connected to the internal fen-water-tracks (Figure 3-3 [b]) and the peatlands were contributing quick flow to basin discharge. Similar processes are also assumed to occur within the upper reaches. Early in the season, for the period of reliable discharge measurement, the upper and lower reaches had similar discharge (Figure 3-11 [b]), and decreased as water tables and transmissivity declined within the bogs (Figure 3-2 and Figure 3-4 [b], respectively) and the bogs no longer contributed water to the fens (30-July and 15-August, Figure 3-3 [b]). With the onset of a wetter period (fall), the bogs re-established connectivity with the fens (Figure 3-3 [b]), and flow at the lower reach increased, without a corresponding increase at the upper station (Figure 3-11 [b]). The low-lying fen-water-tracks of the upper reaches include a series of large systematic pools and ridges (much larger than the internal fen-water-tracks) which require filling in order to re-establish quick flow response (Quinton and Roulet, 1998). As such, streamflow contributions recovered earlier within the lower reaches. The lower reaches were also found to be more responsive to small rainfall events (<5 mm), as compared to the upper reaches (Figure 3-11 [a. and b.]). This is not surprising as bogs tend to have flashier hydrological regimes than fens (Holden, 2005) on account of highly permeable upper layers and rigid peat matrix which allow for large changes in water table position (Figure 3-2) and subsequent effluxes of water (Figure 3-4) with a relatively small amount of precipitation (Price et al., 2003).

3.5.2.2 Peatland Contributions to Stream DOC Concentrations

Lowest stream water DOC concentrations were observed in the early spring (Figure 3-11 [c]). Typically, stream channel DOC levels may be diluted at the beginning of spring due to the mixing of stream water with DOC-poor snowmelt passing minimally within the peat profile and rapidly across the frozen peat surface (Dyson et al., 2011). Leclair et al. (submitted) noted snowmelt water contributed to surface flows during April-2011, when frost table depths were shallow. The increase in DOC concentrations obtained during April and May-2012 (Figure 3-11 [c]) probably reflects the smaller snowmelt input in 2012 (Table 3-1), and the increasing depth to frost, which increases the availability of DOC. This also explains the large flux of water and DOC, without an increase in DOC concentration, which was observed during the large rainfall event of mid-May. As discharge declined through the summer, DOC concentration was generally maintained, thus the flux of DOC declined (Figure 3-11 [d]). Upon the fall wet-up period, as the peatlands became reconnected and the newly decomposed matter that had accumulated over the summer was available for export, DOC was flushed from the peatlands and caused an increase in DOC concentration within the stream channel (Figure 3-11 [c]). The higher DOC flux in the lower section reflects the earlier increase of flow along that reach.

Overall, stream discharge and DOC chemistry indicate the importance of the lower reaches of the NNGC watershed as being key contributors of runoff and DOC to the NNGC stream. However, based on calculated groundwater flow (Equation 3-3), the upper 75 cm of the internal fen-water-tracks provided less than 1% of the total basin discharge and DOC flux throughout the study period. By comparison, the area of these four FWTs is ~8% of the NNGC watershed. Although this calculation does not include contributions below 75 cmbgs, flow through the deeper layers is severely constrained by its very low hydraulic conductivity (Whittington and Price, 2013). Our estimate of discharge contributions of <1% from the internal fen-water-tracks corroborates earlier estimates calculated by Leclair et al. (submitted) who accounted for deep peat flow. Unfortunately, neither of these studies included surface flow measurements through rivulets or flow through soil pipes, for reasons noted in Section 3.3.1.

Surface flow through rivulets was observed within the southern portion of the internal fen-water-tracks during the major wet-up events of spring and fall 2012. A flume was constructed at the southern outlet of FWT4 following this study (in 2014) and a stage-discharge rating curve was generated based on stage measurements obtained from the flume throughout 2014 (McCarter, personal communication). Substituting water table measurements obtained from the southern fen nest of FWT4 on 20-October-2012 into the stage-discharge relationship reveals a surface discharge measurement of 3.4×10^{-4} m^3 s^{-1}; this is substantially higher than the calculated groundwater flow through the upper 75 cm of FWT4 on 20-October-2011 (1.7×10^{-6} m^3 s^{-1}). A surface water sample was also collected at the time of this measurement which had a DOC concentration of 33.38 mg L^{-1}; therefore, the estimated surface water flux of DOC from FWT4 (11.3 mg s^{-1}) was also substantially higher than the flux of DOC calculated within the upper 75 cm of the peat profile (0.04 mg s^{-1}). These values therefore indicate that rivulet flow could represent a large flux of water and DOC from the internal fen-water-tracks to the stream channel which has been unaccounted for in the calculations of this paper.

3.5.3 Influence of Aquifer Depressurization on Peatland Hydrology, DOC Production and Transport

FWT1 and FWT 4 are of equivalent shape and size, have similar patterns of pond convergence and comparable thicknesses of underlying marine sediments. FWT1 is closer to the mine and the cone of aquifer depressurization (~8 m) is greater than at FWT4 (~2 m), based on 2012 data (Figure 3-1 [b]). With the substantial differences in aquifer depressurization for these otherwise similar systems, differences in their behaviour have been noted. Given the pseudoreplication inherent in this and many basin-scale field studies (Hurlbert, 1984), the differences noted between these two systems may be an artefact of the treatment, so the broader applicability of these results must be viewed accordingly. Nevertheless, observations here are consistent with interpretations made by Leclair et al. (2015) and Whittington and Price (2012; 2013), and along with data from FWT2 and FWT3 provide insight into the function of fen water track and their response to dewatering.

At southern fen nest of FWT1, the depth of water table decline over the summer was greater than at the southern fen nest of FWT4 (Figure 3-5), which exacerbated the decrease and variability of transmissivity at FWT1 (Figure 3-7). The higher water table variability at FWT1 was in part, due to the stronger mid-summer downward gradients (Figure 3-6) enhancing deep seepage. However, the peat at FWT1 was also more rigid, and underwent less deformation with drainage (water table depth and elevation more similar; Figure 3-5). Peat subsidence, which occurred at both sites, can occur seasonally due to changes in soil water pressure, and decreases the hydraulic conductivity of peat (Price, 2003), and hence its transmissivity. As noted previously, it also reduces the apparent depth to water table, which partially offsets the reduced hydraulic conductivity since more water is available to flow in the relatively permeable upper layers, and makes the system more responsive to runoff events than it otherwise would be (Whittington and Price, 2006). It is likely that earlier and greater dewatering of the peat at FWT1 has pre-consolidated the peat matrix, and this is only partly reversible (Kennedy and Price, 2005). At monitoring stations located proximal to FWT1, Whittington and Price (2013) found peatland surface subsidence (up to 7.3 cm) occurred over the first 5 years of aquifer depressurization.

