Impacts on Water Quality at a Newly Opened and Extracted Peatland: Influence of Internal Processes and Hydrological Connectivity in Horticultural Peat Harvesting

by

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Abstract

Horticultural peat harvesting continues to expand in Canada, yet the impact to downstream water quality during the initial stages of peatland opening and peat extraction are poorly understood. Major changes can occur to the hydrology and internal processes within a peatland during the transition to a drained, extracted, peat field that can influence the availability and export of nutrients. In addition to changes occurring within the harvested field, drainage ditch composition and beaver (*Castor canadensis*) dams can further modify the water quality exported to downstream ecosystems. The objective of this study was to investigate changes to the in-field processes and hydrological connectivity that may influence the outflow water quality during the understudied period when a peatland is opened and initially extracted in the continental Boreal Forest in Alberta, Canada. Water levels, volumetric flow, depth of ice, and aeration depth were measured along with water chemistry (electrical conductivity (EC), major ions, pH, and nutrients (dissolved and particulate nitrogen (N), phosphorus (P), and carbon (C))) in surface water and groundwater in natural and harvested peat fields, and at the outflows of the peatland complex. Sampling was initiated in the fall of 2018, prior to peatland disturbance, and continued monthly from March through October in 2019 and 2021, capturing the perimeter ditching, vegetation mulching, internal ditching, and peat extraction. *In situ* ion availability was measured in surface peat layers, alongside surface and below ground temperature, soil moisture, and peat aeration in 2021. Perimeter ditching had little impact on the peat field hydrology; however, ditching increased the catchment contributing area and magnitude of flow to the outflow and extended the flow duration throughout the year compared to the reference. Mulching the vegetation increased water levels, surface temperatures, and initial concentrations of total dissolved N and P in porewaters within the harvested peat field, although N and P concentrations were not sustained at peak levels. Internal ditching and ongoing extraction decreased water levels below 100 cm, but soil moisture remained high and aeration was shallow. Nutrient concentrations in the harvested field were elevated relative to the reference, likely due to decomposition. Internal ditches did not consistently mirror the harvested field; higher nutrient concentrations were common and water was sourced from precipitation, surface, and deep peat porewaters in variable proportions throughout the year. Compared to natural peatland areas, harvesting activities greatly decreased peat water storage capacity, encouraged ice formation, and increased spring runoff in a landscape dominated by summer runoff. Transport of high stores of nutrients in the harvested field and internal ditches was dependent on hydrological connectivity that varied both seasonally and interannually. Thus, nutrient concentrations were often poor indicators of mass discharge and leaching risk. When ditches reached underlying mineral sediments, EC, pH, and P concentrations in outflow waters differed drastically relative to reference and harvested peatland porewaters, and drainage ditches acted as a source of predominantly mineral sediments. Waters exiting the main outflow ditch into the downstream outflow swamp had higher P and TSS concentrations, but similar N and DOC concentrations relative to an adjacent reference outflow swamp. Increased flow encouraged beavers to establish a dam at the harvesting operation outflow. Downstream of the beaver dam, ammonium and dissolved P concentrations rose above peat field outflow levels, but sediment concentrations were reduced. Although internal processes increased nutrient availability in the harvested fields, this study shows that hydrological connectivity, ditch substrate, and presence of beaver are key players governing the water quality exported downstream.

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Chapter 1: General Introduction

1.1 OVERVIEW

1.1.1 Horticultural Peat Harvesting in the Boreal Forest

As horticultural peat harvesting continues to expand in Canada (ECCC, 2021), there is concern that the activities required to extract peat may increase the concentration of dissolved carbon and nutrients entering downstream waterways (Betis et al., 2020; Donahue et al., 2022; Kløve, 2001; Pschenyckyj et al., 2023; St-Hilaire et al., 2004). Natural peatlands are ubiquitous across the Canadian boreal forest. They are primary source areas for runoff to the larger landscape (Devito et al., 2017; Hokanson et al., 2020; Karlsen et al., 2016; Van Der Velde et al., 2013; Van Huizen et al., 2020), and thus, can provide receiving aquatic ecosystems with a reliable source of water containing dissolved organic carbon (DOC) (Strack et al., 2015; Waldron et al., 2009), inorganic and organic nitrogen (N) (Devito et al., 1999; Westbrook & Devito, 2004), phosphorus (P) (Plach et al., 2016), and other biologically significant major ions (Bourbonniere, 2009). As important players in the boreal forest landscape water connectivity, peatlands can be negatively impacted by land use disturbance. There are concerns that alterations to the availability and mobility of nutrients in the outflow water leaving peat harvesting sites may cause eutrophication of receiving lakes and streams (Conley et al., 2009; Kløve, 2001; Westbrook et al., 2006) and jeopardise water quality. Assessment of such impacts are complicated by spatially and temporally dependent cumulative effects that can occur over several decades as the harvested peatland transitions through several operational stages: opening the peatland in preparation for harvest, extracting the peat, and restoring the peatland after all merchantable peat has been removed.

Literature on the impact of peat harvesting on outflow nutrient concentrations and export remains equivocal. Many studies on drained and harvested peatlands have observed elevated concentrations of carbon (C), N, and P in peat pore and outflow waters (Betis et al., 2020; Clausen & Brooks, 1983; Edokpa et al., 2017; Haapalehto et al., 2014; Joensuu et al., 2002; Kløve, 2001; Marttila et al., 2018; Menberu et al., 2017; Moore, 1987; Munir et al., 2017; St-Hilaire et al., 2004). However, recent studies have found no discernible difference between nutrient concentrations or exports leaving peatlands that have been drained or where the surface peat has been extracted compared to natural reference peatlands (Harris et al., 2020; Macrae et al., 2013; Palviainen et al., 2022; Pschenyckyj et al., 2023). The variability in nutrient concentrations and export observed at peatland outflows across such a broad range of studies is likely influenced by pre-existing nutrient concentrations, and may vary significantly with hydrological connectivity, soil, vegetation, and weather patterns (Kreutzweiser et al., 2008).

Some variability in the reported impacts to water quality at harvested peatlands may reflect differences in the hydro-biogeochemical function of sites that may be in early, compared to late, stages of peat harvest. These stages include: 1) opening the peatland by digging ditches and removing the woody vegetation layer, followed by 2) physical extraction of dried surface peat which can occur over decades, and more recently 3) restoration of old extracted peat fields by re-flooding the site and establishing peatland vegetation (Landry & Rochefort, 2012; Rochefort et al., 2003). Variability may also reflect differences in the diversity of biogeoclimatic settings in which peat harvest activities occur. Hydro-biogeochemical processes and, thus, nutrient concentration and export, can vary with local and regional variation in the surface relief, depth of peat deposits, hydrologic pathways, seasonality and volume of flow, and the source and chemical makeup of the precipitation, surface water, groundwater, and outflow water chemistries at a given peatland (Buttle et al., 2005; Devito, Creed, Gan, et al., 2005; Price et al., 2005). For extrapolating research findings and interpreting potential impacts of peat harvesting activities, placing peat harvesting research in the context of the different stages of harvest operations and the biogeoclimatic location of the site is imperative.

1.1.2 Importance of Biogeoclimatic Setting

Peatlands and thus, current and potential peat harvesting operations, extend across boreal Canada in a wide band (Tarnocai et al., 2011), spanning different climates, bedrock and surficial geologies, soil textures, and topographies (Devito, Creed, Gan, et al., 2005; Stralberg et al., 2020). Regional variations in the impact of peat harvest operations on peatland and outflow biogeochemistry would be expected (Dahl et al., 2007; Price et al., 2005), and may help explain some contrast in findings noted in the previous section. However, more studies representing a range of biogeoclimatic settings are needed to understand the effect of peat harvesting, assist in interpreting the relationship with outflow nutrient concentrations and export, and highlight key differences in hydrology and nutrient transformations when predicting water quality.

Previous studies examining the impact of peat harvesting have largely been conducted in snowmelt-dominated, humid climates in the crystalline bedrock region of boreal Canada (Ketcheson et al., 2012; Moore, 1987; Price, 1997; Waddington et al., 2008; Waddington & Price, 2000) and Europe (Kløve, 2001; Kløve et al., 2010; Lepistö et al., 2006; Marttila et al., 2018; Sallantaus, 1992). Here, high runoff volumes and continuous hydrological connectivity between harvested peatland fields and outflows are expected. This contrasts with glaciated Boreal Plain regions that are characterized by a sub-humid climate, much lower runoff, and summer dominated flow (Brown et al., 2014; Dahl et al., 2007; Devito, Creed, & Fraser, 2005; Devito et al., 2017; Wind-Mulder & Vitt, 2000). When combined with low relief, heterogeneous glacial deposits, and complex groundwater-surface water interactions (Mulqueen, 1986; Price et al., 2005; Winter, 2001), the relationship between the impacted harvested peat field and the outflow hydro-biogeochemistry quickly become complicated within the Boreal Plain.

Although some research on the impact of peat harvesting operations conducted in sub-humid continental areas within Canada exist (Wind-Mulder et al., 1996; Wind-Mulder & Vitt, 2000), key information required to understand alterations to water quality is still lacking, and even less information is known about the impacts of opening a peatland and early peat extraction within the Boreal Plain. This is especially critical, as water quality best management practices are similar across the country, and it is currently unknown if they should account for these biogeoclimatic differences.

1.1.3 Importance of Accounting for Different Harvesting Stages

Major changes occur to the peatland hydrology and physicochemical properties that can have an effect on outflow water chemistry during the opening, extraction, and restoration stages within the overall peat harvesting operations. Opening and initial extraction of peat only represent a small window of time relative to the lifespan of an extracted peat field (10 – 40 + years) and the years following peatland restoration. When opened, the impact of initially digging a perimeter ditch, mulching and removal of surface vegetation, and installing internal drainage ditches within the peat field that drain into the perimeter ditch, may be substantial compared to later harvesting stages when hydro-geomorphic and biogeochemical processes have stabilized. These drastic alterations to the initial peatland homeostasis alter newly and previously available chemical constituents in the peat porewater within the peatland (Menberu et al., 2017) and outflow water (Holden et al., 2004; Prevost et al., 1999). Further, the initial site characteristics and the depth, substrate, and length of the perimeter and internal ditches within the field will define short- and long-term drainage. The opening activities can potentially set future trajectories of chemical cycling within peatlands, and geomorphological processes within the installed drainage networks, that influence outflow concentrations and export over both short and long-term (Nieminen, Sallantaus, et al., 2017) that may vary over time (Pschenyckyj et al., 2023).

However, the impact on water quality during opening and early extraction is a crucial knowledge gap that may have significant implications downstream. Very few peer-reviewed studies exist describing the initial changes to the water chemistry at newly drained peatlands and, of these, results are inconsistent. Prevost et al. (1999) observed increased nutrient concentrations in surface water at a forested, drained, peatland that did not return to pre-ditching levels five years post-ditching. In contrast, Moore (1987) observed increased ammonium and P concentrations at drained, newly extracted peatlands in Quebec, Canada; however, the impact was very short lived and concentrations returned to non-harvested levels after approximately one week. Thus, there is a need to place peat harvesting research in the context of harvesting stage (opening, extraction, restoration). Additionally, biogeoclimatic location is imperative to highlight key differences in hydrology, nutrient transformation, and nutrient exports between studies. This may help predict water quality outcomes when a peatland is opened and subsequently extracted, especially in an area as biogeoclimatically diverse as the circumboreal forest within Canada.

1.2 PROJECT CONTEXT, OBJECTIVES, AND THESIS STRUCTURE

This thesis is part of a much larger project on the impact of horticultural peat harvesting on water quality within Canada (Figure 1.1). The larger project aims to address the impact on water quality at peatlands in both continental (Alberta) and maritime (New Brunswick) regions over the three stages of peat harvest operations. The focus of the overall project is on investigating changes when natural peatlands are prepared for harvesting, during early and late active extraction periods, and following restoration after harvesting is complete.

This MSc. study addresses a key part of the overall project by reporting on changes to the in-field processes and hydrological connectivity that may influence the outflow water quality during the understudied period when a peatland is opened and initially extracted in the continental Boreal Forest in Alberta, Canada (Figure 1.1, highlighted star). This MSc. study aims to better understand the potential drivers altering water quality by assessing if nutrient concentrations and exports in outflow water are driven largely by (1) changes to peatland internal processes, or (2) alterations to water availability and hydrological flow path.

This thesis is separated into four main chapters. The current chapter (Chapter 1) serves to provide a general synthesis of the existing literature and develops conceptual models that hypothesize the potential impacts of peat harvesting operations on peatland physicochemistry, hydrology, and biogeochemistry. It also gives a detailed overview of the study area and describes the peat harvesting chronology for the study.

Chapter 2 focuses on the first study objective by comparing the alterations that occurred within the harvested peatland as it was opened and extracted, relative to a natural reference peatland. To infer potential changes to, and primary drivers of, nutrient availability, the physicochemical parameters, hydrology, and water quality within the harvested peatland were assessed prior to disturbance, following the installation of the perimeter ditch and vegetation mulching, after internal ditch installation, and during peat extraction. Findings were compared with an adjacent reference peatland. Additionally, the water quality within the peat field was compared with the water in the internal ditches within the harvested field to begin to understand if the main flow path from the peat field to the internal ditch was via surface and shallow flow, or deep flow.

Chapter 3 focuses on the second study objective by examining how hydrological flow paths are integrated with the controls on nutrient availably in both the undisturbed and harvested peatland areas discussed in Chapter 2, with a focus on outflow water quality. The potential for changes to nutrient concentrations and export rates at the outflow following extraction activities was explored, and the question of whether the water quality at the harvested outflow swamp was an expression of (a) alterations that occurred within the harvested field, (b) a result of different source waters intersected by ditching, or (c) external processes within the outflow channels was addressed. To explore this question, water quality was assessed leaving the harvested field, along the perimeter ditch, and at the end of the outflow channel. This was compared with water collected pre-harvest, as well as with the water observed in the harvested peat field and reference peat field from Chapter 2. Additionally, the influence of beaver dam construction that occurred during the study at the harvested outflow swamp was investigated by following water chemistry above and below the beaver dam following dam construction. Finally, the overall influence of peat extraction at a regional scale was assessed by comparing stream flow, water quality, and nutrient concentrations and export rates of water downstream of the harvested field operations with water downstream of a nearby unharvested outflow swamp.

Chapter 4 gives a summary of the key findings from this study and outlines the research applications. Future research and study limitations are also addressed.

1.3 LITERATURE REVIEW AND CONCEPTUAL MODEL

In order to predict the impacts of early peat extraction on downstream water quality, a conceptual model of the changes to the internal processes within the peatland was developed (Figure 1.2) that hypothesizes the alterations to the hydrology, physicochemical processes, and biogeochemical processes that my occur during peat harvesting stages, including: (1) the period of perimeter ditching, vegetation removal, internal ditch installation, and (2) active peat extraction. Many peat harvesting operations have a range of harvesting stages operational at a given time, resulting in different spatial and temporal scales that may impact the water quality downstream. Further, other site-specific differences, such as the seasonal and interannual precipitation variability, the ditch substrate material, and the presence or absence of beaver dam establishment at the outflow must be addressed in order to understand the quality of water leaving a given harvested peatland. Here, the existing literature is discussed and key knowledge gaps are identified in determining in-field and outflow water quality, with a focus on the transition from a natural peatland to an extracted peat field as summarized in Figure 1.2.

1.3.1 Alterations to Peatland Internal Processes

1.3.1.1 Perimeter Ditching

During the initial stages of peat harvesting, a perimeter ditch is dug around the area slated for harvest, the existing surface vegetation is mulched, and vegetative reestablishment is prevented over the life of the peat extraction field (Landry & Rochefort, 2012; Figure 1.3). This results in complex changes to the in-field internal processes that govern nutrient availability and mobility. Installing the perimeter ditch lowers the water table and re-directs water flow to a centralized outlet. At this stage, only minimal changes in porewater nutrient availability within the harvested field may occur because the perimeter ditch will have little effect on the water table at distances greater than 30 m (Landry & Rochefort, 2012; Prevost et al., 1997). Without any major alterations to the peatland water level beyond 30 m, it is predicted that it is unlikely that the overall chemistry within the harvested field will be affected by ditching and water table alterations.

1.3.1.2 Vegetation Removal

In contrast, permanently removing forest vegetation could result in major changes to the peatland ecosystem. Vegetation removal has been shown to increase the peat surface temperature by inhibiting shade, and raise the water table by decreasing evapotranspiration (Sarkkola et al., 2010; Walbridge & Lockaby, 1994). Elevated temperatures have been shown to encourage decomposition (Holden, Chapman, et al., 2006; Mulqueen, 1986; Williams & Crawford, 1983), and the flooding that follows woody vegetation removal may boost the soluble reactive phosphorus (SRP) concentrations released via cell lysis (Reddy et al., 2005) or, by iron (Fe) hydroxide reduction as anaerobic conditions persist (Aldous et al., 2007; Carlyle & Hill, 2001; Niedermeier & Robinson, 2009). The freshly mulched material may be high in labile nutrients (Hyvönen et al., 2000; Tolvanen et al., 2020); yet, because new vegetation is not allowed to establish, the removal of nutrients via root and rhizoid uptake will be greatly reduced (De Mars et al., 1996; Jabłońska et al., 2021; Walbridge & Lockaby, 1994), increasing porewater nutrient concentrations and the potential for nutrient leaching.

Alternatively, internal mechanisms may be active during the early stages of peatland opening that slow the rate of nutrient availability and reduce the potential for export from peat fields. The higher water table present after mulching and before the internal drains are installed could maintain anoxic conditions. This may result in no observable alteration to the peat porewater nutrient concentrations following the initial peatland opening because poor aeration has been shown to significantly reduce decomposition (Holden et al., 2004).

The lack of actively growing vegetation on extracted peat fields may also slow decomposition. Extensive studies on drained peatlands converted to forest plantations (Lepistö et al., 2006; Nieminen, 2004; Nieminen, Sarkkola, et al., 2017; Palviainen et al., 2022) or agricultural land (Hemond & Benoit, 1988; Łachacz et al., 2023; Liu et al., 2023; Yli-Halla et al., 2022) have shown increases in N and P concentrations in outflow water chemistry. However, unlike in extracted peat fields, living vegetation remains or is planted in the drained peat, and fertilizers are often added to enhance vegetative growth. By initially removing the peatland vegetation and preventing regrowth, fresh, highly labile organic matter is no longer deposited by living vegetation while the peat field is operational (De Mars et al., 1996; Jabłońska et al., 2021), and exudates from live roots are no longer secreted that encourage microbes to rapidly break down the peat (Mastný et al., 2021). The absence of these mechanisms may alternatively slow decomposition and could limit nutrient availability in peat porewater during the first few years of extraction.

1.3.1.3 Internal Ditching and Early Peat Extraction

When the internal ditches are installed within the opened field, and the water table begins to lower in earnest by draining into the perimeter ditch, internal processes occurring within the peat field likely change (Figure 1.2; Figure 1.3). Previous studies have shown that when the water table is lowered, the ground and surface water flow paths become altered, the peat bulk density increases, and the depth of aeration from the peat surface deepens over time (Minkkinen & Laine, 1998; Price et al., 2003; Van Seters & Price, 2002; Waddington & Price, 2000). Increased nutrient availability, elevated temperatures, and aerobic conditions may encourage decomposition (Holden, Chapman, et al., 2006; Mulqueen, 1986; Williams & Crawford, 1983) and increase nutrient availability in peat porewater (Munir et al., 2017), especially near drainage ditches (Tolvanen et al., 2020) would be predicted. As decomposition accelerates, mineralization and nitrification increase, and DOC, and available N and P have been shown to leach from the peatland via subsurface drainage (Holden, Chapman, et al., 2006; Marttila et al., 2018; Nieminen, Sallantaus, et al., 2017; Waddington et al., 2008; Wells & Williams, 1996). Aerobic peat soils found in extracted and drained peatlands have been associated with increased N and P turnover (Bridgham et al., 1998; Croft et al., 2001; Holden et al., 2004; Munir et al., 2017), and the flush of nutrients provided by the newly mulched vegetation, lack of root uptake, and oxygen-rich conditions initiated by drainage, could account for the reported increases in nutrients following peatland harvesting.

However, aerobic conditions within the peat field profile may not be as complete as previously conceptualized. Even once the internal drains were installed and the water table was lowered, Price (1997) found that the surface soil moisture was maintained in a cutover bog. Therefore, anoxic conditions may persist above the water table in much of the extracted field even after intensive drainage, which could discourage widespread decomposition deep into the peat profile. In areas where the peat becomes aerobic, P can be precipitated as Fe and aluminum (Al) phosphate and adsorbed onto organic matter, reducing its mobility (Carlyle & Hill, 2001; De Mars et al., 1996; Sah & Mikkelsen, 1986). Soils high in organic matter and reducible Fe that were flooded, then drained, have been shown to adsorb high concentrations of P (Sah & Mikkelsen, 1989), rendering it immobile. Therefore, the initial peatland chemistry and the peat aeration status should be monitored when the peatland is being opened and during the first few years of extraction. This may provide valuable insights into the environmental conditions present and inform the available nutrient status.

Despite seemingly improved environmental conditions for microbial activity, extracted peatlands have been associated with decreased microbial diversity and biomass (Andersen et al., 2013; Croft et al., 2001). The peat desired for horticultural peat extraction is initially very poorly decomposed, has a high carbon to nitrogen ratio (C:N), low pH, and contains phenols that slow decomposition (Moore & Basiliko, 2006; Waddington et al., 2015). Studies on newly opened, young peat harvesting operations are limited, yet, the time since extraction began and, consequently, the quality of the peat being extracted, may have major impacts on the availability and mobility of nutrients transported to the outflow. Although drainage may increase oxygen availability and temperature (Aaltonen et al., 2021; Prevost et al., 1999), other key factors controlling decomposition rates, such as pH and substrate quality (Holden, Chapman, et al., 2006; Holden et al., 2004) may continue to hinder decomposition. This could result in little to no detectable change in nutrient concentrations at harvested and reference locations during the first few years of extraction.

1.3.1.4 Peat Availability and Extracted Field Age

In order for mobile nutrients to be leached from peat into downstream waterways, peat substrate that is capable of leaching nutrients must be physically present. Vacuum peat harvesting extracts peat in thin layers over decades (Landry & Rochefort, 2012); thus, the nutrient-laden peat exposed to prime decomposition conditions at the ground surface is continuously removed and stockpiled elsewhere to be processed, packaged, and sold. Therefore, the *in situ* exposed peat may not be subjected to decomposition conditions for a long enough time period to leach nutrients out of the peat field during active operations. Instead, new, low-quality peat is continuously uncovered at the ground surface, that must re-start the mineralization process. The continuous removal of nutrient-rich vegetation grown in riparian areas adjacent to streams (Jabłońska et al., 2021) and on drained peat (De Mars et al., 1996) has been shown to decrease the availability of leachable nutrients in other systems. By removing the surface layer of peat, a similar phenomenon may be occurring that removes the layer of peat rich and mobile nutrients before they can be leached into ditches. This could potentially explain some of the variability in nutrient exports observed in previous studies.

Understanding the physicochemical conditions present at opened and newly extracted peatlands is especially critical since peat harvesting areas often have an assortment of extraction fields of variable age and peat substrate quality. However, most studies on the effect of peat harvesting on water quality examine older sites that have been undergoing extraction for extended periods of time or, are awaiting restoration (Haapalehto et al., 2014; Wind-Mulder et al., 1996; Wind-Mulder & Vitt, 2000). Here, the peat has been exposed to prolonged conditions favoring decomposition, and is comprised of minerotrophic, higher quality substrate from lower in the peat profile (Graf et al., 2008; Nieminen, Sallantaus, et al., 2017; Wind-Mulder et al., 1996; Wind-Mulder & Vitt, 2000) that likely accounts for the large volume of literature reporting increased nutrient concentrations. An understanding of how opened and newly extracted peatlands contribute to nutrient exports is needed to begin to understand the total impact of peat harvesting, inclusive of all stages of the operation. This is especially important since integration between areas and cumulative influences from different water sources may complicate the interpretation of downstream water quality.

1.3.2 Alterations to Hydrology

1.3.2.1 Hydrological Connectivity

The mechanisms governing peatland discharge in extracted peatlands are poorly understood; yet discharge volume and chemistry have a direct impact on downstream water quality. Ditching has been shown to increase (Prevost et al., 1999), have no effect on (Åström et al., 2001), and decrease (Holden, Evans, et al., 2006) runoff and base flow volumes. Holden et al. (2004) tabulated numerous studies reporting alterations to peatland hydrology following drainage; however, they noted an overall lack of hydrological and hydrochemical process-based measurement. Similarity, Shotyk (1986) noted a need for peatland hydrological description prior to, during, and after peat harvesting activities in future studies. Understanding how different flow paths may vary over space and time during early peat extraction begins to address this critical knowledge gap, and is needed to assess whether changes to the water quality within the peat field are reflected at the outflow.

The overall influence of peat harvesting on outflow stream concentrations and export ultimately dependent on the type, seasonality, and magnitude of the hydrologic flow paths in connection with the harvested peat field nutrient dynamics. When hydrologically connected, ditches allow for a direct pathway between the surface peat, deeper peat pore water chemistry, and the outflow, which can alter the chemical makeup of outflow waters by enabling nutrient-rich water, previously in contact with the harvested peat field, to be exported off site (Marttila et al., 2018; Nieminen, Sallantaus, et al., 2017). Peatland water entering ditches can move in a variety of ways, including infiltration-excess overland flow, saturation-excess overland flow, rapid acrotelm flow, and pipe flow (Evans et al., 1999; Holden & Burt, 2003) (Figure 1.2.C). If deep enough, ditches may also intersect groundwater, potentially creating a direct connection between the harvested outflow and groundwaters of various scales and chemistries (Marttila et al., 2018; Pschenyckyj et al., 2023; Tóth, 1999; Winter et al., 2003). However, little study has been devoted to understanding the origin of ditch water at vacuum harvested peatlands, especially during the early stages of extraction.

1.3.2.2 Peat Water Storage

Determining the dominant flow path is likely controlled by the availability of water and the peat storage capacity. Knowing how the water got to a ditch, whether by running directly over the peat field surface or via subsurface flow, is important because it can have a major impact on the water chemistry (Holden & Burt, 2003). Studies completed in areas with sufficient precipitation to frequently return saturation to the near surface have concluded that runoff from natural peatlands is often generated by saturation-excess overland flow (Evans et al., 1999; Holden & Burt, 2003). In these instances, the peat water storage capacity is very low and easily overcome. Evans *et al.* (1999) observed quick surface and near-surface runoff when water tables were within 5 cm of the surface at a blanket bog in the United Kingdom; however, runoff was limited when the water table was lowered during drought and peat storage exceeded precipitation. In peat harvesting operations, conditions for surface overland flow are most likely to occur following vegetation removal and before the internal drains are installed because the water table will be near the ground surface (Komulainen et al., 1999; Walbridge & Lockaby, 1994) and the acrotelm layer of peat is still present. In contrast, once extensive ditching commences, water tables near the ground surface that allow for near-surface saturated conditions are rarely present during initial stages of extraction.

Internal ditching lowers the water table in the peat field, increasing the available water storage area in the peat (Evans et al., 1999; Marttila & Kløve, 2010). The extent to which the water level drops during peatland drainage depends on the hydraulic conductivity of the peat and the subsequent ditch spacing (Boelter, 1972; Holden et al., 2004; Prevost et al., 1997; Rothwell et al., 1996), as well as the sub-peat geohydrological conditions (Mulqueen, 1986). Once the water table deepens, water will take longer to move vertically down the peat profile and will come in prolonged contact with peat of various levels of decomposition, altering its chemistry (McCarter et al., 2020). Due to the reduced ability for saturation-excess overland flow to occur, water observed in the internal ditches may originate from deep in the peat profile or could be connected to peatland groundwater. Haapalehto et al. (2014) observed higher DOC concentrations and evidence of minerogenic water influence in peat ditches compared to the peat field porewater in a drained peatland in Finland. Price (1996) observed lateral drainage from the peat field of a partially restored cutover bog into the ditch when conditions were wet; however, the flow reversed and water moved from the ditch back into the peat field during dry conditions. This may further increase, or initiate, the base flow water volume and alter the magnitude and seasonality of outflow chemistry. Prevost et al. (1999) saw increased base flows that were sustained throughout the study period in a drained peatland in Quebec, Canada. The increased concentration of nutrients observed in previous studies could, therefore, be the due to higher volumes of water channeled towards the outflow that were in contact with deep peat containing mobile nutrients that would otherwise not be exported under natural conditions.

Alternatively, ditching may deplete storage, decreasing the magnitude and frequency of base flow in favor of rapid responses to storm events (Holden, Evans, et al., 2006). Higher bulk density and reduced hydraulic conductivity has been associated with peat compression and decomposition following drainage (McCarter et al., 2020). This could lessen the amount of water required to return the water table to the ground surface and result in rapid responses to precipitation events. Price (1996) found that the peat at a harvested, cutover bog had substantially lower specific yield than the natural reference peatland. Thus, as the saturated hydraulic conductivity within the peat field diminishes, the available storage capacity of the peat is reduced, making the field more susceptible to flashy hydrological responses. Further, dried peat can become hydrophobic (Evans et al., 1999), increasing the chance of infiltration-excess overland flow to occur during intense rainfall events (Hayashi, 2013).

1.3.2.3 Precipitation Variability

Peat saturation and moisture content that govern the type of flow path, as discussed in the previous section, are also highly dependant on local site conditions, including the amount of precipitation (Holden, Chapman, et al., 2006; Mulqueen, 1986), as well as the overarching biogeoclimatic setting. Peatland field studies that occur over multiple years and are subject to seasonal and interannual variability, often observe alterations in volume of flow as a result of non-uniform precipitation (Bay, 1969; Evans et al., 1999; Stenberg et al., 2018). Yet, the variability in precipitation and associated degree of peat saturation is rarely considered when determining the impact of peat harvesting on water quality at a given harvesting location. More importantly, different biogeoclimatic settings will have distinct seasonal or inter-annual weather patterns that alter the type (either snow or rain), volume, and timing of precipitation, and may seriously alter the hydrological connectivity between the peat field and the outflow (Price et al., 2005), especially once drainage is initiated. Therefore, an understanding of the range of potential flow paths through a peatland, and their subsequent effect on water chemistry, is critical, and will likely need to be addressed differently depending on the peatland biogeoclimatic setting (Devito, Creed, Gan, et al., 2005; Winter, 2001).

In biogeoclimatic settings with drier climates and lower runoff, such as in the Canadian Boreal Plains (Devito, Creed, & Fraser, 2005; Devito et al., 2017, 2023), low precipitation inputs may hydrologically isolate the peat field from the outflow for portions of the year and result in variable water chemistry. Variable nutrient concentrations between wet and dry periods have been observed in peatlands across the boreal (Burd et al., 2018; Howson et al., 2021; Muller & Tankéré-Muller, 2012; Vitt et al., 1995); however, there are limited studies on the effect of ditch dryness on water discharge volume and chemistry. Dry ditches have been reported at a drained peatland used for agriculture in Indonesia during the dry season (Putra et al., 2021), suggesting that this phenomenon may occur routinely during water limited portions of the year, but has been under reported. If the internal ditches within the extracted field occasionally dry up, the peat field would be disconnected from the outflow and the continuous transmission of peat field water impacted by harvesting activities would cease until sufficient water is available to transport nutrients. This could result in non-uniform transmission of nutrients to the outflow that demonstrate a wide range of nutrient concentrations that vary throughout the year. Marttila and Kløve (2010) noted that when ditches were dry, the discharge and suspended solid concentrations generated from a rainstorm were low; however, a second rain event of similar size that occurred while ditches were moist resulted in higher suspended solid exports. This may require flexible mitigation strategies that consider the seasonal variation in magnitude of flow when monitoring the quality of water leaving peat harvesting operations.

1.3.2.4 Ice Dynamics

In biogeoclimatic settings where peatlands routinely freeze during the winter months, seasonal frozen ground has been shown to act as an impermeable layer that enables infiltration-excess overland flow from melting snow, greatly increasing runoff and the export of accumulated nutrients in the spring (Brown et al., 2010; Hayashi, 2013; Price, 1987; Van Huizen et al., 2020). The peat field is milled to uniformly dry the surface and expedite vacuum harvesting, so it may behave similarly to a cultivated agricultural field during the winter months. Van der Kamp et al. (2003) observed that cultivated soils, compared to grassland soils, had less infiltration and contributed more runoff, likely because of a lack of macro pores that result in a continuous ice layer. In addition, drained peatlands have been shown to have colder winter soil temperatures (Prevost et al., 1997) and stay frozen longer than their natural counterparts (Lieffers & Rothwell, 1987). Therefore, harvested peatlands may have thick, uniform ice lenses that persist for longer into the spring and summer months. This not only extends periods of overland flow, but extended freeze-thaw cycles have been shown to accelerate decomposition and mineralization of leachable nutrients found in soil (Campbell et al., 2005; Hayashi, 2013; Joseph & Henry, 2008). Thus, frozen ground-induced overland flow could increase the potential for elevated pulses of nutrient concentrations to be transported to the outflow.

1.3.3 Influence of Drainage Ditch Material

Few studies that determine the downstream impact of peat harvesting have addressed the ditch substrate composition when discussing outflow water chemistry. The climate, relief, and geologic setting will influence the depth, decomposition, and shape of the peat deposit (Ivanov, 1981; Packalen et al., 2016). This effects what substrate the ditch is cut into, as well as the depth and gradient required to maintain suitable drainage during extraction (Boelter, 1972; Mulqueen, 1986). Low relief areas require ditching into deep, minerotrophic peat or the underlying sub-peat, mineral sediments (Joensuu et al., 1999, 2002; Nieminen, Piirainen, et al., 2018). Deeper peat deposits have been associated with higher electrical conductivity (EC), pH, and nutrient concentrations (Pschenyckyj et al., 2023; Wind-Mulder et al., 1996), while mineral sediment chemical composition will vary based on the local surficial geology (Figure 1.2.C).

Surface and deeper peat waters entering the ditch interact with the ditch substrate, which can alter the chemical composition of the water. Åström et al. (2001) observed different hydrochemistry downstream of a drained forest with ditches cut into the underlying mineral till compared to a non-ditched reference, and attributed the changes to interactions with the mineral ditch and alterations to the hydrological flow path. Joensuu et al. (2002) observed increased pH, EC, suspended solids, and ammonium as a result of ditch maintenance in Finnish peatlands. Reynolds and Hughes (1989) observed elevated Al concentrations associated with the mineral soil in a forest drainage ditch, and Pschenyckyj et al. (2023) attributed higher pH, EC, and calcium (Ca) stream water concentrations to contact with the regional groundwater and sub-peat parent material as a result of peatland drainage. This suggests that the ditch substrate composition can have a significant influence on the water chemistry, and may drastically alter the final water quality leaving a site.

Ditch construction and maintenance exposes fresh erosive surfaces, and changes the magnitude and type of sediments exported downstream, either organic or mineral, as they are cut deeper into the ground over time (Joensuu et al., 1999; Nieminen, Piirainen, et al., 2018; Tuukkanen et al., 2017). Elevated suspended sediment concentrations have been observed during periods of high discharge, such as during spring melt and following summer storms (Marttila & Kløve, 2010). In addition, ditches cut into underlying mineral sediments have been shown to generate higher suspended sediment loads compared to ditches cut into deep peat (Stenberg et al., 2015). Therefore, accounting for the ditch substrate may assist in accurately predicting the chemical composition of outflow waters, especially as ditches are cut deeper over time, and are subjected to fluctuations in the magnitude of water flow.

1.3.4 Influence of Beavers (*Castor canadensis*)

Beavers (*Castor canadensis*) are ubiquitous across boreal Canada and can often be found in low gradient areas with flowing water (Touihri et al., 2018), such as peatland drainage ditches (Kalvīte et al., 2021). They readily mobilize to construct extensive dams, ditches, and lodges in response to changes in water flow magnitude and seasonality (Rosell et al., 2005). Beaver activity at constructed outflow locations alters stream residence time, hydraulic head, and hyporheic interactions that influence the magnitude and seasonality of stream nutrient transformation, concentration, and export (Devito et al., 1989; Devito & Dillon, 1993; Klotz, 1998; Rosell et al., 2005). They further influence the outflow water chemistry by retaining N and P during low flow events, as well as exporting these nutrients during high flow (Devito et al., 1989; Devito & Dillon, 1993). Considering the extent to which beavers can alter the water chemistry of a given system, and their range across boreal Canada, it is necessary to examine the background influence of beaver activity when predicting the short and long-term effects of opening a peatland and harvesting peat on the chemistry of receiving waters. This is also applicable to places in Europe where beaver re-introduction efforts are ongoing (Brazier et al., 2021).

1.3.5 Knowledge Gaps

Based on the available research, there are several key knowledge gaps when assessing the full scale of impact to downstream ecosystems at newly opened horticultural peat harvesting locations. Data on newly opened peatlands is very limited, especially in sub-humid, continental climates. Research is needed to assess how the perimeter ditching, vegetation removal, internal ditching, and first few years of peat extraction influence the in-field nutrient availability and subsequent outflow water quality. To do this, data on changes to the internal processes operating within the peatland are required in order to help pinpoint the mechanisms guiding nutrient availability (Figure 1.2). These include measurements of temperature, anoxia, pH, C:N, and bulk density within impacted and reference peatlands in order to gauge the likelihood of decomposition, which could lead to nutrient leaching downstream. Additional mechanisms that may control nutrient availability, such as vegetation uptake, removal of mineralized surface peat via vacuum harvester, and the time since extraction commenced should be addressed.

Knowing whether nutrients are present and have the potential to move from the peat field, into a ditch, and off site at newly extracted peatlands is important to understand when planning for water quality management. Hydrological alterations to the peatland following opening and early extraction need to be understood within the context of the peatland biogeoclimatic setting (Figure 1.2). Alterations to the hydrological flow paths could allow for water from different sources to be expressed at the outflow, but this has not been addressed in opened and newly extracted peatlands. Further, differences in precipitation and ice dynamics need to be assessed in order to understand the seasonal and interannual variability in hydrological connectivity between the harvested field and the outflow.

The type of ditch substrate material in which the water is flowing could have huge consequences for the water quality at the outflow. The influence of ditches on water quality is substantially underreported, yet almost all operations involving peatlands, be it related to agriculture, forestry, or horticultural peat harvesting, require ditches. This understanding is required in order to accurately predict the impact to downstream ecosystems, especially considering the range in geologic material with which the ditch may come into contact. Finally, beavers are ecosystem engineers that rapidly inhabit areas with flowing water across Canada. Yet, their manipulations to the outflow water quality leaving peat harvesting locations has not been studied. Considering beavers are being re-introduced across Europe (Brazier et al., 2021), an understanding of their potential influence on the outflow water quality is of interest.

1.4 STUDY AREA

1.4.1 Biogeoclimatic Setting

The study was conducted on a small, forested peatland that was slated for commercial horticultural peat harvesting near Avenir, Alberta (54.992349°, -112.412725°) approximately 210 km northeast of Edmonton, Alberta, Canada (Figure 1.3). In the Canadian national-scale ecological framework, the study site is situated within the Boreal Plains Ecozone and sits on the border between the Mid-Boreal Uplands Ecoregion and the Boreal Transition Ecoregion (Ecological Stratification Working Group, 1996) (Figure 1.3). Ecological units are further divided on a provincial-scale, thus, in Alberta, the peatland in this study is found within the Central Mixedwood Natural Subregion of the Boreal Forest Natural Region (Natural Regions Committee, 2006) (Figure 1.3).

This region has a sub-humid, continental climate, and is characterized by cold winters and warm summers with a long-term mean annual temperature of 0.2 °C (Devito, Creed, & Fraser, 2005; Natural Regions Committee, 2006). The mean annual potential evapotranspiration (Thornthwaite 520 mm) (Devito et al., 2012, 2017; Donnelly et al., 2016) exceeds the mean annual precipitation (478 mm) in most years (Natural Regions Committee, 2006). Of the total annual precipitation, 70 % arrives during the growing season between April and August (Natural Regions Committee, 2006), largely as higher intensity, short-duration, convective rain storms (Buttle et al., 2005; Devito, Creed, & Fraser, 2005). Precipitation can vary substantially between years and decades, resulting in wet and dry patterns that strongly influence water availability (Buttle et al., 2005; Devito et al., 2023). During winter, snow depths are shallow due to low precipitation and sublimation (Devito, Creed, & Fraser, 2005; Ferone & Devito, 2004; Ireson et al., 2015); frost depths in organic soils are extensive from November though to June (Brown et al., 2010; Petrone et al., 2008; Van Huizen et al., 2020). In the Boreal Plains Ecozone, the long-term annual runoff is generally less than 100 mm yr^{-1} and varies annually from < 5 mm yr^{-1} to 300 mm yr^{-1} (Devito, Creed, & Fraser, 2005; Devito et al., 2017; Donnelly et al., 2016). Runoff depends on the soil water storage availability, which is determined by spatial differences in surficial geology, and seasonal and interannual weather variability (Devito, Creed, & Fraser, 2005; Devito et al., 2017, 2023).

The Central Mixedwood Natural Subregion is characterized by level to gently rolling plains containing expanses of forests, lakes, peatlands, mineral wetlands, and hummocky uplands (Natural Regions Committee, 2006). The natural subregion bedrock is dominated by Cretaceous shales, sandstones, and siltstones; and Devonian limestones, shales, and siltstones (Natural Regions Committee, 2006). Quaternary deposits on top of the bedrock range from 50 to 100 m thick, with massive tills that form aquitards where fine-textured deposits are present (Fenton et al., 1994). The surficial geology is predominantly moraine and glaciolacustrine in origin, with lesser proportions of glaciofluvial deposits (Natural Regions Committee, 2006). Organic deposits cover 40 % of the Central Mixedwood Natural Subregion, but can be as high as 80 % in low relief lacustrine areas, and as low as 20 % in areas with hilly terrain (Natural Regions Committee, 2006). The soils are approximately 60 % mineral, mainly Grey Luvisols (35 %), and 40 % organic, dominated by Organic Mesisols (Natural Regions Committee, 2006). The vegetation in the upland is a mix of aspen (*Populus tremuloides*) and white spruce (*Picea glauca*) forests, while the poorly drained wet areas are predominantly fens and bogs with black spruce (*Picea mariana*) and peat mosses (Natural Regions Committee, 2006).

1.4.2 Local Site Description

The main peatland study area is a continental treed bog to poor-fen peatland (Alberta Environment and Sustainable Resource Development, 2015) that developed on a high elevation plateau between the Athabasca River and Lac La Biche, Alberta. Prior to disturbance, the ~100 ha peatland had a shallow dome that shed surface water towards two, unchanneled swamps (Alberta Environment and Sustainable Resource Development, 2015) with surface flows that drained to the northwest and southeast (Figure 1.3). The focus of this study is on the southeast area of the peatland because the constructed drainage from the peat harvesting activities was designed to exit at this location. The area within the peatland where surface flow is not obvious is referred to as the peatland “field”, and the area where the water flows out of the peatland as surface flow is referred to as the “outflow” in this study.

1.4.2.1 Climate

The thirty-year climate normal (1991 – 2020), measured at the Atmore meteorological station (approximately 35 km southwest from the study area), had a mean annual temperature of 1.9 °C, with monthly ranges from -13.9 °C to 16.4 °C in January and July, respectively (Alberta Climate Information Services, n.d.). The mean annual precipitation (412 mm) is non-uniformly distributed over the year, with, on average, 80 mm (20 %) falling during winter (November 1 – March 31), 66 mm (16 %) falling during spring (April 1 – May 31), 211 mm (51 %) falling during summer (June 1 – August 31), and 55 mm (13 %)

falling during autumn (September 1 – October 31) (Alberta Climate Information Services, n.d.). Runoff values at rivers near the study area vary greatly. The long-term runoff measured at Logan River (~ 50 km northeast) from 1986 – 2010 was 96 mm yr^{-1}, while the runoff at the Amisk River (~ 65 km southeast) was only 5 mm yr^{-1} (Devito et al., 2017; Environment Canada, n.d.). Both river catchments are within the Central Mixedwood Natural Subregion in Alberta, contain large peatland areas, and represent catchments with either high groundwater inputs (Logan River) or high surface water inputs (Amisk River) (Devito et al., 2023).

Investigation and sampling occurred from 2018 – 2021, with limited sampling in 2020 due to the COVID-19 global pandemic. During this study, major fluctuations in precipitation and temperature were observed (Figure 1.4). During the 2018 hydrologic year (November 1, 2017 – October 31, 2018), the mean annual temperature was 0.9 °C, the mean precipitation was 369 mm, and Penman-Monteith estimate of evapotranspiration (PET) using grass as a reference surface was 659 mm (Alberta Climate Information Services, n.d.). During the 2019 and 2020 hydrologic years, the mean annual temperatures were slightly below the 30-year average (1.2 °C and 1.4 °C, respectively), and the mean precipitation was above average (488 mm and 511 mm, respectively), with PETs of 634 mm and 644 mm, respectively (Alberta Climate Information Services, n.d.). The 2021 hydrologic year had an above average mean annual temperature (2.8 °C), below average precipitation (283 mm), and a high PET (728 mm). Runoff in the nearby rivers also fluctuated over the study (Figure 1.5). The runoff at Amisk River during the 2018, 2019, 2020, and 2021 years was 24 mm yr^{-1}, 25 mm yr^{-1}, 78 mm yr^{-1}, and 9 mm yr^{-1} respectively (Environment Canada, n.d.). The runoff at Logan River during the 2018, 2019, 2020, and 2021 years was 84 mm yr^{-1}, 96 mm yr^{-1}, 192 mm yr^{-1}, and 72 mm yr^{-1} respectively (Environment Canada, n.d.).