The southern fen nests at FWT1 and FWT4 had similar DOC concentrations at the beginning of the study period, but the patterns became distinct through time. Results obtained within the southern fen nest of FWT4 generally corresponded the average fen values reported in Figure 3-9 [b], with similar DOC concentrations in the spring and summer sampling events, but significantly higher in the fall (Figure 3-10). As previously noted, the similarity in DOC concentrations at all measured depths in FWT4 reflect better mixing than in FWT1 (Figure 3-10) and in bogs (Figure 3-9 [a]). This mixing is associated with greater water flow as suggested by its higher transmissivity (Figure 3-7). Conversely, by the summer sampling event, DOC concentrations at the southern fen nest of FWT1 began to stratify as concentrations increased in the shallower layers, but remained unchanged within the 50-75 cm layer. Similar to the bogs, these values are a likely result of increased aerobic decomposition (Section 3.5.1.2). The similarity of DOC concentration in the southern fen nest of FWT1 to that of the bogs probably reflects the greater impact of aquifer depressurization at FWT1, which like bogs, had larger water table recession and greater downward hydraulic gradients during the summer (Figure 3-5 and Figure 3-6, respectively). Leclair et al. (submitted) also determined that fens located within the zone of influence are displaying more *bog-like* hydrologic signatures by acting as recharge zones (rather than discharge areas) due to an increase in downward hydraulic gradients.

As a consequence of aquifer depressurization, the higher DOC concentrations noted within the southern fen nest of FWT1 during the summer and fall are a result of increased water table variability, increased vertical hydraulic gradients and lowered seasonal water tables. These processes caused an increase in decomposition during the summer and a subsequent surge in DOC to the stream networks during the fall *wet-up-period*. And although DOC concentrations remained generally high throughout the *fall wet-up-period*, concentrations were diminishing by the end of the study period as DOC continued to be flushed from the system, coincident with the increased transmissivity during that period. Unfortunately sample collection ceased prior to freeze-up. It is likely, that DOC concentrations would have continued to decline to near spring values prior to freeze-up. This is an indication that DOC production and transportation to downstream ecosystems is increasing as a result of aquifer depressurization.

3.6 Conclusion

Until recently, very little research has been completed on internal fen-water-tracks. Victor Mine has provided a venue for research into the hydrology and geochemistry of these features, as well as the potential impact of aquifer depressurization on dissolved organic carbon (DOC) production and export to downstream ecosystems.

Similar to large low-lying fen-water-tracks, our results indicate that internal fen-water-tracks function as conveyors of water to downstream ecosystems. And, located atop domed bogs and recipients of their DOC laden waters, these systems also act to export DOC from the domed bogs toward receiving surface waters. Although our results indicate that discharge and the flux of DOC to stream networks was greatest following a large rain event during the spring, spring-time stream DOC concentrations were generally lower than in the fall; this is important to note as it is the concentration of DOC, rather than the quantity, that affects stream ecology (Steinberg, 2003).

Warmer temperatures and low water tables associated with summer conditions result in an increase in DOC production and a decline in connectivity between the bogs and fens, thus allowing for DOC accumulation within the bogs. Upon the fall wet-up-period, water tables recover, the bogs and fens are reconnected, and the newly decomposed matter is available for suspension and transportation to stream networks. As a result, DOC concentrations within the North Granny Creek stream channel generally increased throughout the fall wet-up period. We therefore suggest that studies regarding the export of DOC from peatlands to receiving stream networks within the HJBL should focus on the fall wet-up period to account for DOC flushing during this time, a time period often omitted in studies. Corresponding peaks in water tables, stream discharge and DOC concentrations within the North Granny Creek, emphasizes the need to closely monitor individual storm events during this time to capture important pulses of DOC to stream networks (Hinton et al., 1997).

However, the increased need for fall monitoring does not discount or replace the need for spring measurements, particularly under climate change scenarios. Since climate change is expected to result in the warming of northern regions and a subsequent shift in the timing and magnitude of the spring flood event, as well as more intense precipitation events, our results indicate that DOC export during the spring may also change. Although Dyson et al. (2011) concluded that the timing and magnitude of the spring flood event is not likely to affect the annual flux of DOC to stream channels; our results indicate that an earlier snowmelt and a subsequently intense precipitation event could result in a large flux of DOC from the peatlands.

Within proximity of Victor Mine, the production of DOC during the summer is increased as a result of aquifer depressurization. Increased vertical hydraulic gradients and lowered seasonal water tables (as a result of aquifer depressurization) (Leclair et al., 2015; Whittington and Price, 2013) promote the production of DOC, as aerobic decomposition and labile carbon turnover increases. Since this is coincident with lowered discharge values, DOC export could be reduced regardless of the increased internal DOC production (Pastor et al., 2003). As of 2013, however, large rain inputs during the fall wet-up period were sufficient to reconnect the impacted system permitting newly decomposed organic matter to be dissolved and transported into downstream ecosystems. Since hydraulic gradients are expected to increase with continuing aquifer depressurization, it is unclear whether annual weather conditions will continue to be sufficient to re-establish connectivity during the wetter seasons (spring and fall) and subsequently flush the accumulated decomposed organic matter from the system.

3.7 Acknowledgments

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4 Conclusion

The main objectives of this thesis were to evaluate the impact of aquifer depressurization on modifications in hydrological processes within the peatland which are components contributing to runoff, as well as on the production and transportation DOC into surface waters within inland Hudson/James Bay Lowlands. Specifically, the effect of lower seasonal water tables and increased vertical hydraulic gradients on recharge and subsequent discharge within an impacted system, and how this could influence the production and mobility of dissolved organic carbon (DOC) to receiving surface waters.