1.4.2.2 Geology and Soils

The bedrock geology within the study area is part of the Lea Park Formation which is characterized by medium to dark grey mudstone, with fine-grained siltstone and sandstone, bentonite clay, concretions of siderite, and veins of calcite, and is part of the Upper Cretaceous Belly River Group (Prior et al., 2013). It is overlain by at least 50 m of geologically recent Quaternary deposits (Fenton et al., 1994), thus, the bedrock is too deep below the ground surface to influence the local hydrology and water chemistry. In peatland areas, the surficial geology is dominated by Holocene organic deposits that are underlain by poorly drained, fine-textured glaciolacustrine sediments (Fenton et al., 2013). A patchwork of Pleistocene glaciolacustrine and moraine deposits are interspersed with the organic deposits (Fenton et al., 2013), and are often topped with mineral wetlands and aspen forest, respectively. The glaciolacustrine deposits are comprised of fine sand, silt, and clay, with some areas containing silty sand, pebbly sand, and minor gravel (Fenton et al., 2013). The moraine deposits are comprised of diamicton (till) made up of a mixture of sand, silt, clay, minor pebbles, cobbles, and boulders (Fenton et al., 2013).

The organic soils within the main peatland field are Typic Mesisols with peat deposits ranging from 0.75 m to 2.65 m deep, underlain by heavy clay rich in carbonates (CAESA, 1998). At the peatland outflow swamp, the soils range from Typic Mesisols in areas with deeper, less decomposed organic material, to Orthic Humic Gleysols containing shallower mesic or humic organic layers. Parent material ranges from clay to clay-loam and is rich in carbonates. The mineral soils in the surrounding non-wetland forest are Orthic Grey Luvisols underlain with carbonate rich heavy clay, that tend towards Gleysols near the outflow (CAESA, 1998).

1.4.2.3 Vegetation

Within the main peatland field, the vegetation is dominated by stunted black spruce, Labrador tea (*Rhododendron groenlandicum*), bog cranberry (*Vaccinium vitis-idaea*), and cloudberry (*Rubus chamaemorus*), with intermittent tamarack (*Larix laricina*). *Sphagnum* peat mosses are the dominant ground cover, and form distinct hummock and hollow microtopography. However, feather mosses and lichens are also prevalent. The vegetation at the outflow swamp is indicative of minerotrophic waters and is dominated by willows (*Salix* spp.), paper birch (*Betula papyrifera*), sedges (*Carex* spp.), water tolerant grass species, and marsh marigold (*Caltha palustris*) with intermittent balsam poplar (*Populus balsamifera*) and white and black spruce along the outflow edges. The ground cover is dominated by brown moss species, with sedge tussocks forming hummocks and patches of bare ground in the hollows. There are some occurrences of *Sphagnum* species and surface flow is prevalent. The adjacent, non-wetland area vegetation is dominated by aspen, balsam poplar, white spruce, red-osier dogwood (*Cornus stononifera*), bracted honeysuckle (*Lonicera involucrata*), and prickly rose (*Rosa acicularis*). Following the construction of beaver dams at the harvested outflow swamp in 2019, the width of the area flooded by outflow waters expanded, and large sections of neighboring trees and shrubs died.

1.5 STUDY DESIGN, SITE SELECTION, AND PEAT HARVESTING CHRONOLOGY

This investigation chronologically followed the sequence of peat harvesting activities, starting before harvesting commenced (pre-harvest), during the time when the peat field was being prepared for harvest (opening), and during the first two years of initial peat extraction (Figure 1.3). A before-after-control-impact (BACI) reference treatment comparison (Green, 1979) was used to assess alterations to hydrological flow path, physicochemical processes, and nutrient concentrations of waters within the peatland proper (peat field) and outflow swamps (outflow) as harvesting activities progressed. Sites established within the peat field were used to address research questions concerning changes to the hydrological flow paths, physicochemical processes, and nutrient concentrations. Ditch and outflow sites were used to address changes to water quantity and quality at receiving outflows to see if alterations to the harvested peat field were visible at the outflow. Ditch locations were also used to observe any changes to water quality associated with ditch substrate.

1.5.1 Pre-Harvest and Field Reference

In the autumn of 2018, prior to the commencement of any harvesting activities, peat field and outflow swamp sites within the proposed harvest area were selected to establish pre-harvest conditions. These sites are referred to as "harvested field" (Site H1; Figure 1.3) and "harvested outflow" (Sites H5 and H6; Figure 1.3). Site H5 was used for chemistry sample collection and flow rate measurements, and Site H6 was used to assess water levels and physicochemical conditions at the outflow swamp (Figure 1.3). A peat field reference location was selected within the peatland proper, but outside of the proposed harvest area, to serve as a natural control once harvesting activities commenced. This site is referred to as "reference field" (Site R1, Figure 1.3). Peat cores and vegetation species compositions were conducted at Sites R1 and H1 (Table 1.1).

1.5.2 Opening

In 2019, a 70-ha area within the peatland was opened for commercial horticultural peat extraction (Figure 1.3). In this study, "opening" refers to all activities prior to extraction that were done to prepare the field for harvesting peat. These include: excavating a perimeter ditch around the area to be harvested, clearing and mulching the surface vegetation, and installing ditches within the newly mulched area (referred to as internal ditches in this study).

The perimeter ditch was dug around the delineated area to be harvested between January and March 2019. This was followed by the clearing and mulching of the surface vegetation from April to August 2019. Due to low relief, the perimeter ditch was cut into the underlying mineral subsoil in order to establish an appropriate gradient to drain the peatland. Where the perimeter ditch ended, a small settling pond was excavated. Sampling sites were established within the perimeter ditch at the primary outlet from the fields (Site H4; Figure 1.3), within the ditch constructed at the pre-harvest outflow swamp above the settling pond (Site H5; Figure 1.3), and after the settling pond within the outflow swamp (Site H6; Figure 1.3). The mulching removed all living vegetation on the peat surface and replaced pre-existing peatland hummock and hollow microtopography with ridges of mulched plant material. In July 2019, shortly after ditching began and while mulching was underway, beavers established a lodge and multi-tiered dam below the settling pond, expanding the flooded area at the harvested outflow swamp and flooding Site H6 (Figure 1.3). Site H6 was converted to a "beaver pond" site, and a new sampling location was selected below the last beaver dam (Site H7, Figure 1.3) for comparison with Site H5 above the beaver pond. The beaver lodge and dam remained active for the remainder of the study.

Lastly, internal ditches were incised into underlying deep peat every 30 m, draining directly into the larger perimeter ditch in late November 2019, after all sampling was finished for the year. Thus, for all measurements conducted in 2019, sites were influenced by perimeter ditching and vegetation removal only. In 2020 and 2021, measurements were influenced by the perimeter ditch, vegetation removal, and internal ditches.

1.5.3 Extraction

Vacuum extraction activities commenced in late summer 2020. Limited data were collected due to restrictions associated with the COVID-19 pandemic. The harvested field was routinely harrowed starting in 2020 to encourage the exposed surface peat to dry, provide a uniform surface for vacuum extraction, and discourage vegetation from establishing. In 2021, Site H2 was set up in an internal ditch within the harvested field to compare the water in contact with deep peat in the ditch with the water in the adjacent field (Site H1) (Figure 1.3). Site H3 was established at the mouth of the same internal ditch where it drained into the perimeter ditch (Figure 1.3). It was used to measure the quantity and quality of the surface water leaving the harvested peat field. Regular ditch maintenance to facilitate drainage and allow for extraction activities occurred during the spring, summer, and autumn months of 2020 and 2021.

1.5.4 Reference Outflow

An adjacent, natural, surface-flow swamp system that drained a similar treed bog / poor-fen peatland approximately 500 m to the east of the harvested outflow was established in 2019 to serve as a reference outflow swamp (Figure 1.3). This reference is referred to as "reference outflow" in this study (R2, Figure 1.3).

1.5.5 General Sampling Schedule

In 2018, prior to disturbance, samples and measurements were conducted in late August and early October. Samples and measurements were collected approximately once per month from March through October in 2019 and 2021. Sampling was conducted in March and October only during 2020 due to logistical constraints from the COVID-19 pandemic; however, continuous water level recording at Site R1 occurred. Detailed descriptions of the methods used for data collection can be found in the methods section in Chapter 1 and Chapter 2.

Table 1.1 Peat core characteristics for pre-harvested, reference, and extracted peat fields

		Soil Texture[a]			K_{sat} (m s^{-1})[c]			ρ_b (g cm^{-3})		C:N		pH		EC (µS cm^{-1})	
From	To	Pre-Harvest	Ref.	Extracted[b]	Pre-Harvest	Ref.	Extracted[b]	Ref.	Extracted[b]	Ref.	Extracted[b]	Ref.	Extracted[b]	Ref.	Extracted[b]
0	-10	VP1	VP1	-	2.4×10^{-3}	2.4×10^{-3}	-	0.04	-	72	-	3.52	-	378	-
-10	-20	VP1	VP1	-	1.0×10^{-3}	1.0×10^{-3}	-	0.05	-	63	-	3.61	-	258	-
-20	-30	VP2	VP2	-	3.6×10^{-4}	3.6×10^{-4}	-	-	-	37	-	3.66	-	203	-
-30	-40	VP3	VP2	-	1.5×10^{-4}	2.8×10^{-4}	-	-	-	65	-	3.52	-	259	-
-40	-50	VP3	VP2	VP3	1.2×10^{-4}	2.3×10^{-4}	1.2×10^{-4}	-	0.09	71	37	3.61	3.56	213	184
-50	-60	VP3	VP3 - 5	VP1 - 5	1.0×10^{-4}	5.4×10^{-5}	1.0×10^{-4}	-	0.09	46	38	3.65	3.57	228	187
-60	-70	VP3	VP3	VP2	9.0×10^{-5}	9.0×10^{-5}	1.7×10^{-4}	-	-	-	43	-	3.71	-	120
-70	-80	VP3	VP6 - 7	VP3 - 5	8.0×10^{-5}	1.2×10^{-5}	4.2×10^{-5}	-	-	39	41	4.02	3.85	172	119
-80	-90	VP3	VP6	VP5	7.3×10^{-5}	1.1×10^{-5}	2.0×10^{-5}	-	-	-	35	-	3.82	-	232
-90	-100	VP4	VP6	VP4	3.5×10^{-5}	9.9×10^{-6}	3.5×10^{-5}	-	-	29	30	4.15	3.98	281	276
-100	-110	VP7	VP5	VP4	4.8×10^{-6}	1.7×10^{-5}	3.3×10^{-5}	-	-	-	-	-	-	-	-
-110	-120	VP8	VP5	VP4	2.4×10^{-6}	1.6×10^{-5}	3.0×10^{-5}	-	-	-	-	-	-	-	-
-120	-130	Clay	VP6	Clay	-	7.9×10^{-6}	-	-	-	18	-	4.35	4.69	262	723
-130	-140	Clay	Clay	-	-	-	-	-	-	-	-	4.24	-	648	-
-140	-150	Clay	Clay	-	-	-	-	-	-	-	-	-	-	-	-

Where: K_{sat} is the saturated hydraulic conductivity, ρ_b is the dry bulk density, C:N is the carbon to nitrogen ratio, EC is the electrical conductivity

[a] Organic soils classified using the von Post scale of decomposition

[b] Depth profile corrected to the estimated original ground elevation

[c] Estimated K_{sat} values calculated using Model 2 from Morris *et al.* (2022) with poor-fen as the trophic type and lawn as the surface microform type

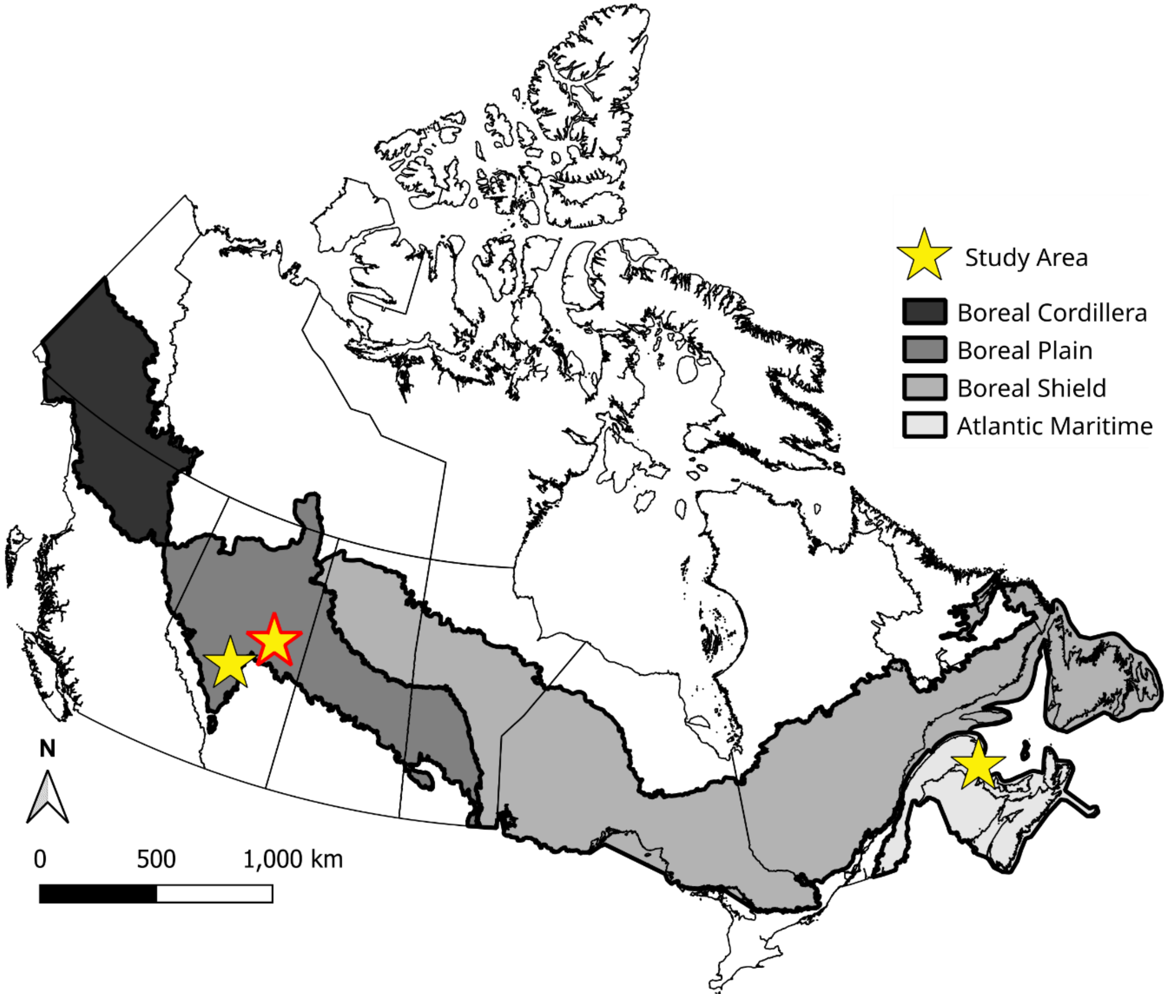

Figure 1.1 Biogeoclimatic setting of the study and larger parent study. Map shows the Boreal Cordillera, Boreal Plain, Boreal Shield, and Atlantic Maritime ecozones of Canada. The study areas for the overall peatland water quality project are demarcated with stars. This thesis is focused on the influence of peatland opening and early peat extraction at the Avenir, Alberta study area, outlined in red.

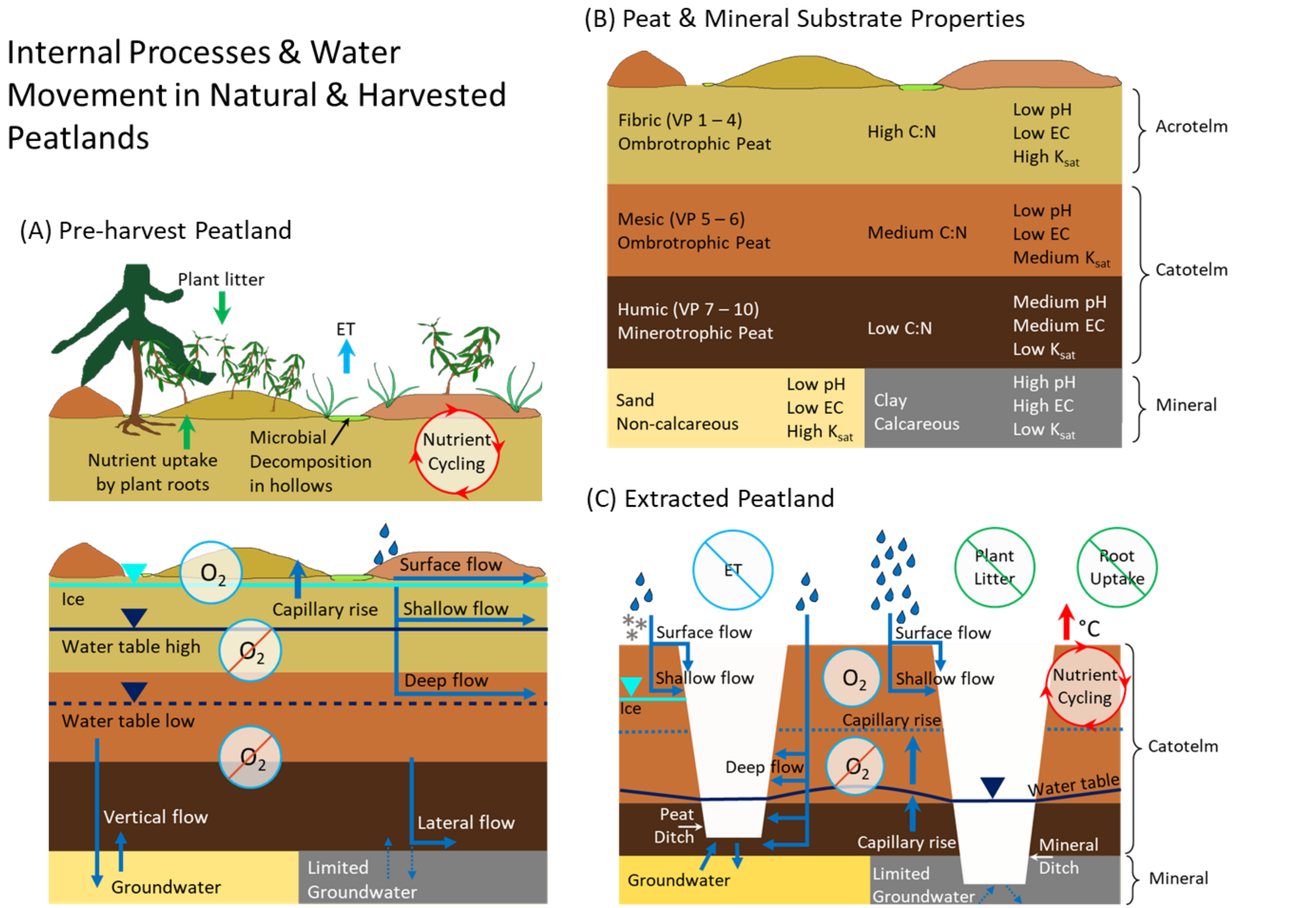

Figure 1.2 Conceptual models and hypotheses for (A) internal processes and water movement in a natural peatland, (B) peat and mineral substrate properties for a generalized peat profile, and (C) internal processes and water movement with internal ditches. See text, section 1.3.

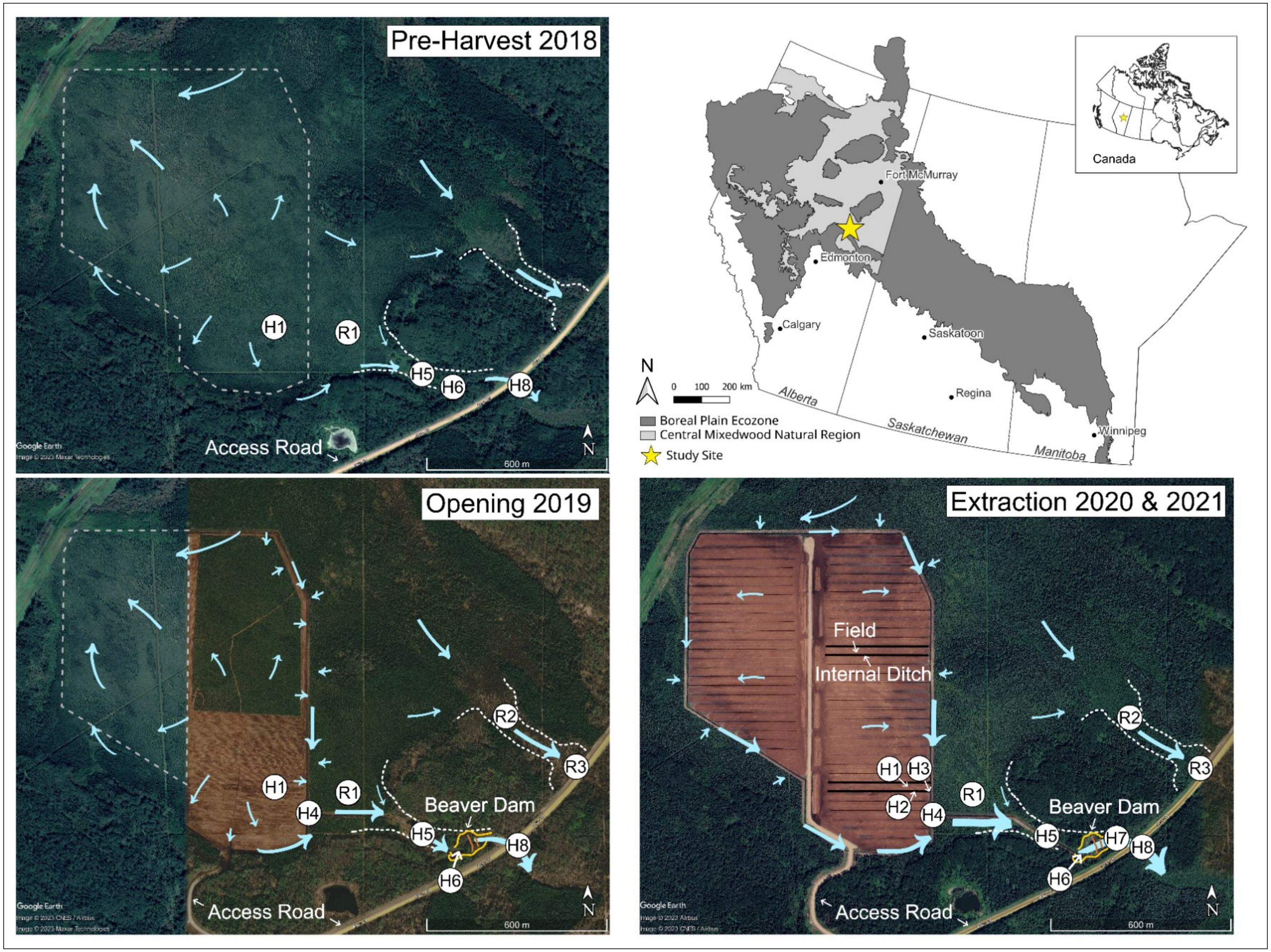

Figure 1.3 Study site overview. Location of Sun Gro Avenir bog within Canada (top right), and changes during pre-harvest (top left), initial opening (bottom left), and extraction (bottom right). Site locations shown with white circles; estimated water flow shown with blue arrows.

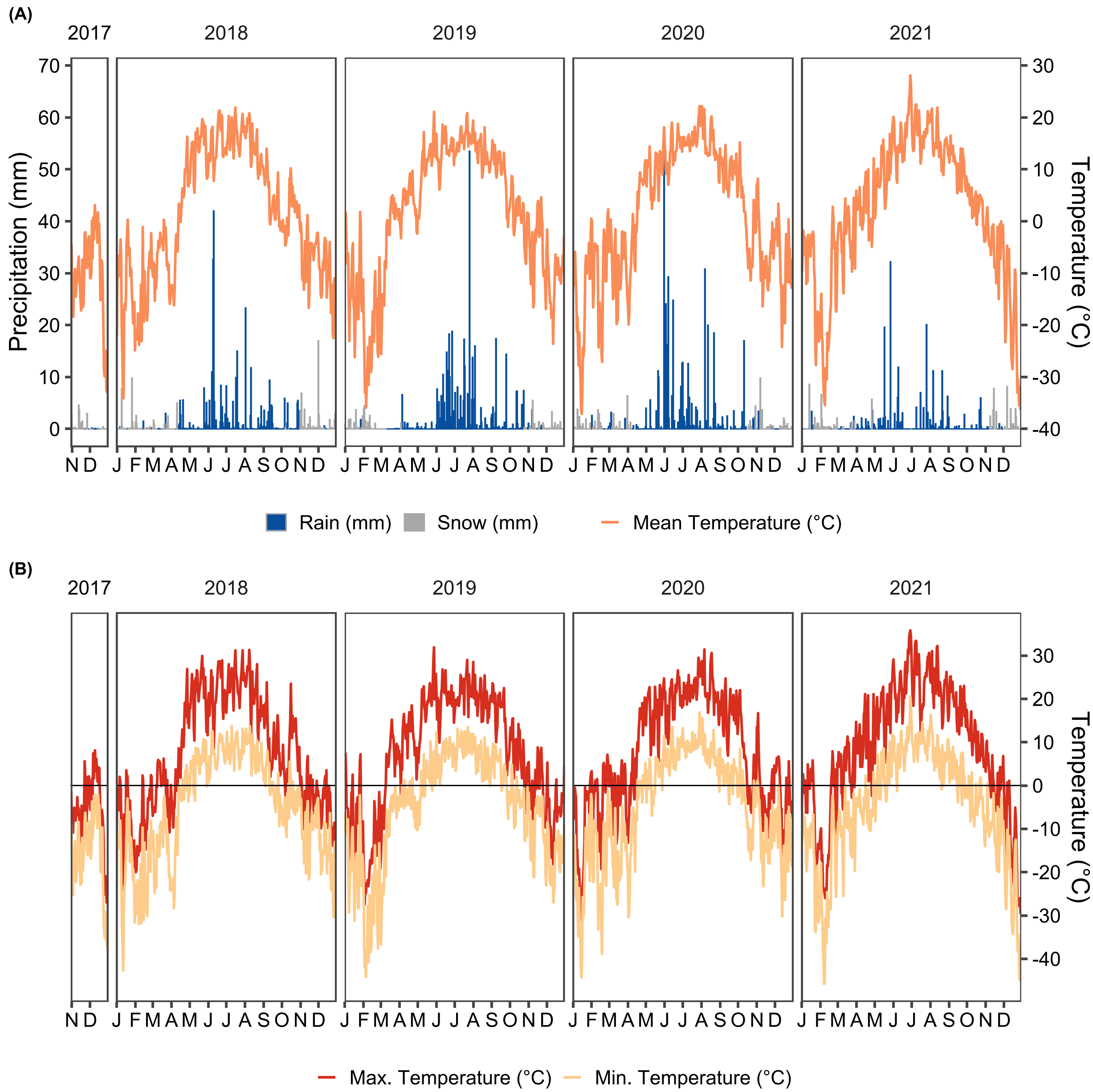

Figure 1.4 Precipitation and temperature during the study period. (A) Mean daily precipitation and temperature and (B) maximum and minimum daily temperatures at the Atmore AGDM meteorological station from November 2017 to December 2021 (Alberta Climate Information Services, n.d.).

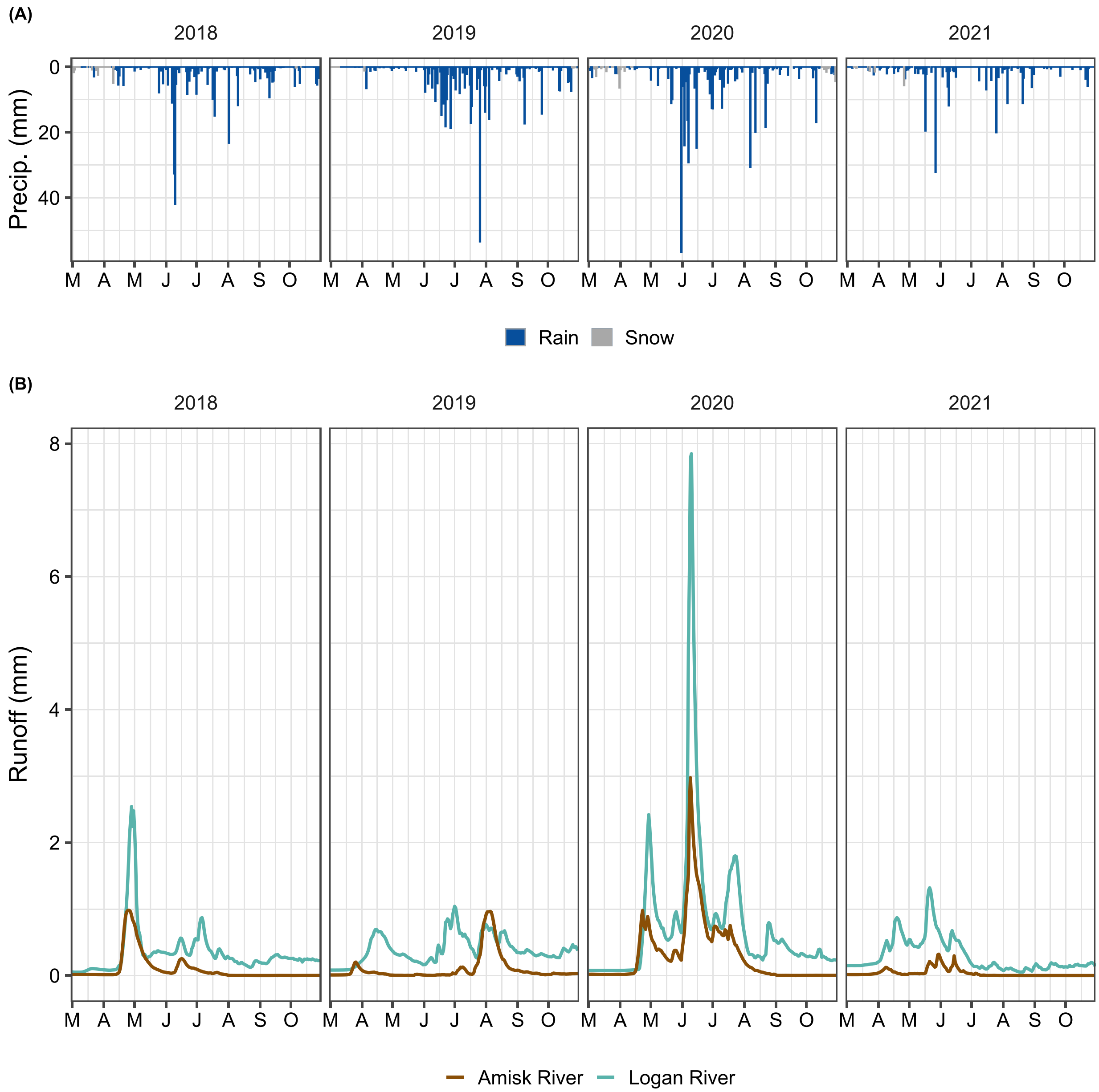

Figure 1.5 Precipitation and runoff at Amisk River and Logan River during the study period. (A) Mean daily precipitation at the Atmore AGDM meteorological station from March 2018 to December 2021 (Alberta Climate Information Services, n.d.). (B) Mean runoff at Amisk River and Logan River gauging stations (Environment Canada, n.d.). Runoff values were calculated using the government provided catchment areas for each river.

Chapter 2: Alterations to In-field Physicochemical and Hydrological Processes

2.1 INTRODUCTION

Major changes occur to the hydrology and physicochemical properties within a peat field when a peatland is opened for harvesting and the peat is subsequently extracted. These changes have the potential to increase nutrient concentrations in receiving water bodies, which may negatively impact the water quality (Betis et al., 2020; Donahue et al., 2022; Kløve, 2001; Pschenyckyj et al., 2023; St-Hilaire et al., 2004). Understanding the processes that may control how the water quality is altered within the peat field is essential for determining what chemical constituents are present and which nutrients are available to leach downstream. This is needed in order to apply mitigation strategies to a range of peatlands across different biogeoclimatic settings. However, a working knowledge of the changes to the water chemistry within the peat porewater during the transition from a natural peatland to an extracted peat field, the underlying processes that may govern their alteration, and the availability and mobility of nutrients present therein is limited.

Very few studies have investigated in-field peat porewater water quality in extracted peatlands, and observations are almost always conducted on peat fields that have been drained and extracted for a long period of time (Haapalehto et al., 2014; Kløve, 2001; Menberu et al., 2017; Wind-Mulder et al., 1996; Wind-Mulder & Vitt, 2000). This is especially concerning as many studies make the assumption that what is sampled at a peatland outflow is indicative of the water quality in the peat field porewater, without measuring it directly. Except for Moore (1987), no other studies on recently opened and extracted peatlands have been conducted in Canada, and the changes to the in-field porewater chemistry during the transition from a natural peatland to an extracted peat field have not been documented. While the older peat extraction fields appear to have higher nutrient concentrations (Haapalehto et al., 2014; Menberu et al., 2017), recently drained peatlands (Prevost et al., 1999) and newly extracted peatlands (Moore, 1987) did not show consistent results. This incongruency between previous studies suggests that internal mechanisms may be operating within the peat field that could both encourage or limit excess nutrients from being exported, leading to variable nutrient availabilities both within the harvested field and at the outflow. This may be especially important during the transition from a natural peatland to an extracted field during the first few years of harvesting activity.

Several factors can control in-field nutrient concentrations, as discussed in Chapter 1 (Figure 1.2). Changes to the ground elevation when the surface vegetation is mulched and peat is extracted exposes deeper peat layers that could have different structural properties and, in addition to alterations following ditching, could result in lower carbon to nitrogen ratios (C:N) (Moore & Basiliko, 2006), higher bulk densities (Minkkinen & Laine, 1998; Van Seters & Price, 2002), and lower saturated hydraulic conductivities (K_{sat}) (McCarter et al., 2020). These could alter water transmission, influence microbial decomposition, and have been associated with higher nutrient concentrations.

Likewise, the hydrological response to vegetation removal and ditching may affect the porewater chemistry by altering aeration and peat pore saturation, as snow and ice dynamics (Hayashi, 2013; Price, 1987; Van Der Kamp et al., 2003), water level depths (Evans et al., 1999; Marttila & Kløve, 2010), soil moisture (Price, 1997), and peat aeration (Price et al., 2003; Waddington & Price, 2000) are modified. The modifications to the peatland could also introduce new source waters to the harvested field via ditching. The electrical conductivity (EC) and calcium (Ca), magnesium (Mg), and sodium (Na) concentrations could demonstrate geochemical influence from deep minerotrophic peat or groundwater, which can be a source of nutrients that did not originate from alterations to the peat field surface (Haapalehto et al., 2014; Joensuu et al., 2002; Marttila et al., 2018; Pschenyckyj et al., 2023), especially when paired with chloride (Cl) (Lockwood et al., 1995) and silica (Si) (Khan et al., 2015) as conservative tracers to identify water source. Likewise, aluminum (Al) and iron (Fe) may indicate geochemical influence (Reynolds & Hughes, 1989) as well as serve as indicators for phosphorus (P) mobilization under variable redox and pH conditions (Aldous et al., 2007; Niedermeier & Robinson, 2009; Sah & Mikkelsen, 1989).

Physicochemical conditions within the peat could be the primary driver of in-field nutrient availability. In addition to the peat C:N, aeration, and soil moisture, the pH, and ground surface and porewater temperatures can also influence microbial decomposition (Aaltonen et al., 2021; Holden, Chapman, et al., 2006; Holden et al., 2004). Higher dissolved organic carbon (DOC) (Strack et al., 2008, 2011) and potassium (K) (Damman, 1978) concentrations can be expected and could serve as an indicator of in-field decomposition. Lack of root uptake (De Mars et al., 1996; Jabłońska et al., 2021; Walbridge & Lockaby, 1994), and increased fresh plant material immediately after vegetation mulching (Hyvönen et al., 2000; Tolvanen et al., 2020) could add nutrients that can both increase the risk of leaching and accelerate microbial decomposition. These processes are likely to change depending on the seasonal and interannual precipitation variability (Evans et al., 1999), as well as across biogeoclimatic settings.

In addition to changes to the porewater chemistry within the harvested peat field, how water travels from the peat field into the internal ditch once the peatland is opened and ready for extraction could influence the water chemistry and nutrient concentrations. Different hydrological pathways through the peat, including infiltration-excess overland flow, saturation-excess overland flow, rapid acrotelm flow, and pipe flow (Evans et al., 1999; Holden & Burt, 2003) (Figure 1.2.C) can modify the porewater chemistry when water comes into contact with peat of various levels of decomposition (McCarter et al., 2020). It is therefore important to understand if water movement from the harvested field to the internal ditch is dominated by surface and shallow, or deep flow because the hydrological pathway could govern the nutrient concentrations exported downstream (Figure 1.2).

2.1.1 Objectives and Hypotheses

This chapter is the first of two chapters within this thesis with the overall objective of understanding whether water quality at the outflow of peat harvest operations is a function of actual alterations to nutrient cycling and availability (of lack thereof) within the harvested peat fields or, a result of influences of, or interactions with, different source waters and processes external to the harvested fields that contribute to the peat harvest operation outflow. The objective of this chapter is to examine, relative to a natural reference peatland, the influence of opening a peatland and extracting peat on in-field peat structural properties, hydrology, physicochemical properties, and porewater chemistries in order to understand if nutrient (nitrogen (N) and P) concentrations and availability rates within the peat field are elevated and able to leach out of the peat field following harvesting activities. These objectives are explored during the transition from a natural peatland to an extracted field, and encompass the effects on the harvested field following the installation of the perimeter ditch, vegetation mulching, internal ditches, and early peat extraction.

To better understand the potential control on nutrient availability within the opening and extraction periods in the harvested peat field, the following alternative hypotheses are proposed:

(1) Nutrient concentrations and availability will be <u>higher</u> in the harvested peat field during the opening and extraction periods relative to the natural peatland reference because of:

(a) favourable conditions for decomposition introduced by ditching and vegetation mulching. These conditions would be: higher peat and porewater temperatures, deeper zones of aeration, and higher nutrient availability from mulched vegetation and lack of plant uptake are present in the harvested field, which would encourage peat mineralization (Figure 1.2).

As such, increases in indirect measures of decomposition (nutrient concentrations and availability rates) will be observed relative to the reference peat field.

(b) upward migration of source waters in contact with deeper peat containing higher nutrient concentrations following ditching and vegetation mulching or, altered ground surface elevation that intersects deeper peat layers containing higher nutrient concentrations within the harvested field. If shallower water levels are present (possibly due to lack of transpiration), and the EC and pH values, and Ca, Mg, Na, Si, and Cl concentrations in the harvested field are similar to deep porewaters observed within the reference peatland, then this would indicate that porewater has moved upwards or, the surficial peat has been removed exposing deeper, more decomposed peat. As such, increased nutrient concentrations in the harvested peat field would be observed.

(2) Nutrient concentrations and availability rates will be similar in the harvested peat field relative to the natural peatland reference if internal ditching and vegetation mulching have little effect on the conditions governing decomposition or, alternatively, if peat composition and environmental conditions limit short-term nutrient release in the harvested field porewaters during opening and extraction. If peat soil moisture values are maintained via peat capillary rise and increased aeration is maximized, together with the original higher C:N and low pH values, then peat mineralization will be tempered and indirect evidence of decomposition (nutrient concentrations and availability rates) will resemble the reference peatland (Figure 1.2).

To further understand whether available nutrients within the peat field are present within the internal ditches, which would increase the potential for leaching downstream, two further questions are asked: Does the dominant water movement occur via:

(1) surface and shallow flow from the harvested field to the internal ditches? If so, the internal ditch nutrient concentrations would resemble the harvested field surface and shallow porewaters.
(2) deep flow down the peat profile within the harvested field and is expressed into the internal ditch after contacting deeper peat? If so, the nutrient concentrations (and geochemistry) of the water in the internal ditch will resemble deep porewaters within the harvested field and reference peatland.

2.2 METHODS

2.2.1 Hydrological Measurements

2.2.1.1 Precipitation, Temperature, and Depth to Ice

Daily temperature, precipitation, and potential evapotranspiration (PET) values were obtained for 2018 – 2021 from a nearby (< 35 km) government meteorological station, Atmore AGDM (Alberta Climate Information Services, n.d.). Snow depth, snow density, and snow water equivalent (SWE) were measured in March 2020 and 2021 along 50 m transects through the reference field and the harvested field (Table 2.1). Transects within the harvested field intersected both the peat field and the internal ditches. SWE values represent net values (snowfall minus sublimation). Snow depth was also reported during visits to each sample site. To detect the presence of an impermeable ice lens that could impede vertical water movement, a steel rod was inserted into the ground to determine the depth to ice from the ground surface during each snow transect measurement in March, and at all sample sites from March through to mid-July. The presence of ice surface depth and thickness was confirmed periodically by excavating a pit or coring with an auger. To estimate the water storage capacity of the peat during snowmelt, storage was calculated (SWE plus mean depth to ice) using the snow and ice transect data (Table 2.1). Negative storage values indicate an ability to store water; positive values indicate no storage capability.

2.2.1.2 Water Levels and Groundwater

Water level was assessed using a network of wells, stilling wells, and shallow pits in key locations within the peat field and outflow (Figure 1.3). All wells and stilling wells were constructed using two-inch inside diameter, polyvinyl chloride (PVC) pipes of variable length. Stilling wells installed in ditches were fully slotted, inserted 50 cm below the sediment, and extended 100 cm above the sediment. Wells were fully slotted from the ground surface and were installed to depths that ranged from 50 cm and 130 cm deep below ground. Wells installed to 50 cm were used to assess both surface and near surface water. Wells installed to > 50 cm were used to assess deeper water, but were also used to prevent surface freezing and observe the potential for perched water levels during late winter and early spring. To minimize the risk of water sample contamination, no back fill was used on wells or stilling wells. When installed, the 2-inch inside diameter PCV well pipe was inserted into a slightly shallower and smaller diameter core hole, then tapped to depth to ensure a tight fit and continual contact with the peat. All slotted sections were fitted with a polypropylene nonwoven geotextile filter sock to reduce the amount of sediment entering the pipe.

Water level depth was measured using a water level meter (Model 107 TLC Meter, Solinst® Canada Ltd.). In selected wells and stilling wells, pressure transducers (HOBO® U20L-04 Water Level

Logger, Onset® Computer Corporation) were installed to determine continuous (4-hour interval) water levels. Where wells could not be installed due to operational constraints, such as on the active peat extraction field, a shallow pit or borehole was dug using a serrated knife, shovel, or Dutch auger. The water level in the pit or borehole was allowed to stabilize, and the depth of water relative to the ground surface was measured.

2.2.2 Hydrochemical Sampling

2.2.2.1 In-field Porewater and Groundwater

The water chemistry assessed within the peat field was based on porewater at two generalized depths: surface to near surface water (within the highly fibric, active peat layer) and deep peat water (within the decomposed (mesic – humic) peat column proper) (Figure 1.2.B for peat depth characteristics). Here, surface water refers to samples taken from visible standing water in shallow pools or hollows. Near surface water refers to samples collected within 50 cm from the ground surface, originating from shallow pits dug into the peat or from shallow wells (50 cm), respectively. If surface water was not visible in hollows or pools, and a shallow well was not present, a hole was cut in the peat to intersect the water table using a serrated knife or shovel (Shotyk et al., 2023). The block of peat was removed from the hole, the water was purged, and the pit was allowed to refill before taking a sample. The block of peat was replaced once a sample was taken. If no water table was intersected within 50 cm of the surface by a pit or shallow well, then the surface water portion of the porewater sampling was considered dry. Water samples were collected from shallow wells using a 1.5-inch (3.8 cm) diameter weighted bailer. When surface waters and near-surface waters collected from shallow wells (50 cm) and near surface pits were compared, the temperature and chemical compositions were similar.

Deeper peat water is operationally defined here as water collected from deeper than 50 cm from the ground surface. This included water collected from wells installed deeper than 50 cm below the ground surface and excavated peat pits > 50 cm deep. The water column below 50 cm from the ground surface within a well was collected using a 1.5-inch (3.8 cm) diameter weighted bailer. Wells were not bailed dry and allowed to re-fill on the same day as sample collection. Rather, following sample collection, each pipe was purged dry in preparation for the subsequent sampling date approximately one month in the future. This was done to allow sediments that may have entered the well during bailing to settle, and to allow for the water at depth to equilibrate with adjacent groundwater outside the well.

Where deeper wells could not be installed, such as in the harvest field, a pit was dug using a shovel or Dutch auger to intersect the water table. As with the shallow peat pits, the peat was cleared from the pit, the water was purged, and the pit was allowed to refill before taking a sample. The hole was refilled with peat after the sample was taken. All well and pit samples, from both surface and deep locations, were pre-filtered in the field with a 250 µm nylon mesh to remove coarse particulates.