Based on the results of this study, the peatlands located within the zone of influence of aquifer depressurization are experiencing hydrological alterations that are affecting peatland recharge, internal production of DOC, discharge and DOC export. Isotopic analysis revealed that drier antecedent moisture conditions (lowered seasonal water tables) are permitting an increase in infiltration of event water, which is subsequently reducing surface flow to stream channels. Isotopic enrichment (due to an increase in isotopically enriched spring rains) indicate that this is occurring most prominently during the spring, as increased vertical hydraulic gradients are resulting in the greatest decline in water levels over the course of the winter, when incoming precipitation is unavailable for recharging the system. During the summer, lower water tables and increased vertical hydraulic gradients provide opportunity for greater aerobic decomposition, increased labile carbon turnover and ultimately a greater accumulation of DOC within the peatlands. Although isotopic analysis indicates a shift in the proportion of surface water and groundwater contributions to the stream during the summer and into the fall wet-up period, incoming precipitation is currently sufficient to reconnect the peatlands and provide some degree of surface runoff at this time. Therefore, DOC accumulated over the summer is currently still available for transportation to the stream networks during the fall. It remains unclear however, whether annual weather conditions will continue to suffice in reconnecting the hydrologically depleted system, as hydraulic gradients are expected to increase with continuing aquifer depressurization. As Whittington (2013) noted, with a simple calculation of peat depth (2.5 m) and specific yield (0.15; unpublished field data), a completely dry system would only require 375 mm (2500 mm x 0.15) of precipitation to completely wet back up. While this is reassuring from a hydrological point of view, from a geochemical viewpoint our results indicate that this could result in large surge of potentially harmful constituents into downstream ecosystems. And, since vertical seepage losses are delayed with rapidly decreasing transmissivity with depth (4 orders of magnitude within the upper 50 cm), a large rainfall event (less than 375 mm) could be sufficient to re-establish temporary connectivity within the upper, highly conductive active layer and result in a subsequent flushing of organic matter without having a fully wet system. This could have large implications on fisheries, as well as on the peatlands themselves, as a flushing of recalcitrant carbon and a turnover of fresh water could promote further peatland decomposition and deterioration following subsequent water table recessions.

In general, this thesis provides insight into the hydrological and geochemical function of bogs, low-lying fens and internal fen-water-tracks within the HJBL. It demonstrates recharge, connectivity and discharge processes across the various landscapes, and emphasizes the important roles bogs and fens play in the fall wet-up period on the delivery of DOC to stream networks within the HJBL.

In terms of resource development, this work highlights the importance of increased vertical hydraulic gradients and lowered seasonal water tables on modifications to recharge/discharge relationships and the proportion of surface water/groundwater contributions to stream networks, as well as on the increased production and mobility of DOC during the early stages of aquifer depressurization.

This thesis demonstrates the usefulness of hydrogeochemical analyses on understanding the hydrological and geochemical implications of aquifer depressurization. Environmental tracers have successfully provided insight into alterations in hydrological processes occurring within the system, despite inputs from upper non-impacted regions of the watershed and annual variations in weather conditions. From a hydrological stand point, these inputs are mitigating the hydrological effects of aquifer depressurization; however, with the use of geochemical tracers and DOC analyses, it is apparent that they are not isolating the peatlands ecologically from the ramifications of aquifer depressurization through an altered hydrological regime.

Caution is advised when interpreting these results in a climate change scenario; lowered seasonal water tables are expected with warmer temperatures associated with climate change, but increased vertical hydraulic gradients are not likely to be as prevalent as with aquifer depressurization.

5 References

Ali, K., 2013. The Role of the Tyrrell Sea Aquitard and Bioherms in the Hydrogeology of the James Bay Lowlands Under Dewatering Conditions, Queen's University, Kingston.

Auterives, C., Aquilina, L., Bour, O., Davranche, M., Paquereau, V., 2011. Contribution of climatic and anthropogenic effects to the hydric deficit of peatlands. *Hydrological Processes*, 25(18): 2890-2906.

Baird, A.J., Belyea, L.R.B., Comas, X., Reeve, A.S., Slater, L.D., 2013. Carbon Cycling in Northern Peatlands. Wiley.

Ballard, C.E., McIntyre, N., Wheater, H.S., Holden, J., Wallage, Z.E., 2011. Hydrological modelling of drained blanket peatland. *Journal of Hydrology*, 407(1–4): 81-93.

Beer, J., Blodau, C., 2007. Transport and thermodynamics constrain belowground carbon turnover in a northern peatland. *Geochimica Et Cosmochimica Acta*, 71(12): 2989-3002.

Belyea, L.R., 1996. Separating the effects of litter quality and microenvironment on decomposition rates in a patterned peatland. *Oikos*, 77: 529-539.

Blodau, C., 2002. Carbon cycling in peatlands — A review of processes and controls. *Environmental Reviews*, 10(2): 111-134.

Blodau, C., Basiliko, N., Moore, T.R., 2004. Carbon turnover in peatland mesocosms exposed to different water table levels. *Biogeochemistry*, 67(3): 331-351.

Blodau, C., Siems, M., 2012. Drainage-induced forest growth alters belowground carbon biogeochemistry in the Mer Bleue bog, Canada. *Biogeochemistry*, 107(1-3): 107-123.

Bouffard, J.-S., 2014. A comparison of conceptual rainfall-runoff modelling structures and approaches for hydrologic predication in ungauged peatland basins of the James Bay Lowlands, Carleton University, Ottawa.

Bowling, L., Kane, D., Gieck, R., Hinzman, L., Lettenmaier, D., 2003. The role of surface storage in a low-gradient Arctic watershed. *Water Resources Research*, 39(4): 1087.

Branfireun, B.A., Roulet, N.T., 1998. The baseflow and storm flow hydrology of a precambrian shield headwater peatland. *Hydrological Processes*, 12: 57-72.

Canadian Council of Ministers of the Environment, 2011. Protocols manual for water quality sampling in Canada.

Clark, I., Fritz, P., 1997. Environmental isotopes in hydrogeology. CRC Press, 328 pp.

Clark, J.M. et al., 2009. Increased temperature sensitivity of net DOC production from ombrotrophic peat due to water table draw-down. *Global Change Biology*, 15(4): 794-807.

Clark, J.M., Heinemeyer, A., Martin, P., Bottrell, S.H., 2012. Processes controlling DOC in pore water during simulated drought cycles in six different UK peats. *Biogeochemistry*, 109(1-3): 253-270.

Clymo, R., 1984. The Limits to Peat Bog Growth. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences (1934-1990)*, 303(1117): 605-654.

Dansgaard, W., 1964. Stable isotopes in precipitation. *Tellus*, 16(4): 436-468.

Davies, G.M., Gray, A., 2015. Don't let spurious accusations of pseudoreplication limit our ability to learn from natural experiments (and other messy kinds of ecological monitoring). *Ecology and Evolution*, 5(22): 5295-5304.

Déry, S., Stieglitz, M., McKenna, E., Wood, E., 2005. Charateristics and Trends of River Dicharge into Hudson, James and Ungava Bays, 1964-2000. *Journal of Climate*, 18: 2540-2556.

Difebo, A., Richardson, M., Price, J., 2015. Fusion of Multispectral Imagery and LiDAR Digital Terrain Derivatives for Ecosystem Mapping and Morphological Characterization of a Northern Peatland Complex, Remote Sensing of Wetlands. CRC Press, pp. 399-412.

Dinsmore, K.J. et al., 2011. Stream water hydrochemistry as an indicator of carbon flow paths in Finnish peatland catchments during a spring snowmelt event. *Science of the Total Environment*, 409(22): 4858-4867.

Dredge, L.A., Cowan, W.R., 1989. Quaternary geology of the southwestern Canadian Shield; in Chapter 3 of Quaternary Geology of Canada and Greenland. In: Fulton, R.J. (Ed.). Geological Survey of Canada, Geology of Canada no. 1.
Dyson, K. et al., 2011. Release of aquatic carbon from two peatland catchments in E. Finland during the spring snowmelt period. *Biogeochemistry*, 103(1-3): 125-142.
Eimers, M.C., Buttle, J., Watmough, S.A., 2008. Influence of seasonal changes in runoff and extreme events on dissolved organic carbon trends in wetland- and upland-draining streams. *Canadian Journal of Fisheries and Aquatic Sciences*, 65(5): 796-808.
Environment Canada, 2008. Canadian Climate Normals or Averages 1971-2000, accessed June 15, 2011 http://www.climate.weatheroffice.ec.gc.ca/climate_normals/index_e.html.
Faure, G., Mensing, T.M., 2005. Isotopes : principles and applications. Wiley, Hoboken, N.J.
Fetter, C., 1994. Applied Hydrogeology. Macmillan New York (USA).
Freeman, C., Evans, C., Monteith, D., Reynolds, B., Fenner, N., 2001. Export of organic carbon from peat soils. *Nature*, 412(6849): 785.
Frey, K.E., McClelland, J.W., 2009. Impacts of permafrost degradation on arctic river biogeochemistry. *Hydrological Processes*, 23(1): 169-182.
Gibson, J., 2001. Forest-tundra water balance signals traced by isotopic enrichment in lakes. *Journal of Hydrology*, 251(1-2): 1-13.
Gibson, J.J., Edwards, T.W.D., Prowse, T.D., 1993. RUNOFF GENERATION IN A HIGH BOREAL WETLAND IN NORTHERN CANADA. *Nordic Hydrology*, 24(2-3): 213-224.
Glaser, P., 1987. The development of streamlined bog islands in the continental interior of North America. *Arctic and Alpine Research*: 402-413.
Glaser, P.H., Hansen, B., Siegel, D., Reeve, A.S., Morin, P., 2004a. Rates, pathways and drivers for peatland development in the Hudson Bay Lowlands, northern Ontario, Canada. *Journal of Ecology*, 92(6): 1036-1053.
Glaser, P.H., Seigel, D.I., Reeve, A.S., Janssens, J., Janecky, D., 2004b. Tectonic drivers for vegetation patterning and landscape evolution in the Albany River region of the Hudson Bay Lowlands. *Journal of Ecology*, 92(6): 1054-1070.
Gorham, E., 1991. Northern peatlands: Role in the carbon cycle and probable responses to climatic warming. *Ecological Applications*, 1(2): 182-195.
Hayashi, M., Quinton, W.L., Pietroniro, A., Gibson, J.J., 2004. Hydrologic functions of wetlands in a discontinous permafrost basin indicated by isotopic and chemical signatures. *Journal of Hydrology*, 296: 81-97.
HCI, 2004. Dewatering of the Victor Diamond Project. Predicted Engineering, Cost, and Environmental Factors. HCI-1779, Prepared by Hydrologic Consultants Inc of Colorado.
Hinton, M.J., Schiff, S.L., English, M.C., 1997. The significance of storms for the concentration and export of dissolved organic carbon from two Precambrian Shield catchments. *Biogeochemistry*, 36(1): 67-88.
Holden, J., 2005. Peatland hydrology and carbon release: why small-scale process matters, 363, 2891-2913 pp.
Holden, J., Chapman, P.J., Labadz, J.C., 2004. Artificial drainage of peatlands: hydrological and hydrochemical process and wetland restoration. *Progress in Physical Geography*, 28(1): 95-123.
Hribljan, J.A., Kane, E.S., Pypker, T.G., Chimner, R.A., 2014. The effect of long-term water table manipulations on dissolved organic carbon dynamics in a poor fen peatland. *Journal of Geophysical Research-Biogeosciences*, 119(4): 577-595.
Hunt, R.J., Bullen, T.D., Krabbenhoft, D.P., Kendall, C., 1998. Using stable isotopes of water and strontium to investigate the hydrology of a natural and a constructed wetland. *Ground Water*, 36(3): 434-443.
Hurlbert, S.H., 1984. Pseudoreplication and the Design of Ecological Field Experiments. *Ecological Monographs*, 54(2): 187-211.

Hvorslev, M.J., 1951. Time lag and soil permeability in groundwater observations, Waterways Experimental Station Bulletin 36. US Army Corps of Engineers, Vicksburg, Mississippi.

Itasca Dever Inc, 2011. Dewatering of Victor Diamond Project, March 2011 update of March 2008 Groundwater flow model, Denver, Colorado.