2.2.3 Peat Soil Measurements and Sampling

2.2.3.1 Peat Soil Characteristics and Physicochemical Properties

As an indicator of reduction-oxidation conditions, a pre-etched steel rod was inserted into the ground at each site, where operationally possible, to observe rust accumulation as an indicator of aerobic conditions (Bridgham et al., 1991; Carnell & Anderson, 1986; Owens et al., 2008; Silins & Rothwell, 1999). The aerobic zone within the peat was estimated by measuring the distance from the ground surface to the deepest extent of iron oxide accumulation on the rod (Carnell & Anderson, 1986; Silins & Rothwell, 1999). Steel rods were assessed, scraped free of rust using steel wool, and re-installed after each site visit. Where microtopography was present, a rod was installed into both the hummock and hollow to capture any variation in aeration.

Soil depth profiles were assessed using hand texturing methods at Sites R1 and H1. At Site H1, a core was taken in 2018, pre-harvest, before any opening activity commenced. Another core was taken in 2021 during active extraction. A Dutch auger was used to core until mineral soil was reached. Organic soil decomposition levels were estimated using the von Post scale of decomposition (Soil Classification Working Group, 1998), and mineral soil textures were assessed using hand texturing (Thien, 1979). Peat and mineral samples from cores collected in 2021 were air dried for 5 days at 20 °C, then oven dried at 105 °C until a constant weight was reached (Dettmann et al., 2021; O'Kelly & Sivakumar, 2014) before analysing soil C:N, EC, and pH.

Peat samples from 0 – 10 cm and 10 – 20 cm below the peat surface were taken in October 2021 at randomly selected PRS® probe locations (see section below). Five locations were selected in the harvested field and reference field, respectively. In the reference field, a sample was taken from both the hummock and hollow associated with the location. To measure bulk density, a known volume of peat was collected using either a hand core or by cutting a cube of peat using a serrated knife; the dimensions of the peat block were recorded in the field. Samples were weighed, air dried for 5 days at 20 °C, then oven dried at 105 °C until a constant weight was reached (Dettmann et al., 2021; O'Kelly & Sivakumar, 2014). Once dried, a final weight was recorded, and bulk density was calculated by dividing the dry weight by the known volume of peat. The leftover peat from the bulk density calculation was ground using a stainless-steel grinder and used to measure peat C:N, EC, and pH.

Saturated hydraulic conductivity (K_{sat}) was estimated using the *model 2* equation described by Morris et al. (2022), which incorporated von Post and peat depth measurements from the harvested and reference fields (Table 1.1). For the purposes of the equation, the peatland type was classified as a poor-fen and the microform was characterized as lawn.

2.2.3.2 Peat Field Nutrient and Major Ion Availability

A soil nutrient availability study was conducted in late June to late July 2021, while the vegetation was most active, to investigate the peat physicochemical parameters and *in situ* relative nutrient and major ion availability within the near-surface peat. Plant Root Simulator (PRS®) probes (Western Ag Innovations, Saskatoon, Canada) were used to measure the relative rate of nutrients and major ions available for root and microbial uptake over a 17.5 cm^2 surface area over a four week period (Jun 26, 2021 – July 25, 2021) (Harris et al., 2020; Munir et al., 2017). During the sampling period, active vacuum extraction occurred in the harvested field. Thus, a systematic, stratified sampling design was selected to capture the natural variability within the field. This design covered as much distance as possible while taking up the smallest amount of area and allowed for vacuum extraction to continue unencumbered. Transects were established in the reference and harvested peat fields.

In the reference field, a randomly assigned transect starting point and direction were selected before entering the area. Five sample sites were spaced approximately 20 m apart along the transect. Each sample site was comprised of three locations spaced 5 m apart to capture a range of conditions. At each location, one hummock and one hollow were selected to capture variability between microforms in the reference field.

In the harvested field, sample sites were placed in the field, adjacent to an internal ditch and at the end of the peat field, near the perimeter ditch. Ongoing peat extraction with heavy equipment prevented placement within the field, far away from ditches. A 30 m temperature and soil moisture transect across the width of the peat field (encompassing the ditch edge, centre of the field, and the following ditch edge) was conducted prior to the study and showed negligible differences in temperature and soil moisture within the top 10 cm. Thus, the locations along the field edge, next to the ditch were determined to be adequately representative of the peat field surface. Three sample sites were spread along the length of harvested peat field, parallel to the internal ditch. Each sample site was comprised of three locations spaced approximately 10 m apart to capture a wider range of variability along the length of the field. Locations were placed in the harvested field, 1.5 m away from the internal ditch edge. Two sample sites were spread across the width of two harvested field ends, parallel to the perimeter ditch and bordered perpendicularly on either side by internal ditches. Locations were placed in the harvested field, 1.5 m away from the internal ditches and 5 m away from the perimeter ditch edge.

At each location, a pair of PRS® probes (one anion and one cation) were installed vertically, the resin sitting between 5 – 10 cm below the ground surface. Probes were placed into a slit cut into the peat with a serrated knife, following standard protocol (Western Ag Innovations, Saskatoon, Canada). Care was taken to assure good contact was made between the probe and the peat. Steel rods estimating aeration were placed at each location and assessed at the end of the study to allow for sufficient time for rust to potentially develop. At the start and end of the study, soil moisture at 6 cm below the ground surface using a soil moisture probe (ML3 ThetaProbe, Delta-T Devices Ltd.) set to the organic soil general calibration setting (ML3 User Manual 2.1), surface temperature using an infrared thermometer (IRK-2 Infrared Thermometer, ThermoWorks Inc.), peat temperature 10 cm below the ground surface (Therma Waterproof Type K Thermometer, ThermoWorks Inc.), and depth to ice were measured.

2.2.4 Chemical Analyses

2.2.4.1 Porewater, Groundwater, and Surface Water Concentrations

For all porewater and surface water samples, EC, pH, and temperature were measured *in situ* using a hand-held meter (HI98129, Hanna® Instruments). In addition, the temperature and EC at the water level surface and at the bottom of each well were measured using an EC meter (Model 107 TLC Meter, Solinst® Canada Ltd.). For all porewaters in peat pits and wells, and all non-flowing surface waters (in pools and non-flowing ditches), filtered (0.45 µm) dissolved fractions (total dissolved nitrogen (TDN), ammonium (NH_4^+-N), nitrite + nitrate (referred to as nitrate and NO_3^--N henceforth), total dissolved phosphorus (TDP), soluble reactive phosphorus (SRP), DOC, and Al, Ca, Cl, Fe, K, Mg, Na, and Si ions) were analyzed. Only the dissolved fraction was analyzed because particulate fractions collected in the sample were assumed to be an artifact of sample collection and not representative. Dissolved fractions were filtered with a 28 mm or 47 mm diameter 0.45 µm Sartorius cellulose acetate syringe filter. Each water sample was collected in high density polyethylene bottles triple rinsed with the sample water. Filled bottles were kept cool, transferred to a 4 °C refrigerator at the end of each field day, and submitted within 48 hours of collection for analysis at the Biogeochemical Analytical Service Laboratory (BASL) at the University of Alberta.

Total dissolved nitrogen, NH_4^+-N, NO_3^--N, TDP, SRP, and Si were analyzed using flow injection analysis (QuickChem® 8500 FIA Automated Ion Analyzer, Lachat Instruments). Analysis followed procedures outlined in the United States Environmental Protection Agency (US EPA) Method 353.2 for TDN, NO_3^--N (US EPA, 1993), and in the American Public Health Association (APHA) Method 4500 for NH_4^+-N, TDP, and SRP (Baird & Bridgewater, 2017). Dissolved organic carbon was measured using a total organic carbon analyser (TOC-5000A Total Organic Carbon Analyzer, Shimadzu Corporation). Analysis followed procedures outlined in the US EPA Method 415.1 for DOC (US EPA, 1999). Aluminum, Ca, Fe, K, Mg, and Na were measured using an inductively coupled plasma optical emission spectrometer (iCAP 6300, Thermo Scientific) following the US EPA Method 200.7 (US EPA, 1994). Chloride was measured using ion chromatography (DX-600 Ion Chromatography System, Dionex Corporation), following the US EPA Method 300.1 (US EPA, 1997a).

2.2.4.2 Soil Physicochemical Properties

Total carbon (TC) and total nitrogen (TN) were analyzed for each dried surface peat sample using dry combustion (Flash 2000 Organic Elemental Analyzer, Thermo Scientific). The C:N were calculated by dividing the TC value by the TN value. For organic samples, EC and pH were measured from a slurry using 1 g of the oven-dried, ground peat (Stanek, 1973) mixed with 12.5 g of milli-Q water (Nilsson et al., 1995). For mineral samples, EC and pH were measured from a slurry using 5 g of the dried, ground mineral soil mixed with 5 g of milli-Q water (Kalra, 1995). The solution was shaken for 10 minutes and left to rest for 30 minutes. The EC, followed by the pH, were measured in the aqueous portion of the slurry using a hand-held meter (HI98129, Hanna® Instruments). The instrument was rinsed with distilled water between samples and checked for calibration after approximately 15 measurements. Duplicates were run on 10 % of the samples.

2.2.4.3 Peat Field Nutrient Availability

The nutrients and major ions assessed using the PRS® probes were nitrogen as ammonium (NH_4^+-N), nitrogen as nitrate (NO_3^--N), phosphorus as phosphate (PO_4^{3-}-P), sulfur as sulfate (SO_4^{2-}-S), Al, Ca, Fe, K, Mg, and manganese (Mn). Each probe contained a permanently charged anion- or cation-exchange resin made of polystyrene cross-linked with divinylbenzene, and held in a 15 cm long by 3 cm wide by 0.5 cm thick plastic support. Probes selected to adsorb anions had R-NH_4^+ as the fixed ionic group within the resin matrix, and probes selected to adsorb cations had R-SO_3^- as the fixed ionic group. Probes were treated by Western Ag. Innovations with easily exchangeable counter ions prior to installation; anion probes were saturated with HCO_3^- and ethylenediaminetetraacetate (EDTA) and cation probes were saturated with Na^+. Probes were kept clean, moist, and cool prior to installation in the peat. Upon retrieval, probes were placed in labelled zip-top bags and stored in a cooler. Probes were cleaned with deionized water within 24 hours of removal, placed in clean, labelled, zip-top bags, and shipped to Western Ag. Innovations for analysis.

2.2.5 Statistical Analyses

The PRS® and *in situ* physicochemical variables measured in the harvested field and reference hummocks and hollows were compared for statistical differences between groups for each variable. The harvested field, reference hummocks, and reference hollows were treated as independent samples. Data were screened for outliers prior to analysis and were removed for the statistical analysis. Each variable was grouped into the pre-determined location (reference hummock, reference hollow, and harvested field) and normality was assessed using graphical representations of the data and the Shapiro-Wilk test ($\alpha = 0.05$, variables with p values > 0.05 were assumed to be normally distributed). Variables that were normally distributed were tested for significant difference using a one-way analysis of variance (ANOVA) followed by a Tukey Honest Significant Difference post-hoc test ($\alpha = 0.05$) for multiple comparisons. Variables that were not normally distributed were tested for significant difference using a Kruskal-Wallis rank sum test followed by a post-hoc Dunn's Test for multiple comparisons. P-values < 0.05 were deemed significant.

Direct gradient analysis with nonmetric multidimensional scaling (NMDS) was selected to visualize the interactions between peat physicochemical variables (surface temperature, sub-surface temperature, depth to ice, rust depth, C:N, bulk density, pH, EC, and soil moisture) and nutrient availability rates (NH_4^+-N, NO_3^--N, PO_4^{3-}-P, SO_4^{2-}-S, Al, Ca, Fe, K, Mg, and Mn) in peat from reference field hummocks and hollows, and from the harvested field. Physicochemical variables were used for the initial ordination and PRS® nutrient availability rates were overlain onto the plot using calculated ordination scores. Data were scaled and a Euclidean distance metric was used. NMDS is a non-parametric, distance-based approach, and was selected because the data did not meet assumptions necessary for parametric techniques. Stress values associated with the NMDS ordinations were stable. All data manipulations and statistical analyses were conducted in RStudio (2023.06.01+524), using R Statistical Software (v.4.2.2).

2.3 RESULTS

2.3.1 Peat Structural Properties

2.3.1.1 Relief and Ground Elevation

The relief at the reference (Site R1) and harvested (Site H1) peat fields, pre-harvest, had a range of microtopography, with hummocks approximately 30 cm tall and hollows approximately 5 cm below the arbitrary ground surface (Figure 2.1.B). The elevation at Site R1 was not altered by harvesting activities for the duration of the study.

Alterations to the relief within the harvested field commenced with the installation of the perimeter ditch in winter 2019, which was incised into the mineral subsoil to depths between less than 1 m and >2 m deep, depending on the location. Aside from the immediate impact to the area converted to ditch, which removed vegetation and exposed mineral soil to the open air, there was no additional impact to the relief or elevation within the harvested area (Figure 2.1.C). Vegetation removal and mulching in spring 2019 altered the relief considerably by removing the surface microtopography. The hummocks were replaced by linear ridges of plant material 5 – 10 cm lower than the original hummock elevation, while the elevation of the hollows remained roughly the same (Figure 2.1.C).

Drastic alterations to the relief continued when the internal ditches were installed in the harvested field during late November, 2019, when the ground was frozen (Figure 2.1.C). The deeper peat approximately 90 cm below the original ground surface was exposed to the air and was put in contact with the surface water moving through the ditch and toward the outflow (Figure 2.1.C). In preparation for extraction, the ground surface relief was levelled and harrowed; the ridges of plant material were removed and the entire area, including hollows, was flattened below the original ground elevation (Figure 2.1.C). By the time vacuum extraction was fully operational in 2021, the ground elevation in the harvested field appears to have been lowered 20 – 40 cm below the original hummock surface and 5 – 10 cm below the original hollow surface as a result of opening activities and vacuum removal (Table 1.1, Figure 2.1.C).

2.3.1.2 Peat Stratigraphy, Saturated Hydraulic Conductivity, Bulk Density, and Carbon:Nitrogen

The general peat stratigraphy was similar across the reference and pre-harvest fields, but there were subtle differences in peat structural properties within the microtopography. The peat deposit at Sites R1 and H1, pre-harvest, was approximately 120 cm deep (Table 1.1). The top 60 cm of the peat profile were comprised of fibric peat with von Post levels of decomposition between VP1 – VP3, while the bottom 60 cm were comprised of fibric, mesic, and humic peat between VP3 – VP8 (Table 1.1). Peat cores taken at Site H1, pre-harvest, showed a higher proportion of fibric peat in the lower 60 cm compared to Site R1 (Table 1.1). Estimated K_{sat} values were similar in the reference field and pre-harvested fields, and reduced by three orders of magnitude with depth in the profile (Table 1.1). The median bulk density in the reference hummocks was < 0.02 g cm^{-3} in the surface peat 0 – 10 cm from the top of the hummock, and 0.03 g cm^{-3} 10 – 20 cm below the hummock surface (Figure 2.2.A,B). The median bulk density in the reference hollows was greater than in the hummocks; ~0.04 g cm^{-3} at 0 – 10 cm and 10 – 20 cm below the hollow surface (Figure 2.2.A,B). The median C:N from 0 – 10 cm was ~60:1 and ~75:1 10 – 20 cm below the top of the hummock (Figure 2.2.C,D). In the hollows, the C:N was lower than in the hummocks, with median C:N of ~50:1 and ~52:1 at 0 – 10 cm and 10 – 20 cm below the hollow surface, respectively (Figure 2.2.C,D).

Alterations to the relief and ground elevation resulted in different near-surface peat properties in the harvested field compared to the reference field. In October 2021, the peat depth at Site H1 was approximately 80 cm deep, and had predominantly fibric with some mesic peat decomposition rates (VP1 and VP5) (Table 1.1). The stratigraphy was similar to Site H1 pre-harvest and Site R1, once consideration for change in depth and spatial variability was made. Estimated K_{sat} values within the surface peat were an order of magnitude lower than the reference and pre-harvest values, and resembled the peat ~40 – 50 cm below ground (Table 1.1). Peat collected within the extracted harvested field from the 0 – 10 cm and 10 – 20 cm below ground had significantly higher bulk densities (median ~0.09 g cm^{-3}) compared to both hummocks and hollows in the reference field (Figure 2.2.A,B). The median C:N at the extracted field was ~48:1 at 0 – 10 cm, and ~55:1 at 10 -20 cm (Figure 2.2.C,D). This was similar to the reference field hollows but significantly lower than reference field hummocks.

2.3.2 Hydrological Responses

2.3.2.1 Ice and Snow

The depth to ice in the reference hummocks and hollows at Site R1 varied spatially with microtopography and showed extended seasonal patterns throughout the study (Figure 2.1.B). On average, the depth to ice in the reference field was 18 cm below the ground surface during the winter (Table 2.1). At the end of the winter, the depth to ice in the hollow was 10 – 20 cm below the ground surface (Figure 2.1.B). Once the snow began melting in the late-winter to early-spring, the hollows were frozen at the ground surface; in contrast, the ice was approximately 40 cm below the tops of the hummocks (Figure 2.1.B). Ice was detected at Site R1 until mid-July in 2021, and began to establish again at the end of October (Figure 2.1.B). During the start of the intensive study in late-June, 2021, ice was ~45 cm below the ground in the reference hummocks and ~30 cm below the ground in the reference hollows (Figure 2.3.A). By the end of the study, in late-July, 2021, the hummocks had fully thawed, while the hollows still had ice remnants detected within 35 cm of the ground surface (Figure 2.3.B). The snow distribution at Site R1 was variable during the study, but was generally minimal and ranged from 30 – 40 cm deep in 2020 and 2021 (Figure 2.1.B, Table 2.1). The snow water equivalent (SWE) over both years was 6 cm, and the storage capability of the reference field was -11 cm at the end of the winter (Table 2.1).

During spring 2019, the depth to ice was similar in the harvested field (Site H1) when compared to reference field (Site R1); however, the ice depth and duration began to vary relative to the reference field once extraction was underway (Figure 2.1.C). During winter 2020, the depth to ice was 10 cm below the ground surface at Site H1 (Figure 2.1.C). In winter 2021, the ground at Sites H1 and H2 was fully frozen, with a thin layer of ice that accumulated above the ground surface (Figure 2.1.C). Considering both years, the depth to ice was much closer to the ground surface at H1 and H2 compared to R1 (Table 2.1). The ice lens at Site H1 began melting more rapidly compared to Site R1, and was fully melted by June near the sample site (Figure 2.1.C). Within the internal ditch at Site H2, the ground was fully frozen until April 2021, when the ice rapidly melted (Figure 2.1.C). During the start of the intensive study in late June 2021, remnants of ice were still observed near the edge of the internal ditch along the entire length of the harvested field; however, no ice was detected by the end of the study in late-July (Figure 2.3.A,B). Ice began to establish at Sites H1 and H2 during late October (Figure 2.1.C). Snow depths were shallower but more uniform at Site H1 compared to Site R1 and ranged from 15 – 37 cm deep in 2020 and 2021 (Table 2.1, Figure 2.1.C). In contrast, the internal ditches (Site H2) collected 86 - 100 cm of snow depth that remained protected within the ditch several days after the snow at Site H1 had melted (Table 2.1, Figure 2.1.C). The SWE at Site H1 was the same as Site R1; however, the SWE at Site H2 was much higher (24 cm) relative to Site R1 (Table 2.1). Compared to Site R1, the storage capability at Sites H1 and H2 were drastically reduced, with mean values of 4 cm and 24 cm, respectively (Table 2.1).

2.3.2.2 Water Levels

The weather patterns over the 4-year study period (2018-2021), as shown in Figure 1.4, were highly variable and had a profound impact on the reference peatland water levels at Site R1 (Figure 2.1.B). In autumn 2018, prior to the commencement of harvesting activities, the water levels at Sites H1 and R1 were within 20 cm of the ground surface (Figure 2.1.B,C). In 2019, reference water levels at Site R1 were maintained within 20 cm below the ground surface throughout the summer and autumn (Figure 2.1.B). Water levels began to taper off during early winter 2019, and dropped further as the peat continued to freeze (Figure 2.1.B). During the winter in 2020 and 2021, the water froze in place and dried out the well resulting in a water level deeper than 100 cm below the ground (Figure 2.1.B). Water levels appeared to be closely tied to the depth to ice in the late winter and spring at Site R1 (Figure 2.1.B). As the snow began to melt in late winter 2021, a perched water table was observed within 30 cm of the ground surface at first, then as ponded water on top of ice in the surface hollows (Figure 2.1.B). During this time, a deeper secondary water level, 80 – 100 cm below the ground, was present (Figure 2.1.B). By early June in 2020 and 2021 a singular water table was at or near the ground surface (Figure 2.1.B). Water levels at Site R1 were maintained within 20 – 30 cm below the ground surface throughout the summer and autumn in 2020 (Figure 2.1.B). However, in 2021, the water level at Site R1 continued to drop after peaking in June, coinciding with the melting ice and lack of rain (Figure 2.1.B). In contrast to 2018, 2019, and 2020, the water table at Site R1 was 100 cm below the ground surface in autumn 2021 (Figure 2.1.B).

The installation of the perimeter ditch did not appear to have any draw-down effect on the water levels at Site H1, within the harvested peat field (Figure 2.1.C). In the late-spring and early summer, after the perimeter ditch was installed and the vegetation mulching was initiated, the water level at Site H1 was similar to Site R1 (Figure 2.1.C). However, as mulching progressed and vegetation was further removed, water levels were higher relative to the ground surface at Site H1 compared to Site R1 during the summer and autumn in 2019 (Figure 2.1).

Once the internal ditches were installed and vacuum extraction commenced, water levels in the harvested peat field (Site H1) began to drop relative to the reference peat field (Site R1) (Figure 2.1.C). Similar to Site R1, the water level at Site H1 appeared to be closely tied to the depth to ice. In early-April 2021, the water level was at the ground surface and dropped lower over time with the melting ice lens (Figure 2.1.C). Summer water levels at Site H1 dropped more rapidly than at Site R1, and water levels remained at the ditch level or deeper into autumn (Figure 2.1.C). During summer, the ditches frequently dried up, with the water level dropping below the ditch bottom, then rapidly re-wetting in response to rainfall events (Figure 2.1.C). The internal ditch remained dry into the autumn (Figure 2.1.C); however, when loose peat was cleared from an adjacent internal ditch at the end of August, water was observed in the ditch and the flow was re-established.

2.3.2.3 Soil Moisture and Rust Depth

Soil moisture measurements were collected in 2021 only. Soil moisture, measured during the hottest and most vegetatively active part of the year, revealed major differences between the hummocks and hollows in the reference field (Figure 2.3.G,H). Measurements taken 6 cm below the ground at the reference hummocks and hollows during the start of the intensive study in late-June, 2021, showed that hummocks were significantly drier than hollows (Figure 2.3.G). However, at the end of July, following a month with very little precipitation, the hollows had dried substantially and were no longer statistically wetter than the hummocks (Figure 2.3.H).

In contrast, soil moisture appeared to remain high in the extracted field, despite the comparably low water level. Soil moisture measurements taken within 6 cm of the ground surface at the end of June, 2021, when the water table was deeper than 60 cm below the ground surface, showed that the extracted harvested field had similar mean percent volume soil moisture estimates (~60 %) to the reference hollows (Figure 2.3.G). Soil moisture measurements taken at the same locations at the end of July, 2021, showed that then extracted harvested field remained moist, while the reference hollows had dried to levels on par with the reference hummocks (Figure 2.3.H).

As with the soil moisture, the rust depth at Site R1 was variable depending on microtopography (Figure 2.1.B). Site R1 hummocks were aerated near the surface only when ice was present in June, then the rust depth deepened to 20 – 40 cm below the hummock surface once the ice melted (Figure 2.1.B). Site R1 hollows had shallower rust depths (0 – 20 cm) compared to the hummocks and appeared to rest at or just above the water table when water levels were high (Figure 2.1.B). However, when water tables lowered beyond 20 – 30 cm, the rust depth did not deepen beyond 20 cm below the hollow regardless of how low the water table dropped (Figure 2.1.B). Pre-harvest, Sites H1 and R1 had similar rust depths (Figure 2.1.C). At the end of the intensive study in July 2021, rust depths were greatest in the reference hummocks and lowest in the reference hollows (Figure 2.3.I).

Following the removal of surface vegetation in 2019, the rust depth at Site H1 in the harvested peat field was similar to the hollows at Site R1 in the reference field, resting within 20 cm of the ground surface, at the approximate water table depth (Figure 2.1.C). Once extraction was fully underway in 2021, the rust depth was between 25 – 40 cm below the ground surface at Site H1 (Figure 2.1.C; Figure 2.3.I), and remained relatively shallow at Site H2, even when the water level dropped (Figure 2.1.C) During July 2021, the hottest and driest part of the year, the median rust depth in the harvested field was ~25 cm below the ground surface, in between the reference hummock and hollow medians (Figure 2.3.I).

2.3.3 Physicochemical Conditions

2.3.3.1 Indicators of Decomposition

2.3.3.1.1 pH

Seasonal and interannual variations in the reference peatland (Site R1) are illustrated in Figure 2.4. The surface and deep porewater pH values at Site R1 in the reference field varied between years, but were generally between 3.5 – 4.0 and 4.5 – 5.5, respectively (Figure 2.4.A). Pre-Harvest, Site H1 had slightly lower pH values in surface and deep porewaters compared to Site R1 (Figure 2.4.A). The pH measurements conducted on dried peat sampled from 0 - 10 cm in the reference hummocks and hollows had similar median values (~3.8) (Figure 2.2.E). While the median pH values at 10 – 20 cm were higher in the reference hollows (~3.8) compared to the hummocks (~3.4), they were not found to be statistically different (Figure 2.2.F).

Following the commencement of vegetation mulching in 2019, the pH values in the harvested field (Site H1) surface and deep porewaters were similar throughout the year (~3.5 – 4.0) (Figure 2.4.A). During this time, surface porewater pH values at Site H1 resembled Site R1; however, as observed pre-harvest, the deep porewater pH values at Site H1 remained lower than Site R1 (Figure 2.4.A).

Once the internal ditches were installed and the harvested field was operational, the pH values varied considerably depending on spatial location within the harvested field (Figure 2.4.A). At Site H1, within the harvested field, the pH of the surface porewater was ~4.0 (Figure 2.4.A). During spring, the pH values of the Site H1 surface porewaters were lower than Site R1 values, and did not resemble the snow water; however, the pH values reduced and resembled Site R1 surface porewaters during the summer and autumn (Figure 2.4.A). Deep porewater at Site H1 hovered near 4.5 and was similar to Site R1 in the summer, but were lower than Site R1 in the autumn (Figure 2.4.A). In contrast, the pH values in the water at Site H2 had wider range compared to Sites R1 and H1. Surface waters ranged from pH 3.9 – 6.3, and deep waters ranged from pH 4.7 – 5.7 (Figure 2.4.A). While the deep porewaters at Site H2 rose gradually over the season, the surface porewaters resembled snow water in spring, lowered to values on par with Sites R1 and H1 in late-June and July, then rose to their peak in the autumn (Figure 2.4.A). Dried peat samples taken from the top 10 cm and from 10 -20 cm below the ground within the harvested field in October 2021 had median pH values of 3.6, and were statistically insignificant compared to peat from the reference hummocks and hollows (Figure 2.2.E,F).

2.3.3.1.2 Temperature

Reference (Site R1) porewater temperatures were fairly stable and predictable over the study period. Surface and deep porewater temperatures at Site R1 ranged from 0 °C – 10 °C and 2.5 °C – 7 °C, respectively, and temperatures during the pre-harvest autumn at Sites R1 and H2 were indistinguishable from each other (Figure 2.4.B).

Following the installation of the perimeter ditch and removal of surface vegetation, the water temperatures at the harvested field were greater than the reference field (Figure 2.4.B). The temperature of the surface porewater at Site H1 was substantially higher (2.5 °C – 15 °C) relative to Site R1 during the spring and summer months (Figure 2.4.B). To a lesser extent, the deep porewater temperatures at Site H1 were also higher than deep porewaters at Site R1 (Figure 2.4.B).

After internal ditches were installed and extraction was underway, the porewater temperatures in the harvested field remained higher than the reference field, and varied by site location (Figure 2.4.B). Site H1 surface porewaters had warmer temperatures compared to Site R1; however, no water was available for collection between the middle of June and the end of September for further comparison during the summer and early autumn (Figure 2.4.B). Deep porewater temperatures at Site H1 were slightly lower than Site H2 within the internal ditch, but were still higher than Site R1 throughout the year (Figure 2.4.B). Surface water temperatures at Site H2 were substantially higher than both Site R1 and H1, with values between 16 °C – 21 °C during June and July (Figure 2.4.B). Deep water temperatures at Site H2 (~10 °C) were higher than Site R1 (2.5 °C – 7 °C), but did not fluctuate substantially until the end of October (Figure 2.4.B).

The surface temperatures measured in the reference during the intensive study were highly influenced by the above average air temperatures experienced during late June and July in the summer of 2021, yet, significant micro-topographical differences were still observed. In late-June, 2021, median surface temperatures on the reference hummocks were higher (~30 °C) than the reference hollows (~23 °C), although they were not found to be statistically different (Figure 2.3.C). At the end of the intensive study in late-July, 2021, the median surface temperatures on the reference hummocks were ~25 °C, while the hollows were ~20 °C (Figure 2.3.D). Temperatures 10 cm below ground in the reference hummocks were constant at the start and end of the intensive study (~20 °C), and were significantly higher than the reference hollows at the start (~5 °C) and end (~8 °C) of the intensive study (Figure 2.3.E,F).

Peat surface temperatures in the harvested field were consistently higher than the reference field, but below ground temperatures did not follow the same trend (Figure 2.3). At the start of the intensive study in late-June, peat surface temperatures on the surface of the harvested field (mean ~33 °C) were similar to temperatures on the reference hummocks, and significantly higher than temperatures in the reference hollows (Figure 2.3.C). Temperatures remained high at the harvested field over the intensive study period (~36 °C), and were significantly higher than both reference hummocks and hollows by the end of July 2021 (Figure 2.3.D). In contrast, temperatures measured 10 cm below the ground in the harvested field in late-June (~8 °C), were significantly lower than the reference hummocks, and similar to the reference hollows (Figure 2.3.E). By late-July, the temperatures 10 cm below the ground at the harvested field (~12 °C), were still significantly lower than the reference hummocks, but were statistically greater than the reference hollows (Figure 2.3.F).

2.3.3.1.3 Dissolved Organic Carbon and Potassium

The DOC and K concentrations in the reference field (Site R1) had little seasonal and interannual variation. Throughout the study, DOC concentrations in surface and deep porewaters at Site R1 were similar (75 – 100 mg L^{-1}) over time, except for the surface water during the spring, which was closer to the snow water DOC concentration (Figure 2.4.C). Pre-harvest, Site H1 DOC concentrations were indistinguishable from Site R1 (Figure 2.4.C). The K concentrations in surface and deep porewaters at Site R1 remained low over all years (< 1 – 1 mg L^{-1}), and pre-harvest, Sites H1 and R1 were comparable (Figure 2.4.D).

Following the vegetation removal, the DOC and K concentrations in the harvested field (Site H1) increased relative to Site R1 (Figure 2.4.C,D). Dissolved organic carbon concentrations in Site H1 surface and deep porewaters were higher (100 – 160 mg L^{-1}) than at Site R1, with the highest values occurring during May when mulching commenced (Figure 2.4.C). The K concentrations in surface and deep porewaters at Site H1 were also elevated (2 – 7 mg L^{-1}) compared to Site R1 (Figure 2.4.D). Potassium concentrations spiked in the surface porewater after mulching commenced, and in the deep porewater after a late-July rain event in 2019 (Figure 2.4.D).

Once internal drains were installed and extraction was active, the DOC and K concentrations continued to be greater in the harvested field (Site H1) and internal ditch (Site H2) relative to the reference field (Site R1) (Figure 2.4.C,D). In early spring, the DOC concentrations in surface porewaters at Sites H1, H2, and R1 were similar to snow, but concentrations increased at Sites H1 and H2 starting in June relative to Site R1 (Figure 2.4.C). Deep porewater DOC concentrations remained elevated at Site H1 compared to Site R1, with even higher concentrations found at Site H2 in the internal ditch water (Figure 2.4.C). The K concentrations in the surface porewater within the internal ditch at Site H2 were similar to snow water during the spring, while the surface porewaters at Site H1 had higher concentrations (3 mg L^{-1}) compared to Site R1 (Figure 2.4.D). However, once the snow melted, both the surface and deep waters at Sites H1 and H2 had higher K concentrations compared to Site R1, with the highest values occurring in the Site H2 surface porewater (7 mg L^{-1}) in mid-June (Figure 2.4.D).

2.3.4 Indicators of Geochemical Influence in Porewaters

2.3.4.1 Electrical Conductivity

Except for during the early spring, when surface porewaters resembled snow water, the surface and deep porewater EC values were fairly constant in the reference field (Site R1), and ranged from 50 – 125 µS cm^{-1} during the study period (Figure 2.5.A). Preharvest, Site H1 had slightly higher surface porewater EC values and slightly lower deep porewater EC values compared to Site R1 (Figure 2.5.A).

Following the removal of surface vegetation during 2019, EC values increased slightly in the harvested field (Site H1) surface porewater (~100 µS cm^{-1}) relative to Site R1 (50 – 75 µS cm^{-1}) (Figure 2.5.A). In contrast, EC values in the deep porewater at Site H1 remained similar to Site R1 (Figure 2.5.A).

During extraction, after internal ditches were installed, the EC values varied between the harvested field (Site H1) and internal ditch (Site H2) locations. The EC values in the surface and deep porewaters at Site H1 were similar to Site R1 (Figure 2.5.A). Similarly, Site H2 had comparable EC values in the surface and deep porewaters to both Site H1 and R1 until the end of June, 2021 (Figure 2.5.A). However, after June 2021, the surface porewaters at Site H2 increased to 150 – 300 µS cm^{-1} and the deep porewaters increased to 150 – 200 µS cm^{-1} (Figure 2.5.A).

At the reference (Site R1), the low EC values observed in surface and deep porewaters contrasted with the EC measured in dried peat samples taken from the surface hummocks and hollows, which were comparably higher (Figure 2.2.G,H). The median EC values obtained from dried peat sampled from the top 10 cm in the hummocks and hollows was significantly higher (~400 µS cm^{-1}) than in the hollows (~325 µS cm^{-1}) (Figure 2.2.G). Although the EC values from dried peat collected 10 – 20 cm below the ground surface were still high compared to the porewater EC, the hummocks and hollows were similar (~350 µS cm^{-1}) (Figure 2.2.H). In contrast to the reference field, EC values from dried peat collected from 0 – 10 cm below the ground in the harvested field more closely resembled the porewater EC from the same relative location. Surface peat from the harvested field had EC values statistically lower than peat from the reference hummocks and hollows, with median EC values of ~150 µS cm^{-1} in both the top 10 cm and from 10 – 20 cm below the ground surface (Figure 2.2.G,H).

2.3.4.2 Calcium, Magnesium, and Sodium

The concentrations of Ca and Mg followed similar trends throughout the study (Figure 2.5.B,C). There was minimal seasonal variability in Ca and Mg concentrations in the reference field (Site R1), except for in surface porewaters during the spring, which resembled snow water (Figure 2.5.B,C). The Ca and Mg concentrations at Site R1 were slightly higher in deep compared to surface porewaters during 2018 and 2019, but the difference between surface and deep porewaters was more pronounced during spring and autumn 2021 (Figure 2.5.B,C). Surface porewaters resembled snow during the spring, then rose to ~5 mg L^{-1} of Ca and ~1.5 mg L^{-1} of Mg by June (Figure 2.5.B,C). For the duration of 2021, deep porewaters had Ca and Mg concentrations between 7 – 15 mg L^{-1} and 2 – 3.5 mg L^{-1} for the duration of 2021 (Figure 2.5.B,C). Pre-harvest, Site H1 had slightly lower Ca and Mg concentrations compared to Site R1 (Figure 2.5.B,C). Sodium concentrations did not appear to vary seasonally at Site R1. Aside from the high Na concentration (4 mg L^{-1}) in the deep porewater in 2018, the surface and deep porewaters were maintained near 1 mg L^{-1} for the duration of the study (Figure 2.5.D).

After vegetation removal was underway, surface and deep porewaters in the harvested field (Site H1) had similar Ca and Mg concentrations relative to R1 (Figure 2.5.B,C). There was a strong Na concentration peak (~7 mg L^{-1}) following vegetation removal at Site H1, then the concentration lowered on par with Site R1 (Figure 2.5.D).

Once internal ditches were installed and extraction was ongoing, the Ca and Mg concentrations for both surface and deep porewaters were similar in the harvested (Site H1) and reference (Site R1) fields (Figure 2.5.B,C). The internal ditch (Site H2) behaved similarly to Site R1 until mid-July, when the concentrations of Ca and Mg were much higher in the surface and deep porewaters compared to Sites R1 and H1 (Figure 2.5.B,C). The Na concentrations at Site H1 surface porewaters were the same as Site R1; however, the deep porewater at Site H1 were much higher (Figure 2.5.D). At Site H2, the Na concentrations in the surface porewaters were similar to Site R1 during the spring and early summer (0.5 – 1 mg L^{-1}), but rose in comparison to Site R1 during the late summer and autumn (2 -2.5 mg L^{-1}) (Figure 2.5.D). Deep porewaters at Site H2 were slightly higher relative to Site R1 (Figure 2.5.D).

2.3.4.3 Aluminum and Iron

There was minimal seasonal variability in Al and Fe concentrations in the reference field (Site R1), except for in surface porewaters during the spring, which resembled snow water (Figure 2.6.A,B). The Al concentrations in surface and deep porewaters at Site R1 were similar (~100 – 200 µg L^{-1}) over the study period (Figure 2.6.A). Pre-Harvest, Site H1 surface porewaters were similar to Site R1 surface porewaters, while the Site H1 deep porewaters had slightly higher Al concentrations than Site R1 deep porewaters (Figure 2.6.A). The Fe concentrations were slightly higher in deep porewater compared to surface porewaters at Sites R1 and H1, pre-harvest (Figure 2.6.B).

After vegetation removal was underway, surface and deep porewaters at Site H1 had higher Al concentrations compared to Site R1 (Figure 2.6.A). The Fe concentrations remained unchanged between Site H1 and Site R1, with no seasonal fluctuation (Figure 2.6.B).

Once internal ditches were installed and extraction was ongoing, Al and Fe concentrations increased in the internal ditch but not in the harvested field, relative to the reference field. Surface porewaters at Sites H1 and H2 had similar Al concentrations to Site R1 (Figure 2.6.A). However, in deep porewaters, Site H1 had slightly lower Al concentrations compared to Site R1, while Site H2 had substantially higher concentrations in July compared to Site R1 (Figure 2.6.A). The springtime surface porewater at Sites H1, H2, and R1 had Fe concentrations resembling snow; concentrations remained low (< 2 mg L^{-1}) for Sites H1 and R1, but increased (5 mg L^{-1}) during the summer at Site H2 (Figure 2.6.B). Deep porewaters at Sites H1 and H2 had higher Fe concentrations compared to R1, and levels were especially high within the internal ditch at Site H2 in July (> 15 mg L^{-1}) (Figure 2.6.B).

2.3.4.4 Chloride

The chloride concentrations in the reference field (Site R1) surface and deep porewaters were fairly constant over the study period (Figure 2.6.C). Except for increases in the surface porewater during June 2019 and deep porewater in July 2021, the concentration was maintained ~0.25 mg L^{-1} (Figure 2.6.C). Pre-harvest, the Cl concentrations at Site H1 were lower relative to Site R1 (Figure 2.6.C).

Once the perimeter ditch was installed and the vegetation was removed in the harvested field, the Cl concentrations followed similar trends to K and, to a lesser extent DOC concentrations (Figure 2.4.C,D; Figure 2.6.C). The Cl concentrations in surface and deep porewaters at Site H1 were higher (0.1 – 1.4 mg L^{-1}) relative to Site R1 (Figure 2.6.C). Chloride concentrations peaked in the surface porewater in May, and in the deep porewater in late-July, 2019 (Figure 2.6.C).

After the internal ditches were installed and peat extraction was underway, the harvested field continued to have higher Cl concentrations relative to the reference field. Chloride concentrations in Site H1 surface and H2 surface and deep porewaters were similar in the spring and early summer (~0.5 – 0.9 mg L^{-1}), and were much higher compared to the snow and Site R1 porewaters (Figure 2.6.C). Site H2 continued to have higher Cl concentrations over the summer, but returned to levels comparable to Site R1 in the autumn (Figure 2.6.C). In contrast, the deep porewater Cl concentrations at Site H1 were considerably lower relative to Site H2, and resembled Site R1 deep porewaters during the driest part of the summer (Figure 2.6.C).

2.3.4.5 Silica

Silica concentrations were only measured in 2021. Surface porewaters in the reference peat field (Site R1) were much lower in the spring during snow melt, and rose gradually over the summer to 3 mg L^{-1}, whereas deep porewaters stayed fairly constant (3 – 5 mg L^{-1}) regardless of the season (Figure 2.6.D).

Silica concentrations appeared to differ between the surface and deep porewaters in the harvested field; however, ice in the spring and the low water table in the summer precluded sampling and direct comparison of deeper and surface pore water, respectively. Harvested field (Site H1) and internal ditch (Site H2) surface porewaters had low Si concentrations that resembled Site R1 surface porewater in the spring and summer; however, the surface porewater at Site H2 increased above 8 mg L^{-1} in autumn after the internal ditches were cleared (Figure 2.6.D). The deep porewaters at Site H2 resembled Site R1 deep porewater in late spring, but the Si concentrations at Sites H1 and H2 were considerably higher (~8 mg L^{-1}) relative to Site R1 deep porewater in the summer (Figure 2.6.D).

2.3.5 Porewater Nutrients

2.3.5.1 Dissolved Phosphorus

Dissolved P concentrations in the reference peat field (Site R1) were consistently low, and exhibited little, if any, seasonal and interannual variability (Figure 2.7.A,B). Over the study, the TDP concentrations within the surface waters were between 0 – 50 µg L^{-1} (Figure 2.7.A), and the SRP values were at or near 0 µg L^{-1} (Figure 2.7.B). Deep porewaters had higher TDP values during autumn 2018, and spring in 2019 (~130 µg L^{-1}), but otherwise remained near 50 µg L^{-1} for the remainder of the study (Figure 2.7.A). The SRP concentrations in deep porewater at Site R1 were more variable. In 2018 and 2019, nearly all of the TDP was comprised of SRP, whereas comparatively little SRP was measured in the deep porewater in 2021 (Figure 2.7.A,B). Pre-harvest, Site H1 resembled Site R1 (Figure 2.7.A,B).

Once the vegetation was mulched in the harvested field, dissolved P concentrations increased relative to the reference field. At site H1, the TDP concentrations in the surface porewaters were elevated during May sampling (~250 µg L^{-1}), immediately following mulching, compared to Site R1, but returned to reference concentrations by summer (Figure 2.7.A). Deep porewater TDP concentrations at Site H1 peaked at the end of July (~275 µg L^{-1}) compared to Site R1 and dropped back to ~100 µg L^{-1} in autumn (Figure 2.7.A). The SRP concentrations at Site H1 followed a similar trend to the TDP concentrations (Figure 2.7.B). Surface porewaters had high SRP concentrations in May and constituted approximately 50 % of the TDP, whereas the deep porewater SRP concentrations were highest in July, and made up approximately 75 % of the measured TDP (Figure 2.7.A,B).

Dissolved phosphorus concentrations were considerably higher in the harvested peatland relative to the reference field once the internal ditches were installed and the field was being actively extracted. In early spring, the TDP and SRP concentrations in the surface porewater in the internal ditch (Site H2) were seasonally low, indicating influence from low concentrations in the snow (Figure 2.7.A,B). However, once the snow melted, the TDP concentrations in the surface porewater at Sites H1 and H2 rose substantially, and peaked at ~950 $\mu g\ L^{-1}$ and in the summer (Figure 2.7.A). The proportion of SRP included within the surface porewater TPD concentration at Sites H1 and H2 was substantial. As with TDP, the surface porewater SRP concentrations peaked in the summer at ~650 $\mu g\ L^{-1}$ and ~750 $\mu g\ L^{-1}$, at Sites H1 and H2 respectively (Figure 2.7.B). At Sites H1 and H2, the deep porewater TDP and SRP concentrations were lower than the surface porewaters at the same site, but were still considerably higher relative to Site R1 (Figure 2.7.A,B). The deep porewater TDP and SRP concentrations peaked in mid-July at Site H1 (~575 $\mu g\ L^{-1}$ and ~375 $\mu g\ L^{-1}$) and at Site H2, (~250 $\mu g\ L^{-1}$ and ~175 $\mu g\ L^{-1}$), respectively (Figure 2.7.A,B).