Johansen, O.M., Pedersen, M.L., Jensen, J.B., 2011. Effect of groundwater abstraction on fen ecosystems. *Journal of Hydrology*, 402(3–4): 357-366.

Keller, J.K., White, J.R., Bridgham, S.D., Pastor, J., 2004. Climate change effects on carbon and nitrogen mineralization in peatlands through changes in soil quality. *Global Change Biology*, 10: 1053-1064.

Kennedy, G.W., Price, J.S., 2005. A conceptual model of volume-change controls on the hydrology of cutover peats. *Journal of Hydrology*, 302: 13-27.

Kværner, J., Snilsberg, P., 2011. Groundwater hydrology of boreal peatlands above a bedrock tunnel – Drainage impacts and surface water groundwater interactions. *Journal of Hydrology*, 403(3–4): 278-291.

Leclair, M., Whittington, P., Price, J., 2015. Hydrological functions of a mine-impacted and natural peatland-dominated watershed, James Bay Lowland. *Journal of Hydrology: Regional Studies*, 4, Part B: 732-747.

Leclair, M., Whittington, P., Price, J.S., submitted. Groundwater connectivity of bog and fen landscapes in a mine impacted region, James Bay Lowland. *Journal of Hydrology*.

Levy, Z.F., Siegel, D.I., Dasgupta, S.S., Glaser, P.H., Welker, J.M., 2014. Stable isotopes of water show deep seasonal recharge in northern bogs and fens. *Hydrological Processes*, 28(18): 4938-4952.

Limpens, J. et al., 2008. Peatlands and the carbon cycle: from local processes to global implications – a synthesis. *Biogeosciences*, 5(5): 1475-1491.

McDonald, B.C., 1989. Glacial and interglacial stratigraphy, Hudson Bay Lowland. In: Fulton, R.J. (Ed.), Quaternary Geology of Canada and Greenland, Geology of Canada. Geological Survey of Canada.

McLaughlin, J., Webster, K., 2014. Effects of Climate Change on Peatlands in the Far North of Ontario, Canada: a Synthesis. *Arctic Antarctic and Alpine Research*, 46(1): 84-102.

Minkkinen, K., Laine, J., 1998. Effect of forest drainage on the peat bulk density of pine mires in Finland. *Canadian Journal of Forestry Resources*, 28: 178-186.

Mitsch, W.J., Gosselink, J.G., 2000. Wetlands. John Wiley, New York, xiii, 920 p. pp.

Moore, T.R., 2013. Dissolved Organic Carbon Production and Transport in Canadian Peatlands, Carbon Cycling in Northern Peatlands. American Geophysical Union, pp. 229-236.

Moore, T.R., Matos, L., Roulet, N.T., 2003. Dynamics and chemistry of dissolved organic carbon in precambrian shield catchments and an impounded wetland. *Canadian Journal of Fisheries and Aquatic Sciences*, 60(5): 612-623.

Morris, P.J., Waddington, J.M., 2011. Groundwater residence time distributions in peatlands: Implications for peat decomposition and accumulation. *Water Resources Research*, 47.

National Wetlands Working Group, 1997. The Canadian Wetland Classification System - Second Edition. University of Waterloo, Waterloo, Ontario.

Orlova, J., Branfireun, B.A., 2014. Surface Water and Groundwater Contributions to Streamflow in the James Bay Lowland, Canada. *Arctic, Antarctic, and Alpine Research*, 46(1): 236-250.

Pastor, J. et al., 2003. Global warming and the export of dissolved organic carbon from boreal peatlands. *Oikos*, 100(2): 380-386.

Philben, M. et al., 2015. Temperature, oxygen, and vegetation controls on decomposition in a James Bay peatland. *Global Biogeochemical Cycles*, 29(6): 729-743.

Price, J.S., 2003. Role and character of seasonal peat soil deformation on the hydrology of undisturbed and cutover peatlands. *Water Resources Research*, 39(9): 1241.

Price, J.S., Fitzgibbon, J.E., 1987. Groundwater storage – streamflow relations during winter in a subarctic wetland, Saskatchewan. *Canadian Journal of Earth Sciences*, 24(10): 2074-2081.

Price, J.S., Heathwaite, A.L., Baird, A.J., 2003. Hydrological processes in abandoned and restored peatlands: An overview of management approaches. *Wetlands Ecology and Management*, 11: 65-83.

Price, J.S., Maloney, D.A., 1994. Hydrology of a patterned bog-fen complex in southeastern Labrador, Canada. *Nordic Hydrology*, 25: 313-330.

Price, J.S., Schlotzhauer, S.M., 1999. Importance of shrinkage and compression in determining water storage changes in peat: the case of a mined peatland. *Hydrological Processes*, 13: 2591-2601.

Price, J.S., Woo, M.-K., 1988. Origin of salt in coastal marshes of Hudson and James bays. *Canadian Journal of Earth Sciences*, 25(1): 145-147.

Quinton, W.L., Roulet, N.T., 1998. Spring and summer runoff hydrology of a subarctic patterned wetland. *Arctic and Alpine Research*, 30(3): 285-294.

R Development Core Team, 2009. R: A language and environment for statistical computing. R Foundation for Statistical Computing.

Reddy, K.R., DeLaune, R.D., 2008. Biogeochemistry of wetlands : science and applications. CRC Press, Boca Raton; London; New York.

Reeve, A., Siegel, D., Glaser, P., 1996. Geochemical controls on peatland pore water from the Hudson Bay Lowland: a multivariate statistical approach. *Journal of Hydrology*, 181(1-4): 285-304.

Reeve, A., Siegel, D., Glaser, P., 2000. Simulating vertical flow in large peatlands. *Journal of Hydrology*, 227(1-4): 207-217.

Reeve, A., Siegel, D., Glaser, P., 2001. Simulating dispersive mixing in large peatlands. *Journal of Hydrology*, 242(1-2): 103-114.

Richardson, M., Ketcheson, S.J., Whittington, P., Price, J., 2012. Runoff generation in a northern peatland complex: the influences of catchment morphology and scale *Hydrological Processes*, 26(12): 1805-1817.

Riley, J.L., 2011. Wetlands of the Ontario Hudson Bay Lowland: An Regional Overview. Nature Conservancy of Canada, Toronto, Ontario, Canada, 156 pp.

Rouse, W.R. et al., 1997. EFFECTS OF CLIMATE CHANGE ON THE FRESHWATERS OF ARCTIC AND SUBARCTIC NORTH AMERICA. *Hydrological Processes*, 11(8): 873-902.

Rouse, W.R., Woo, M., Price, J.S., 1992. Damming James Bay: I. Potential impacts on coastal climate and the water balance. *The Canadian Geographer*, 36(1): 2-7.

Schot, P.P., Dekker, S.C., Poot, A., 2004. The dynamic form of rainwater lenses in drained fens. *Journal of Hydrology*, 293(1): 74-84.

Siegel, D.I., Glaser, P.H., 1987. Groundwater Flow in a Bog-Fen Complex, Lost River Peatland, Northern Minnesota. *Journal of Ecology*, 75(3): 743-754.

Siegel, D.I., Glaser, P.H., 2006. The Hydrology of Peatlands. In: Wieder, K., Vitt, D. (Eds.), Boreal Peatland Ecosystems. Ecological Studies. Springer-Verlag, Berline.

Singer, S.N., Chen, C.K., 2002. An assessment of the groundwater resources of Northern Ontario, Environmental monitoring and reporting branch, Ministry of Environment, Toronto, Ontario, Canada.

Sjörs, H., 1959. Bogs and fens in the Hudson Bay Lowlands. *Arctic*, 12(1): 2-19.

Steinberg, C., 2003. Ecology of Humic Substance in Freshwaters. Springer, New York.

Strack, M. et al., 2008. Effect of water table drawdown on peatland dissolved organic carbon export and dynamics. *Hydrological Processes*, 22(17): 3373-3385.

Strack, M., Waddington, J., Rochefort, L., Tuittila, E., 2006. Response of vegetation and net ecosystem carbon dioxide exchange at different peatland microforms following water table drawdown. *J. Geophys. Res*, 111.

Straková, P. et al., 2011. Litter type affects the activity of aerobic decomposers in a boreal peatland more than site nutrient and water table regimes. *Biogeosciences*, 8(9): 2741-2755.

Sui, J., Thring, R., Karney, B.W., Wand, J., 2007. Effects of river ice on stage-discharge relationships - A case study of the Yellow River. *International Journal of Sediment Research*, 22(4).

Turner, K.W., Wolfe, B.B., Edwards, T.W.D., 2010. Characterizing the role of hydrological processes on lake water balances in the Old Crow Flats, Yukon Territory, Canada, using water isotope tracers. *Journal of Hydrology*, 386(1–4): 103-117.

Ulanowski, T., 2014. Hydrology and biogeochemistry of a bog-fen tributary complex in the Hudson Bay Lowlands, Ontario, Canada. University of Western Ontario, London, Ontario.

Waddington, J., Roulet, N., 1997. Groundwater flow and dissolved carbon movement in a boreal peatland. *Journal of Hydrology*, 191(1-4): 122-138.

Webster, K.L., McLaughlin, J.W., 2010. Importance of the Water Table in Controlling Dissolved Carbon along a Fen Nutrient Gradient. *Soil Sci. Soc. Am. J.*, 74(6): 2254-2266.

Whittington, P., 2013. The Impacts of Diamond Mining to Peatlands in the James bay Lowlands. PhD Thesis, University of Waterloo, Waterloo, 111 pp.

Whittington, P., Price, J., 2012. Effect of mine dewatering on peatlands of the James Bay Lowland: the role of bioherms. *Hydrological Processes*, 26(12): 1818-1826.

Whittington, P., Price, J.S., 2006. The effects of water table draw-down (as a surrogate for climate change) on the hydrology of a patterned fen peatland near Quebec City, Quebec. *Hydrological Processes*(20): 3589-3600.

Whittington, P., Price, J.S., 2013. Effect of mine dewatering on the peatlands of the James Bay Lowland: the role of marine sediments on mitigating peatland drainage. *Hydrological Processes*, 27(13): 1845-1853.

Whittington, P.N., Ketcheson, S.J., Price, J.S., Richardson, M., Di Febo, A., 2012. Areal differentiation of snow accumulation and melt between peatland types in the James Bay Lowlands. *Hydrological Processes*, 26(17): 2662-2671.

Wieder, R.K., Vitt, D.H., 2006. Boreal Peatland Ecosystems. Springer London, Limited.

6 Appendix A

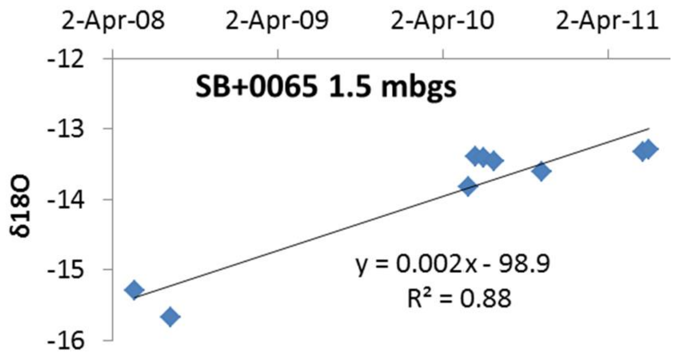

Figure A-1: Example of linear regression performed on all peat-water extracted from piezometers located along the main research transect, Trib5A and MS15. The example shown is from the 1.5 mbgs piezometer located at nest SB+0065B.

Table A-1: Contextualized variables for slope of $\delta^{18}O$ versus time graphs, for remote sites (left) and the main research transect (right), including: nest location, peatland type, depth of measurement (meters below ground surface), slope, y-intercept, R^2 value, number of samples (n), and years containing data.