2.3.5.2 Dissolved Nitrogen

Dissolved nitrogen concentrations varied little over the study period in the reference field. Except for during the spring, when surface waters resembled concentrations measured during the snow survey, there was very little seasonal, and no discernable interannual, variability in TDN concentrations in the reference peat field (Site R1) (Figure 2.7.C). The TDN at Site R1 was between 1000 – 1500 $\mu g\ L^{-1}$ for both surface and deep porewaters (Figure 2.7.C). Very little of the dissolved nitrogen within the TDN concentration could be attributed to NH_4^+-N. Ammonium concentrations were low (25 – 50 $\mu g\ L^{-1}$) and resembled the snow water concentrations for both surface and deep porewaters at Site R1; there was no clear seasonal or interannual variability (Figure 2.7.D). Nitrate concentrations were detected in the snow water in early spring, but were otherwise not detected in the surface or deep porewaters at Site R1 during the study period (Figure 2.7.E). Nitrate concentrations at Site H1 resembled Site R1 during the fall, pre-harvest (Figure 2.7.E).

Immediately following vegetation mulching in May 2019, increased dissolved N concentrations were observed in the harvested field relative to Site R1. The TDN concentrations in the surface and deep porewaters behaved similarly to the DOC and K concentrations during the same time period (Figure 2.4.C,D; Figure 2.7.C). The TDN concentrations at Site H1 surface porewater were greater than Site R1, with a peak around 3000 µg L^{-1} in May immediately following the initiation of vegetation mulching (Figure 2.7.C). Deep porewaters at Site H1 were also higher than Site R1, and were ~2000 µg L^{-1} in summer and autumn (Figure 2.7.C). The NH_4^+-N concentrations increased very slightly in Site H1 surface porewater compared to Site R1 in May 2019, while deep porewaters at Site H1 remained similar to Site R1 during summer and autumn (Figure 2.7.D). No nitrate was detected at Site H1 in 2019 (Figure 2.7.E).

After internal ditches were installed and the harvested field was under active extraction, the TDN, NH_4^+-N, and NO_3^--N concentrations increased dramatically after spring snow melt in March in the harvested field relative to Site R1. Site H1 TDN concentrations in the surface porewaters were higher than snow at the end of March (1100 µg L^{-1}), resembled snow and Site R1 TDN concentrations in the beginning of April (~250 µg L^{-1}), but were elevated compared to Site R1 by June (~3000 µg L^{-1}) (Figure 2.7.C). Site H1 deep porewater TDN concentrations were higher relative to Site R1 deep porewater in spring and summer, and peaked at ~2750 µg L^{-1} in July (Figure 2.7.C). At Site H2, surface porewater during spring resembled Site R1 and snow waters; however, concentrations rose above 3250 µg L^{-1} in July (Figure 2.7.C). The deep porewater at Site H2 had much higher concentrations of TDN (5000 µg L^{-1}) relative to Site R1 (Figure 2.7.C). The NH_4^+-N concentrations in Site H1 surface porewaters were higher than snow at the end of March (300 µg L^{-1}), resembled snow and Site R1 in the beginning of April, and were elevated compared to Site R1 in June (~600 µg L^{-1}) (Figure 2.7.D). Deep porewater at Site H1 had higher NH_4^+-N concentrations compared to Site R1, with the highest concentrations in July (~950 µg L^{-1}) (Figure 2.7.D). Site H2 surface water resembled Site R1 throughout the summer, but NH_4^+-N concentrations increased substantially in the autumn (~450 µg L^{-1}) (Figure 2.7.D). Site H2 deep porewater NH_4^+-N concentrations remained greater than concentrations at Site R1, and were constant (~600 µg L^{-1}) through spring and summer (Figure 2.7.D). Nitrate surface porewater concentrations at Site H1 were more than twice as high as the snow in late March (~400 µg L^{-1}), but were not detected for the rest of the year (Figure 2.7.E). The NO_3^--N concentrations in Site H2 surface porewaters in early March resembled concentrations measured during the snow survey, but, were also not detected for the rest of the year (Figure 2.7.E). Deep porewaters at Sites H1 and H2 had no detectable NO_3^--N concentrations (Figure 2.7.E).

2.3.6 In-field Chemical Availability Study

2.3.6.1 Major Ion and Nutrient Availability

During the in-field chemical availability study, from late-June to late-July 2021, the nutrient availability measured by PRS® probes showed clear differences between hummock and hollow microtopography, and higher nutrient availability was measured within the harvested field relative to the reference hummocks and hollows (Figure 2.8). Ammonium, PO_4^{3-}-P, K, Al, Mn, Ca, and Mg availabilities were highest in the harvested field (Figure 2.8). Ammonium and K availabilities showed similar trends; both had significantly higher availability rates in the harvested field and very low rates in the reference hummocks and hollows (Figure 2.8.A,C). The PO_4^{3-}-P and Al availability in the harvested field was statistically higher than the reference hummocks, but was not significantly different from the reference hollows (Figure 2.8.B,D). The Mn availability in the harvested field was highly variable, but was similar to the reference hollows and significantly greater than the reference hummocks (Figure 2.8.F). The Ca and Mg availabilities were similar to each other, with the highest rates found in the harvested field and the lowest in the reference hummocks (Figure 2.8.H,I). For Ca and Mg, the harvested field, reference hummocks, and reference hollows availabilities were all significantly different (Figure 2.8.H,I).

In contrast, Fe and SO_4^{2-}-S availabilities were highest in the reference hollows and comparatively low in the reference hummocks and harvested field (Figure 2.8.E,G). The Fe availability within the surface peat was significantly higher in reference hollows compared to reference hummocks, while the harvested field had Fe availability rates in between the hummocks and hollows (Figure 2.8.E). Sulfate availability was significantly higher in the reference hollows and appeared to have similar trends in availability as Fe; no SO_4^{2-}-S was detected in the reference hummocks, and very little was available in the harvested field (Figure 2.8.G). Nitrate (not shown) was not detected in the surface peat within the harvested field, or the reference hummocks and hollows.

2.3.6.2 Integrating Peat Physicochemical Properties and Chemical Availability

When considering the physicochemical conditions that may control the nutrient and ion availability in the surface peat, the harvested field, reference hummocks, and reference hollows appear to represent distinct populations, with the harvested field more closely resembling the reference hollows (Figure 2.9). During the intensive study from late-June to late July, 2021, the reference hummocks were associated with warmer subsurface temperatures, higher C:N, deeper ice, and deeper rust depths (Figure 2.9). The reference hollows, by contrast, were associated with shallower ice and rust depths, cooler subsurface temperatures, and lower C:N, which corresponded to higher SO_4^{2-}-S and Fe availability rates (Figure 2.9).

The extracted field was characterized by higher surface temperatures, bulk density, and soil moisture, and lower EC and pH within the peat 0 – 10 cm below the ground surface (Figure 2.9). High surface temperatures, in the harvested field were associated with higher PO_4^{3-}-P availability rates in the extracted field (Figure 2.9). Similarly, high bulk density was associated with higher K, NH_4^+-N, Mg, and Ca availability rates in the extracted field (Figure 2.9). Finally, the higher soil moisture observed in the extracted field was associated with higher NH_4^+-N, Mg, and Ca availability rates (Figure 2.9).

2.4 DISCUSSION

The harvested peatland showed little change in in-field nutrient concentrations during opening, but nutrient concentrations and availability increased following internal ditching and early extraction. This said, the nutrient concentrations in the internal ditches during extraction were dynamic and did not mirror the in-field values. This suggests that additional processes, such as accelerated decomposition within the internal ditch, could govern the outflow nutrient concentrations. Therefore, internal ditches may be a major contributing source of nutrients downstream, particularly following large rain events.

This study presents some of the first in-field water chemistry data associated with the opening and early extraction periods of horticultural peat harvesting. In addition, potential processes that may govern the chemical concentration and availability of nutrients were explored, addressing critical knowledge gaps and providing much needed information to help understand the nutrient dynamics within harvested peatlands. Due to the lack of literature on this topic, comparison with additional studies was difficult. Thus, this research provides a starting point to explore both the potential nutrient concentrations as well as the processes that may govern nutrient availability and mobility when assessing nutrient leaching risk at opened and newly extracted peatlands within the context of their biogeoclimatic setting.

2.4.1 Perimeter Ditching and Vegetation Mulching

2.4.1.1 Effect of Perimeter Ditching Within the Peat Field

The influence of the perimeter ditch on the overall water table draw-down within the entire harvested field appeared to be negligible during 2019 because water levels within the harvested field were higher, not lower, relative to the reference field. The impact of initial opening a peatland field was largely influenced by mulching surface vegetation and leaving the biomass in place, and is discussed in the next section. Although minimal impact to the water levels in open peat field was observed, perimeter ditching did impact the magnitude and quality of the receiving outflow waters. This is discussed in Chapter 3.

Other studies have observed a similar lack of ditch influence on the water table within peatlands, albeit with large variations in the distance affected by drainage. Prevost et al. (1997) found that water tables were lowered within 15 m of drainage ditches in a black-spruce forested peatland in Quebec, Canada, but that water levels measured at distances beyond 15 m remained unaffected. Boelter (1972) observed a range in ditch effectiveness based on the hydrologic conductivity of the peat. Ditches in moderately decomposed peat with a bulk density of 0.17 g cm^{-3} and a hydraulic conductivity of 2.2 x 10^{-5} cm s^{-1} had no effect on the water table 5 m away from the ditch, while ditches in poorly decomposed peat with a bulk density of 0.08 g cm^{-3} and a hydraulic conductivity of 7.2 x 10^{-2} cm s^{-1} saw lowered water tables 50 m away from the ditch (Boelter, 1972). Given these previous findings, the estimated saturated hydraulic conductivity and bulk density of the peat in this study suggests that water tables could be affected up to 50 m away from the perimeter ditch. However, the perimeter ditch would still have little effect on the majority of the harvested field due to the large area and the comparatively small edge effect. With the limited hydrological connectivity between the harvested field area and the perimeter ditch, it is unlikely that any changes to the porewater nutrient concentrations within the harvested field would be reflected at the harvesting operation outflow.

2.4.1.2 Effect of Vegetation Mulching Within the Peat Field

2.4.1.2.1 In-field Water Level and Temperature Alterations

Mulching the surface vegetation was associated with slightly higher water levels within the harvested field relative to the reference field. This was likely due to a combination of light compaction from the mulching equipment (Gauthier et al., 2022), and from the elimination of actively transpiring living vegetation (Leppä et al., 2020; Sarkkola et al., 2010, 2013). Previous studies have underscored the role of vegetation as water table regulators in peatland forests. For example, Leppa et al. (2020) found that cutting up to 50% of the surface vegetation caused the water table to rise by 15 – 40% in a forested peatland with drainage ditches spaced 50 m apart. Considering that the entire harvested field in this study was cleared and mulched while the ground was frozen, and only a perimeter ditch was installed around the border of the field for drainage, it is therefore likely that the increase in water level observed in the harvested field relative to the reference field was due to the cessation of transpiration.

The total removal of shade after the vegetation was mulched resulted in notably higher surface porewater temperatures in the harvested field, while the deep porewaters were only slightly higher than the reference and remained fairly constant. Surface temperatures have been shown to significantly increase following vascular vegetation removal in other peatlands. Leonard et al. (2018) observed a significant increase in surface temperature when all vascular vegetation was removed, relative to sites where only trees were removed or where vegetation remained intact in a poor fen in Alberta, Canada. Similarly, wooded peatlands that had been clearcut within 1 – 4 years in southern Manitoba, Canada had higher surface water temperatures compared to non-harvested control peatlands (Locky & Bayley, 2007). Several studies have suggested that higher soil and water temperatures after vegetation clearing have resulted in increased decomposition and accumulations of nutrients in peatland soils and water (Locky & Bayley, 2007; D. M. Morris, 2009). However, in this study, sustained accumulations of nutrients were not measured in the harvested field porewater, suggesting that higher levels of decomposition did not occur immediately after vegetation removal.

2.4.1.2.2 In-field Porewater Nutrient Concentrations

In response to the addition of fresh, mulched vegetation, short pulses of DOC and K that corresponded with similar trends in TDP, SRP, and TDN concentrations were measured directly following mulching in the spring, and after a large rain event at the end of July 2019. However, these peaks were not sustained over the 2019 sampling season, and showed clear decreases in concentrations following each peak. The source of DOC, K, TDP, SRP, and TDN was almost certainly from fresh plant material and cell lysis following the mechanical mulching of the surface vegetation in the harvested field. Damman (1978) observed that P and K were associated with living sphagnum and were found in high concentrations in surface peat, and fresh peatland forest residue has been shown to be high in N and P (Hyvönen et al., 2000; Tolvanen et al., 2020). Further, fresh litter has been shown to augment concentrations of DOC and DON in clearcut peatland forests (D. M. Morris, 2009).

However, the pulses of TPD and SRP concentrations in the harvested field surface waters were only moderately higher than the reference, whereas the TDN, DOC, and K concentrations in surface porewaters were much higher relative to the reference. The presence of P in general could be due in part to the dissociation of P and Fe complexes under anoxic conditions. Rust depths, as an indicator of peat aerobic conditions, and redox-sensitive Fe concentrations were similar within the harvested and reference fields. Given the low oxygen environment and the higher water tables present in the harvested field, iron hydroxide reduction in anaerobic waters, as previously observed in peatlands and riparian areas (Aldous et al., 2007; Carlyle & Hill, 2001; Niedermeier & Robinson, 2009), may have been responsible for the peaks in TDP and SRP. However, the Fe concentrations in the harvested field were low, and did not vary alongside the dissolved P in the harvested field. This suggests that the P detected in the harvested field porewaters was not due to Fe reduction. Instead, the pulse of P observed in the harvested field was likely a result of cell lysis following the mulching of the surface vegetation (Reddy et al., 2005), especially considering that P has been shown to readily leach from logging residues in clearcut peatlands (Palviainen et al., 2004). However, P pulses were moderate compared to other nutrients, which could be because P availability is extremely limited in nutrient poor peatlands (Damman, 1978). Thus, any P released during mulching may have been rapidly immobilized by microbes in high C:N substrates and removed from the porewater.

The lack of sustained, high concentrations of TDP, SRP, and TDN, coupled with the very low NH_4^+-N concentrations present in the porewater, suggests that nutrients remained limiting within the harvested field even after the vegetation was removed, and that available nutrients were likely rapidly immobilized by microbes. Although live vegetation has been shown to play a key role in nutrient removal in wetland ecosystems (De Mars et al., 1996; Jabłońska et al., 2021; Walbridge & Lockaby, 1994), it appears that their role in removing nutrients from the porewater may not be as important as previously assumed because nutrient concentrations decreased in the harvested field after the vegetation was removed instead of increasing as predicted. Instead, the rapid decrease in nutrient concentrations was likely due to immobilization by microbes within the nutrient limiting environment, as described in a clearcut peatland system by Westbrook and Devito (2004).

The lack of sustained high nutrient concentrations could also be due to poor substrate quality. Forested peatlands on the Canadian Boreal Plains are dominated by black spruce which is reportedly high in phenolic compounds that slow decomposition (Alshehri et al., 2020) and could contribute to low evidence of decomposition in the harvested field. Initial boosts in microbial activity due to the labile material directly following mulching, followed by slowed decomposition were observed in a microcosm study by Alshehri et al. (2020). Thus, the phenolics in mulched black spruce could be quickly muting any potential increase in microbial decomposition and may limit the nutrients detected in peat porewaters. Similarly, Palviainen et al. (2004) found that logging residues did not consistently result in higher decomposition and nutrient release. The low evidence of decomposition could also be because insufficient time had elapsed for microbial populations to increase, or the environmental conditions (C:N, pH, anoxia) were still unfavorable for rapid microbial population growth (Holden et al., 2004). Despite the influx of nutrients from the mulched vegetation, mulched peat hummocks mixed with wood chips would still have high C:N ratios, the water was still very acidic (pH 3.5 – 4), and the surface conditions were anoxic, all of which may have drastically limited the microbial population growth.

Further study on the microbial biomass and speciation during the transition from natural peatland to extracted peat field is needed to further understand the underlying mechanism.

2.4.1.3 In-field Source Waters

It is unlikely that different water sources were intersected that resulted in the short-term increase in nutrients in the harvested peat field following vegetation mulching. While previous studies have also observed higher EC values in flooded areas, they attributed the increase in EC to vertical solute transport from deeper soils to the surface (Biagi & Carey, 2022; Kelln et al., 2008). However, in this study, this is unlikely because the EC in the surface porewater was consistently higher than the deep porewater. This indicates that the increase in EC observed in the harvested field was not solely due to translocation of solutes from deeper peat or mineral subsoils. The increase in EC values following mulching could instead be due to the influx of ions added to the system when the vegetation was mulched, or from the dissolution of solutes that accumulated in hollow and surface vegetation that were mobilized when the water levels rose into the layer of mulched vegetation.

Further, the pH of the surface porewaters did not change when the vegetation was mulched, nor did the concentrations of Ca and Mg, suggesting that deep porewaters did not migrate upwards. There was an increase in Na concentrations following vegetation mulching, but this was likely due to inputs from the freshly mulched vegetation, and not from new water sources altering the porewater chemistry. Although Na is easily exchangeable and has been shown to be readily removed from ombrotrophic peats (Damman, 1978), the newly added Na from mulching may have been removed from the porewater through adsorption onto the surrounding peat.

Although chloride has often been used as a conservative tracer in forested ecosystems to determine water sources (Lockwood et al., 1995), Cl concentrations fluctuated alongside the K and DOC concentrations in this study, suggesting that Cl may have been influenced by biological factors. Lovett et al. (2005) found that mineralization of soil organic matter can release Cl, and Cl exports increased after vegetation disturbance. They concluded that when atmospheric Cl deposition was low, Cl was mostly governed by biology, not geochemistry (Lovett et al., 2005). Likewise, Svensson et al. (2012) found that Cl was a suitable tracer in areas with high Cl deposition, but areas with low Cl deposition had a net release of Cl, possibly due to interactions with vegetation, which made using Cl as a tracer unreliable. Thus, Cl concentrations could be used in maritime areas to indicate water source, but may not be as useful in continental areas with lower Cl concentrations, such as in this study. This said, the higher Cl concentrations that mimicked the K and DOC concentrations observed in the harvested field porewaters relative to the reference field porewaters in this study suggest that changes to the porewater chemistry were likely from biological processes or leaching from vegetation remnants and not upwelling of deeper water following vegetation removal. Therefore, Cl alongside K concentrations could be used to indicate water that has been in contact with crushed or decomposing organic matter.

2.4.2 Internal Ditching and Extraction

2.4.2.1 In-field Snow and Ice

Without vegetation and microtopography to increase the surficial elevation variability, the distribution of snow on the harvested field was shallower, more uniformly distributed, and began melting sooner than the reference field, with deep accumulations collecting in the internal ditches. Cleared areas have conversely been shown to accumulate more snow compared to forested areas (Golding & Swanson, 1986). However, the shallower depth of snow observed in the fields, and the accumulation in the ditches, indicates saltation and drifting of snow from the field to the ditches (Pomeroy & Gray, 1990), or, increased sublimation from exposure to higher levels of incident radiation (Redding & Devito, 2011). Schelker et al. (2013) found that exposed snow in a clearcut boreal forest in Sweden melted earlier compared to the non-clearcut reference. Therefore, snow that melts during warm winter days may refreeze and form ice layers at the surface. Schelker et al. (2013), also observed higher SWE in clearcut relative to uncut peatlands. However, the SWE in this study was similar in the reference and harvested fields, but much higher in the internal ditches. It is possible that increased saltation and sublimation in the harvested field counteracted the lack of vegetation interception that would otherwise limit the amount of snow accumulating on the ground. This could account for the similar SWE values relative to the reference field. The higher SWE in the internal ditches is very likely due to saltation because the ditches contained close to three times the depth of snow compared to the harvested and reference fields.

Below the snow in the extracted field, concrete ice was present above the ground surface in the late-winter and early spring. A study by Redding and Devito (2011) observed that sparsely vegetated southern sandy slopes with high hydraulic conductivity developed concrete soil frost and generated surface runoff. Unvegetated, exposed surface peat could encourage freeze-melt cycles over the winter and spring and, together with reduced drainage from higher bulk densities observed at the harvested field peat surface, explains the thick concrete ice formation observed in this study. The concrete ice in the harvested field and internal ditches enabled melting snow in contact with the surface peat to pool above the frozen ground and could be rapidly transported off site.

In contrast, the depth to ice in the hummocks was close to 40 cm below the hummock surface in the reference field, resulting in ample unfrozen area to store melting snow in April. Hummocks in a study by Bubier et al. (1998) were also found to be dry, were ice-free, and had live moss actively photosynthesizing while the hollows remained frozen and wet until early June. This occurred while the harvested field was frozen right to the ground surface. Therefore, melted snow that would otherwise infiltrate into the peat could quickly move to the frozen internal ditch and be transported to the outflow, increasing the potential for runoff and transportation of nutrients from the peatland surface. While the importance of frozen ground in snow melt runoff transportation has been previously discussed in other ecosystems (Hayashi, 2013; Price, 1987; Redding & Devito, 2011; Van Huizen et al., 2020), the stark change from a peatland system that can retain snow melt to one that cannot may have major consequences for the volume of water transported downstream in harvested peatlands.

The concentrations of nutrients in the snow meltwater sitting on top of ice in the harvested field were greater than the reference field. Surface water samples collected in early March, 2021 from both the harvested and reference fields resembled the snow, and contained both NH_4^+-N and NO_3^--N concentrations, but TDP, SRP, and TDN concentrations were below detection. However, when water samples were taken from just below the accumulated surface ice in the harvested field in late-March 2021, concentrations of TDP, SRP, TDN, NH_4^+-N, and NO_3^--N were much higher than the snow. Although cold temperatures have been shown to slow microbial activity (Williams & Crawford, 1983), NO_3^--N concentrations were more than two times higher, potentially due to freeze-thaw cycles in the surface peat that caused cell lysis (Campbell et al., 2005; Hayashi, 2013; Joseph & Henry, 2008). Therefore, with the thick, concrete ice lens present in the harvested field, the likelihood of nutrient enrichment and increased availability through physical (cell lysis) or biological processes that occurred before the peat froze, is increased. This has important implications for nutrient export during spring melt because snowmelt water in contact with the nutrient-rich surface peat can easily travel over the frozen ground and be transported to the outflow.

2.4.2.2 In-field Water Levels and Physicochemical Alterations

Water levels in the harvested field dropped rapidly following the melting ice, as observed in a previous study in the same biogeoclimatic setting near Utikuma Lake, Alberta, Canada (Petrone et al., 2008). This underscores the importance of ice as a mechanism for maintaining water tables near the ground surface in peatlands found in this sub-humid, continental climate. It is also notable that the water levels within the reference field dropped significantly during the summer of 2021 due to lack of summer rainfall. This drop in water table has been observed previously in bogs and poor fens subjected to drought conditions in the United Kingdom (Evans et al., 1999), and has been documented within the Boreal Plains Ecozone (Hokanson et al., 2020; Thompson et al., 2017). However, it appears that the water level within the harvested field declined more rapidly and remained lower relative to the reference field.

Although water levels were deeper than 90 cm below ground in the harvested field by late-July, 2021, the soil moisture measured 6 cm from the ground surface remained close to 60% saturation and the rust depth did not drop deeper than 35 cm below the ground. These data, collected after a month of very limited precipitation, suggest that water level may not be a suitable measure of the extent of peat anoxia and soil moisture, and that peat in the harvested field has substantial resiliency to drying out, likely due to capillary rise (Macrae et al., 2013). Similar results were observed by Price et al. (1997) who reported a poor relationship between the water level and the surface soil moisture during dry conditions in a drained bog. They found that even when water levels were approximately 100 cm below the ground surface, the volumetric soil moistures ranged from less than 20% to over 70% within 3 cm of the peat surface (Price, 1997).

Once internal ditches were installed and extraction had commenced, temperatures in surface and deep porewaters in both the harvested field and internal ditch were higher relative to the reference field, particularly in the internal ditch surface porewater. Although the sub-surface temperatures measured 10 cm below the ground in the summer were in between the reference hummocks and hollows, the surface temperatures at the harvested field were higher, and, when paired with moist conditions, may have enabled accelerated decomposition in the harvested field. High temperatures were better indicators of increased DOC and DON concentrations in clearcut peatlands in Ontario, Canada (D. M. Morris, 2009), and higher surface water temperatures were credited with increasing concentrations of NH_4^+-N, TP, and SRP in recently clearcut peatlands (Locky & Bayley, 2007).

The aerobic conditions, moist soil, and warm temperatures appear to have provided prime conditions for decomposition in the harvested field and internal ditches. While direct measurements of decomposition were not measured in this study, K and Cl concentrations were much higher in the harvested field relative to the reference field, and the DOC and K concentrations were even higher in the internal ditches. This suggests that peat decomposition was likely occurring. Potassium has been shown to rapidly leach from the peat profile (Damman, 1978), so sustained, elevated concentrations could indicate ongoing release of K from within the peat via decomposition. The Cl concentrations were also greater in the harvested field relative to the reference field, possibly due to mineralization of soil organic matter (Lovett et al., 2005).

2.4.2.3 In-field Porewater Nutrient Concentrations

The trends in K and Cl concentrations were very closely mirrored by TDP, SRP, TDN, and NH_4^+-N in the harvested field porewaters, suggesting that higher nutrient concentrations in the harvested field, relative to the reference field, were due to internal changes within the peat field that may have increased decomposition. This said, the spatial distribution of N compared to P was different in the harvested field porewaters.

The higher TDP and SRP concentrations in the surface relative to the deep porewaters within the harvested field suggests that accelerated decomposition occurred predominantly in the top 50 cm of peat when temperatures were high and water was plentiful. Contrary to what has been reported in the literature, the concentrations of dissolved P were lower in the deep, anaerobic porewaters in this study. Kløve (2001), found that P concentrations increased when the water table decreased because the older water deep in the peat had higher P concentrations. In this study, deep waters do not appear to be high in TDP, so the pulse observed could be a result of decomposition present near the peat surface that was not as prevalent deeper in the peat profile. It is possible that the lack of P detected in the harvested field as the summer progressed and the water levels dropped was due to poor P mobility in unsaturated soils that prevented downward P migration from the surface. Phosphorus may have adsorbed onto the existing organic material or been immobilized by microbes when the water table lowered, leading to accumulations of P within the zone of water table fluctuation (Damman, 1978), that would not be detected in the dissolved fraction of a water sample. This suggests that the mobility of P may be tied to the hydrological connectivity within the harvested field, which could result in inconsistent export rates from the field to the outflow.

In contrast, the TDN and NH_4^+-N concentrations in the harvested field porewaters were similar at depth and increased over the summer. The greater dissolved N concentrations in the harvested field porewaters relative to the reference field porewaters could be due to higher temperatures at depth that may have improved the conditions for decomposition, as observed by Kløve (2001). Both Haapalehto et al. (2014) and Menberu et al. (2017) also measured higher N concentrations in the peat field relative to a pristine peatlands. Nitrate was not detected except for during snowmelt in this study. This suggests that nitrification was hindered, possibly due to lack of oxygen, and low pH within the peat profile (Kieckbusch & Schrautzer, 2007).

2.4.2.4 Surface Peat Nutrient Availability

Nutrient availability rates measured using PRS® probes corroborated the surface porewater chemistry results in general. They indicated similar trends during a drier period when water was less abundant, and augmented the porewater data in addition to providing further experimental information on the influence of ditching and extraction on peat fields. Availability of NH_4^+-N, PO_4^{3-}-P, and K were higher in the harvested field surface peat relative to the reference, indicating that decomposition likely continued through the summer. This also suggests that nutrients are available for potential flushing once hydrological flow paths are reconnected during rain or snow melting events.

Nutrient availability rates followed similar trends to previous studies on drained peatlands, although NH_4^+-N and K rates were notably higher in this study. The PO_4^{3-}-P availability rate observed in this study was similar to a study conducted by Munir et al. (2017), who observed higher rates of PO_4^{3-}-P in newly drained hollows relative to the control in a recently drained peatland in northern Alberta, Canada. As in this study, Munir et al. (2017) also observed higher NH_4^+-N rates in the recently drained peatland relative to the control, although the amount of available NH_4^+-N found in the harvested field in this study was almost twice as high as the rate observed in their recently drained peat. Munir et al. (2017) also found NO_3^--N at all sites, whereas this study was unable to detect any NO_3^--N availability. This discrepancy could be due to the presence of living vegetation that was active in the drained peatland studied by Munir et al. (2017) that could have removed some of the N, or differences in peat quality. High NH_4^+-N values have also been found in drained *Sphagnum*-sedge or bare peat pools, which were associated with increased mineralization (Harris et al., 2020). Harris et al. (2020) found higher K availability in the drained pools, which is consistent with the elevated K availability at the harvested field observed in this study. However, the K rate observed in the harvested field in this study was approximately ten times higher than the availability observed by Harris et al.(2020), indicating that higher rates of decomposition may have occurred, or freeze thaw cycles in the winter and spring and/or mechanical harrowing may have increased cell lysis.

When determining the peat properties that were primary indicators of nutrient availability, the gradient analysis revealed that the higher bulk densities, soil moistures, and surface temperatures observed in the harvested field were associated with increased NH_4^+-N, PO_4^{3-}-P, K, Ca, and Mg relative to the reference hummocks and hollows. While Ca and Mg concentrations were not high in the harvested field surface porewaters, their availability rates were significantly higher in the harvested field. The reason for this discrepancy is unclear and requires further investigation. The analysis also suggested that the harvested field more closely resembled the reference hollows relative to the hummocks, but that all three microforms were distinct. Previous studies have observed clear chemical differences based on microtopography (Harris et al., 2020; Munir et al., 2017), and drained peatlands have been shown to more closely resemble hollows rather than hummocks (Munir et al., 2017).

It is important to consider the nature of the peat extraction process, which is to remove the dried surface peat in layers, when interpreting these findings. The results of this study suggest that high stores of nutrients are available in the harvested field surface peat, yet the same peat is continuously removed via vacuum extraction. Due to the high carbon content of the peat soils, it is reasonable to expect decomposition to be slow and for microbial populations to require time to grow large enough to release sufficient nutrients that can be detected in porewaters. This is reset each time the surface peat is removed. While this study has shown that higher nutrient concentrations are present after just two years of extraction in both the harvested field and internal ditches, the extent to which accelerated decomposition can be extrapolated over time must include this context. Thus, continued investigation is required.

2.4.2.5 Internal Ditch Source Waters and Potential Export

Measurements of nutrient concentrations and trends within the harvested field porewaters were not always reflected in the internal ditches. While the TDN concentrations were similar between the harvested field and the internal ditch over the 2021 sampling year, the TDP, SRP, and NH_4^+-N showed unsynchronized trends. This incongruency indicates that the water quality within the internal ditch is dynamic, and is likely influenced by several processes. An understanding of the range of nutrient sources is essential for predicting the impact to downstream waterbodies.

The difference between the harvested field porewater chemistry and the internal ditch water chemistry in the spring could be due to dilution from melting snow. In the spring, the internal ditches were frozen and filled with snow, with water chemistry similar to snowmelt. It does not appear that the high TDP and SRP concentrations observed in the harvested field surface porewater had a substantial influence on the internal ditch water chemistry at this time, possibly due to low SWE in the harvested field and lack of rain to flush the dissolved P from the field into the ditch. The TDN and NH_4^+-N concentrations were also similar to the snow, with slightly more nutrients that were likely picked up by the water moving along the ditch. However, once the snow in the ditch melted, dilution from precipitation cannot explain the continued differences between the harvested field porewaters and the internal ditch water chemistry in the summer and autumn.

It is possible that the variability in nutrient concentrations is reflective of water in contact with different peat compositions or mineral subsoil accessed by the internal ditches. The pH, EC, Ca, Mg, and Si concentrations in the internal ditches became increasingly higher relative to the harvested field porewaters as the summer progressed and the water level in the harvested field declined. This may indicate that the water within the internal ditches was predominantly fed by precipitation or surface porewaters in the spring, with some influence of deep porewaters. As the water level declined, the deeper porewaters became the predominant water source in the summer and fall, barring lack of rain. This also implies that the hydrological connectivity between the harvested field and the internal ditch may be very important. Previous studies have observed flushing of nutrients from the peat field to the ditch after heavy rainfall, with a mixture of precipitation and groundwater observed in the ditches (Kløve, 2001). However, in this study, the concentrations of TDP, SRP, TDN, DOC, and K observed in the internal ditches were much higher than the concentrations measured in the pre-harvest and post-mulched surface and deep porewaters. Further, the nutrient concentrations in the adjacent deep porewater in the harvested field were generally lower than the concentrations in the internal ditch. This indicates that water sourced from deep porewater alone was not solely responsible for the increased nutrient concentrations.

Unlike in previous studies that observed lower nutrient concentrations in the ditch water relative to the adjacent harvested field (Haapalehto et al., 2014; Kløve, 2001), this study observed higher concentrations in the internal ditch relative to the harvested field. Differences in P concentrations between the groundwater and ditches in Kløve (2001) were attributed to the removal of P in ditch waters through sorption, complexes with Fe, or algal uptake (Kløve, 2001). The increased nutrient concentrations in this study suggests that mechanisms that would otherwise remove P and N from the internal ditches were not dominant. It does not appear that Fe played a critical role in P availability in the internal ditches because the Fe concentrations were low and do not follow the same trend as P. In addition, the water depth in the ditches was fairly low and periods of stagnation and lack of water were detected on several occasions in May, July, and September. The dry conditions at this site, characteristic of the Boreal Plain, could inhibit the establishment of algae, as well as provide prime conditions for decomposition within the internal ditch.

Although it appears that the internal ditches received some proportion of water from the deeper peat porewater, the higher nutrient concentrations within the internal ditch relative to the harvested field observed in the summer months was most likely due to accelerated decomposition. Compared to the harvested field, the deeper peat exposed by the internal ditch had lower C:N, and the surface water temperatures within the internal ditch were much higher. The elevated DOC and K concentrations relative to the harvested field porewaters also suggests that decomposition may have increased. Strong correlations between TDN and DOC have been reported in previous peatland studies (Burd et al., 2018; Pinsonneault et al., 2016), and older peat from deeper underground and subjected to drainage has been shown in to contain lower C:N and have an overall higher N content (Damman, 1978; Leifeld et al., 2020). In addition, TN concentrations have been positively correlated with higher temperatures in drained peatlands in Northern Europe (Nieminen et al., 2021), as have concentrations of DOC (D. M. Morris, 2009). The high temperatures and anoxic conditions may have accounted for the low NH_4^+-N and high TDN concentrations in the internal ditch, as biological uptake may have rapidly converted NH_4^+-N to organic nitrogen (Kløve, 2001). These processes could explain the higher TDP, SRP, TDN, and DOC, and lower NH_4^+-N concentrations observed in the internal ditches in summer. Regardless of the processes, the higher concentrations observed in this study indicate the internal ditches may be a major contributing source of nutrients that have high potential for export, especially during rain events following periods of water stagnation and warm temperatures that likely accelerate decomposition.

This said, it appears that the risk of exporting nutrients downstream is not uniform over the year and between years. Understanding the chemical composition of the surface and deep porewaters in the harvested field, in addition to considering the range in weather patterns for a given year, is required for accurately predicting the potential water quality within the harvested field and internal ditches. Although parts of this study occurred during a time period with below-average summer rainfall, these weather conditions are not uncommon and occur roughly every four years on the Boreal Plain (Devito, Creed, & Fraser, 2005; Devito et al., 2023). During mesic rainfall weather patterns, hydrologic connectivity with the field is highly probable, and the concentrations observed within the internal ditch may be more reflective of the surface peat, or show evidence of dilution as seen in previous studies (Haapalehto et al., 2014; Kløve, 2001). Thus, additional data should be collected at other harvested peatlands located in the Boreal Plain under different weather patterns to observe the range in nutrient variability both within the harvested field and the internal ditches. This will aid in predicting the range of potential nutrients that could leach from a newly harvested peat field under a range of seasonal and interannual weather patterns

2.5 CONCLUSIONS

This study suggests that the initial perimeter ditching appeared to have little influence on the water levels, aeration, and hydrological connectivity within the newly opened harvested field. However, vegetation mulching removed transpiring vegetation and altered the ground surface elevation and relief, resulting in a shallower water table. Although the nutrient concentrations within the harvested peat field were initially increased, this was from leaching plant material when the surface vegetation was mulched, and not from upward migration of nutrients in deeper porewaters or accelerated peat decomposition. Observed increases in porewater nutrient concentrations were short lived, possibly due to immobilization by microbes, sustained poor environmental conditions for decomposition, or insufficient time for microbial populations to increase enough to initiate rapid decomposition of the exposed surface peat. In fact, evidence of decomposition (elevated N and P concentrations) during the opening period was not observed in peat porewaters in the short term, despite higher surface temperatures and increased nutrient inputs from the fresh vegetation mulch.

In contrast, installing internal ditches and extracting peat appears to have influenced the internal processes within the harvested field and internal ditches, altered the hydrological flow paths, and has the potential to transport higher concentrations of nutrients downstream. Alterations to the snow and ice dynamics within the harvested field and internal ditches could enable nutrient-rich melt waters to be more readily transported off site. Although the water levels within the harvested field were lower than the reference peatland, the soil moisture remained high and the depth of aeration did not extend deep into the peat profile, likely due to peat capillary rise. The near-surface aerobic conditions, moist peat, and elevated surface and porewater temperatures likely encouraged decomposition in the harvested field, resulting in higher nutrient concentrations. Internal ditches were dynamic and did not consistently mirror the nutrient concentrations observed in the harvested field. They appeared to be influenced by precipitation, surface, and deep peat porewaters in variable proportions throughout the year, but the nutrient concentrations are likely governed by increased decomposition enabled by low C:N and elevated summer temperatures. Therefore, internal ditches may be a major contributing source of nutrients downstream, particularly following large rain events.

Table 2.1 Snow and ice survey conducted in March 2020 & 2021 in reference and harvested peat fields

	Snow Depth (cm)			Depth to Ice (cm)			SWE (cm)			Storage (cm)[d]		
	Site R1[a]	Site H1[b]	Site H2[c]	Site R1	Site H1	Site H2	Site R1	Site H1	Site H2	Site R1	Site H1	Site H2
n	28	32	4	20	21	4	28	32	4	28	32	4
Mean	32	24	93	-18	-3	0	6	6	24	-11	4	24
SD	12	6	6	16	7	0	3	2	1	2	2	1
Med	29	23	94	-15	0	0	6	6	24	-12	3	24
Min	11	15	86	-50	-25	0	2	4	23	-16	1	23
Max	53	37	100	-1	0	0	11	10	25	-7	8	25

Where: n is the number of samples, SD is one standard deviation from the mean, Med is the median, Min is the minimum value, Max is the maximum value, and SWE is the snow water equivalent. Negative (-) values indicate an ability to store water, positive (+) values indicate no storage capability.

[a]Reference field

[b]Harvested field

[c]Internal ditch within the harvested field

[d]Estimated storage capability (SWE + mean ice depth)

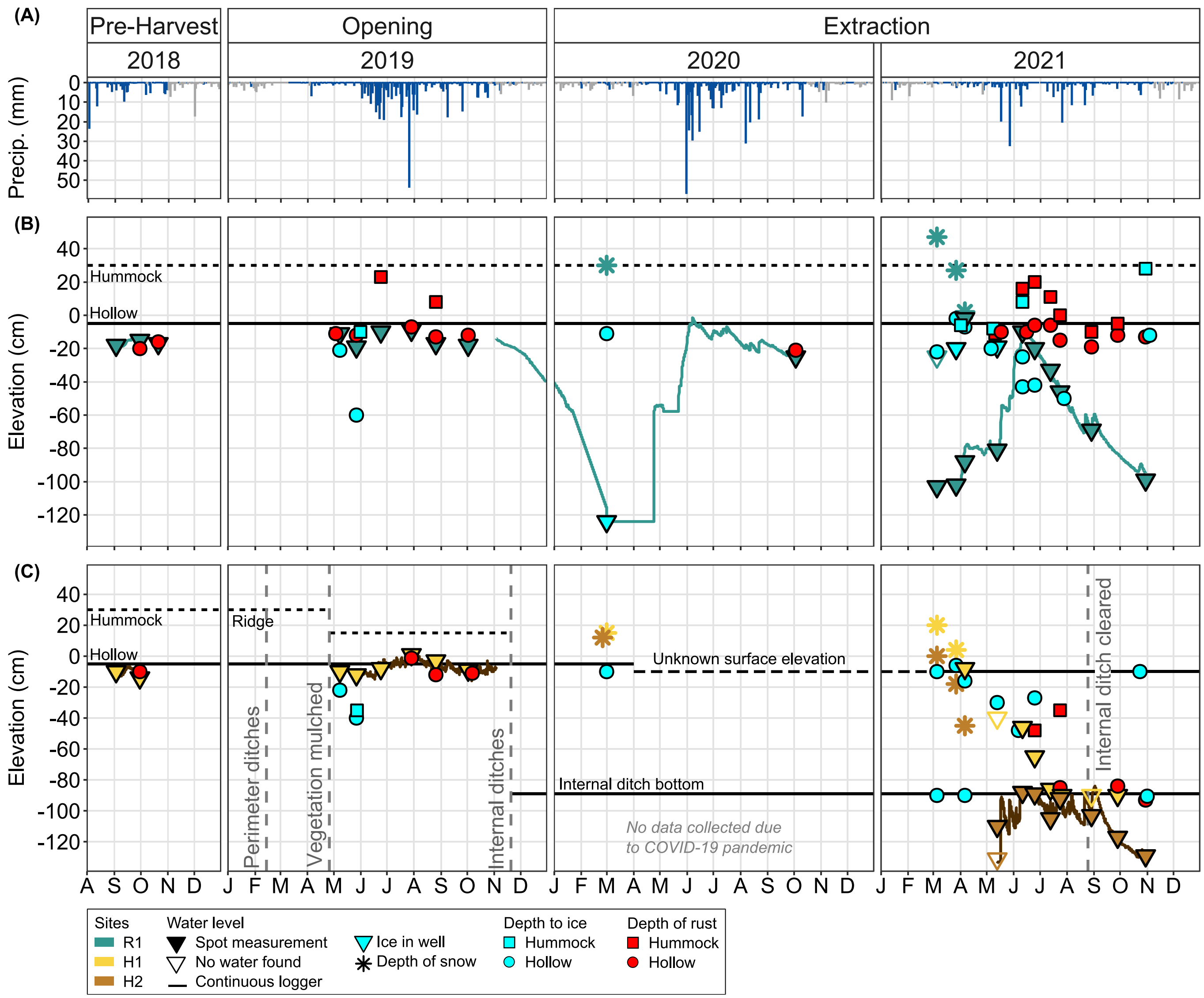

Figure 2.1 (A) Mean daily precipitation (precip.); (B) in-field water level, snow, rust, and depth to ice in the reference field (Site R1); and (C) in-field water level, snow, rust, and depth to ice in the harvested field (Site H1, yellow) and internal ditch (Site H2, brown).

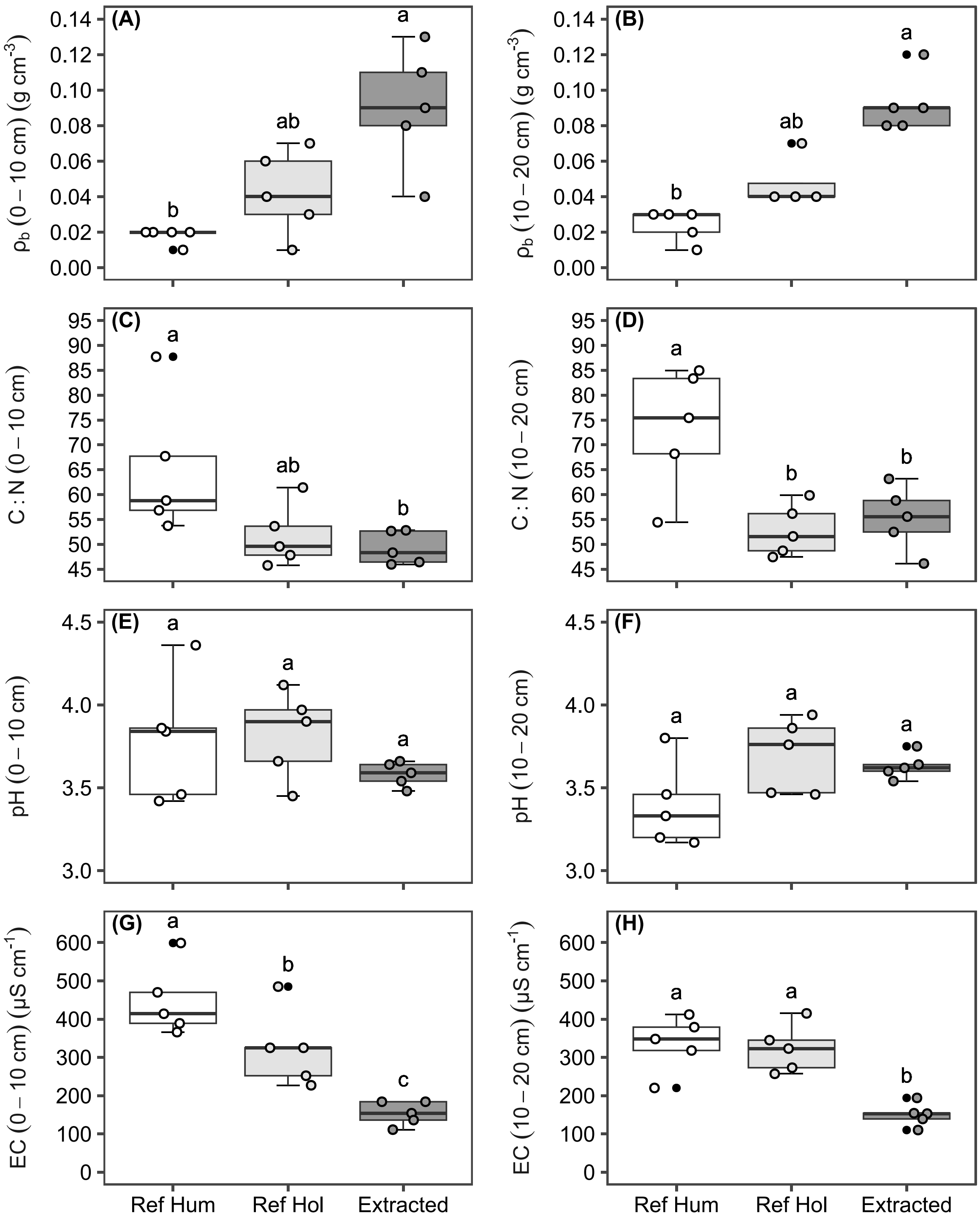

Figure 2.2 Surface peat properties for reference hummocks (Ref Hum), hollows (Ref Hol), and two year-old extracted peat field locations taken 0 – 10 cm and 10 – 20 cm from the ground surface. Outliers are represented by black dots; shaded dots represent individual samples. Different letters indicate significant differences among reference microforms and extraction locations (ANOVA or Kruskal-Wallis, $p < 0.05$). Outliers were removed for statistical analysis. (A & B) Dry bulk density, (C & D) carbon:nitrogen ratios, (E & F) pH of dried peat, and (G & H) electrical conductivity of dried peat.