Nest Location	Type	Depth (mbgs)	Slope	Y Intercept	R^2	p-Value	n	Years with Data
5A+0035	Bog	0.9	-7.0E-04	14.1	0.04	0.61	9	08,10,11
		1.5	-1.7E-04	-8.2	0.10	0.35	11	08-11
5A+0100	Bog	0.9	-5.5E-04	7.2	0.65	2.8E-03	11	08,10,11
		1.5	7.4E-06	-15.5	2.4E-04	0.97	9	08,10,11
		2.14	6.7E-05	-17.8	0.01	0.83	8	08-11
5A+0210	Bog	1.5	-6.4E-04	11.6	0.80	1.1E-03	9	08,10,11
		2.26	1.3E-04	-19.2	0.11	0.43	8	08-11
CB+0025	Bog	0.9	1.5E-04	-20.2	0.05	0.60	8	08,10,11
		1.5	-9.3E-05	-10.9	0.03	0.69	8	08,10,11
		2.3	-4.3E-04	2.8	0.32	0.40	6	08,10,11
CB+0040	Bog	0.9	1.6E-04	-20.5	0.03	0.67	8	08,10,11
		1.5	-8.0E-07	-14.3	4.7E-06	1.0	7	08,10,11
		2.6	-1.1E-04	-9.6	0.06	0.63	6	08,10,11
CB+0080	Fen	0.9	-1.4E-03	44.8	0.23	0.17	10	08,10,11
		1.5	2.8E-04	-25.0	0.02	0.67	10	08,10,11
		2.72	-1.4E-03	44.1	0.76	0.02	6	08,10,11
CB+0150	Fen	0.9	-2.0E-04	-4.8	0.13	0.38	7	08,10,11
		1.5	-2.9E-04	-1.9	0.15	0.33	8	08,10,11
		2.5	-9.9E-04	26.2	0.77	0.02	6	08,10,11

Nest Location	Type	Depth (mbgs)	Slope	Y Intercept	R^2	p-Value	n	Years with Data
SB+0045	Bog	2	4.7E-04	-33.7	0.81	0.29	3	08,11
SB+0065	Bog	0.9	2.5E-03	-114.6	0.45	0.22	5	08,10,11
		1.5	2.1E-03	-98.9	0.88	1.6E-04	9	08,10,11
		2.25	1.4E-03	-70.0	0.89	1.1E-04	9	08,10,11
SB+0100	Bog	0.9	9.6E-04	-53.3	0.67	6.8E-03	9	08,10,11
		1.5	2.8E-04	-26.7	0.05	0.52	11	08,10,11
		2.43	8.3E-06	-15.9	1.3E-04	0.98	9	08,10,11
SB+0180	Fen	0.9	5.1E-04	-36.3	0.15	0.34	8	08,11
		1.5	1.0E-04	-19.9	0.03	0.71	7	08,11
SB+0365	Fen	0.9	1.8E-04	-20.0	0.02	0.72	9	08-11
		1.5	6.8E-05	-15.6	0.03	0.57	12	08-11
SB+0430	Fen	0.9	-1.2E-04	-7.1	0.01	0.82	10	08-11
		1.5	4.2E-04	-29.2	0.22	0.14	11	08-11
SB+0540	Fen	0.9	9.4E-04	-52.3	0.84	3.7E-03	7	08,10,11
		1.5	6.1E-04	-39.0	0.82	2.1E-03	8	08,10,11
SB+0645	Bog	0.9	1.3E-03	-68.9	0.85	1.1E-03	8	08-11
		1.5	-1.3E-04	-10.5	0.08	0.45	9	08-11
SB+0875	Fen	0.9	1.1E-03	-55.1	0.19	0.21	10	08-11
		1.4	1.3E-03	-65.4	0.14	0.29	10	08-11
SB+1005	FWT	0.9	-1.5E-03	48.0	0.29	0.09	11	08-11
		1.5	-7.5E-05	-8.6	3.7E-04	0.95	15	08-11
SB+1130	FWT	0.9	-1.0E-03	30.2	0.30	0.20	7	08-11
		1.5	-2.8E-04	-0.6	0.27	0.23	7	08-11
		2.4	1.3E-03	-66.3	0.50	0.12	6	08,10,11
SB+1295	Bog	0.9	8.3E-04	-45.2	0.06	0.63	6	08,11
		1.5	3.8E-04	-27.7	0.01	0.83	7	08,10,11
SB+1445	Bog	0.9	9.5E-04	-51.0	0.53	0.01	11	08-11
		1.5	1.1E-03	-58.4	0.61	4.6E-03	11	08-11
		2.57	2.8E-04	-25.1	0.03	0.71	7	08,10,11
SB+1485	Bog	0.9	-7.2E-04	14.4	0.09	0.53	7	08,10,11

Table A-2: Contextualized variables for slope of δ^2H versus time graphs, for remote sites (left) and the main research transect (right), including: nest location, peatland type, depth of measurement (meters below ground surface), slope, y-intercept, R^2 value, number of samples (n), and years containing data.

Nest Location	Type	Depth (mbgs)	Slope	Y Intercept	R^2	p-Value	n	Years with Data
5A+0035	Bog	0.9	-1.5E-03	-45.0	0.02	0.72	9	08,10,11
		1.5	-3.1E-04	-95.9	0.03	0.62	11	08-11
5A+0100	Bog	0.9	-3.2E-03	21.1	0.80	1.9E-04	11	08,10,11
		1.5	-1.2E-03	-61.2	0.49	0.04	9	08,10,11
		2.14	-9.0E-04	-72.4	0.11	0.42	8	08-11
5A+0210	Bog	1.5	-2.6E-03	1.2	0.57	0.02	9	08,10,11
		2.26	-1.2E-03	-56.9	0.47	0.06	8	08-11
CB+0025	Bog	0.9	-6.9E-04	-77.0	0.08	0.51	8	08,10,11
		1.5	-4.6E-03	78.4	0.78	3.7E-03	8	08,10,11
		2.3	-3.6E-03	38.8	0.31	0.25	6	08,10,11
CB+0040	Bog	0.9	-1.4E-03	-47.0	0.11	0.42	8	08,10,11
		1.5	-3.5E-03	36.0	0.78	8.7E-03	7	08,10,11
		2.6	-3.0E-03	12.3	0.65	0.05	6	08,10,11
CB+0080	Fen	0.9	-1.6E-02	542.1	0.40	0.05	10	08,10,11
		1.5	3.0E-03	-222.1	0.06	0.49	10	08,10,11
		2.72	-1.0E-02	314.6	0.78	0.02	6	08,10,11
CB+0150	Fen	0.9	3.1E-04	-110.9	0.01	0.84	7	08,10,11
		1.5	-2.1E-03	-16.9	0.20	0.27	8	08,10,11
		2.5	-8.2E-03	226.6	0.67	0.05	6	08,10,11