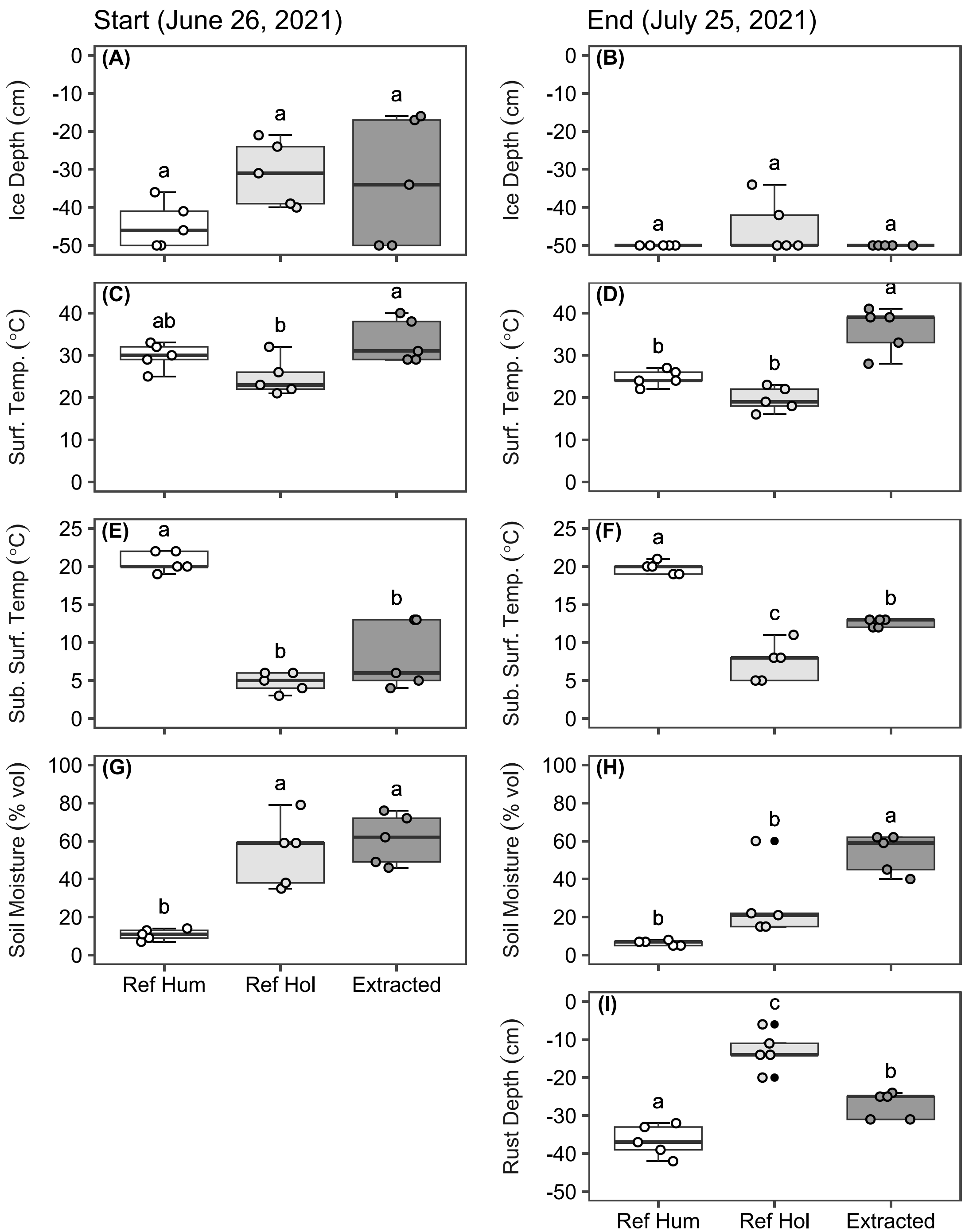

Figure 2.3 Surface peat properties for reference hummocks (Ref Hum), hollows (Ref Hol), and two year-old extracted peat field locations taken at the start and end of the PRS® ion availability study. Outliers are represented by black dots; shaded dots represent individual samples. Different letters indicate significant differences among reference microforms and extraction locations in the same time period (ANOVA or Kruskal-Wallis, $p < 0.05$). Outliers were removed for statistical analysis. (A & B) Depth to ice from the ground surface (negative values are depth below ground), (C & D) ground surface temperature, (E & F) subsurface temperature 10 cm below the ground surface, (G & H) volumetric soil moisture, and (I) rust depth (negative values are depth below ground).

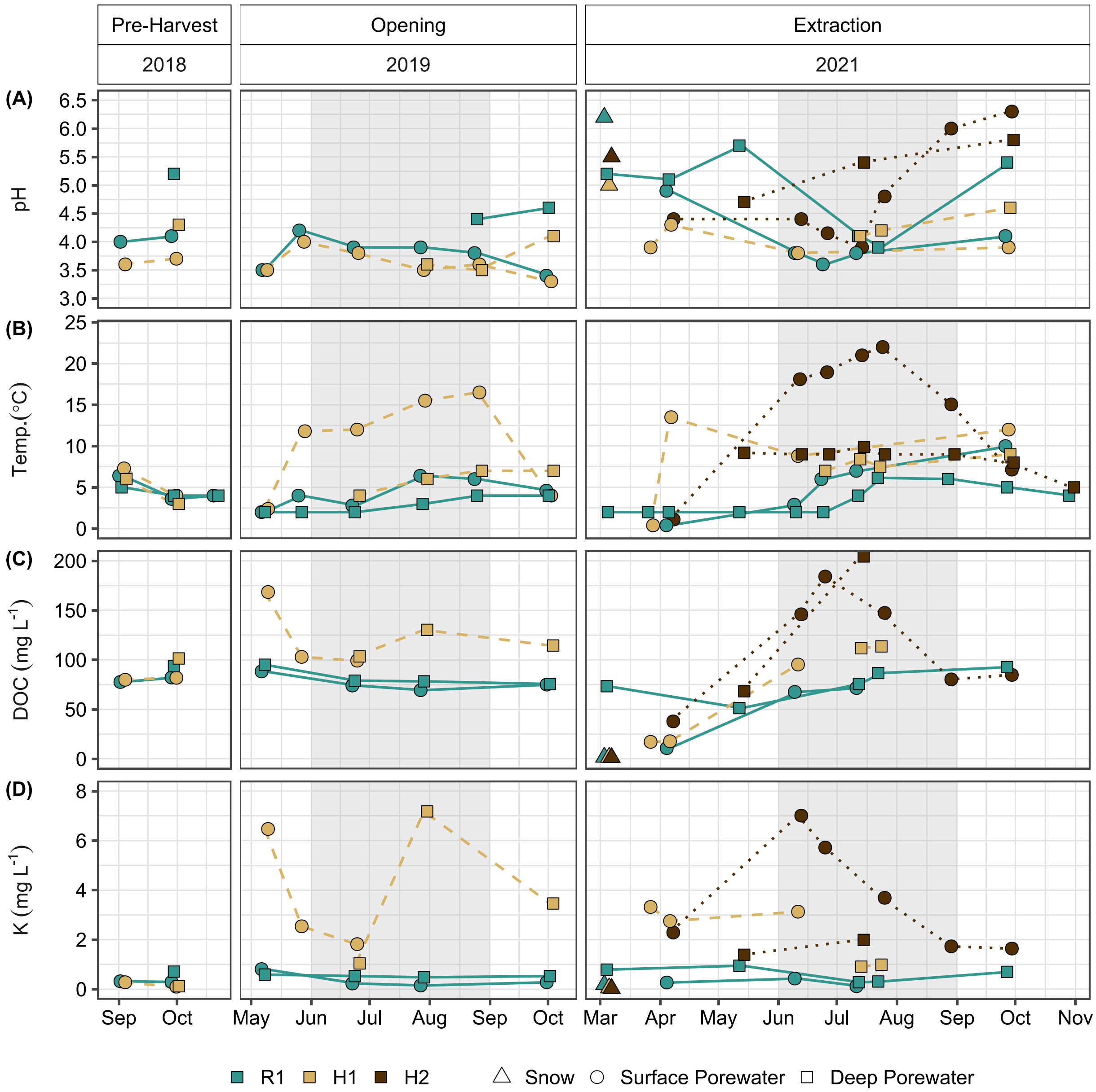

Figure 2.4 Comparison of seasonal variations in (A) pH, (B) temperature, (C) dissolved organic carbon (DOC), and (D) potassium (K) concentrations collected from snow, surface porewaters, and deep porewaters at Sites R1 (reference peat field), H1 (harvested peat field), and H2 (internal ditch within the harvested peat field). Grey area demarcates summer months in 2029 and 2021. May 2019 sample followed the start of vegetation mulching.

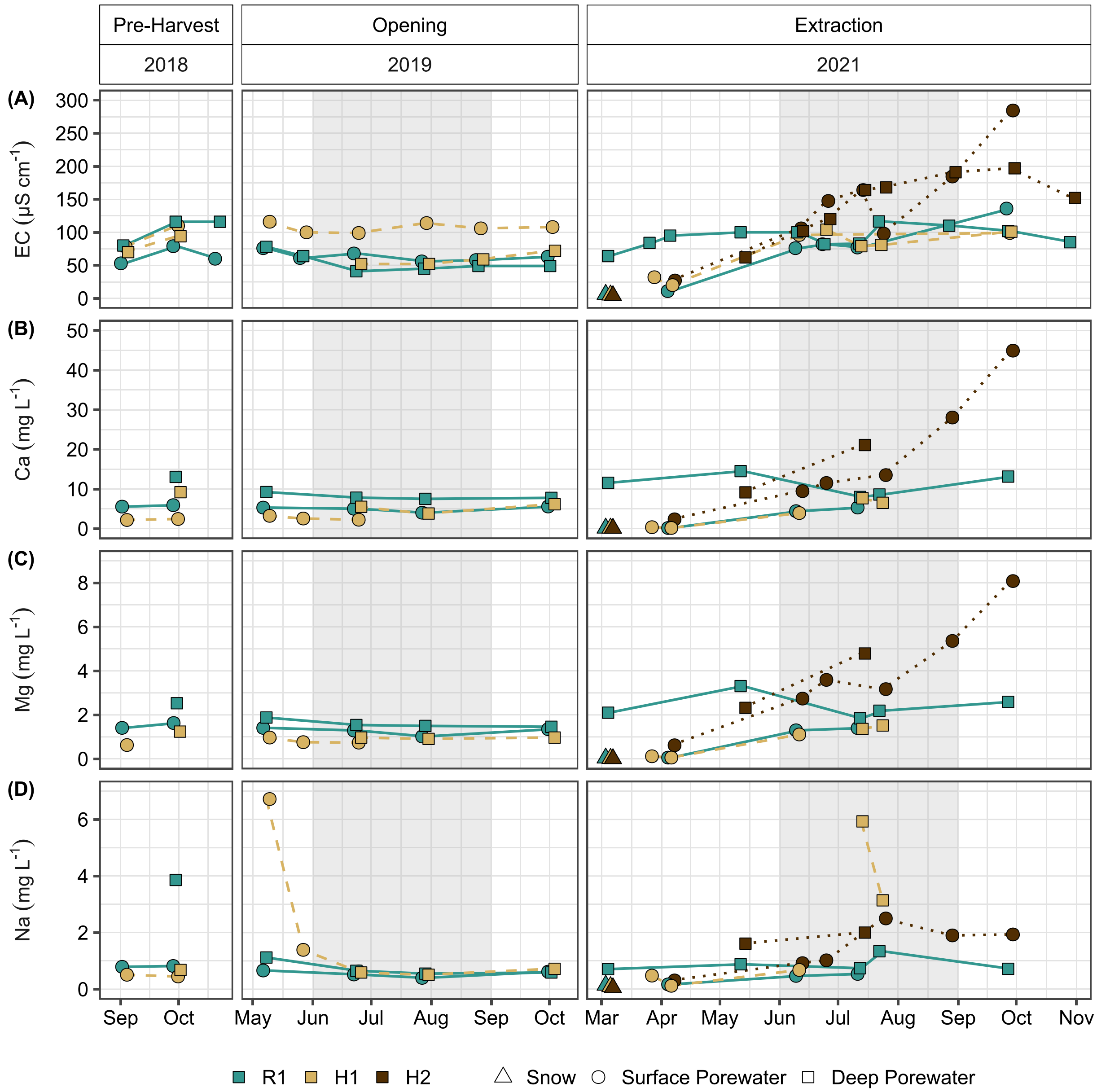

Figure 2.5 Comparison of seasonal variation in (A) electrical conductivity (EC), (B) calcium (Ca), (C) magnesium (Mg), and (D) sodium (Na) concentrations collected from snow, surface porewaters, and deep porewaters at Sites R1 (reference peat field), H1 (harvested peat field), and H2 (internal ditch within the harvested peat field). Grey area demarcates summer months in 2029 and 2021. May 2019 sample followed the start of vegetation mulching.

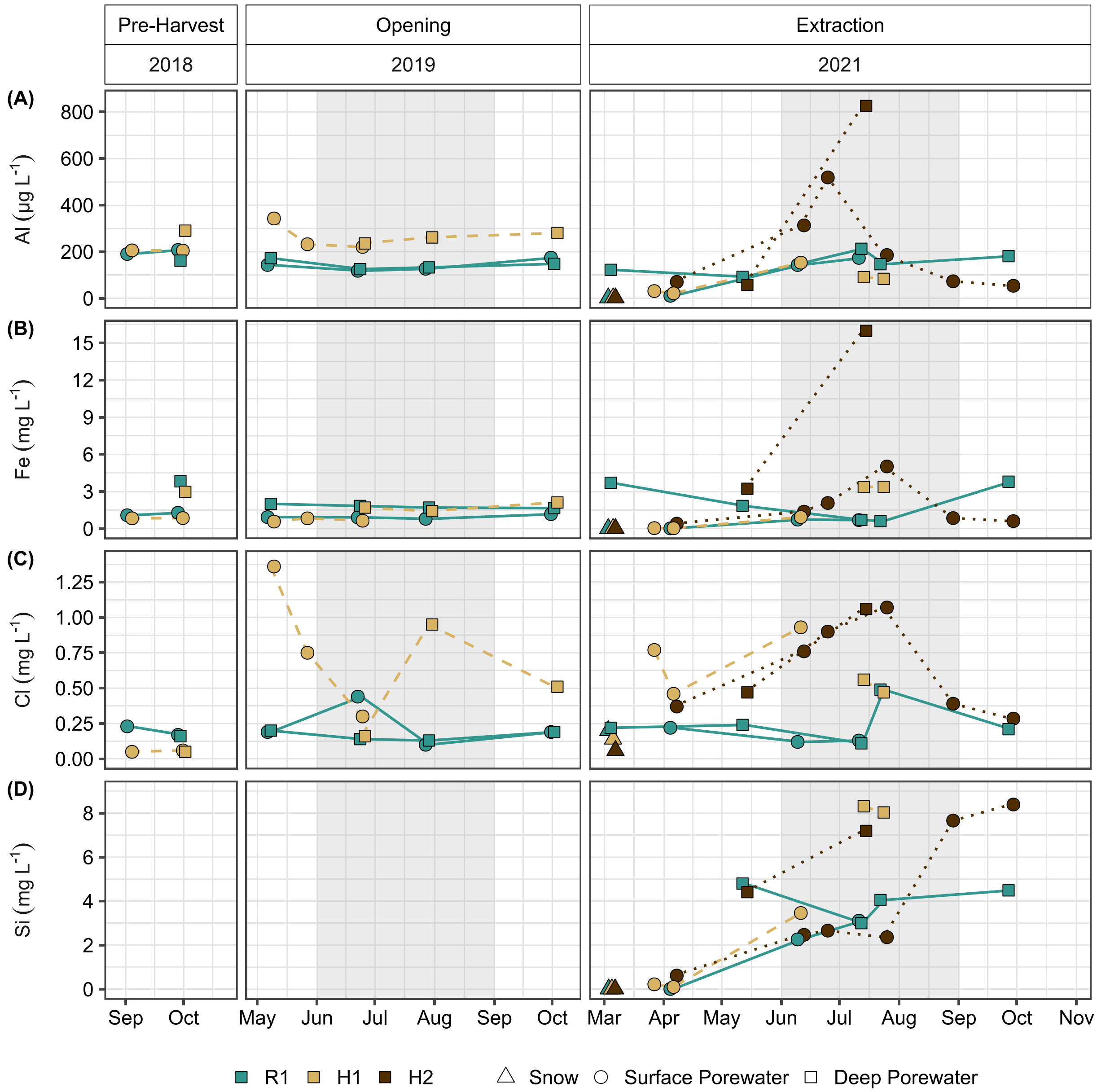

Figure 2.6 Comparison of seasonal variation in (A) aluminum (Al), (B) iron (Fe), (C) chloride (Cl), and (D) silica (Si) concentrations collected from snow, surface porewaters, and deep porewaters at Sites R1 (reference peat field), H1 (harvested peat field), and H2 (internal ditch within the harvested peat field). Grey area demarcates summer months in 2029 and 2021. May 2019 sample followed the start of vegetation mulching.

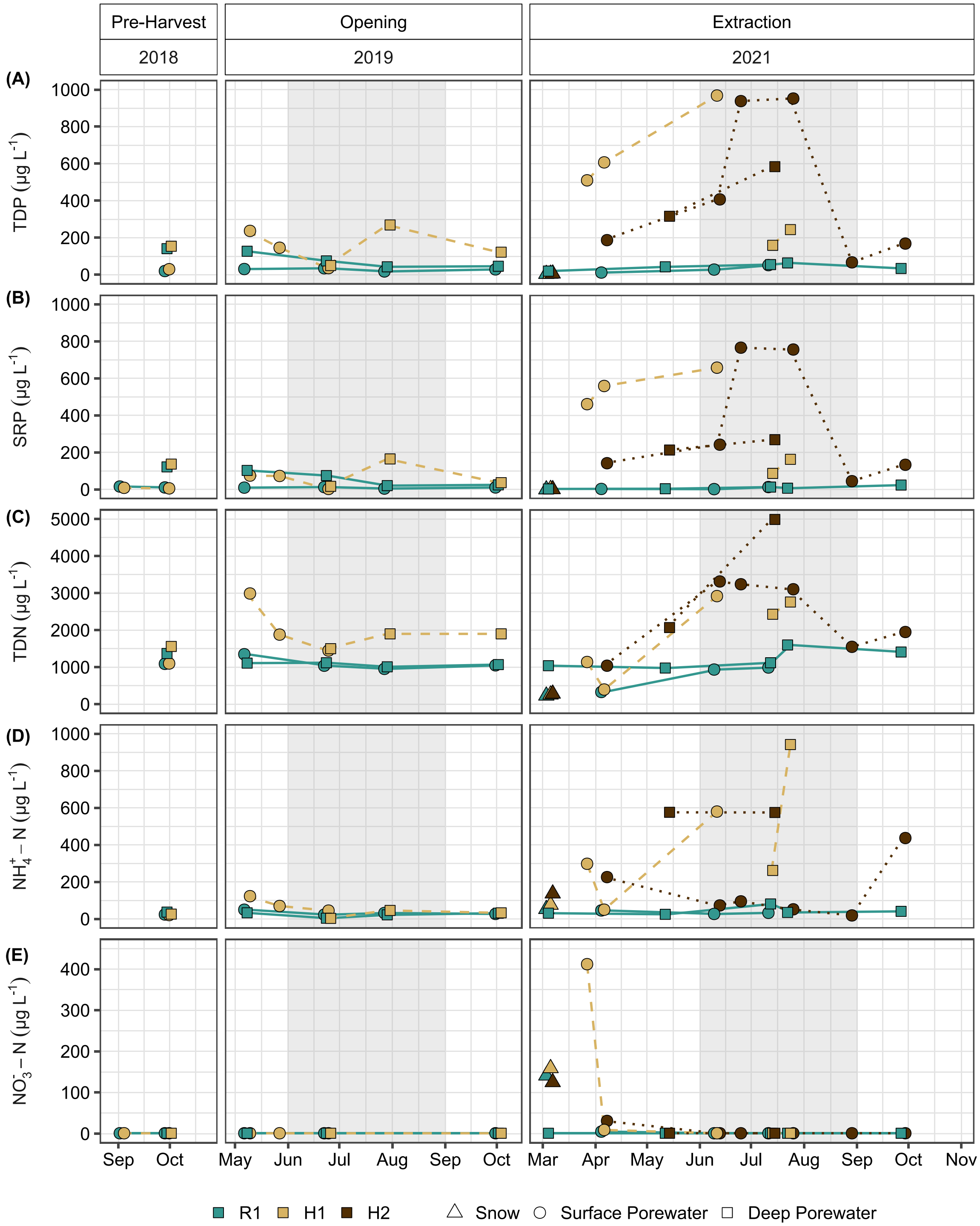

Figure 2.7 Comparison of seasonal variation in (A) total dissolved phosphorus (TDP), (B) soluble reactive phosphorus (SRP), (C) total dissolved nitrogen (TDN), (B) ammonium (NH_4^+-N), and (C) nitrite + nitrate (NO_3^--N) concentrations collected from snow, surface porewaters, and deep porewaters at Sites R1 (reference peat field), H1 (harvested peat field), and H2 (internal ditch within the harvested peat field). Grey area demarcates summer months in 2029 and 2021. May 2019 sample followed the start of vegetation mulching.

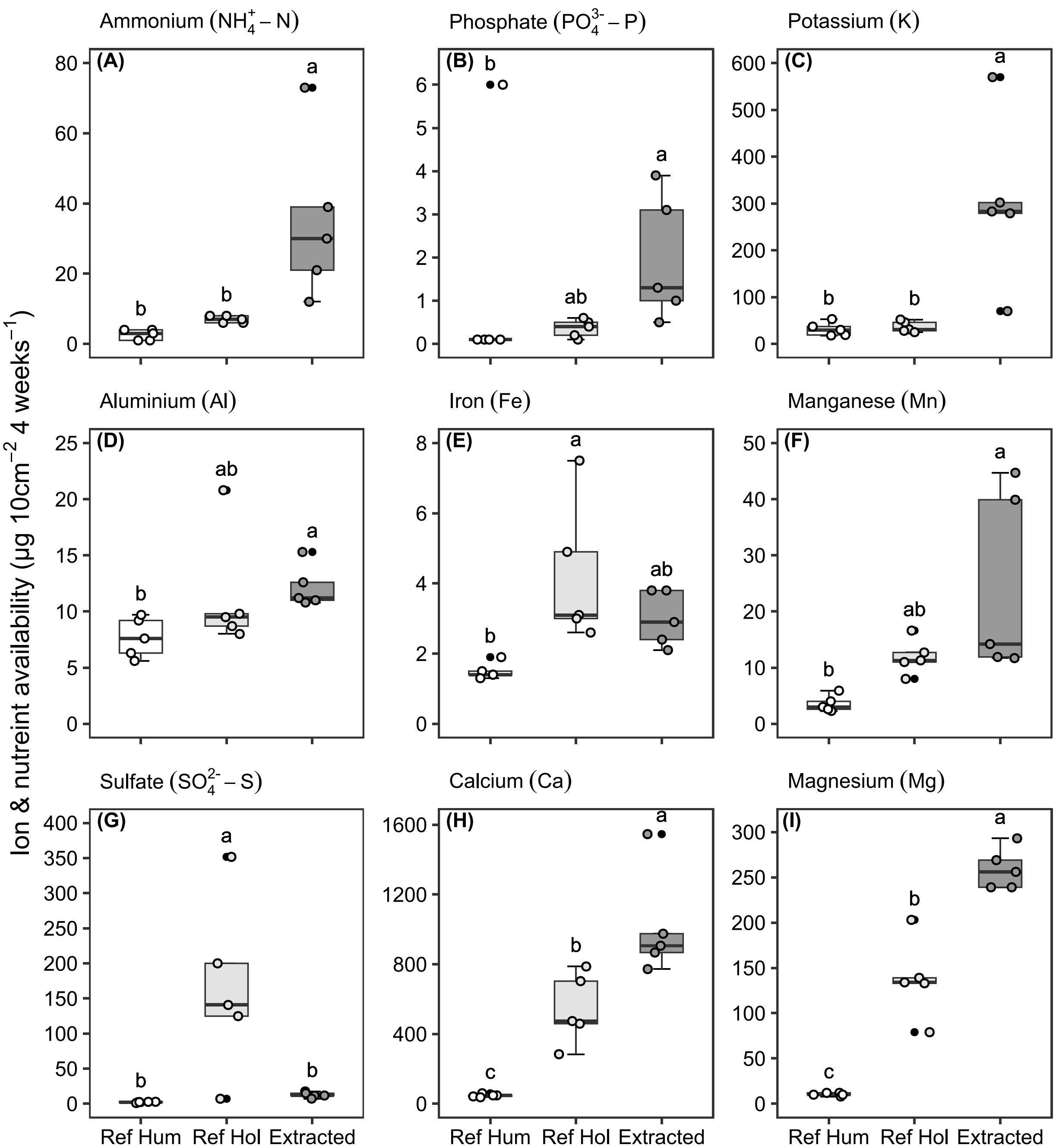

Figure 2.8 Chemical availability (supply rate in μg per 10 cm^2 over 4 weeks, PRS® probes, n = 5 (with 3 probes grouped together per sample)) for reference hummocks (Ref Hum), hollows (Ref Hol), and two-year-old extracted peat field sample locations (July 2021). PRS® probes were placed 5 to 10 cm below the ground surface. Outliers are represented by black dots; shaded dots represent individual samples. Different letters indicate significant differences among reference microforms and extraction locations in the same time period (ANOVA or Kruskal-Wallis, $p < 0.05$). Outliers were removed for statistical analysis. Nitrogen available as nitrate (NO_3^--N) was sampled, but values were below detection for all locations.

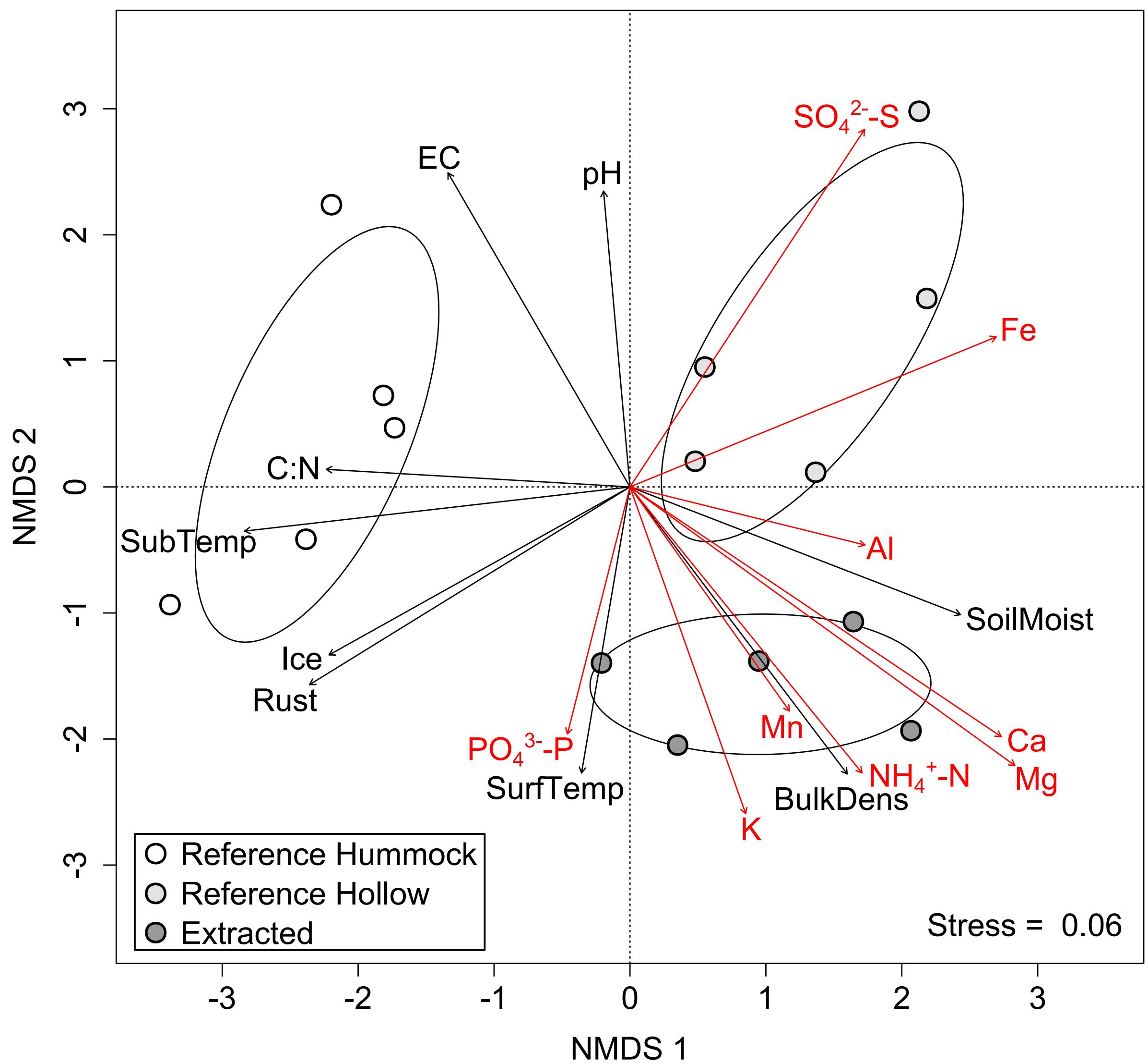

Figure 2.9 Direct gradient analysis with NMDS ordination showing the interactions between peat in situ physicochemical variables (vectors with black text) and in situ chemical availability using PRS® probes (vectors with red text). Nitrate (NO_3^--N) values were below detection limits for all sites and were not included in the analysis. Physicochemical variables were scaled and a Euclidean distance matrix was used for the initial ordination; chemical availability vectors were overlaid onto plot using calculated ordination scores. Different shades represent peatland microforms and extraction treatments. Ellipses represent 60 percent confidence intervals.

Chapter 3: Alterations to Outflow Water Quality

3.1 INTRODUCTION

A working knowledge of the water quality leaving horticultural peat harvesting operations is essential for determining the relative impact to downstream waterbodies. As discussed in Chapter 1 of this thesis, several studies have observed increased nutrient concentrations and sediment associated with drained peatlands (Betis et al., 2020; Clausen & Brooks, 1983; Edokpa et al., 2017; Haapalehto et al., 2014; Joensuu et al., 2002; Kløve, 2001; Marttila et al., 2018; Moore, 1987; Munir et al., 2017; St-Hilaire et al., 2004); however, other studies have found little impact (Harris et al., 2020; Macrae et al., 2013; Moore, 1987; Palviainen et al., 2022; Pschenyckyj et al., 2023). While it is essential to first understand how nutrient availabilities may differ chronologically in the early periods of horticultural peat extraction following ditching, vegetation removal, and peat extraction (addressed in Chapter 2 of this thesis), it is also important to clarify the potential alterations in hydrology and biogeochemistry as water is transported from the peat fields to the outflow via the drainage infrastructure, as well as changes that occur within the downstream receiving ecosystem. The overall general impact of peat harvesting operations requires consideration of changes in biogeochemistry and hydrological linkages at different scales through time and space. In order to address this key knowledge gap, an understanding of where the water at the outflow originated, the water quality associated with the water source, and any alterations to the water moving to the outflow of a harvested peat operation compared to adjacent reference outflows is critical.

As water from the harvested field moves toward the harvested outflow through a network of ditches, additional water sources may be intersected that could mix with water impacted by the harvested peat field and could alter the overall outflow nutrient concentrations and export rates. Water chemical composition may be drastically different from the surface peat porewaters if the internal and perimeter ditches are deep enough to intersect local or regional groundwater (Pschenyckyj et al., 2023), or deeper porewaters from the adjacent non-harvested peatland. Depending on the volume and source of the additional water, the effect of peat harvesting may be overwhelmed.

The internal and perimeter ditch substrate (either peat or mineral) may further modify the chemistry of waters originating within the harvested field, as well as water moving along hydrological flow paths that are intersected by the ditch network and make up the outflow water. Relative to surface peat porewaters, several studies have observed alterations to the water chemistry when samples were collected in peat (Pschenyckyj et al., 2023; Wind-Mulder et al., 1996) and mineral (Åström et al., 2001; Reynolds & Hughes, 1989) ditches. Chemical transformations within the ditch may mobilize or retain nutrients of various forms, depending on the ditch chemical properties. Also, as water travels along the ditch, organic or mineral sediments can be entrained and transported downstream, especially during periods of high flow (Marttila & Kløve, 2010). Considering the importance of ditches intersecting the substrate, the range in peat depths and geology across Canada may greatly influence outflow water quality.

Finally, the scale and magnitude of the impact to downstream waterbodies should be considered. The relative risk to downstream ecosystems will vary depending on the location of the harvested peatland within the broader landscape, the type of ecosystem into which the outflow water discharges (for example, an additional wetland versus a flowing stream), and the influence of other ecosystem modifiers, such as *Castor canadensis* (beaver) on the hydrochemical influence or transformations of final downstream water quality (Devito et al., 1989; Devito & Dillon, 1993; Klotz, 1998; Rosell et al., 2005).

3.1.1 Objectives and Hypotheses

The objective of this chapter is to compare the water quality and quantity observed at an outflow swamp downstream of peat harvest operations with a nearby reference outflow swamp to determine: (1) the potential overall impact of harvesting activities on downstream ecosystems and (2) the relative role of changes to nutrient concentrations in the harvested peat field, channel networks draining the peat field, and stream processes within the drains on water quality. This study chronologically examines the discharge rates and water quality within and from a peatland before harvesting activities commenced, after the installation of a perimeter ditch and vegetation mulching, and following the installation of internal drainage ditches and vacuum peat harvesting. Water quality and quantity from a reference peatland field and downstream outflow swamp were compared with water moving from the peat harvesting operation. Water was sampled in the internal ditch leaving the harvested peat field, in the perimeter ditch, and in the main outflow channel before entering the receiving outflow swamp (Figure 1.3). The impacts of peatland harvesting on the water quality in the receiving outflow swamp ecosystem were also compared with water modified by the presence of beaver activities (dam construction) immediately downstream, as well as with a nearby reference outflow swamp (Figure 1.3).

The following alternative hypotheses are proposed. The water quality entering the aquatic ecosystem downstream of the peat harvesting operations is a function of:

(1) changes that have occurred within the harvested peatland field as a result of vegetation mulching, internal ditching, and ongoing peat extraction;
(2) different source waters that are hydrologically connected during internal and perimeter ditch installation;
(3) chemical transformations to the water in contact with the mineral substrate within the perimeter ditch or main outflow channel;
(4) hydrological and soil biogeochemical modifications within the outflow stream or outflow swamp ecosystem due to presence of beaver activities, such as dam construction, that
    (a) reduce downstream nutrient transport, or, alternatively,
    (b) are a source of nutrients to downstream waterways.

3.2 METHODS

3.2.1 Hydrological Measurements

3.2.1.1 Stream Flow

Spot measurements of water discharge (L s^{-1}) were estimated at naturally confined areas or in constructed ditches. Flow was estimated at outflow locations (Sites H3 (field internal ditch), H4 (perimeter ditch), H5 (end of main outflow ditch), R2 (reference outflow)) as well as at the road culverts (Sites H8 and R3; Figure 1.3) by timing the volume of water caught in a graduated bucket or, by averaging stream velocity with a surface float multiplied by the measured channel cross-sectional area (Dingman, 2015). Continuous flow was determined at selected sites using stage height-discharge relationships established using the installed stilling wells with continuous water level recorders.

As there were no ditches at the reference (Site R2; Figure 1.3) and pre-harvest (Site H5; Figure 1.3) outflow locations, the occurrence of sheet flow across the small valley was noted by the presence of surface water determined visually at first, then later with manual well water level measurements and automated water level recorders. Actual sheet flow at the reference outflow (Site R2) was also estimated using a combination of: 1) measuring surface depth and velocity across the small valley area, and 2) a general laminar flow equation containing a laminar flow equivalent to the Chezy equation and the tortuous flow in wetlands (Dingman, 2015; Ferone & Devito, 2004).

There is considerable uncertainty in estimates of sheet flow volume. However, these estimates are constrained by water level measurements relative to ground surface at each peatland outflow, and flow measurements at downstream culverts (Sites H8 and R3) that confined estimates of outflow discharge. Together, they provide reasonable estimates of the occurrence of flow, the timing of surface flow initiation, and relative seasonal variation.

Estimates of continuous discharge (L s^{-1}) over the entire study for both outflow swamp sites (R2 and H5) were conducted by extrapolating government stream flow data from the nearby Logan and Amisk rivers (Environment Canada, n.d.). The extrapolations were estimated by using the relationship between spot measurement discharge rates and mean daily discharge rate measurements for each peatland outflow (Sites R2 and H5), paired with the mean daily discharge rates (offset by a lag of 2 days) for Logan River and Amisk River. Flow was measured at multiple locations along the harvested outflow ditch and outflow swamp (Sites H4, H5, H7, H8) to observe any major changes in water volume and to amass multiple measurements for quality control.

3.2.1.2 Precipitation, Temperature, Depth to Ice, and Rust Depth

See Chapter 2, section 2.2.1.1 for a detailed description of precipitation, temperature, and depth to ice procedures. In addition to the depth to ice, rust depth was measured using a steel rod at Sites R2 and H7. Please see Chapter 2, section 2.2.3.1 for a detailed description of the rust depth procedure.

3.2.1.3 Water Levels

Water levels in ditches and outflow fen locations were assessed using stilling wells constructed as described in Chapter 2, section 2.2.1.2. Water level depth was measured using a water level meter (Model 107 TLC Meter, Solinst® Canada Ltd.), and pressure transducers (HOBO® U20L-04 Water Level Logger, Onset® Computer Corporation) were installed to determine continuous (4-hour interval) water levels.

3.2.2 Hydrochemical Sampling

3.2.2.1 Flowing Surface Water

Stream or ditch water samples were collected from water above the surface of hollows that was visibly flowing at the reference and harvested outflow swamps or, from moving water within a constructed channel or ditch. In order to quantify both particulate and dissolved concentrations and exports, flowing water samples were screened in the field with a 750 µm nylon mesh to remove large debris such as leaves and twigs that would not fit in the sample bottles. No other filtering was applied. When samples were collected, care was taken to not disturb the sediment so that an overestimation of the particulate fraction was avoided. Ditch substrate composition was noted when sampling and representative samples of water moving through ditches cut into both peat and mineral substrates were obtained.

3.2.2.2 Surface and Deep Porewaters

To help determine whether water observed in ditches and outflow swamps was sourced from surface or deep porewaters within the peatland, water was extracted from the reference (Site R1) and harvested (Site H1) fields. Detailed sampling descriptions can be found in Chapter 2, section 2.2.2.1.

3.2.3 Chemical Analyses

3.2.3.1 Flowing Surface Water Concentrations

For all flowing surface water samples, electrical conductivity (EC), pH, and temperature were measured *in situ* using a hand-held meter (HI98129, Hanna® Instruments). In addition, the temperature and EC at the water level surface and at the bottom of each stilling well were measured using an EC meter (Model 107 TLC Meter, Solinst® Canada Ltd.). For stream flow samples, both the non-filtered particulate (total nitrogen (TN), total phosphorus (TP), total particulate carbon (PC), total suspended solids (TSS)), and filtered (0.45 µm) dissolved fractions (total dissolved nitrogen (TDN), ammonium (NH_4^+-N), nitrite + nitrate (referred to as nitrate and NO_3^--N henceforth), total dissolved phosphorus (TDP), soluble reactive phosphorus (SRP), dissolved organic carbon (DOC), and aluminum (Al), calcium (Ca), chloride (Cl), iron (Fe), potassium (K), magnesium (Mg), and silica (Si) ions) were analyzed. Dissolved fractions were filtered with a 28 mm or 47 mm diameter 0.45 µm Sartorius cellulose acetate syringe filter. Each water sample was collected in high density polyethylene bottles triple rinsed with the sample water. Filled bottles were kept cool, transferred to a 4 °C refrigerator at the end of each field day, and submitted within 48 hours of collection for analysis at the Biogeochemical Analytical Service Laboratory (BASL) at the University of Alberta.

Total nitrogen, TDN, NH_4^+-N, NO_3^--N, TP, TDP, SRP, and Si were analyzed using flow injection analysis (QuickChem® 8500 FIA Automated Ion Analyzer, Lachat Instruments). Analysis followed procedures outlined in the US EPA Method 353.2 for TN, TDN, NO_3^--N (US EPA, 1993), and in the American Public Health Association (APHA) Method 4500 for NH_4^+-N, TP, TDP, and SRP (Baird & Bridgewater, 2017). Dissolved organic carbon was measured using a total organic carbon analyser (TOC-5000A Total Organic Carbon Analyzer, Shimadzu Corporation). Analysis followed procedures outlined in the US EPA Method 415.1 for DOC (US EPA, 1999). Aluminum, Ca, Fe, K, and Mg were measured using an inductively coupled plasma optical emission spectrometer (iCAP 6300, Thermo Scientific) following the US EPA Method 200.7 (US EPA, 1994). Chloride was measured using ion chromatography (DX-600 Ion Chromatography System, Dionex Corporation), following the US EPA Method 300.1 (US EPA, 1997a). Total particulate nitrogen (TPN) and total particulate carbon (PC) were assessed using combustion and elemental analysis. The sample was filtered using a 25 mm diameter 0.7 µm GF/F filter and analyzed using an elemental analyzer (CE440 Elemental Analyzer, Exeter Analytical Ltd.), following US EPA method 400.0 (US EPA, 1997b). Total suspended solids were measured by passing the sample through a 47 mm diameter 0.45 µm Sartorius cellulose acetate syringe filter and recording the dried weight of the particles retained on the filter. The TPN and TDN were added together to approximate the TN in the sample for select sampling years when TN was not analysed.

3.2.3.2 Surface and Deep Porewater Concentrations

Methods for determining the surface and deep porewater concentrations at Sites R1 and H1 can be found in Chapter 2, section 2.2.4.1.

3.2.4 Calculations and Statistical Analyses

3.2.4.1 Daily Mass Discharge

A daily mass discharge (kg day^{-1}) of nutrients associated with the water flowing at the reference and harvested outflows was calculated by multiplying the estimated daily stream flow (L s^{-1}) at a given outflow site by the water chemistry concentration (µg L^{-1} or mg L^{-1}) measured at the same site during that day. When more than one chemistry sample was collected at a given sampling site, the average of the samples for that day was used as an instantaneous estimate. Due to challenges associated with defining a contributing catchment area, the export rates per unit between the reference and harvested catchments could not be directly compared. However, the relative seasonal or inter-annual change in magnitude was used as an indicator of similarity between systems.

3.2.4.2 Statistical Analysis

The assessments made in this chapter are descriptive in nature. All data manipulations were conducted in RStudio (2023.06.01+524), using R Statistical Software (v.4.2.2; R Core team, 2022).

3.3 RESULTS

3.3.1 Hydrology

3.3.1.1 Stream Flow

There was significant seasonal and interannual variability in precipitation and associated outflow discharge during the four-year study period. In general, discharge from the adjacent reference outflow swamp (Site R2) had two primary peaks: a smaller peak during spring (April – May), associated with melting snow on ice or frozen ground, and a second, larger peak during summer (June – August) associated with summer rain events (Figure 3.1.B). Estimated discharge rates during spring and summer of 2018, before sampling commenced, indicated that the pre-harvested area had slightly higher peaks compared to the reference area (Figure 3.1.B). During autumn, Site R2 was either dry (as in 2018 and 2021), or had substantially lower flow rates (as in 2019 and 2020) compared to the summer season (Figure 3.1.B). In 2021, the normal depth of summer precipitation did not arrive and flow ceased entirely after mid-July at Site R2 (Figure 3.1). Pre-harvest, the main harvested outflow (Site H5) was dry in the autumn (Figure 3.1.B).

Following the installation of the perimeter ditch and vegetation mulching in 2019, the discharge rates at Site H5 had similar timing of peaks in the spring and summer relative to Site R2, and the relative magnitude of flow was comparable to pre-harvest conditions. During spring and early summer 2019, spot discharge measurements indicated greater responses to rain events in the harvest outflow swamp (H5) compared to the reference outflow swamp (R2) (Figure 3.1.B). Both the reference and harvest outflow responded to large rain events in late July 2019. However, in the autumn, flow rates decreased substantially at Site R2, whereas higher flow rates continued at Site H5, indicating an increase in base flow leaving the harvested area (Figure 3.1.B). This same effect continued after the internal ditches were installed and extraction commenced. Although flow at the reference outflow ceased entirely in July, 2021, water continued to flow at Site H5 for the duration of the study (Figure 3.1.B). Internal ditches within the harvested field were frequently dry during 2021; however, they quickly became wet and began flowing again during rain storms (Figure 3.1.C). As a result, discharge leaving the harvested field internal ditches (Site H3) was flashy in response to rain events (Figure 3.1.C). Internal ditches also began flowing after ditch maintenance, when loose peat that had accumulated in the ditch was removed and the ditch bottom was cut deeper into the peat profile (Figure 3.1.C).

3.3.1.2 Ice and Snow

The depth of snow and ice within the reference (R1) and harvest (H1) peatland fields for 2019-2021 are outlined in Chapter 2. In March and early-April, ice was at or above the ground surface at the outflow swamp site (Site R2, Figure 3.2.B) compared to ice that remained below the hollow surface within in the reference peat field (Site R1, Figure 2.1.B). However, the ice remained much longer (until late July 2021) in the reference peatland compared to the outflow swamp, where ice was no longer detected by mid-June in 2021 (Figure 3.2.B). Ice had not yet established by the end of October in 2021 at Site R2 (Figure 3.2.B). The maximum snow depths were similar in the reference peatland (Site R1) and outflow swamp (Site R2), and ranged from 30 – 40 cm at in 2020 and 2021 (Figure 2.1.B; Figure 3.2.B).

At the harvested outflow swamp, ice was present above the ground surface at Sites H6 and H7 during March and April in 2020 and 2021 (Figure 3.2.C). During 2020, ice was present ~10 cm below the ground surface in the harvested peat field (Site H1), and at (Site H2) or above (Site H1) the ground surface in 2021 (Figure 2.1.C). However, in contrast to the reference outflow swamp (Site R2) and the harvest field (Site H1), the ice had fully melted at Sites H6 and H7 by mid-May (Figure 3.2.C). Maximum snow depths were variable at the harvest outflow swamp (Sites H6 and H7), and were generally shallower relative to the reference outflow (Site R2) with snow depths that ranged from 20 – 40 cm deep (Figure 3.2.C). Maximum snow depths were similarly shallow (< 25 cm) in the harvested field (Site H1).

3.3.1.3 Water Levels and Rust Depths

Water levels were influenced by the seasonal and interannual variability in precipitation experienced during the study period, and ice dynamics appeared to play a role in keeping spring water levels near the ground surface (Figure 3.2.B) and generating outflow discharge (Figure 3.1.B). At the reference outflow swamp (Site R2), water levels were at least 30 cm deep during the winter, with water unable to be detected in the well until early May, indicating that no winter outflow discharge occurred (Figure 3.2.B). As the snow began to melt in the spring at Site R2, water levels were detected 5 – 10 cm above the ground surface, perched on frozen ground (Figure 3.2.B), with limited observed outflow (Figure 3.1.B). Water levels during the summer and autumn were variable and appeared to be influenced by the availability of precipitation. During the summers of 2019 and 2020, when precipitation was plentiful, water levels stayed around 5 cm above the ground, associated with outflow discharge, before dropping slightly to the ground surface as the ground froze (Figure 3.2.B). Outflow discharge occurred throughout each summer (Figure 3.1.B). In contrast, during the 2021 dry summer, outflow discharge ceased (Figure 3.1.B) as the water level dropped to 30 cm below ground in July, and continued dropping for the remainder of the year until it was deeper than 50 cm below the ground in autumn (Figure 3.2.B). Water levels in the reference peatland (Site R1) followed a similar trend (Figure 2.1.B).

The rust depth, indicating potential biogeochemical processes that may occur independent of the source from adjacent peatlands, at Site R2 was at or just below the ground surface when standing water was present (Figure 3.2.B). However, when the water level dropped at Site R2 in mid-July 2021, the rust depth lowered but remained within 10 cm of the ground surface, and did not drop lower than ~25 cm even when the water table was deeper than 50 cm (Figure 3.2.B). Rust depths remained within 20 cm of the hollow surface with in the reference peatland (Site R1), as well.

Water levels at the pre-harvest outflow (Site H6) were at or below the ground surface in the autumn with very low (< 0.01 L s^{-1}) outflow discharge observed at the culvert (Site H8) (Figure 3.1.C;

Figure 3.2.C). During the initial stages of peatland opening, when the perimeter ditch was installed and vegetation mulching had commenced, water levels at in the harvested outflow swamp (Site H6) were slightly higher than Site R2 during the spring and summer, but rose significantly from ~ 10 cm above ground to 50 cm above ground after beaver damming was initiated in late August 2019 (Figure 3.2.C). The water level at Site H6, now within the beaver pond, remained between 55 – 70 cm above the ground, on top of a thick layer of unconsolidated flocculant in 2020 and 2021 after the internal ditches were installed and extraction had begun (Figure 3.2.C). Below the beaver dam, at the new harvest outflow swamp location (Site H7), the water level was maintained at approximately 5 cm above the ground regardless of the year or season (Figure 3.2.C). This area was continuously flooded and rust was not detected below the ground surface during 2020 and 2021 (Figure 3.2.C).

3.3.2 Harvested Outflow: Indicators of Water Source and Flow Path

Comparison of selected physiochemical parameters of flowing water within the constructed drainage network, from the harvested field internal ditch (Site H3), to the cumulative perimeter ditch (Site H4), and the base of the main outflow ditch (Site H5), as well as potential sources from the reference (Site R1) and harvest (Site H1) peatland field sites (see Chapter 2) are illustrated in Figure 3.3 and 3.4.

3.3.2.1 Temperature

Pre-harvest water temperatures in the autumn of 2018, at the future location of the terminal point for the main harvested outflow ditch (Site H5), were similar to temperatures in the surface and deep porewaters within the harvested and reference peatlands (Sites H1 and R1) (Figure 3.3.A). However, once the perimeter and main ditches were installed in 2019, the water temperature at all harvested outflow ditch locations showed a clear hump-shape that mirrored the seasonal air temperature variability (Figure 3.3.A, Figure 1.4). None of the harvested outflow ditch locations had water temperatures that resembled the Site R1 reference surface or deep porewater temperatures (~2.5 – 6 °C; Chapter 2) (Figure 3.3.A). However, temperatures did resemble the surface porewaters in the harvested field (Site H1) in 2019 (Figure 3.3.A). The harvested outflow ditch locations (Site H3, H4, and H5) continued to have warmer water relative to the reference peat field (Site R1) once the internal ditches were cut and extraction was ongoing in 2021. Water temperatures ranged from 1 °C in the early spring, to > 25 °C during the summer months (Figure 3.3.A).

3.3.2.2 pH

The pre-harvest pH values at the future location of the terminal point for the main outflow ditch (Site H5) were between 4.3 – 4.5, and resembled the deep porewater at Sites R1 and H1 (Figure 3.3.B). After the perimeter and main ditches were installed and vegetation mulching was underway within the harvested field in 2019, pH values in the perimeter (Site H4) and main outflow (Site H5) ditches rose considerably relative to the pre-harvest pH, with values ranging between 6.1 and 7.5 (Figure 3.3.B). Once the internal ditches were installed within the harvested field and peat extraction was ongoing, pH values taken within the perimeter and main ditches (Sites H4 and H5) remained high (5.5 – 8.0). In contrast, values taken from within an internal ditch leaving the harvested field (Site H3) were similar to the adjacent harvested field (Site H1; < 4.5) during the spring and summer (Figure 3.3.B). However, the pH at Site H3 began to increase in the late summer and autumn to values between 6 – 6.5 (Figure 3.3.B).

3.3.2.3 Electrical Conductivity

Pre-harvest EC measurements at Site H5 were ~50 µS cm^{-1}, and resembled the reference surface porewaters (Sites R1 and H1) (Figure 3.3.C). After the perimeter ditch was installed and the vegetation was mulched in the harvested field, the EC values in the perimeter and outflow ditches (Sites H4 and H5) varied considerably over the year (150 – 375 µS cm^{-1}), but were generally much higher than EC values measured at the peatland reference field (Site R1) and the newly mulched Site H1 (Figure 3.3.C). Following the installation of internal ditches and active extraction in the harvested field, the EC values at Sites H4 and H5 were substantially higher relative to the reference porewaters during the spring and summer months, except for the large snowmelt event in April, when EC values were < 100 µS cm^{-1} at all sampling sites (Figure 3.3.C). In contrast, the EC values in the internal ditch outflow (Site H3) closely resembled Site R1 and H1 during the spring and summer, but deviated and began to resemble Sites H4 and H5 in the late summer and autumn following clearing and deepening of the internal ditches (Figure 3.3.C). The EC trends throughout this study very closely resembled the changes in Ca, Mg, and Na concentrations (Appendix, Figure A.1).

3.3.2.4 Chloride

The Cl concentrations within the reference and harvested peatlands and at the outflows along the harvested drainage network fluctuated over the study period. Pre-harvest Cl concentrations at Site H5 were ~0.25 – 50 mg L^{-1} and resembled peatland (Sites R1 and H1) porewaters (Figure 3.3.D). During 2019, the newly ditched Sites H4 and H5 had similar Cl concentrations to each other and to Site R1 surface waters (0.25 – 1 mg L^{-1}). However, the concentrations increased at Sites H4 and H5 and resembled the newly mulched peatland Site H1 deep porewaters later in the summer and autumn

(Figure 3.3.D). The high Cl concentrations observed immediately following vegetation mulching in the harvest field (Site H1) surface porewaters were not observed in any of the outflow ditches. Once the internal ditches were installed and peat extraction was ongoing, all outflow ditches (Sites H3, H4, and H5) had Cl concentrations that resembled H1 surface porewaters, and were higher relative to the reference peatland (Site R1) porewaters during the spring and summer (Figure 3.3.D). However, Cl concentrations decreased at all harvested outflow locations as the summer progressed (Figure 3.3.D).

3.3.2.5 Silica

Silica concentrations were measured in 2021 only. The Si concentrations in the perimeter ditch (Site H4) and at the end of the main outflow ditch (Site H5) were high in the early spring during low flow and ice cover, were very low when the snow melted in April, and stabilized at concentrations ~6 mg L^{-1} for the remainder of the summer and autumn (Figure 3.3.E). Much like the pH and EC values, the internal ditch outflow, Site H3, had similar Si concentrations to the peatland Site R1 and H1 surface porewaters in the spring and summer (3 – 5 mg L^{-1}), but the concentrations rose to levels similar to Sites H4 and H5 in the later summer and autumn (Figure 3.3.E).

3.3.2.6 Dissolved Organic Carbon

In autumn 2018, pre-harvest, DOC concentrations at the future terminus of the main outflow ditch (Site H5) were ~60 mg L^{-1}, and resembled peatland field Site R1 and H1 surface porewaters (Figure 3.4.A). Following the installation of the perimeter ditch and mulching of vegetation in the harvested field, the DOC concentrations in the perimeter (Site H4) and main outflow (Site H5) ditches were lower relative to the peatland reference (Site R1) and harvested (Site H1) porewaters in the spring (~50 mg L^{-1}). However, they rose to levels similar to Site R1 surface and deep porewaters in the summer and autumn (~75 – 100 mg L^{-1}) (Figure 3.4.A). After the internal ditches were installed and extraction had commenced in the harvested field, the mineral ditches (Sites H4 and H5) had similar DOC concentrations to the peatland field Sites R1 and H1 (Figure 3.4.A). In contrast, the internal ditch outflow, Site H3, had much higher DOC concentrations (~150 – 175 mg L^{-1}) in the summer relative to all other sites, but levels dropped to concentrations similar to Sites H4 and H5, in the autumn (Figure 3.4.A).

3.3.2.7 Potassium

The pre-harvest K concentrations at the main outflow (Site H5) were near 0.5 mg L^{-1} and resembled peatland field Sites R1 and H1 (Figure 3.4.B). Following the installation of perimeter ditches in 2019, Sites H4 and H5 had low K concentrations that resembled the pre-harvest conditions in the spring and early summer (< 1 mg L^{-1}). However, the concentrations rose to 2 – 3 mg L^{-1} in the late summer and autumn alongside high concentrations observed in the newly mulched peatland harvest Site H1 deep porewaters (Figure 3.4.B). Once the internal ditches were installed and extraction was ongoing, K concentrations at harvested outflow Sites H4 and H5 remained relatively constant (1 – 2 mg L^{-1}) throughout the year at levels slightly higher than those observed at the reference peatland Site R1 (Figure 3.4.B). In contrast, the internal ditch outflow (Site H3) had low K concentrations (1 – 2 mg L^{-1}) in the spring and fall, but very high concentrations in the summer (6 – 7 mg L^{-1}) (Figure 3.4.B).

3.3.2.8 Aluminum

Pre-harvest, the Al concentrations at the future main outflow ditch end point (Site H5) were high (~300 µg L^{-1}) relative to the reference peatland Site R1 porewaters (Figure 3.4.C). After the perimeter ditching and vegetation mulching in the harvested field in 2019, the perimeter ditch (Site H4) had Al concentrations similar to Site R1, while the main outflow ditch (Site H5) had a peak in Al concentration (300 µg L^{-1}) at the end of July (Figure 3.4.C). Once the internal ditches were installed and extraction was underway in the harvested field, Sites H4 and H5 had stable concentrations of Al throughout the year between 50 – 100 µg L^{-1} that was lower relative to peatland reference (Site R1) and harvested field (Site H1) (Figure 3.4.C). In contrast, the internal ditch outflow (Site H3) was similar to Sites H4 and H5 in the spring while the snow was melting, but Al concentrations rose substantially during the summer (> 500 µg L^{-1}) to levels much higher than Sites R1 and H1 before returning back to ~50 µg L^{-1} in autumn (Figure 3.4.C).

3.3.2.9 Iron

Pre-harvest, the Fe concentrations at the main outflow (Site H5) were between 1 – 1.5 mg L^{-1} and resembled the surface porewaters at peatland Sites R1 and H1 (Figure 3.4.D). During 2019, following the installation of the perimeter ditch and vegetation mulching in the harvested field, the mineral ditch outflows (Sites H4 and H5) had similar Fe concentrations to each other, and were greater relative to the peatland Site R1 and H1 porewaters (Figure 3.4.D). The Fe concentrations were highest in the spring and autumn (5 – 5.5 mg L^{-1}), and lowest in mid-summer (~2.5 mg L^{-1}) (Figure 3.4.D). After the internal diches were installed and extraction was active in the harvested field, outflow ditch Sites H4 and H5 had higher Fe concentrations relative to the peatland Site R1 surface and deep porewaters in the spring (~5.7 mg L^{-1}), but resembled Site R1 deep porewaters later in the summer and autumn (Figure 3.4.D). The internal ditch outflow (Site H3) had Fe concentrations at or below 2 mg L^{-1} for the duration of the year and resembled surface porewaters at peatland Sites R1 and H1, except for a peak at the end of July ~5 mg L^{-1}

(Figure 3.4.D). All sites had decreased Fe concentrations in April during snow melt on frozen ground (Figure 3.4.D).

3.3.3 Harvested Outflow: Nutrient and Sediment Concentrations and Export Rates

Comparison of nutrient and sediment concentrations in flowing water within the constructed drainage network, from the harvested field internal ditch (Site H3), cumulative perimeter ditch (Site H4), and at the end point of the main outflow channel (Site H5), as well as the potential sources of dissolved forms of nutrients in the reference (Site R1) and harvested (Site H1) peatland fields (see Chapter 2) are illustrated in Figure 3.5 and Figure 3.6.

3.3.3.1 Nitrogen

Pre-harvest, the concentrations of TN, TDP, NH_4^+-N, and NO_3^--N were ~1 500 µg L^{-1}, ~1 000 µg L^{-1}, ~10 µg L^{-1}, and below detection, respectively, at the future terminus of the main outflow ditch (Site H5) and resembled porewaters at the peatland Sites R1 and H1 (Figure 3.5). After the perimeter ditch was cut and the vegetation was mulched in the harvested field in 2019, the TN concentrations resembled the pre-harvest values, except for a notable peak during the summer in the perimeter ditch (Site H4, ~3 000 µg L^{-1}) and main outflow ditch (Site H5, ~4 700 µg L^{-1}) (Figure 3.5.A). In contrast, the TDN concentrations at Sites H4 and H5 rose during the summer relative to the pre-harvest concentrations and the Site R1 porewaters, but resembled the H1 deep porewaters (Figure 3.5.B). A small increase in the NH_4^+-N concentration relative to Site R1 porewaters, and resembling Site H1 porewaters, was observed at Site H5 during May, but NH_4^+-N was not altered by the perimeter ditch installation otherwise (Figure 3.5.C). The NO_3^--N remained below detection limits at both Sites H4 and H5 (Figure 3.5.D).

Following the installation of internal ditches and extraction within the harvested field, the perimeter and outflow ditches (Sites H4 and H5) had similar TN concentrations to the pre-harvest values (Figure 3.5.A). However, the TN concentrations in the internal ditch (Site H3) were much higher in the summer, with a peak ~10 700 µg L^{-1} (Figure 3.5.A). The TDN concentrations at Sites H4 and H5 rose gradually over the summer, but closely resembled the porewater concentrations at reference peatland Site R1 (Figure 3.5.B). In contrast, the internal ditch (Site H3) resembled the surface and deep porewaters at Site H1 and had substantially higher TDN concentrations in the summer (~3 000 µg L^{-1}) relative to peatland reference Site R1 porewaters (Figure 3.5.B). The NH_4^+-N concentrations were the highest at the main outflow (Site H5) under the ice during early March; the concentration was more than 6 times the concentration found in the snow (Figure 3.5.C). Ammonium concentrations at outflow ditches Sites H3, H4, and H5 were higher in the early spring relative to peatland reference Site R1, but resembled the peatland harvest Site H1 porewater (Figure 3.5.C). However, the concentrations at Sites H3, H4, and H5 lowered to values similar to Site R1 porewaters after the snow melted, and remained low until the end of summer and autumn (Figure 3.5.C). Nitrate was only detected in the early spring, with greater concentrations relative to Site R1 (and snow) found at Site H4 (~310 µg L^{-1}) and H5 (~230 µg L^{-1}) prior to the snow melting (Figure 3.5.D).

3.3.3.2 Phosphorus

The TP, TDP, and SRP concentrations at Site H5, prior to construction of the main outflow ditch, were ~75 µg L^{-1}, ~60 µg L^{-1}, and ~40 µg L^{-1}, respectively, and resembled the peatland Site R1 and H1 porewater concentrations (Figure 3.6.A,B,C). After the perimeter ditch was installed and the vegetation was mulched in the harvested field in 2019, the TP concentrations at the perimeter ditch (Site H4) and the main outflow ditch (Site H5) were similar to the pre-harvest reference concentrations until the middle of summer, when the TP concentrations rose to ~700 µg L^{-1} at Site H4 and ~1 300 µg L^{-1} at Site H5 at the end of July (Figure 3.6.A). The TDP and SRP concentrations at Sites H4 and H5 followed a similar trend, albeit with a less sharp peak at the end of the summer (TDP ~350 µg L^{-1}, SRP ~300 µg L^{-1}) that were similar concentrations to those observed in the deep porewater at peatland harvest Site H1 (Figure 3.6.B,C). Except for during July 2019, the dissolved (TDP) fraction of the TP was higher than the particulate fraction at Sites H4 and H5 (Figure 3.6.A,B).

Once the internal ditches were installed and extraction was active in the harvested field, the TP concentrations at Sites H4 and H5 remained around 150 – 250 µg L^{-1}, except for a peak in April dominated by particulate phosphorus at Site H5 (~650 µg L^{-1}), and peak at the end of October dominated by TDP at Site H4 (~550 µg L^{-1}) (Figure 3.6.A,B). Site H3 had much higher high TP concentrations during the summer, with a peak ~1 800 µg L^{-1} (Figure 3.6.A). The TDP and SRP concentrations at Sites H4 and H5 were slightly higher than Site R1 porewaters (between 100 – 150 µg L^{-1} and 50 – 125 µg L^{-1}, respectively), with the highest concentration of TDP (~330 µg L^{-1}) and SRP (230 µg L^{-1}) observed at Site H4 in autumn (Figure 3.6.B,C). In contrast, the TDP and SRP concentrations at the internal ditch outflow (Site H3) were much higher than Sites H4 and H5 in the summer (TDP ~950 µg L^{-1}, and SRP ~780 µg L^{-1}) and resembled the concentrations observed at the harvest peatland Site H1 surface porewaters (Figure 3.5). Of the TDP measured in 2021, approximately 50 – 75 % was SRP (Figure 3.6.B,C).

3.3.3.3 Total Suspended Solids and Particulate Carbon

Particulate carbon concentrations generally represented less than 10 % of the TSS concentrations at Sites H4 and H5 in both 2019 and 2021 (Figure 3.6.D,E). Further, the concentrations varied between years, with lower concentrations observed in 2021 relative to 2019. Peaks in sediments appeared to be associated with large influxes of water, such as during snow melt and after large rain events (Figure 3.1; Figure 3.6.D,E). Although there were substantial TSS and PC concentrations measured at Site H3 in June 2021, elevated concentrations were not observed downstream at Sites H4 and H5 (Figure 3.6.D,E).

3.3.3.4 Nutrient and Sediment Mass Discharge Rates

Comparison of the daily mass discharge (concentration (kg L^{-1}) multiplied by discharge (L day^{-1})) along the constructed drainage network from the harvested field internal drainage ditch (Site H3), cumulative perimeter ditch (Site H4), and the end of the main outflow channel (Site H5) for selected parameters are shown in Figure 3.7 and Figure 3.8.

Not only were the concentrations variable over the study period, but the timing and volume of water flow fluctuated dramatically, resulting in mass discharge rates that varied considerably by season and year. However, full seasonal sampling was not completed in each year, thus comparison is limited to between the autumn (September – October) of all years, and the summers (May – August) of 2019 and 2021. Nonetheless, measured N, P, TSS, and PC daily mass discharges were generally lower in 2021 relative to 2019 (Figures 3.7, 3.8). Peak export rates for N, P, TSS, and PC were highest during peak water discharge in late-July, 2019, regardless of the water concentration. The exception was NH_4^+-N and NO_3^--N daily mass discharges which had substantial peaks during snowmelt in 2021 in response to both seasonal high discharge and very high concentrations at this time. Unfortunately, no direct comparison with peak spring nutrient mass discharge in 2019 can be made. Comparisons between seasons are possible for 2021, where highest exports were observed during spring melt in response to higher water flow. In summer 2021, the daily mass discharge rates were substantially reduced due to the prolonged low or absence of water flow. Comparison of autumn (September – October) mass discharge rates trended from low to increasing higher daily mass discharge for all forms of nutrients from 2018 to 2021 (Figure 3.7; Figure 3.8). This occurred alongside increased hydrologic connectivity and prolonged autumn outflow discharge (Figure 3.1).

For nitrogen (N) species, the cumulative perimeter ditch (Site H4) and the end of the main outflow ditch (Site H5) had similar TDN, NH_4^+-N, and NO_3^--N daily mass discharge rates in both 2019 and 2021 (Figure 3.7). During most sampling periods, total inorganic nitrogen (TIN; NH_4^+-N and NO_3^--N) represented a minor portion of TDN, and thus dissolved organic N represented 50 % or more of the total nitrogen (TN) export. However, following the large rain event in July 2019 and during the pulse of water after snow melt in April 2021, contributions from TIN remained low but overall daily TN export increased moving from Sites H4 to H5, while the daily TDN export was similar. The remaining portion of TN export would be from particulate N, as also reflected in increased PC daily mass export from Sites H4 to H5 during these two higher flow events (Figure 3.8).

Similarly, TP, TDP and SRP daily exports were comparable between Sites H4 and H5 for most sampling dates. The TDP represented about 50 % of the TP, and thus the remainder was as particulate phosphorus export. In contrast to TN, the TDP daily export was primarily from inorganic phosphorus (SRP). During the July 2019 and April 2021 flow events, the rate of SRP export was much greater at Site H4 compared to downstream at Site H5. However, the daily mass TP export at the downstream Site H5 was more than double the export in the perimeter ditch (Site H4), and this coincided with 5-fold increase in TSS export from Sites H4 to H5. The low PC contribution compared to the TSS export indicates that the increase is due to mineral sediment export. Notably, the high N, P, TSS, and PC concentrations observed at the internal ditch outflow (Site H3) during April or June 2021 did not appear to contribute much to the mass discharge rates (Figure 3.5; Figure 3.6; Figure 3.7; Figure 3.8); Site H3 had consistently low mass discharge rates relative to Sites H4 and H5.

3.3.4 Comparing Outflow Swamp Responses and Influences of Beaver Activity

3.3.4.1 Water Source Indicators

Comparison of seasonal physiochemical characteristics in flowing water at the reference outflow swamp (Site R2) and harvested outflow swamp above (Site H5) and below (Site H7) the beaver dam during the 2018 – 2021 study are illustrated in Figure 3.9 and Figure 3.10.

In 2019, after the perimeter ditch was installed and vegetation was mulched within the harvested field, the harvested main outflow (Site H5) had initially higher water temperatures, pH, EC values, and Fe concentrations in spring and early summer, relative to the reference outflow (Site R2), and were comparable the rest of the year (Figure 3.9.A,C; Figure 3.10.B,D). The Cl concentrations at Site H5 were similar to Site R2, and slightly increased over the sampling period in 2019 (Figure 3.9.D). Aluminum concentrations were variable, but a peak in concentration was apparent at Site H5 in July relative to Site R2 (Figure 3.10.C). The DOC and K concentrations were comparable to Site R2 in the spring and early-summer, but Site R2 DOC concentrations rose in the late summer compared to Site H5, where as K concentrations only increased slightly (Figure 3.10.A).

Comparison of the harvested outflow swamp above (main channel H5) and below (Site H7) the beaver dam in 2021, indicated potential water quality changes from beaver dam construction. It also indicated flooding from water at the main outflow channel (Site H5) that was influenced by the internal (Site H3) and perimeter (Site H4) ditch construction, as well as active peat extraction in the harvest fields. Site H7 had very similar seasonal trends and magnitude of EC measurements and Cl, Si, DOC, and Al concentrations relative to Site H5 (Figure 3.9.C,D,E; Figure 3.10.A,C). In contrast, the temperature and pH values below the dam at Site H7 were lower relative to the flowing water above the dam at Site H5. The K and Fe concentrations were higher below the dam at Site H7 relative to above the dam at Site H5 (Figure 3.9.A,B; Figure 3.10.B,D).

During the extraction phase in 2021, comparisons of physiochemical characteristics of flowing surface water at the harvest outflow swamp (Sites H5 and H7) with the reference outflow swamp (Site R2) were limited to spring and summer because the reference outflow swamp was dry after mid-July. For all parameters, the values and concentrations, with the exception of K, were similar and low at the harvested and reference outflow swamps during peak water levels and discharge during spring snow melt in April (Figure 3.9 and 3.10). Following spring melt, the water temperature, pH, and EC measured at the harvested outflow swamp (Sites H5 and H7) were higher than the reference outflow swamp (Site R2) (Figure 3.9.A,B,C; Figure 3.10.D). Iron concentrations above the beaver dam at the harvested outflow swamp were similar, whilst Fe concentrations below the dam were much higher than Fe concentrations at the reference outflow swamp. The Cl and K concentrations were considerably higher at Site R2 relative to Sites H5 and H7 in the spring, but both concentrations declined substantially in the early summer at Site R2, whereas Sites H5 and H7 remained stable (Figure 3.9.B). The opposite was observed with Al concentrations; Al concentrations were lower at Site R2 relative to Sites H5 and H7 in the early-spring, but rose higher relative to Sites H5 and H7 in the late-spring and early summer (Figure 3.10.C). The Si concentrations rose together at Sites R2, H5, and H7 in the spring, but the Si concentration remained high at the harvested outflow swamp into the summer and autumn, whereas the reference outflow swamp concentrations declined prior to the termination of flow (Figure 3.9.E). The DOC concentrations at Sites R2, H5 and H7 were similar while water was present to sample (Figure 3.10.A).

3.3.4.2 Nutrients and Sediments

During the opening phase in 2019, the TN and TDN concentrations were lower and the NH_4^+-N concentrations were low but similar at the harvested outflow swamp main channel (Site H5) compared to the reference outflow swamp (Site R2) (Figure 3.11.A,B,C). Detectable NO_3^--N concentrations were observed at Site R2 in late May 2019, but were below detection at Site R2 and Site H5 for the duration of the sampling period (Figure 3.11.D). In contrast, the TP, TDP, and SRP were notably higher at the harvested outflow swamp (Site H5) relative to the reference outflow swamp (Site R2) during the summer (Figure 3.12.A,B,C). The TSS concentrations were similar at the main harvest outflow (Site H5) and the reference outflow (Site R2) when sampled during moderate or low discharge. The exception was for a very large peak (~1 600 mg L^{-1}) at the harvest outflow Site H5 at the end of July, following a large rain event (Figure 3.11.D). In contrast, the concentrations of PC were much lower than TSS, and lower at Site H5 relative to Site R2 during July and August; however, peaks in PC concentrations were observed at both the harvest and reference site during the July rain event (Figure 3.12.E).

When comparing the potential influence of the beaver dam on the quality of water received at the main outflow (Site H5) entering the outflow swamp during the extraction stage, the NH_4^+-N, TP, TDP, and SRP concentrations of flowing water below the dam at Site H7 were consistently higher than concentrations at the harvest outflow (Site H5) (Figure 3.11.C; Figure 3.12.A,B,C). In contrast, there was no or little change in the TN and TDN concentrations from the main outflow at Site H5 to below the beaver dam at Site H7 (Figure 3.11.A,B). The NO_3^--N and TSS concentrations measured at Site H5 were similar to Site H7, except for during snowmelt in April, 2021, when concentrations were lower below the beaver dam at Site H7 compared to above the dam at Site 5 (Figure 3.11.D; Figure 3.12.D). The PC concentrations at Site H7 were generally higher and much more stable relative to Site H5; however, as with the TSS concentration, the PC concentration was lower at Site H7 during snowmelt relative to Site H5 (Figure 3.12.E).

For spring to June, when water was available for comparison between the reference outflow swamp and the harvested outflow swamp, the TN and TDN concentrations were similar at Sites R2, H5 (above the dam), and H7 (below the dam). The exception was in May, when concentrations spiked at Site R2 relative to Sites H5 and H7 before returning to similar levels in June (Figure 3.11.A,B). The NH_4^+-N concentrations at the reference outflow (Site R2) were similar to the main outflow (Site H5); however, both sites had lower NH_4^+-N concentrations compared to below the dam at the harvest outflow Site H7 for most summer sampling dates (Figure 3.11.C). The exception was a spike in NH_4^+-N concentrations during May at Site R2 (Figure 3.11.C). The NO_3^--N concentrations at Sites R2 and H7 were similar before and during snowmelt; NO_3^--N concentrations were below detection for sample dates at all sites after April (Figure 3.11.D).

The phosphorus (P) concentrations at the harvested outflow swamp (Site 5, above the dam) were similar to concentration at the reference outflow swamp (Site R2), when flow existed. Phosphorus concentrations at the harvest outflow below the dam at Site H7 were also similar relative to Site R2 in the spring (Figure 3.12.A,B,C). But, while the P concentrations declined at Site R2 (as well as at Site H5) before the water dried up in mid-July, they stayed higher at Site H7 into the fall (Figure 3.12.A,B,C). Both harvested outflow swamp sites (H5 and H7) had higher TSS concentrations compared to the reference outflow swamp (Site R2) during snow melt, but were otherwise similar when water was available for comparison (Figure 3.12.D). In contrast, concentrations of PC did not follow similar trends. Site R2 had similar PC concentrations compared to Site H7 during snow melt, but was much higher than both Sites H5 and H7 in late May (Figure 3.12.E).

3.3.4.3 Nutrient and Sediment Mass Discharge

There were notable differences in the daily mass discharge of nutrients depending on seasonal and interannual differences in precipitation at both the harvested and reference outflows. When comparing the relative change in export between Sites H5 and R2 during the opening phase in 2019, similar magnitudes and seasonal changes in daily mass discharge were observed for all forms of N, as well as PC, at both the harvest outflow swamp Site H5 compared to the reference outflow swamp Site R2, with peaks during the large rain event in July (Figure 3.13, Figure 3.14.E). This contrasts the trends in P and TSS daily mass discharge (Figure 3.14.A,B,C,D). The magnitude of daily mass discharge of TP, TDP, and SRP at the harvest outflow Site H5 was similar to that at the reference outflow (Site R2) during lower flow events in late spring and autumn (Figure 3.14.A,B,C). However, during higher water flow in the summer, the daily export values of total (TP) and dissolved P (TDP and SRP) were greater from the harvest outflow swamp (Site H5) compared to the reference outflow swamp (Site R2) (Figure 3.14.A,B,C). Of note was the large proportion of TP export contributed by dissolved inorganic P (30 – 40 % SRP) at the harvest outflow swamp for most sampling periods (Figure 3.14.A,C). The exception to this trend was during the high flow event in July, 2019. Although TDP and SRP mass discharge from the harvested outflow swamp (Site H5) also peaked during July, daily export of TP increased by more than 5-fold and only 2- to 3-fold for dissolved P (Figure 3.14.A,B,C). Comparison with the daily mass discharge of TSS relative to PC, indicates that the increased P export was primarily mineral particulate P (Figure 3.14).

During the extraction phase in 2021 the daily mass discharge of N, P and TSS from both the harvest and reference outflow swamps were considerably less, reflecting the lower water discharge in

2021 compared to 2019 (Figure 3.13; Figure 3.14). Daily mass discharge of all forms of N and P from the harvested outflow swamp (Site H5) peaked and were greater than mass discharge from the reference outflow swamp (Site R2) during the spring snow melt (Figure 3.13; Figure 3.14). Daily export values of N and P were similar at Sites H5 and R2 during the early summer (Figure 3.13; Figure 3.14). However, once water flow from the reference outflow swamp ceased and daily export was zero, continuous flow resulted in a cumulative increase in mass discharge of N and P from the harvest outflow swamp (Site H5) during the late summer and autumn (Figure 3.13; Figure 3.14).

The establishment of the beaver dam had considerable influence on net export from the harvest outflow swamp. There was a sizable reduction in daily mass discharge of forms (TN, TP, TSS, and PC) below the dam (Site H7) relative to the main outflow ditch above the dam (Site H5) during peaks in export in early spring (Figure 3.13.A; Figure 3.14.A,D,E). In contrast there was a notable increase in daily mass discharge of NH_4^+-N, TDP, and SRP export rates at Site H7 relative to Site H5 throughout the summer and autumn (Figure 3.13.C; Figure 3.14.B,C). When compared to the magnitude of export at Site R2, the N and P export rates were much higher at Site H7 in early spring, but export rates of TN, TDN, and TP show similar trends relative to Site R2 for the remainder of the year (Figure 3.13, Figure 3.14). However, Site R2 was dry after mid-July 2021 resulting in no nutrient exports, while Site H7 continued to export NH_4^+-N, TDP, and SRP during the summer and autumn.

3.4 DISCUSSION

3.4.1 Hydrological Alterations at the Harvested Outflow

3.4.1.1 Discharge and Water Levels

This study showed a large range of flow volumes and water level depths that fluctuated both within and between years at both the harvested outflow and reference outflow swamp. Although alterations to the timing and magnitude of peak flows at the harvested outflow relative to the reference could not be determined exactly, modifications to the persistence and seasonality of flow as a response to ditching was shown. Installing the perimeter ditch prolonged the flow into autumn and, as a result, flowing water was always present at the harvested outflow swamp even when flow had ceased at the reference outflow swamp. Similar alterations to the base flow following peatland harvesting were observed by Prevost et al. (1999), who found summer flows increased by 25 % in a drained forested peatland.

Although water was always flowing at the harvested outflow, the internal ditches frequently dried up and flashy responses to influxes of water from melting snow or rain were observed. Instances of dry peatland ditches have been observed in Finland, and were responsible for alterations to the sediment concentrations when they were re-wetted following sequential rain events (Marttila & Kløve, 2010). Flashy flows have been associated with peatlands that were saturated with water within 5 cm of ground surface (Evans et al., 1999), as well as in drained peatlands from infiltration excess overland flow (Holden, Evans, et al., 2006). While the rapid increase in flow observed in this study could be a result of saturated overland flow when ice was present, once the ice melted, relatively heavy precipitation was required to maintain connectedness. The flashiness later in the season when the water table in the harvested field was low may be due to a slowed infiltration, possibly from higher bulk densities and compression of surface peat (McCarter et al., 2020), as observed in the harvested field in this study (Chapter 2). This suggests that seasonal and interannual variability in precipitation are important indicators of hydrological connectivity and must be accounted for when estimating the nutrient leaching risk. The findings in this study caution that the water sampled from the perimeter and main outflow ditches may not be solely sourced from water in contact with the harvested surface peat at all times, although pulses of water from the harvested field could influence the outflow water quality during select events.

This said, the persistence of the water observed at the harvested outflow, even when the internal ditches and surface peats did not contain free water, indicates that cutting the perimeter and main outflow ditches altered the hydrological flow path from surface dominated to a combination of surface and deeper waters that varied in proportion depending on the water table depth within the peatland. This has been documented in drained blanket bogs in the UK, where flow paths were shown to shift from surface dominated to subsurface following ditching, although the influence of snow melt and ice were deliberately excluded from the study (Holden, Evans, et al., 2006).

3.4.1.2 Source Waters and Flow Paths

The higher water temperatures along the ditch and at the harvested outflow suggests that the water was sourced from the surface peat, and not from cooler deep peat porewater or groundwater intersected by the perimeter ditch. While it is possible that the elevated water temperatures observed at the outflow were because of higher exposure to incident radiation from the lack of shade, water sourced from deep underground has been associated with cool, stable temperatures that do not tend to fluctuate with daily or seasonal air temperature (Kalbus et al., 2006). The water temperature ranges that varied alongside the air temperature therefore indicate that the water found at all outflow sites likely contained a high proportion of surface or near surface water.

Whilst the water temperature at the harvested outflow had a signature resembling the surface peat porewater, the higher pH values, EC measurements, and Si concentrations pointed to a water source influenced by mineral instead of peat as soon as contact was made with the perimeter and main outflow ditches. Previous studies have also noted alterations to pH and EC following mineral contact, either as a result of mingling with groundwater or from contact with mineral ditches (Åström et al., 2001; Joensuu et al., 2002; Pschenyckyj et al., 2023). In this study, it is unlikely that groundwater intersected by the perimeter and main outflow ditches was the dominant water source at the harvested outflow swamp because, in addition to variability of warm water temperatures, the peatland was underlain by clay and was situated on a topographic high. This limits the likelihood of groundwater discharge rates great enough to sustain prolonged flow at the harvested outflow because hydraulic conductivities in clay are very low and the regional and local topographic position does not favor groundwater discharge but, rather, recharge (Winter et al., 2003). In addition, the similar DOC concentrations between the harvested outflow and the reference surface and deep peat porewater concentrations further indicate that the water was sourced from within the peatland and not from a geochemical source. The mineral influence observed in this study could be explained by water coming into contact with the mineral subsoil within the perimeter and main outflow ditches, and transformations occurring as a result.

If transformations within the ditch were the main driver for changes in EC, pH, and Si concentrations observed at the outflow, then the type of mineral soil and rate of mineral weathering is likely a key moderator. The acidity of the peat waters entering the ditch could accelerate weathering of the carbonate-rich clay subsoil, causing the clear increase in pH, EC, and Si concentrations. This was observed by McMahon et al. (1995), who found that Si, Ca, Mg, and Fe dissolution was accelerated in organic acid-rich waters. The DOC concentrations at the harvested outflow represent an abundance of organic acids that could be available for mineral weathering that could be sustained if the water was sourced from both surface and deep peat. However, the proximity to bedrock and the mineral composition of the subsoil will vary across Canada, not to mention places where the peat depth is substantial and ditches do not contact the mineral below. While ditches are clearly a key player in water quality transformation in this study, the influence of ditches in other settings could be inconsequential or have different chemical influence. Therefore, further study is required in settings with different mineral subsoils and deeper peat depths to tailor best management practices.

In addition to the transformations that likely occurred within the mineral perimeter and main outflow ditches, the water at the harvested outflow swamp appears to have been sourced from surface and deep peat waters as indicated by the concentrations of Cl and K. Previous studies have used Cl as a conservative tracer to indicate different water sources (Khan et al., 2015; Lockwood et al., 1995). The Cl concentrations at the harvested outflow suggest a switching of water source during low flow in 2021, from a high proportion of surface water when the harvested field was hydrologically connected to the outflow, to mostly deep peat water during low flow. Similarly, the K concentrations indicate that water within the harvested field contributed significantly to the harvested outflow, especially during wet periods when the surface peat was hydrologically connected. Potassium has been shown to readily leach from the peat profile (Damman, 1978; Prevost et al., 1999). Since the harvested field surface and deep waters had higher K concentrations relative to the reference surface and deep porewaters, and the harvested outflow had K concentrations that were in-between the two, it follows that water from the harvested field was present at the outflow in sufficient quantities to increase the K concentrations relative to the reference baseline. This also indicates that during instances where the water quality at the harvested outflow did not have a chemical signature that reflected mixing between the harvested and reference fields, then transformations to the water in contact with the mineral substrate in the perimeter and main outflow ditches must have been responsible for the change.

3.4.2 Nutrient and Sediment Concentrations and Export Rates

The nutrients and sediments observed at the harvested outflow were clearly influenced by the hydrological connectivity between the harvested field and outflow. During high connectivity, the harvested field appears to have been a major source of P, but when the connectivity was low in 2021, the source of P was not retained at the same concentrations at the harvested outflow. This could be due to the lack of freely available water and a decreased soil anoxia that could prevent P from mobilizing from the harvested field. The TDN looks to have been sourced from both the harvested field and the surrounding non-harvested peatland because the concentrations in the perimeter ditch and at the main outflow followed DOC, and remained at similar levels in both 2019 and 2021. This shows that the source of nutrients can come from both the harvested field and the neighbouring non-harvested surface and deep peat porewaters, but that the connectivity to the harvested field can be transient.

It appears that the installation of internal ditches and early extraction increased the concentrations of TN, TDN, P, TSS, and PC within the internal ditches in the harvested field, but the lack of hydrological connectivity in 2021 prevented much of their transportation downstream. Despite the clearly elevated concentrations in summer 2021 in the internal ditch, concentrations within the perimeter and main outflow ditches remained low, and the export rates extrapolated to all the internal ditches were very small. The huge difference in water availability and flow between 2019 and 2021 highlights the importance of considering export rates when determining the risk to downstream waterbodies because higher concentrations were often not associated with higher export in this study. This cautions that concentration alone is not a suitable indicator of the risk of nutrient transport off site.

Despite the clear influence of water altered within the harvested field and additional water sourced from surface and deep peat, the most significant alteration to the outflow water quality appears to be sediment additions and chemical transformations that occurred within the perimeter and lower main outflow ditches. It appears that the perimeter ditch had an additional impact on the concentration of P, TN, PC, and TSS. When P was transported to the perimeter ditch from the harvested field, the TDP and SRP concentrations at the harvested outflow were equal or greater than the concentrations in the harvested field. Thus, since surface and deep porewaters from the reference peatland made up a portion of the water and were low in P, processes within the mineral perimeter and main outflow ditches must have occurred that increased the concentration of P. The TDP and SRP concentrations were likely modified by the pH in the mineral ditch (pH from 5.5 – 8.0 at Sites H4 and H5; Figure 3.3.B). Orthophosphate concentrations have been found to be the most freely available at pH ranges 5.5 – 7.2, and DOC was found to maintain availability at high pH ranges (Cerozi & Fitzsimmons, 2016). Therefore, if P transport is a concern, the pH and DOC concentrations of the water in the outflow ditch network should be considered.

Higher TP, TN, PC, and TSS concentrations that coincided with increased connectivity, snow melt, and precipitation suggest that sediment was added to the ditch during initial peatland flushing or via erosion as water travelled to the outflow during snow melt and rain events. While the TSS appears to be predominantly comprised of mineral sediment from contact with the mineral perimeter ditch, particulate carbon was also high, and was likely introduced as peat from the upper ditch sides crumbled and fell into the water. These concentrations increased along the perimeter and main outflow ditches during high flow events, indicating that the entire length of the ditch acted as a source of mineral and organic particulate. This is similar to Tuukkanen et al. (2016), who found that peatland ditch cleaning increased suspended sediment concentrations and yields during high runoff events for both mineral an organic particulates. When paired with the new ditch construction that cut through both peat and mineral, and the high volume of water contacting mineral and peat within the same ditch, it is unsurprising that the estimates of daily mass discharge of TSS and PC were very high in 2019. This is similar to Stenberg et al. (2015), who found higher sediment loads in mineral compared to peat ditches, especially during the first year after ditch clearing.