Nest Location	Type	Depth (mbgs)	Slope	Y Intercept	R^2	p-Value	n	Years with Data
SB+0045	Bog	2	5.9E-03	-342.2	0.76	0.32	3	08,11
SB+0065	Bog	0.9	1.4E-02	-665.5	0.31	0.33	5	08,10,11
		1.5	1.5E-02	-704.2	0.91	6.9E-05	9	08,10,11
		2.25	9.0E-03	-472.8	0.98	3.8E-07	9	08,10,11
SB+0100	Bog	0.9	5.8E-03	-339.2	0.74	2.9E-03	9	08,10,11
		1.5	2.5E-03	-210.2	0.23	0.14	11	08,10,11
		2.43	2.5E-03	-213.2	0.55	0.02	9	08,10,11
SB+0180	Fen	0.9	1.9E-03	-187.7	0.04	0.62	8	08,11
		1.5	-5.8E-05	-111.3	4.5E-04	0.96	7	08,11
SB+0365	Fen	0.9	8.8E-04	-135.7	0.01	0.76	9	08-11
		1.5	2.8E-05	-101.5	3.8E-04	0.95	12	08-11
SB+0430	Fen	0.9	1.1E-03	-136.1	0.06	0.48	10	08-11
		1.5	3.5E-03	-236.2	0.56	7.9E-03	11	08-11
SB+0540	Fen	0.9	5.8E-03	-337.7	0.92	6.9E-04	7	08,10,11
		1.5	4.2E-03	-276.6	0.88	5.9E-04	8	08,10,11
SB+0645	Bog	0.9	8.5E-03	-450.8	0.87	6.5E-04	8	08-11
		1.5	-8.7E-05	-110.1	0.01	0.79	9	08-11
SB+0875	Fen	0.9	6.2E-03	-345.7	0.29	0.11	10	08-11
		1.4	9.0E-03	-460.5	0.19	0.21	10	08-11
SB+1005	FWT	0.9	-7.4E-03	210.6	0.49	0.02	11	08-11
		1.5	2.3E-03	-186.4	0.03	0.53	15	08-11
SB+1130	FWT	0.9	-5.2E-03	114.9	0.22	0.29	7	08-11
		1.5	2.6E-04	-103.5	0.01	0.82	7	08-11
		2.4	6.4E-03	-351.2	0.86	0.01	6	08,10,11
SB+1295	Bog	0.9	3.7E-03	-241.9	0.07	0.61	6	08,11
		1.5	1.4E-03	-153.8	0.01	0.87	7	08,10,11
SB+1445	Bog	0.9	2.4E-03	-188.8	0.10	0.34	11	08-11
		1.5	6.0E-03	-340.7	0.45	0.02	11	08-11
		2.57	2.2E-03	-185.3	0.17	0.35	7	08,10,11
SB+1485	Bog	0.9	-4.7E-03	83.8	0.07	0.56	7	08,10,11

Table A-3: Contextualized variables for slope of Chloride Concentration versus time graphs, for remote sites (left) and the main research transect (right), including: nest location, peatland type, depth of measurement (meters below ground surface), slope, y-intercept, R^2 value, number of samples (n), and years containing data.

Nest Location	Type	Depth (mbgs)	Slope	Y Intercept	R^2	p-Value	n	Years with Data	Nest Location	Type	Depth (mbgs)	Slope	Y Intercept	R^2	p-Value	n	Years with Data
5A+0035	Bog	0.9	2.2E-03	-82.1	0.49	0.12	6	09-11	SB+0100	Bog	1.5	4.4E-04	-16.6	0.01	0.88	6	09,11
		1.5	-4.9E-03	261.9	0.03	0.68	8	09-11	SB+0180	Fen	0.9	-7.5E-04	31.0	0.45	0.03	10	09-11
5A+0100	Bog	0.9	5.3E-04	-5.6	0.01	0.79	9	09-11			1.5	-4.5E-03	188.1	0.66	0.02	8	09-11
		1.5	4.2E-03	-128.7	0.34	0.17	8	09-11	SB+0365	Fen	0.9	8.0E-04	-24.8	0.06	0.47	11	09-11
5A+0210	Bog	1.5	1.2E-02	-463.2	0.74	0.01	8	09-11			1.5	-3.4E-03	155.7	0.27	0.08	12	09-11
		2.26	-1.2E-03	89.3	0.00	0.93	6	09-11	SB+0430	Fen	0.9	-2.2E-03	94.2	0.23	0.14	11	09-11
5A+0400	Fen	0.9	6.1E-04	-19.8	0.02	0.72	8	09-11			1.5	-1.2E-02	488.1	0.89	4.7E-06	12	09-11
5A+0475	Fen	0.9	8.3E-03	-330.2	0.99	1.9E-06	7	09-11	SB+0540	Fen	0.9	-3.3E-02	1356.1	0.99	2.5E-08	9	09-11
											1.5	-8.5E-02	3481.0	0.98	1.3E-07	8	09-11
									SB+0645	Bog	0.9	-4.5E-03	195.7	0.49	0.02	11	09-11
											1.5	-2.4E-03	130.6	0.03	0.60	12	09-11
									SB+0875	Fen	0.9	1.3E-03	-53.0	0.34	0.04	13	09-11
											1.4	-1.2E-03	51.7	0.02	0.70	12	09-11
									SB+1005	FWT	0.9	-1.3E-03	52.1	0.91	1.2E-06	12	09-11
											1.5	-4.3E-03	177.5	0.82	2.3E-05	13	09-11
									SB+1130	FWT	0.9	-2.5E-04	10.4	0.39	0.09	8	09-11
											1.5	-5.6E-04	23.4	0.53	0.03	9	09-11
									SB+1295	Bog	0.9	8.5E-05	-3.2	0.09	0.39	11	09-11
											1.5	-3.0E-04	12.5	0.02	0.72	9	09-11
									SB+1445	Bog	0.9	-2.0E-04	8.4	0.06	0.50	10	09-11
											1.5	-1.2E-04	5.1	0.05	0.56	9	09-11