The water expressed from the harvested field internal ditches in 2021 also had high TSS, with a much higher proportion of particulate carbon compared to the water found in the perimeter and main outflow ditches, as expected. This is probably due to accumulations of loose peat in the internal ditch and the lower water volumes with minimal peat contact during low flow in the perimeter and outflow ditches. Phosphorus and N associated with peat particles likely account for the elevated TP and TN concentrations that followed similar peaks as the PC concentrations (Nieminen, Palviainen, et al., 2018). However, the increase in TP could also be associated with mineral particulate within the ditch (Tuukkanen et al., 2017), particularly during higher water flow events. This said, the overall exported TP, TN, TSS, and PC leaving the internal ditches was very low in 2021, likely due to the lack of water connectivity, though ditch clearing and sequential rain events could increase their ability to export sediment (Marttila & Kløve, 2010).

Finally, the NH_4^+-N and NO_3^--N concentrations and daily mass discharge rates highlighted the importance of snow and ice dynamics within the ditches, and the role of ice as a conduit between the harvested field and the outflow. The consistently low NH_4^+-N concentrations in reference porewaters, juxtaposed with high NH_4^+-N and low NO_3^--N concentrations at the harvested outflow prior to snow melt, suggest that mineralization of organic N may have occurred within the field (Daniels et al., 2012), as well as in the ditch below the surface ice under anoxic conditions (Devito & Dillon, 1993). During the freeze-thaw just before the main snow melting event, the prevalence of NO_3^--N increased to levels higher than the snow, possibly due to the cell lysis within the peat and sediment as freeze-thaw cycles persisted (Campbell et al., 2005; Hayashi, 2013; Joseph & Henry, 2008). While the snow melt resulted in lower NH_4^+-N and NO_3^--N concentrations relative to one week prior, the concentrations leaving the internal ditch, along the perimeter ditch, and at the harvested outflow were still higher than the harvested and reference field porewaters, and the overall export rates were substantial indicating that water was moving from the harvested field downstream.

The ability for NH_4^+-N and NO_3^--N to be exported so efficiently was likely due to the prevalence of frozen ground both in the harvested field and within the internal ditches (see Chapter 2) because the concentrations found within the harvested field were similar to concentrations in the perimeter ditch and outflow. This connectivity was not observed when ice was absent; high concentrations of NH_4^+-N observed leaving the internal ditches in autumn 2021 were not found immediately downstream. As discussed in Chapter 2, the ability for the harvested field to store snowmelt waters was substantially impaired, and this study subsequently measured very high NH_4^+-N and NO_3^--N export rates during snow-on-ice melting events. This shows the importance of ice within the peatland from a runoff estimation standpoint, but also highlights the importance of accounting for water quality year-round.

3.4.3 Influence of the Beaver Dam on Outflow Water Quality

The establishment of the beaver dam had a significant impact on the water quality leaving the harvested outflow swamp with lower temperature, pH, NO_3^--N, and TSS values, and higher K, Fe, P, and NH_4^+-N values below the dam relative to above the dam. Shading from the remaining tree canopy cover combined with increased residence time in the beaver pond was probably responsible for the lower temperatures observed in the water below the dam. The remaining alterations appear to be due to redox transformations and mobilization when the surrounding organic-rich forest was flooded, or from the settling effect that the pond provided. The decrease in NO_3^--N concentrations and export rates during the spring were likely due to denitrification that occurred in the flooded beaver pond under anoxic conditions, which also explains the large increase in NH_4^+-N concentrations and exports (Bason et al., 2017; Devito & Dillon, 1993). Phosphorus increases were possibly due to additional P mobilization when the beaver pond flooded the neighboring forest, as well as further mobilization of P from particles within the existing water in the pond and anoxic soil below (Kieckbusch & Schrautzer, 2007). This is supported by the high concentrations of Fe that followed similar trends to the P concentrations; flooding has been shown to mobilize P bound with Fe (Tiemeyer et al., 2005). However, the P release following flooding may be short-lived (Aldous et al., 2007), and further observations at older ponds should be undertaken.

Likewise, the beaver pond appears to have acted as a large settling basin, as the TSS concentrations and exports were consistently lower below the dam relative to above and a thick layer of flocculant was partially suspended in the water column. Bason et al. (2017) also observed a decrease in suspended sediments downstream from beaver dams, with older ponds storing more sediment relative to younger ponds. This suggests that the filtering power of the beaver dam could increase with time, although beaver ponds are dynamic systems and seasonal flow conditions have been shown to alter TN and TP exports (Devito et al., 1989). The pH values were lower below the dam in this study, and settled near a pH of ~7. Although previous studies on beaver dams in acidic soils saw an increase in pH following flooding (Błędzki et al., 2011; Margolis et al., 2001), the decrease in pH is similar to the trend for submerged calcareous soils according to Ponnamperuma (1972). In addition, the lower pH could be a result of flooding the adjacent forest, which may have introduced additional organic material and Fe that, when submerged, have been shown to cause water to approach pH 6.5 within a few weeks (Ponnamperuma, 1972).

3.4.4 Impact of Peat Harvesting within the Larger Landscape

The TN, TDN, and PC concentrations leaving the harvested outflow swamp were often lower or equal to the water in the reference outflow swamp, but the export rates had trends that were very similar, suggesting that there was very little impact to these indicators of water quality even when the increase in water volume was considered during opening and early extraction. A study by Pschenyckyj et al. (2023) on degraded peatland systems in Ireland found opposite TDN trends; TDN concentrations were higher in degraded compared to natural peatlands. The lack of increase in this study could be due to the young age of the extracted field. The concentrations of TDN may increase with time as extraction continues, as seen by the increased concentrations in the internal ditches in 2021 that were not hydrologically connected to the outflow.

Although the beaver dam helped to lower the water temperatures at the harvested outflow, the temperature was still higher than what was observed in the nearby reference outflow swamp. Higher stream temperatures have been found to negatively impact fish (Kurylyk et al., 2013); however, at this location, the harvested outflow drains into another large peatland complex that does not have a defined stream and has several water sources. Therefore, it is unlikely to have a lasting impact the downstream ecosystem.

The higher pH and EC measurements at the harvested outflow could shift the vegetation communities downstream from oligotrophic to minerotrophic. However, peatlands commonly have adjacent swamp wetland ecosystems that are characterized by minerotrophic waters (Alberta Environment and Sustainable Resource Development, 2015). The receiving waters from the central plateau peatlands in this study were dominated by swamp wetlands prior to opening for peat harvest. This is likely reflective of the calcareous soils that the peatland domes drained toward, and explains the little difference in water quality between the adjacent reference outflow swamp and the harvested swamp receiving water from peat harvest operations. Further study at local and regional scales is needed to assess any impact to the community structure.

When comparing the harvested outflow to the adjacent reference outflow, it was difficult to disentangle the effect of vegetation removal, ditching, and extraction from the interannual weather variability and the influence of beaver. While the hydrological connectivity between the harvested field and outflow appears to be responsible for the greater P concentrations and relative export rate magnitudes compared to the reference outflow in 2019, the net increased P concentration and export at the harvested outflow swamp in 2021 seams to be from the expansion of the beaver dam. Likewise, the NH_4^+-N concentrations at the harvested outflow also increased after the beaver dam was established, while TSS and NO_3^--N concentrations were tempered.

It is difficult to ascertain if changes to water quality as a result of beaver activities can be considered an impact of peat harvesting. The beaver dam was established as a direct consequence of the increased flow following peatland drainage; therefore, the increased P and NH_4^+-N concentrations from the dam could be considered downstream pollutants. However, several studies conducted on boreal peatlands have observed variable nutrient concentrations between wet and dry periods (Burd et al., 2018; Howson et al., 2021; Muller & Tankéré-Muller, 2012; Vitt et al., 1995), so it is not unreasonable to expect these ecosystems to regularly experience both high and low nutrients, just not without periods of low to no flow. Dry conditions in the late summer and fall are normal in this area (Alberta Climate Information Services, n.d.) and flow can be drastically reduced or cease altogether starting in late August. The transition from an outflow that frequently dried up, to one that continuously exported water suggests that, regardless of any alterations to the water chemistry following harvesting or beaver activities, the ecosystems downstream of the harvested outflow may be receiving an influx of nutrients and sediments that surpass what they would normally receive due to increased hydrological connectivity.

Yet, beavers are ubiquitous across the Canadian Boreal Forest, and constantly “engineer” ecosystems to suit their needs, creating wetland habitat and altering the downstream environment in the process (Brazier et al., 2021). For example, this study area is situated within a vast peatland complex, with several small streams, marshes, and shallow open water bodies containing beaver dams that can been seen by satellite imagery. This is also compounded by the tendency for peatlands to be part of a larger wetland complex, often encompassing a range of wetland types (bogs, fens, swamps, marshes, and shallow open-water wetlands) adapted to a range of water quality and quantity (Alberta

Environment and Sustainable Resource Development, 2015). Thus, it is not unreasonable to conclude that impacts from increased flow and beaver activities may modify the immediate downstream ecosystem, but not outside of the existing range of natural variability found in the local or regional area. This raises important questions surrounding what should be considered an acceptable range of "normal" nutrient concentrations and export rates from natural, undisturbed peatlands, and makes drawing conclusions about the impact of peatland disturbance difficult. It also highlights the importance of knowing the weather conditions, water table depths, and flow connectivity from several previous seasons before determining whether increases in nutrients are due to peatland disturbance or, simply, from increased or decreased hydrologic flow connectivity following short or long-term weather patterns.

3.5 CONCLUSIONS

Water quality and quality at the outflow swamp downstream of a recently opened and extracted peat field was assessed to determine the impact of peat harvesting activities on downstream ecosystems, and the relative role of changes to nutrient concentrations within the harvested peat field, channel networks draining the peat, and stream processes within the ditches on the outflow water quality. Physical and chemical tracers presented in this study indicated that the water from the harvested field was observed at the harvested outflow, but surface and deep peat porewaters from the adjacent undisturbed peatland were also present at the outflow in different proportions depending on hydrological connectivity. Flashy responses to melting snow or rain events juxtaposed with frequently dry internal ditches within the harvested field suggested that the continuous supply of water sampled from the perimeter ditch, main outflow ditch, and the receiving harvested outflow swamp was sourced from the adjacent undisturbed peatland in addition to the harvested field in variable quantities depending on the season and year. Thus, the concentration of a given chemical parameter observed in the internal ditch within the harvested peat field was often a poor indicator of outflow concentration and relative mass export. Installing the perimeter ditch prolonged the flow (and subsequent nutrient export) at the harvested outflow year-round, compared to the reference outflow swamp where flow ceased or reduced considerably during the non-summer periods. Further, seasonal variability in weather and ice patterns had a large impact on the hydrological connectivity between the harvested field and the harvested outflow, which was exemplified by the high exports of NH_4^+-N and NO_3^--N observed when the ground was frozen.

It also appears that the surface and deep peat waters were chemically transformed when in contact with mineral ditch substrate. This substantially modified the pH, EC, and P concentrations. Further, ditches acted as a source of organic and mineral particulate; concentrations and exports increased along the perimeter ditch towards the outflow, and were highest following snow melt and large rain events. The beaver dam further modified the water quality downstream of the harvesting operations by reducing sediment and NO_3^--N concentrations and exports, and increasing NH_4^+-N and dissolved P concentrations and exports relative to the adjacent reference outflow swamp. The increased P and NH_4^+-N, coupled with the prolonged flow doubtlessly impact the downstream wetland ecosystem in some capacity, although it is unlikely that the transformation will be outside of the existing range of natural variability found in the area.

This study shows that water quality at an outflow swamp downstream of peat harvesting activities was influenced by changes that occurred within the harvested peat field. However, the final harvested outflow water chemistry and export was driven by a combination of the hydrological connectivity between the harvested field and the outflow, the chemical transformations that occurred within the perimeter and main ditches, and the presence of beaver activity brought on by increased flow following ditch installation.

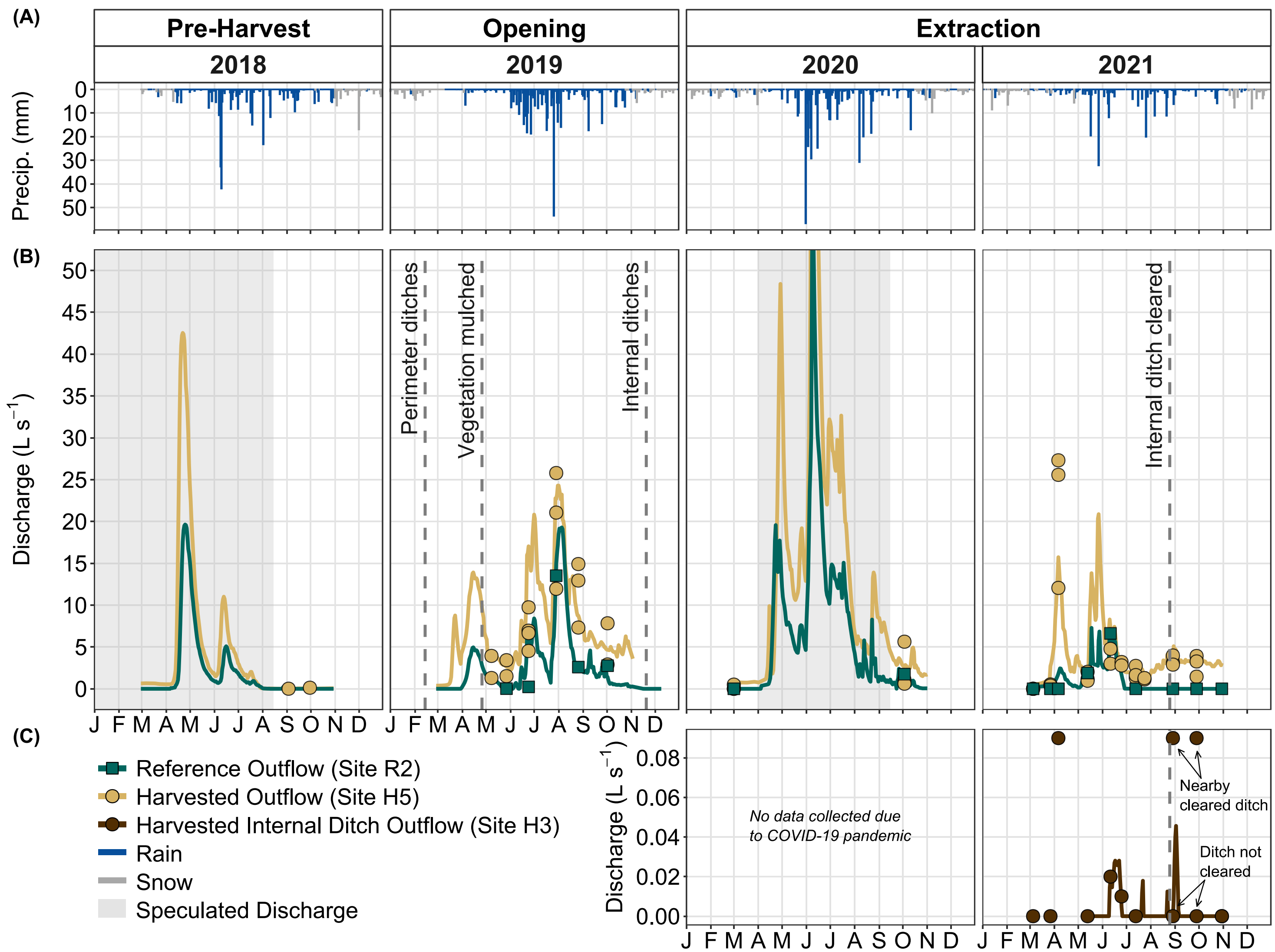

Figure 3.1 (A) Mean daily precipitation, (B) estimated water discharge for reference outflow swamp (Site R2) and harvested outflow swamp (using Site H5)), and (C) discharge for one harvested field internal ditch outflow (Site H3). Points are instantaneous field measurements of discharge; lines are estimates of continuous flow, letters at bottom of graph represent months, grey boxes show estimated discharge.

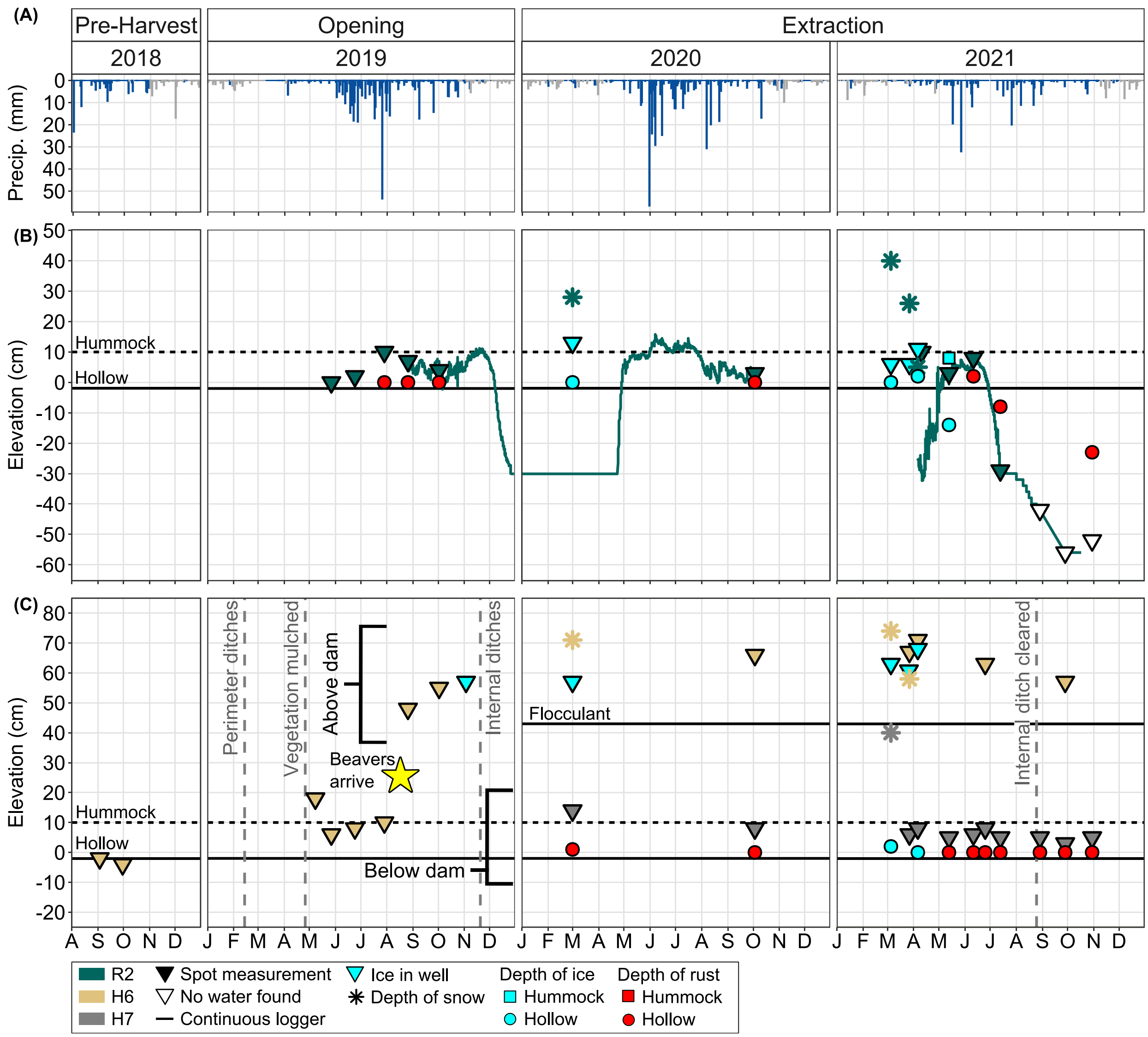

Figure 3.2 (A) Mean daily precipitation, (B) the hydro-physical condition at the reference outflow swamp (Site R2), compared to (C) the harvest outflow swamp. Two separate locations are shown at the harvested outflow swamp: Site H6 represents conditions in the outflow swamp prior to and after the perimeter and main outflow ditches were constructed in early 2019. Site H6 was above the beaver dam and became flooded following damming in late Aug 2019. Site H7 was established in 2020 and represents the conditions below multiple beaver dams. See Figure 1.3 for site locations. Shown are instantaneous spot measurements (triangles) or continuous records (logger, shown as coloured line) for water level elevations in wells or standing water, level of flocculent (sediments accumulated in beaver pond), elevation of ice surface, snow, and depth of rust relative to average surface elevation of hollows and adjacent hummock microtopography.

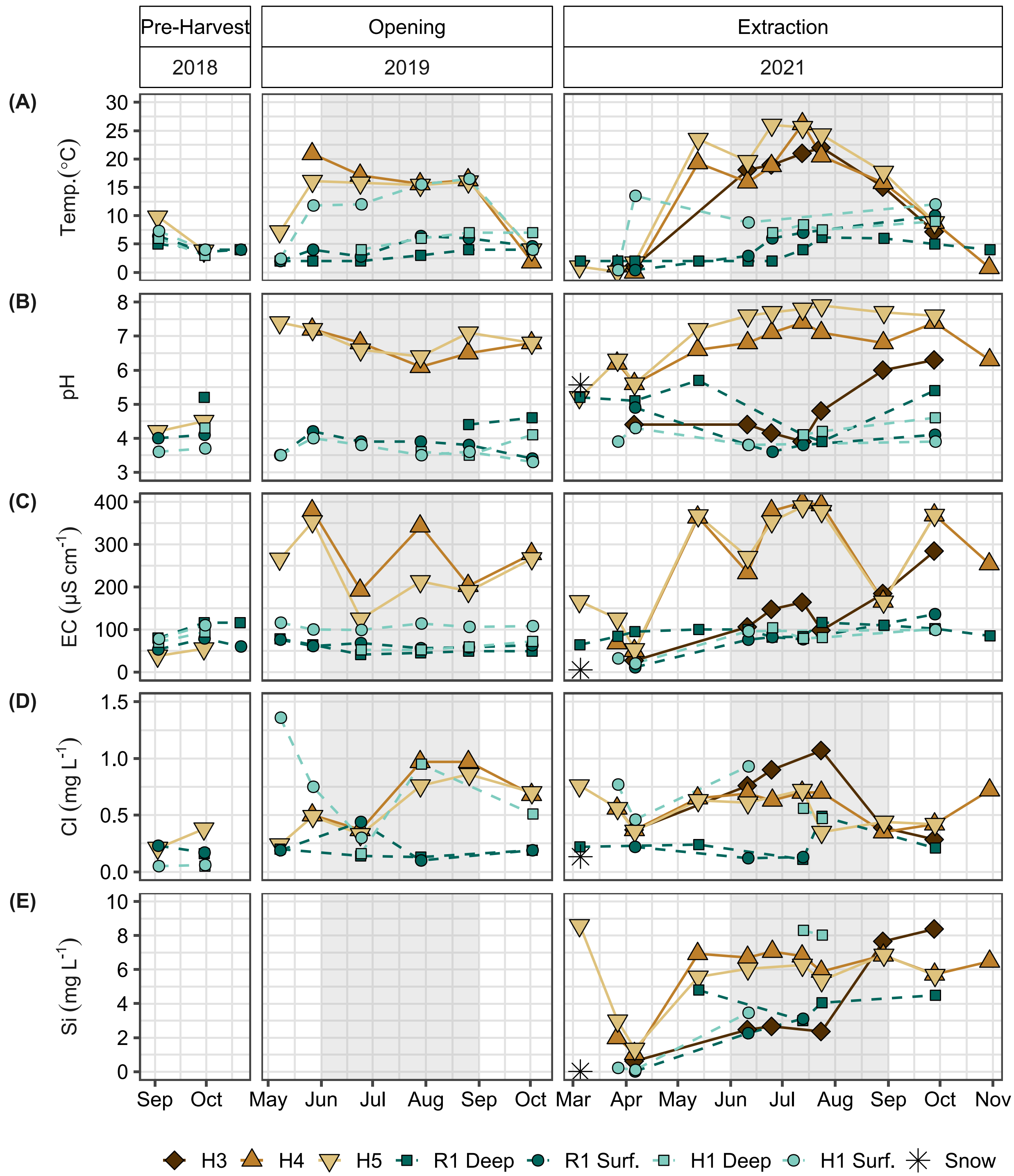

Figure 3.3 Comparison of seasonal variations in (A) water temperature (Temp.), (B) pH, (C) electrical conductivity (EC), and concentrations of (D) chloride (Cl), and (E) silica (Si) during the 2018 – 2021 study. Samples collected from flowing waters within the constructed drainage network from the harvested field internal ditch (Site H3), perimeter ditch (Site H4), and main outflow ditch (Site H5). Potential source areas (including surface (Surf.) and deep (Deep) porewaters) in the reference (Site R1) and harvested (Site H1) peatlands are shown, in addition to snow values. Grey areas represent summer months.

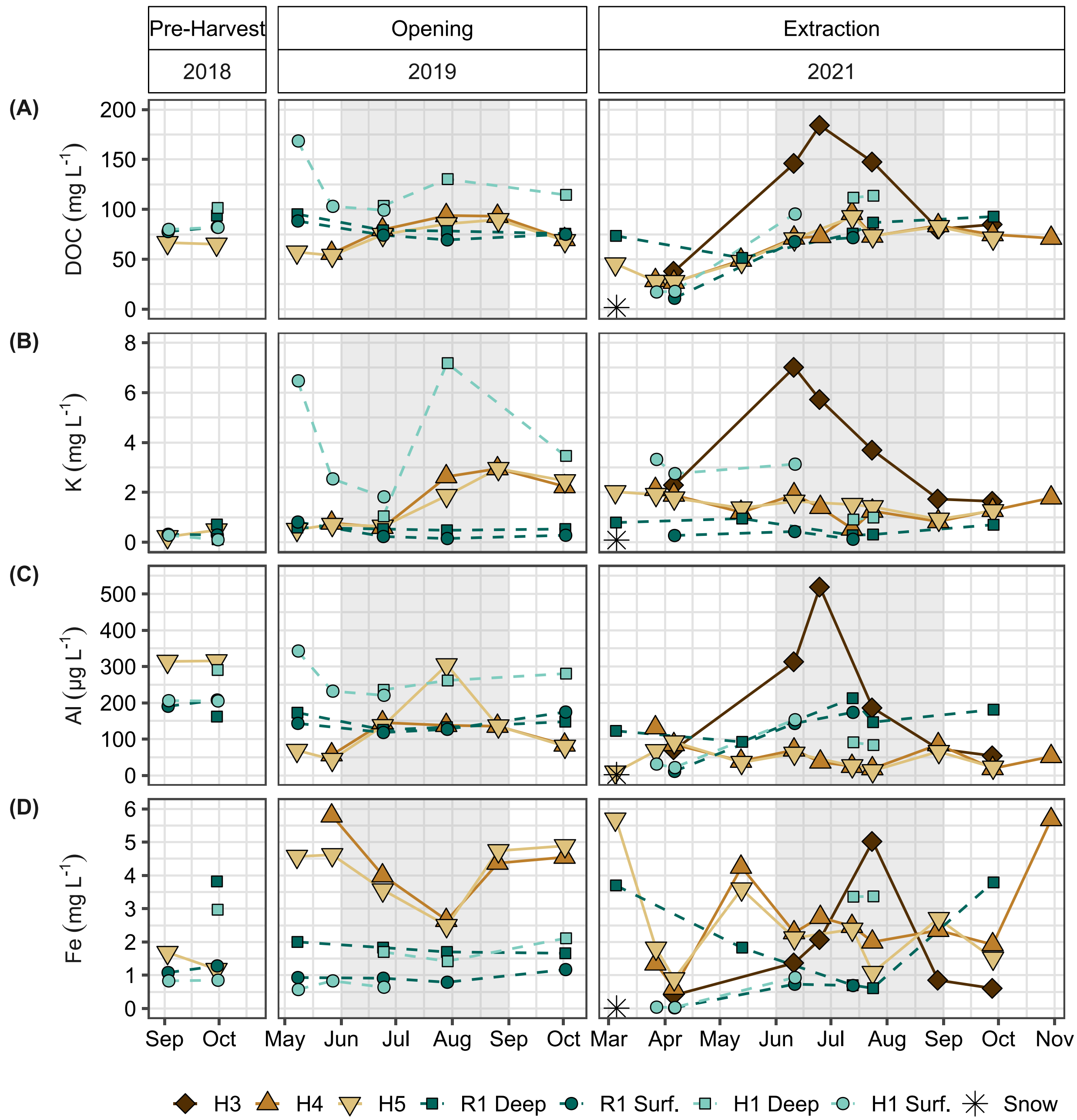

Figure 3.4 Comparison of seasonal variations in concentrations of (A) dissolved organic carbon (DOC), (B) potassium (K), (C) aluminum (Al), and (D) iron (Fe) during the 2018 – 2021 study. Samples collected from flowing waters within the constructed drainage network from the harvested field internal ditch (Site H3), perimeter ditch (Site H4), and main outflow ditch (Site H5). Potential source areas (including surface (Surf.) and deep (Deep) porewaters) in the reference (Site R1) and harvested (Site H1) peatlands are shown, in addition to snow values. Grey areas represent summer months.

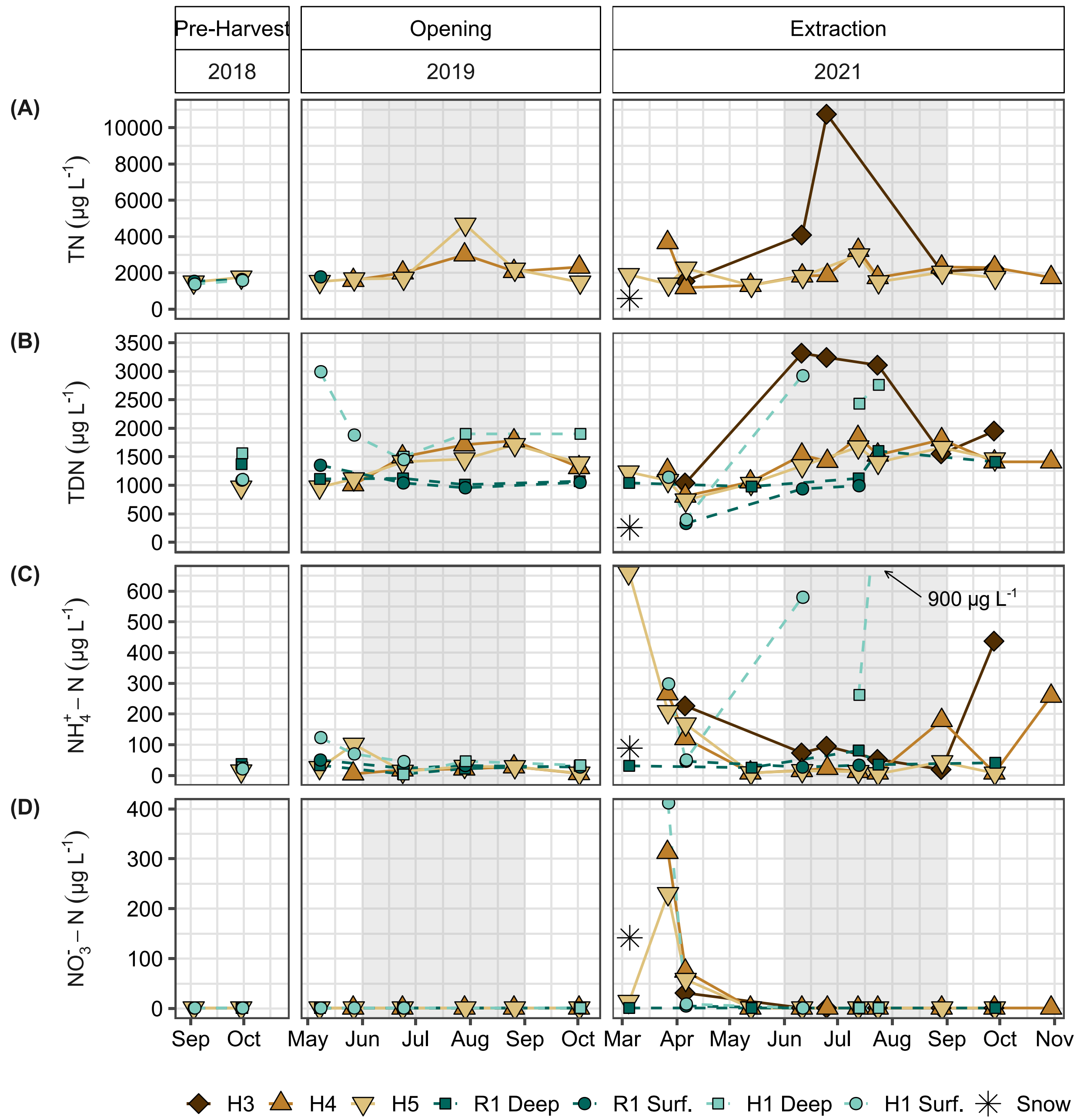

Figure 3.5 Comparison of seasonal variations in concentrations of (A) total nitrogen (TN), (B) total dissolved nitrogen (TDN), (C) ammonium as N (NH_4^+-N), and (D) nitrate + nitrite as N (NO_3^--N) during the 2018 – 2021 study. Samples collected from flowing waters within the constructed drainage network from the harvested field internal ditch (Site H3), perimeter ditch (Site H4), and main outflow ditch (Site H5). Potential source areas (including surface (Surf.) and deep (Deep) porewaters) in the reference (Site R1) and harvested (Site H1) peatlands are shown, in addition to snow values. Grey areas represent summer months.

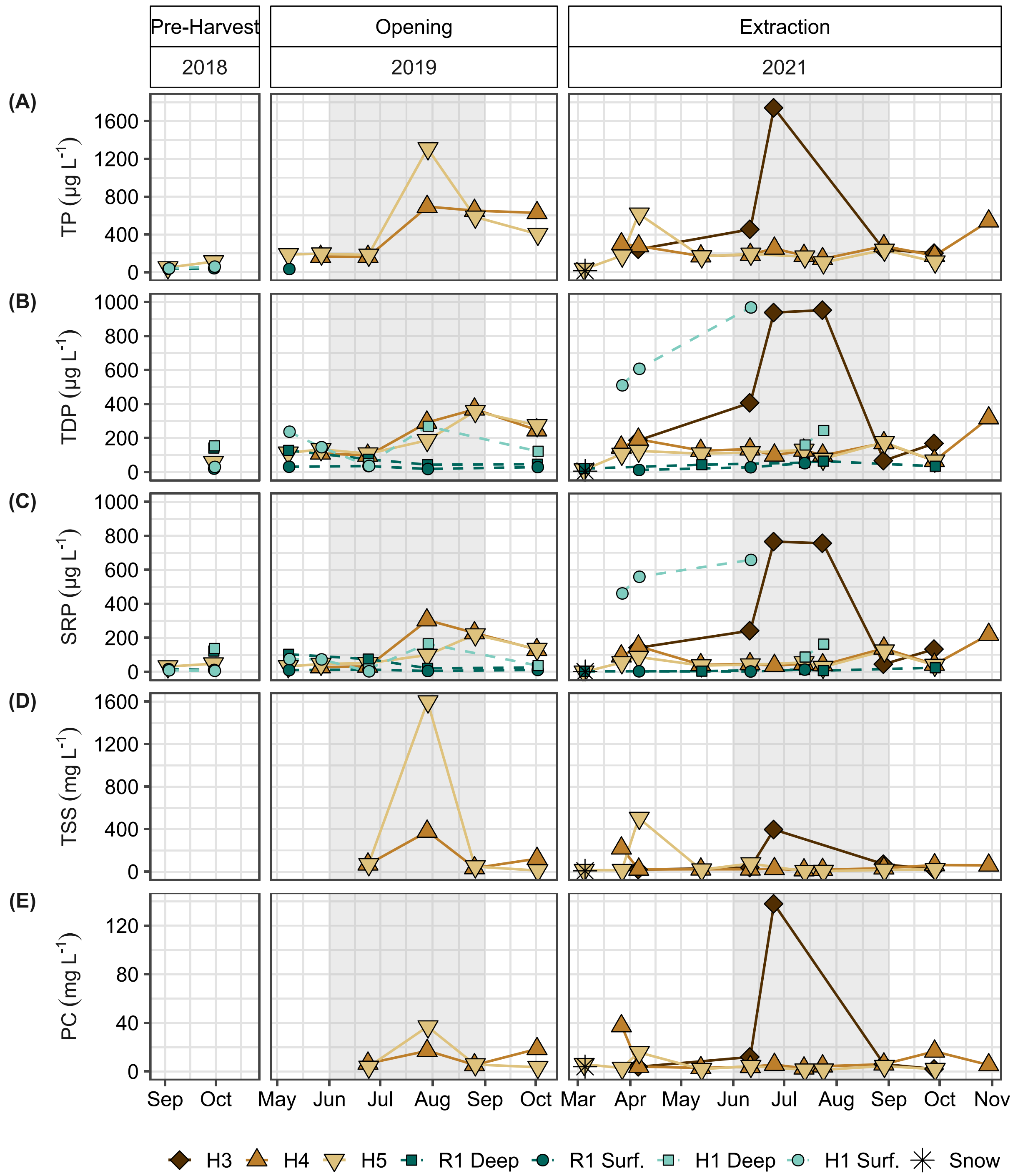

Figure 3.6 Comparison of seasonal variations in concentrations of (A) total phosphorus (TP), (B) total dissolved phosphorus (TDP), (C) soluble reactive phosphorus (SRP), (D) total suspended solid (TSS), and (E) particulate carbon (PC) during the 2018 – 2021 study. Samples collected from flowing waters within the constructed drainage network from the harvested field internal ditch (Site H3), perimeter ditch (Site H4), and main outflow ditch (Site H5). Potential source areas (including surface (Surf.) and deep (Deep) porewaters) in the reference (Site R1) and harvested (Site H1) peatlands are shown, in addition to snow values. Grey areas represent summer months.

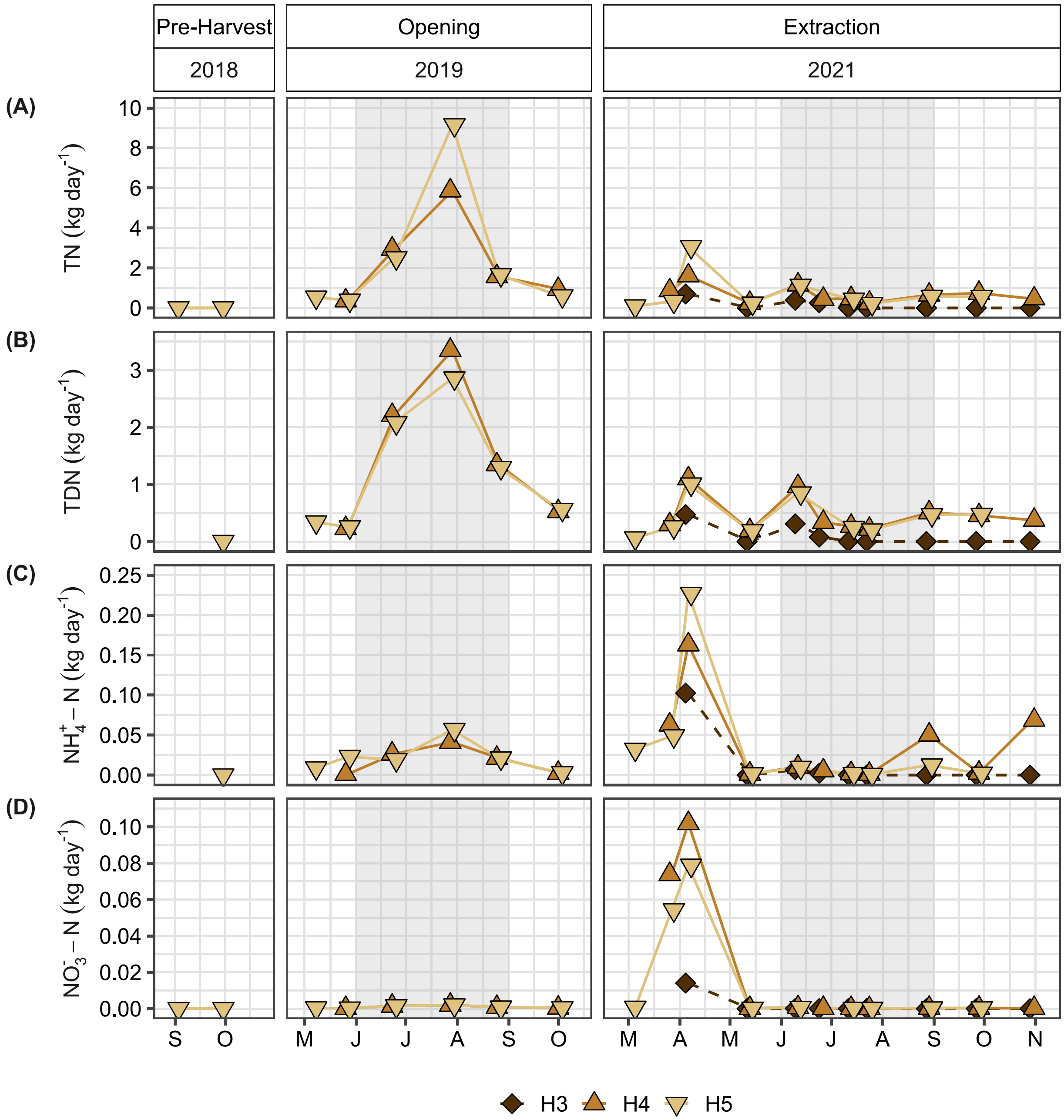

Figure 3.7 Comparison of seasonal variations in daily mass discharge (kilograms per day) within the constructed drainage network for (A) total nitrogen (TN), (B) total dissolved nitrogen (TDN), (C) ammonium as N (NH_4^+-N), and (D) nitrite + nitrate as N (NO_3^--N) during the 2018 – 2021 study. Harvested field internal ditch (Site H3), perimeter ditch (Site H4), and main outflow ditch (Site H5). Grey areas represent summer months.

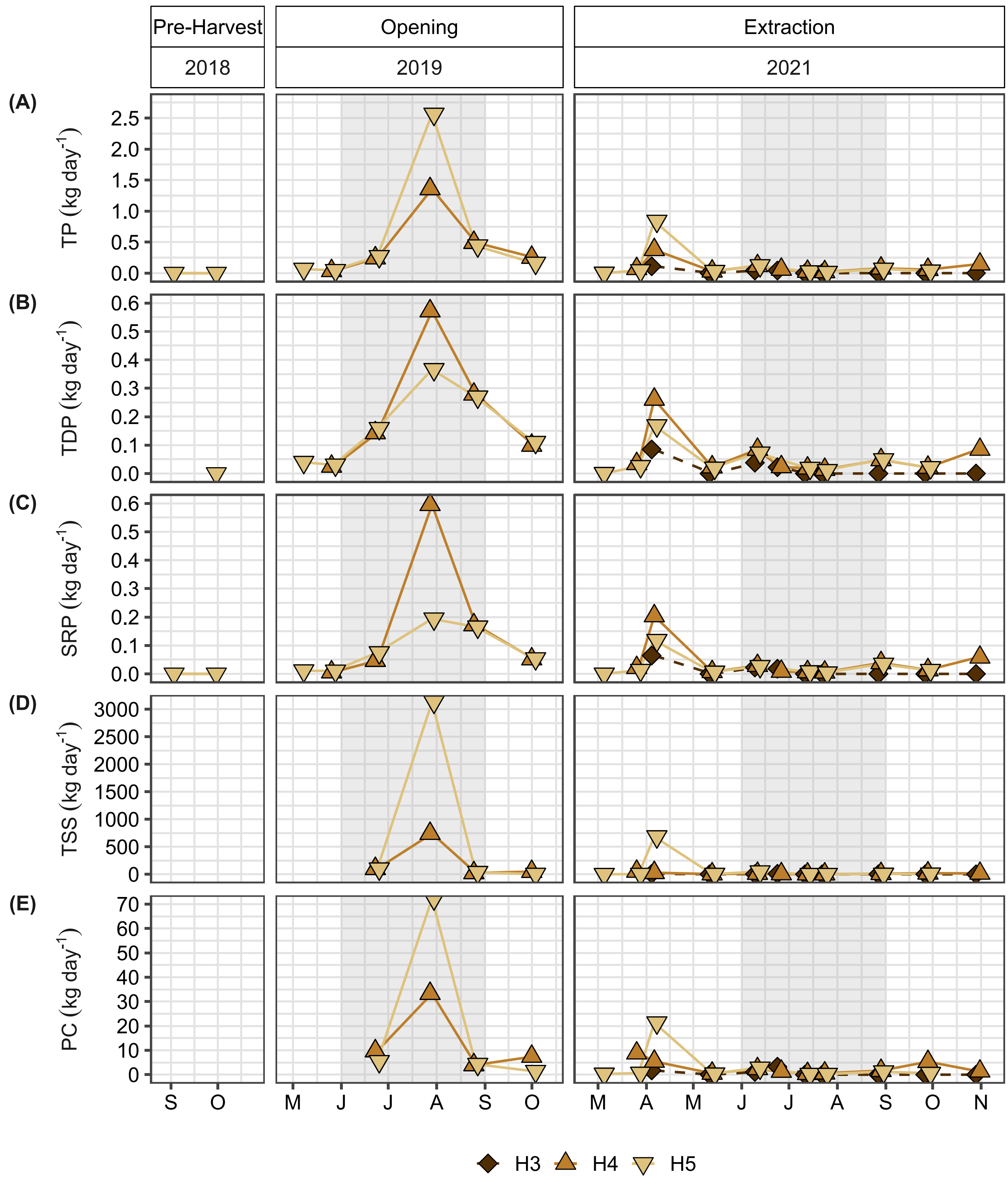

Figure 3.8 Comparison of seasonal variations in daily mass discharge (kilograms per day) within the constructed drainage network for (A) total phosphorus (TP), (B) total dissolved phosphorus (TDP), and (C) soluble reactive phosphorus (SRP), (D) total suspended solid (TSS), and (E) total particulate carbon (PC) during the 2018 – 2021 study. Harvested field internal ditch (Site H3), perimeter ditch (Site H4), and main outflow ditch (Site H5). Grey areas represent summer months.

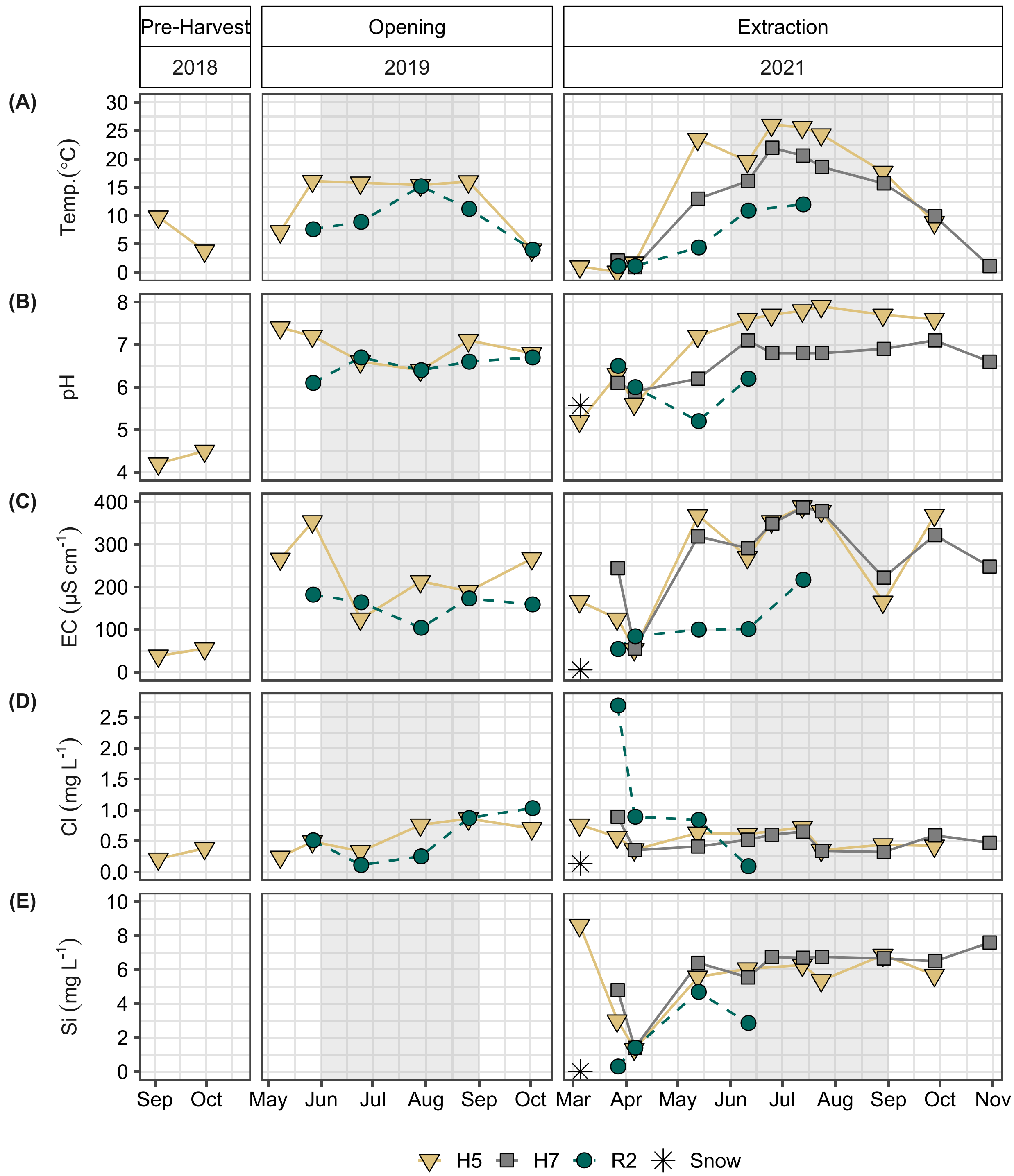

Figure 3.9 Comparison of seasonal physiochemical characteristics in flowing water at the reference outflow swamp (Site R2) and harvested outflow swamp above (Site H5) and below the beaver dam (Site H7) for (A) water temperature, (B) pH, (C) electrical conductivity (EC), and concentrations of (D) chloride (Cl) and (E) silica (Si). Site H5 in 2018 was at the same location prior to ditching the main channel to the outflow, Site H7 was a new site set below the beaver dams in 2021 following initial activity and construction in late August 2019. Grey areas represent summer months.

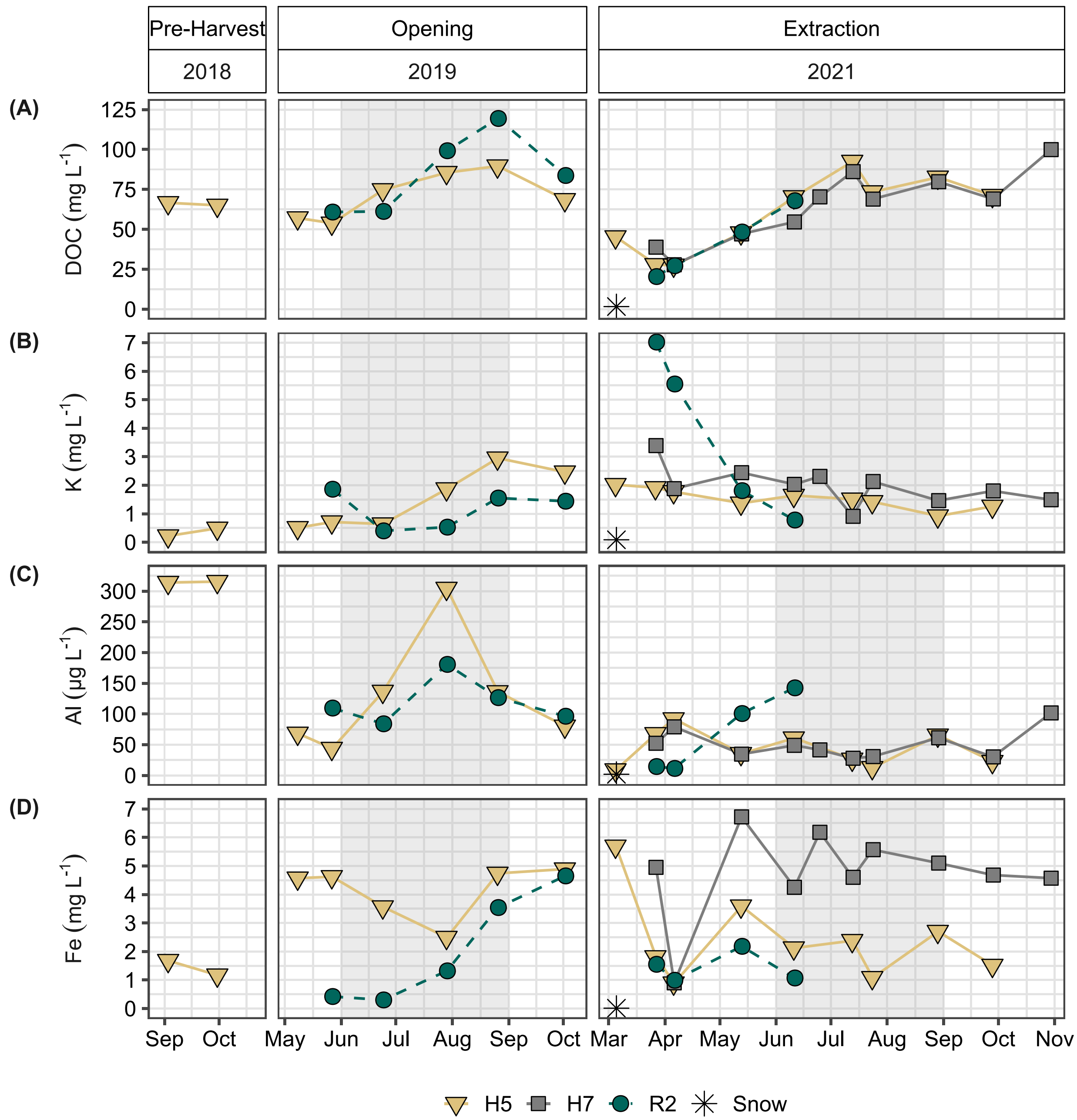

Figure 3.10 Comparison of seasonal concentrations in flowing water at the reference outflow swamp (Site R2) and harvested outflow swamp above (Site H5) and below the beaver dam (Site H7) for (A) dissolved organic carbon (DOC), (B) potassium (K), (C) aluminum (Al), and (D) iron (Fe) concentrations. Site H5 in 2018 was at the same location prior to ditching the main channel to the outflow, Site H7 was a new site set below the beaver dams in 2021 following initial activity and construction in late August 2019. Grey areas represent summer months.

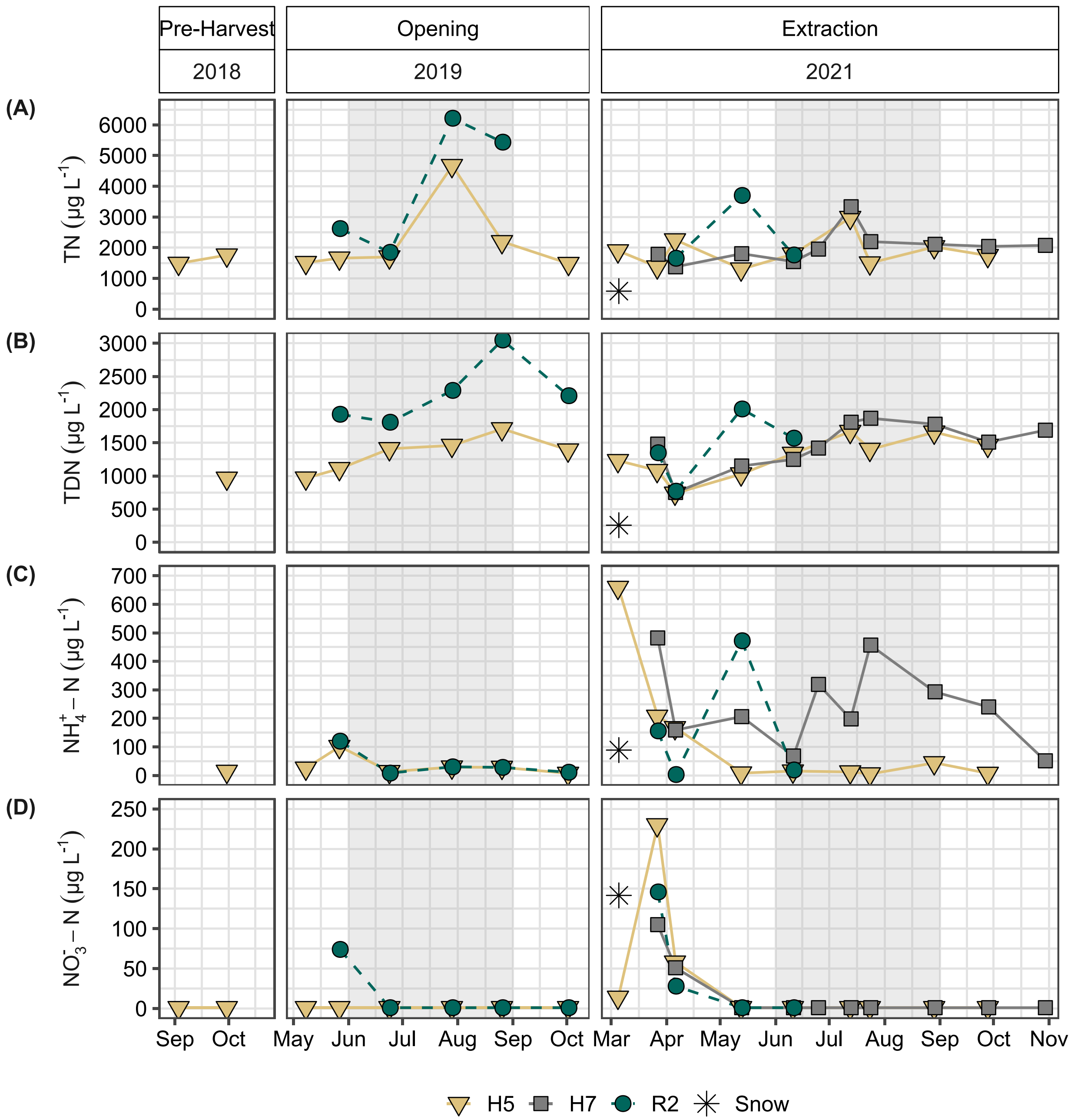

Figure 3.11 Comparison of seasonal concentrations in flowing water at the reference outflow swamp (Site R2) and harvested outflow swamp above (Site H5) and below the beaver dam (Site H7) for (A) total nitrogen (TN), (B) total dissolved nitrogen (TDN), (C) ammonium as N (NH_4^+-N), and (D) nitrite + nitrate as N (NO_3^--N) concentrations. Site H5 in 2018 was at the same location prior to ditching the main channel to the outflow, Site H7 was a new site set below the beaver dams in 2021 following initial activity and construction in late August 2019. Grey areas represent summer months.

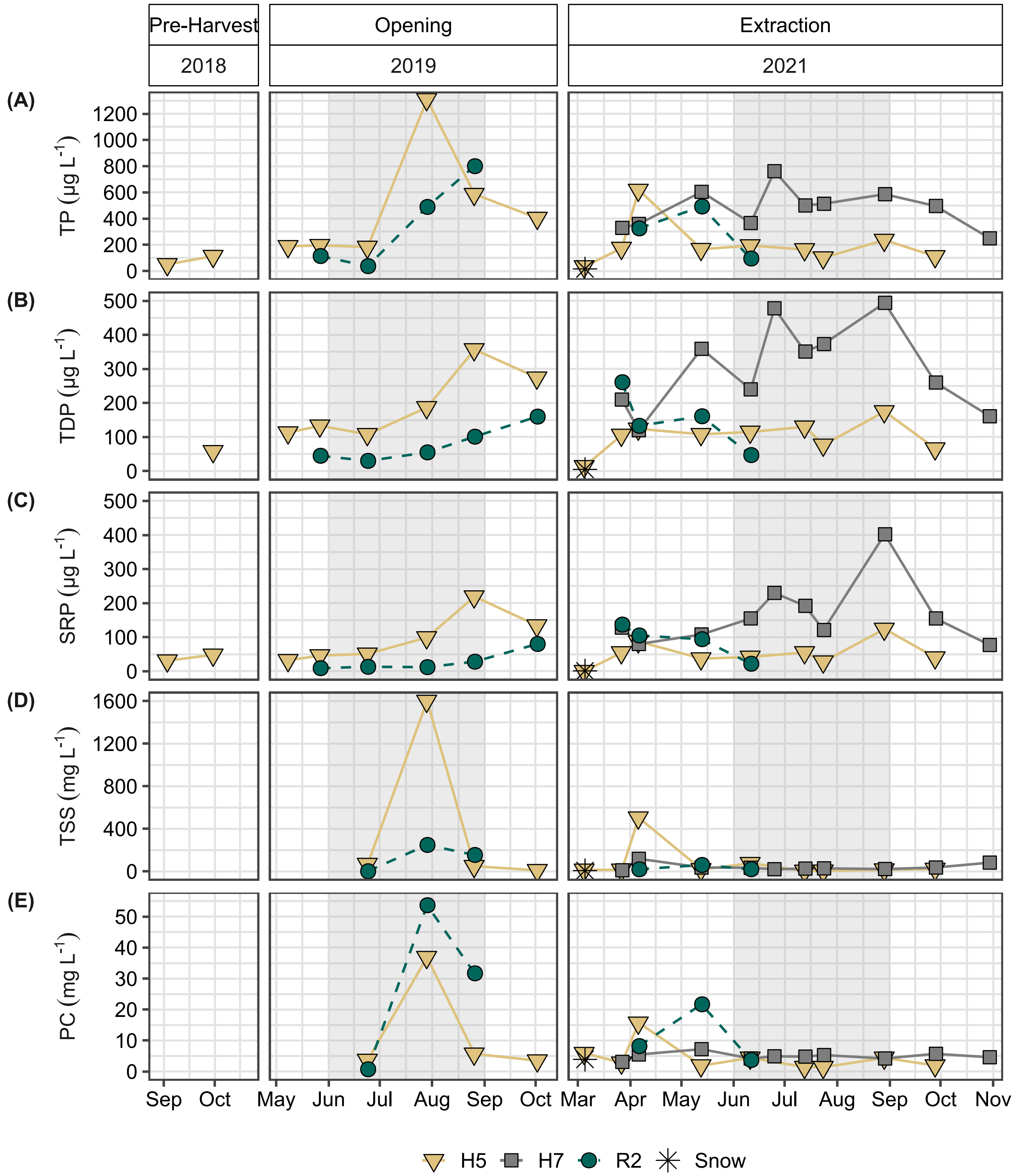

Figure 3.12 Comparison of seasonal concentrations in flowing water at the reference outflow swamp (Site R2) and harvested outflow swamp above (Site H5) and below the beaver dam (Site H7) for (A) total phosphorus (TP), (B) total dissolved phosphorus (TDP), (C) soluble reactive phosphorus (SRP), (D) total suspended solid (TSS), and (E) total particulate carbon (PC) concentrations. Site H5 in 2018 was at the same location prior to ditching the main channel to the outflow, Site H7 was a new site set below the beaver dams in 2021 following initial activity and construction in late August 2019. Grey areas are summer months.

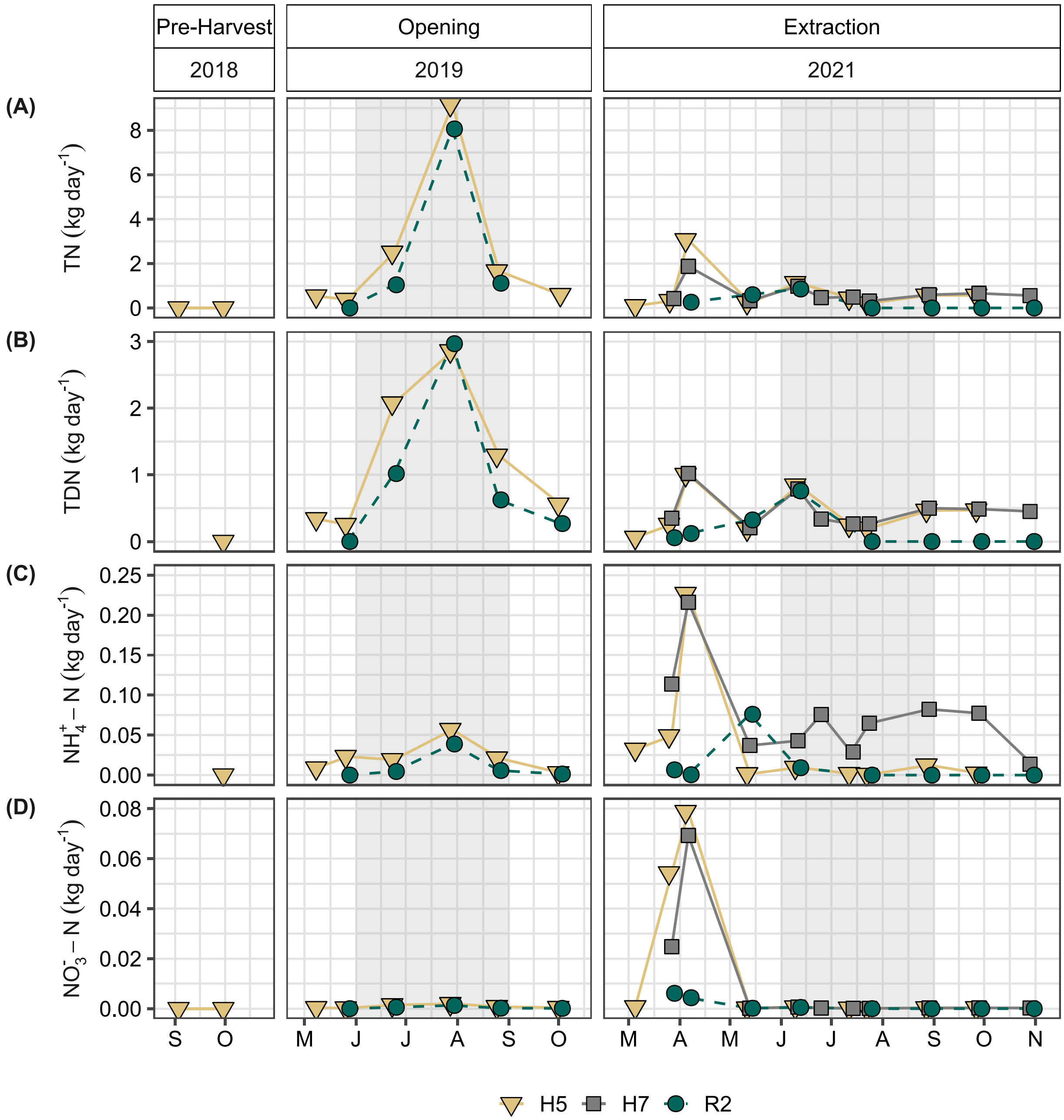

Figure 3.13 Comparison of seasonal variation in daily mass discharge (kilograms per day) at the reference outflow swamp (Site R2) and harvested outflow swamp above (Site H5) and below the beaver dam (Site H7) for (A) total nitrogen (TN), (B) total dissolved nitrogen (TDN), (C) ammonium as N (NH_4^+-N), and (D) nitrite + nitrate as N (NO_3^--N). Site H5 in 2018 was at the same location prior to ditching the main channel to the outflow, Site H7 was a new site set below the beaver dams in 2021 following initial activity and construction in late August 2019. Grey areas represent summer months.

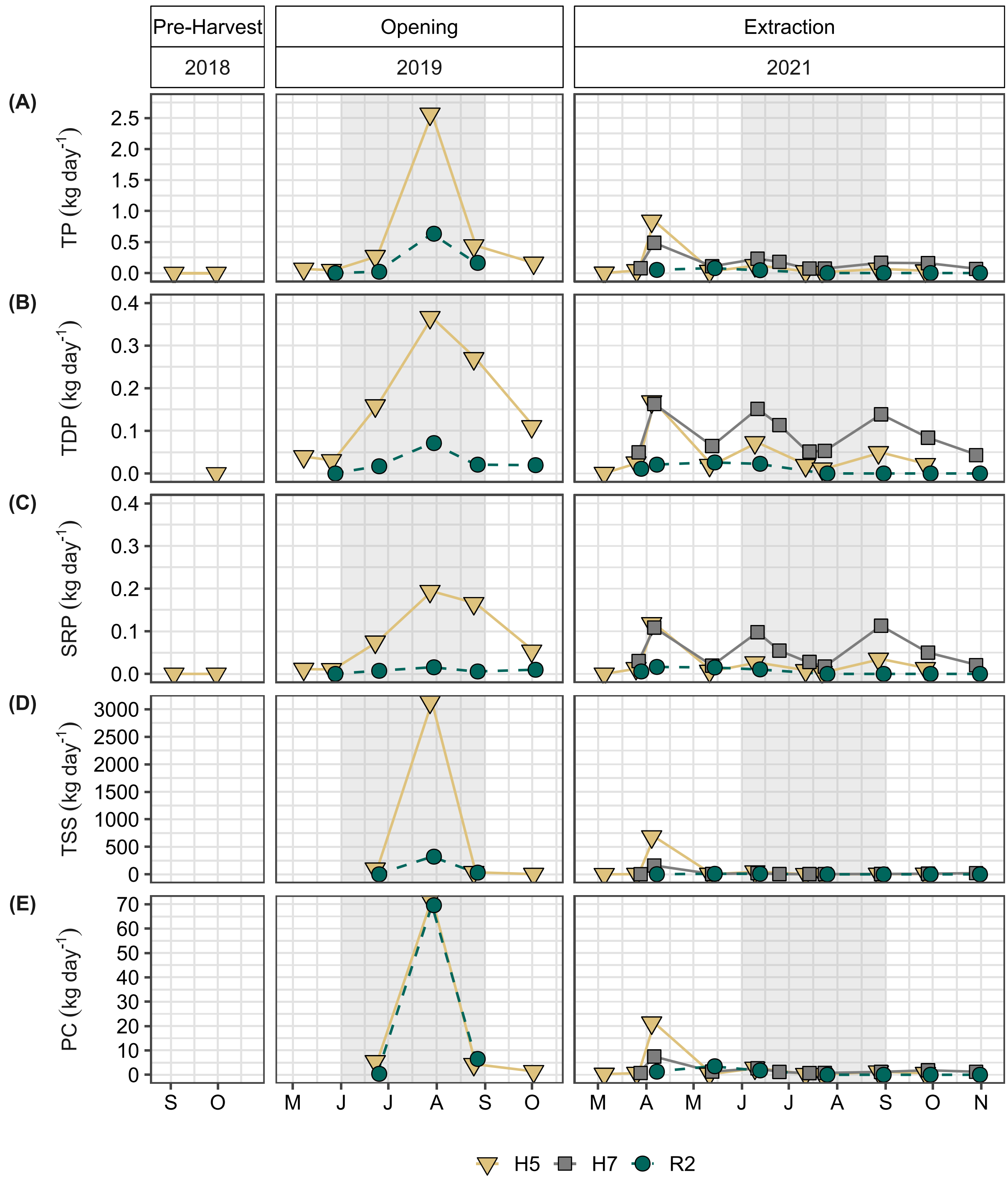

Figure 3.14 Comparison of seasonal variation in daily mass discharge (kilograms per day) at the reference outflow swamp (Site R2) and harvested outflow swamp above (Site H5) and below the beaver dam (Site H7) for (A) total phosphorus (TP), (B) total dissolved phosphorus (TDP), (C) soluble reactive phosphorus (SRP), (D) total suspended solid (TSS), and (E) total particulate carbon (PC). Site H5 in 2018 was at the same location prior to ditching the main channel to the outflow, Site H7 was a new site set below the beaver dams in 2021 following initial activity and construction in late August 2019. Grey areas represent summer months.

Chapter 4: Synthesis and Applications

4.1 RESEARCH SUMMARY

This is some of the first research assessing the water quality associated with newly opened and extracted peatlands in continental Canada. It is also one of very few studies in the world that begins to addresses the impact of perimeter ditching, vegetation mulching, internal ditching, and early peat extraction on alterations to the internal processes within the peatland that may govern nutrient availability, and changes to hydrological flow paths that potentially influence the transport of nutrients to downstream ecosystems. This study is placed within the biogeoclimatic context of the continental Boreal Plain in Alberta, Canada, with timing and volume of precipitation, geology, atmospheric inputs, and peatland forming processes that have an impact on the water quality and quantity. The importance of understanding the ditch substrate material and its impact on transforming water chemistry is additionally explored. Further, this study acknowledges the that beavers are ubiquitous across the Boreal Forest in Canada, and addresses the influence of these ecosystem engineers on the water quality exported from the peat harvesting operation. Finally, this study touches on the importance of comparing water quality at the final harvesting outflow location with similar wetland ecosystems within the local area in order to gain an understanding of the impact of peat harvesting activities within the broader landscape.

This study found that opening peatlands and extracting peat increased nutrients in the peat porewater; increased and extended flow, resulting in higher nutrient exports; and increased the subsurface hydrological connectivity by installing ditches. However, the final outflow water chemistry and exports appear to be driven by the hydrological connectivity between the harvested field and the harvested outflow, which varied at both seasonal and interannual scales; the internal, perimeter, and main channel ditch substrate, resulting in nutrient form transformations and altered concentrations; and the presence of beaver activities, such as dam construction, that trapped sediment and mobilized reduced nutrient forms.

4.1.1 Impact of Opening and Early Extraction on In-field Processes

The perimeter ditch did not appear to lower the overall harvested field water levels, resulting in no discernable impact to the hydrology within the harvested field. Mulching the vegetation cut transpiration and increased the water levels within the harvested field, which enabled anoxic conditions to persist. Mulching also increased surface temperatures and initial concentrations of total dissolved phosphorus (TDP), soluble reactive phosphorus (SRP), total dissolved nitrogen (TDN), dissolved organic carbon (DOC), and potassium (K) in porewaters within the harvested peat field relative to the reference peat field. Despite the influx of nutrients from the mulched vegetation, the newly opened harvested field had pulses of nitrogen (N) and phosphorus (P) that were not sustained at an elevated concentration, likely due to immobilization by microbes. Similar N and P concentrations between the harvested and reference peat fields suggests that decomposition within the harvested field was minimally affected by opening activities, at least in the short-term. Despite the higher water level, it does not appear that there was any change in water source within the harvested field. Therefore, it appears that the initial impact of opening the peatland on the in-field hydrology and physicochemical processes, at least on a short, one-year scale, was minimal.

Following the installation of internal ditches, water levels within the harvested field were lower than the reference, but did not reflect the extent of soil moisture and depth of aeration. Aerobic conditions, moist peat, and elevated surface and porewater temperatures likely encouraged decomposition in the harvested field, resulting in higher nutrient concentrations. However, the internal ditches did not consistently mirror the nutrient concentrations observed in the harvested field, and were influenced by precipitation, surface, and deep peat porewaters in variable proportions throughout the year. The nutrient concentrations within the internal ditches were likely governed by increased decomposition enabled by the low carbon to nitrogen ratio (C:N) peat found at the bottom of the internal ditches, and by the elevated summer temperatures. It appears that installing internal ditches and extracting peat influences the internal processes within the harvested field and internal ditches, alters the hydrological flow paths, and has the potential to transport higher concentrations of nutrients downstream.

4.1.2 Impact of Opening and Early Extraction on Outflow Water Quality and Quantity

Although the harvested field was minimally impacted by perimeter ditching, installing the perimeter ditch prolonged the flow at the harvested outflow swamp into autumn, while the reference outflow swamp remained dry, suggesting that the mass of nutrients discharged into downstream ecosystems increased. Alterations to the snow and ice dynamics within the harvested field and internal ditches enabled nutrient-rich melt waters to be more readily transported off site, and seasonal and interannual variability in weather and ice patterns impacted the hydrological connectivity between the harvested field and the harvested outflow. This means that although high stores of nutrients were available in the harvested field surface peat, the risk of exporting nutrients from the harvested field downstream was not uniform over the year and between years. Thus, the concentration of a given chemical parameter observed in the harvested field internal ditch was often a poor indicator of relative mass discharge.

Whilst water from the harvested field was observed in the perimeter ditch and main ditch entering the harvested outflow swamp, the chemical composition of the water indicated that water was also sourced from surface and deep peat porewaters from the adjacent undisturbed reference peatland. Additionally, water was chemically transformed when in contact with mineral ditch substrate, with substantial modifications to pH, electrical conductivity (EC), and P concentrations observed. Ditches additionally acted as a source of sediments; the highest discharge rates were observed following snow melt and large rain events.

4.1.3 Impact of Opening and Early Extraction within the Larger Landscape

The water entering the outflow swamp from the main outflow ditch had higher P and total suspended solids (TSS) concentrations and mass discharge rates compared to the reference outflow swamp, but N and DOC concentrations and mass discharge rates were comparable. However, once the beaver dam was present, P and NH_4^+-N concentrations and mass discharge rates increased, while TSS decreased. The higher pH, EC, and P concentrations entering the harvested outflow swamp were not indicative of bog to poor-fen ecosystems, and indicates that the water quality was modified by peat harvesting activities. However, although impacts to downstream ecosystems are likely, the range of water quality was not outside of the natural variability found in other outflow swamps in the area.

4.2 RESEARCH APPLICATIONS

Using the findings from this research, several applications can be applied at newly opened and extracted peat harvesting operations. Firstly, the time of year when water quality samples are taken is important for determining the overall impact to downstream ecosystems. Sampling should be conducted at different times during the year to understand the range in hydrological connectivity because it appears to be very important for understanding nutrient exports.

The quality of the peat substrate and the range of conditions conducive to decomposition within the harvested field and the internal drainage ditches must be considered. Although perimeter ditching and vegetation mulching did not result in sustained nutrient availability during the first year when the peatland was opened, nutrients were clearly available once the internal ditches were installed and extraction had commenced. However, the transportation of nutrients available within the harvested field and internal ditches downstream appears to be governed by hydrological connectivity.

Ice and snow dynamics played a large roll in nutrient transport from the harvested field to the outflow, and the timing, frequency, and volume of precipitation further control the hydrological connectivity. Thus, assessments of water saturation within the harvested field, and water flow leaving the internal ditches, within the perimeter ditch, and at the main harvested outflow are needed in order to understand the extent of connectivity during a sampling period. This may also be affected by ditch maintenance, which can re-connect the internal ditches with the perimeter and main outflow ditches.

The substrate composition of the ditches, whether mineral or peat, has a substantial influence on the final water quality leaving the harvesting area, and must be considered. In addition, the continuous flow of water leaving the harvested peatland will inevitably attract beavers, who will further manipulate the outflow and alter the water quality. The influence of beaver must be accounted for because this study suggests that additional nutrient concentrations and exports are associated with beaver dam construction.

Finally, determining where to sample and what constitutes a suitable outflow location for determining the impact to downstream ecosystems is not a simple task. There are several locations within a harvested peat field operational area that can be sampled, all of which could show drastic differences in water quality. Samples collected in the harvested field, from the internal drainage ditch, in the perimeter ditch, at the terminus of the main ditch, or downstream of a beaver dam can indicate both major and minor impacts to water quality following the peatland opening and extraction of peat. As such, the relative risk to downstream ecosystems will depend on the nature of the downstream ecosystem, which should be considered when assessing whether peat harvesting has had a negative impact on water quality. Many peatlands naturally discharge into swamps and moderate to rich fens, both of which have very different water quality parameters relative to the bogs and poor-fens selected for peat harvesting. Thus, a clear idea of what counts as the outflow, and what ecosystem is directly affected by peat harvesting activities will aid in determining the relative downstream risk to water quality.

4.3 FUTURE RESEARCH AND STUDY LIMITATIONS

While this study has made headway in determining the impact on water quality downstream of an opened and newly extracted harvested peatland, more information is needed to continue to understand the parameters governing nutrient exports to downstream ecosystems. Extensive assessments of the peat profile chemistry within the harvested field are needed to further understand the substrate quality and chemical composition of the peat. This may aid in finding a signature indicative of surface and deep porewaters, and could help explain the higher nutrient concentrations found in the internal ditches intersecting deep peat. Likewise, direct measures of decomposition and assessments of biomass and speciation of microbial populations are needed in opened and newly extracted peatlands to determine the speed of decomposition in the early years of peat harvesting.

Continued monitoring during all times of year, including winter and early spring, is needed to further understand seasonal patterns of hydrological connectivity. Additionally, detailed assessments during major rain events will help clarify the findings in this study, and provide better estimates of the of mass discharge and water discharge leaving new harvesting locations during peak flows. Further studies at opened and newly extracted peatlands need to be conducted under different weather patterns to see the observations observed herein vary in wet, normal, and dry years, and how the weather from the previous year influences the nutrient availability and leaching risk.

Further assessment of different harvesting stages is required to understand the similarities and differences between opened, extracted, and restored peatlands, and to determine if the physicochemical and hydrological processes responsible for alterations to the water quality expressed at the outflow are altered with time since extraction began, or are modified by restoration activities. This should be done in several biogeoclimatic settings across Canada in order to understand what processes governing water quality are universal, and which are dependant on climate, geology, atmospheric inputs, and peat quality.

Inventories that account for the prevalence of beavers at peat harvesting operations as well as in natural wetland complexes will help underscore their prevalence. This is needed in order to disentangle what constitutes a peat harvesting impact to water quality, and what is the effect of beaver dam construction. Finally, an assessment of the range of water quality associated with non-harvested bogs, fens, and swamps within the Boreal Plain Ecozone, as well as the prevalence of fen and swamp outflows, is needed to place the water quality leaving the harvested peatland into the local and regional context. This will help assess the potential risk to downstream ecosystems, and help direct approvals for future peat harvesting operations in the future.

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Appendix

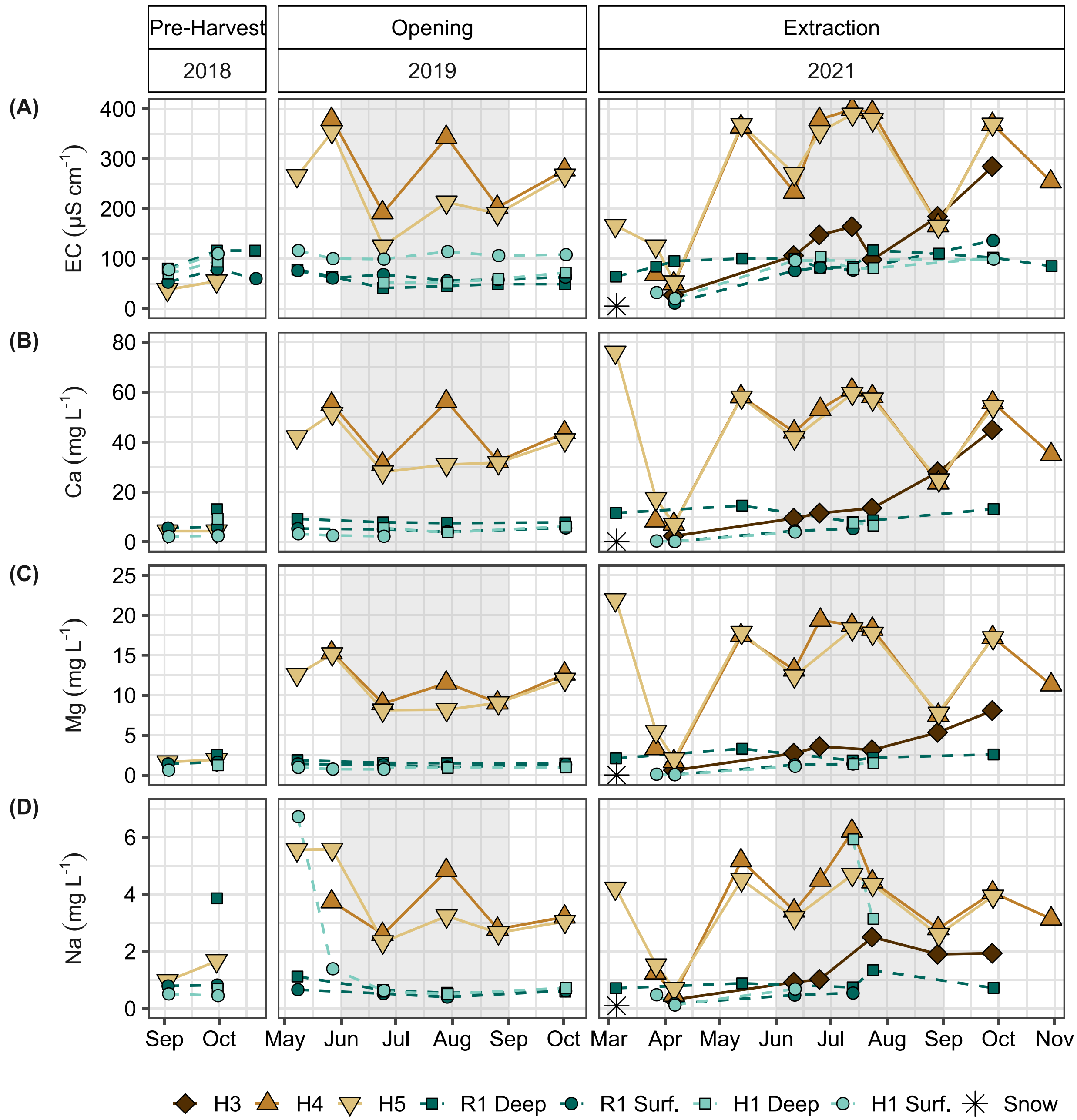

Figure A.1 Comparison of seasonal variations in concentrations of (A) electrical conductivity (EC), (B) calcium (Ca), (C) magnesium (Mg), and (D) sodium (Na) during the 2018 – 2021 study. Samples collected from flowing waters within the constructed drainage network from the harvested field internal ditch (Site H3), perimeter ditch (Site H4), and main outflow ditch (Site H5). Potential source areas (including surface (Surf.) and deep (Deep) porewaters) in the reference (Site R1) and harvested (Site H1) peatlands are shown, in addition to snow values. Grey areas represent summer months.

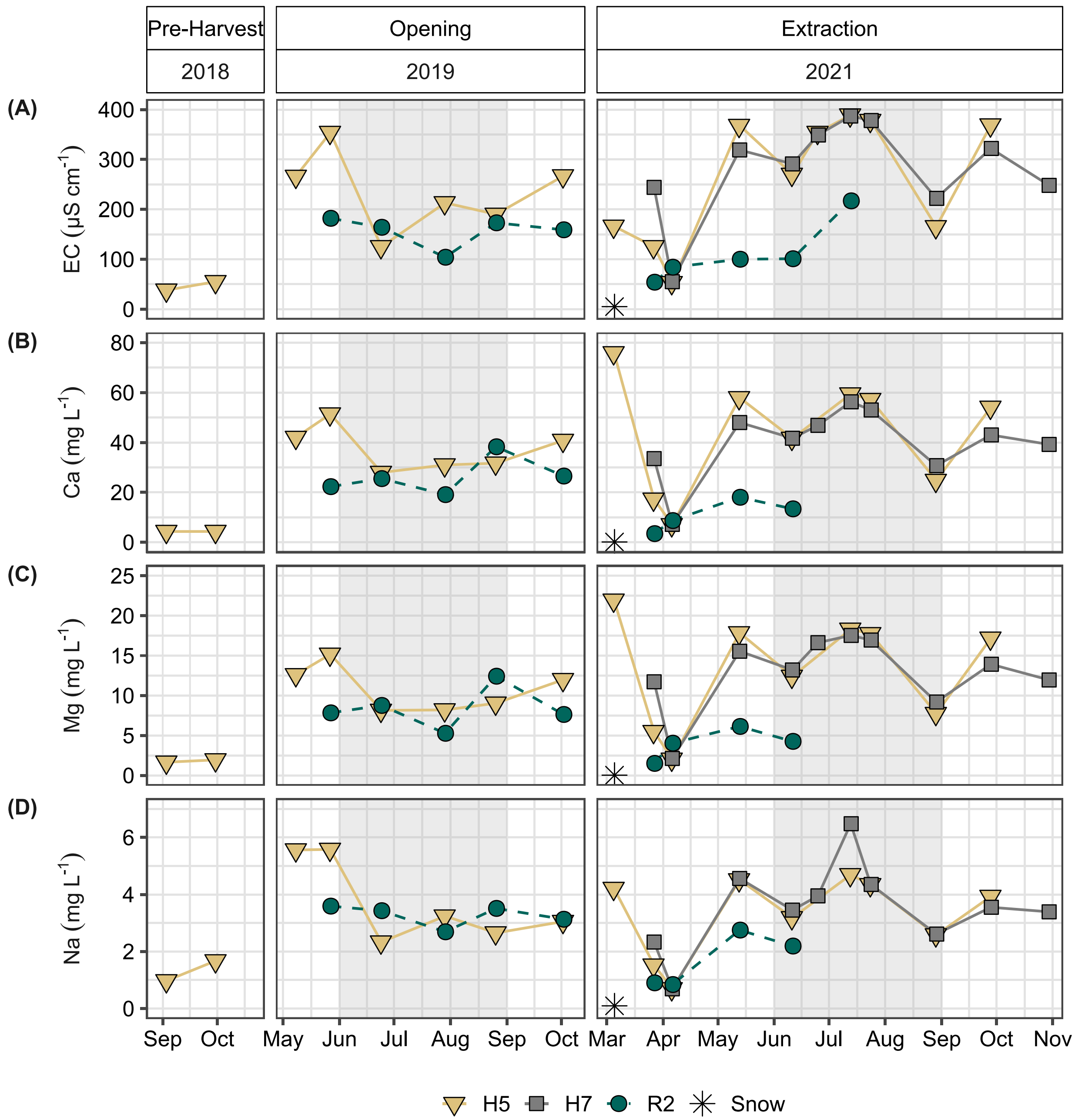

Figure A.2 Comparison of seasonal concentrations in flowing water at the reference outflow swamp (Site R2) and harvested outflow swamp above (Site H5) and below the beaver dam (Site H7) for (A) electrical conductivity (EC) values, and (B) calcium (Ca), (C) magnesium (Mg), and (D) sodium (Na) concentrations. Site H5 in 2018 was at the same location prior to ditching the main channel to the outflow, Site H7 was a new site set below the beaver dams in 2021 following initial activity and construction in late August 2019. Grey areas are summer months.