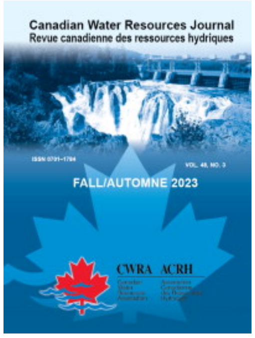

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Advances in wetland hydrology: the Canadian contribution over 75 years

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ABSTRACT

Wetlands are an integral part of the Canadian landscape, providing crucial ecohydrological services with globally significant benefits. Over the past 75 years, Canadian scientists have emerged as international leaders in wetland hydrological research, contributing to a better understanding of wetland form and function. Early Canadian research was instrumental in the development of a classification scheme that provided a foundation for later investigations into vadose zone processes, solute transport, evapotranspiration, ground-ice dynamics, biogeochemical cycling, and modelling. This work has coalesced into a better understanding of the factors that contribute to wetland presence and persistence on the landscape, and the internal processes that result in their unique functions of carbon sequestration, water storage, flood mitigation, water quality enhancement, and wildlife habitat. In Canada and across the world, wetlands are threatened at a range of scales and intensities by disturbances like climate change, resource extraction, wildfire, altered land use, and contamination. In response, Canadian researchers have become global leaders in characterizing the impacts of disturbance on wetland function and been at the forefront of innovative restoration and reclamation techniques. As the value of wetlandisre increasingly acknowledged by stakeholders and decision-makers, the need for evidence-based wetland research will only continue to grow. Canadian scientists are well-positioned to lead wetland hydrology into the next 75 years.

Les milieux humides font partie intégrante du paysage canadien, offrant des services écohydrologiques essentiels avec des avantages d'importance mondiale. Au cours des 75 dernières années, les chercheurs canadiens se sont imposés comme des leaders internationaux dans la recherche hydrologique sur les milieux humides, contribuant à une meilleure compréhension de leur forme et de leur fonction. Les premières recherches canadiennes ont joué un rôle clé dans l'élaboration d'un système de classification qui a servi de fondement à des études ultérieures sur les processus de la zone non saturée, le transport des solutés, l'évapotranspiration, la dynamique de la glace du sol, le cycle biogéochimique et la modélisation. Ces travaux ont mené à une meilleure compréhension des facteurs qui contribuent à la présence et à la durabilité des milieux humides dans le paysage, les processus internes qui donnent lieu à leurs fonctions uniques de séquestration du carbone, de stockage de l'eau, de mitigation des inondations, d'amélioration de la qualité de l'eau, et d'habitats fauniques. Au Canada et partout dans le monde, les milieux humides sont menacés à diverses échelles et intensités par des perturbations telles que le changement climatique, l'extraction de ressources, les feux de forêt, l'utilisation des terres modifiée et la contamination. En réponse, les chercheurs canadiens sont devenus des leaders mondiaux dans la caractérisation des impacts des perturbations sur la fonction des milieux humides et ont été à l'avant-garde de techniques innovantes de restauration et de remise en état. À mesure que la valeur des milieux humides est de plus en plus reconnue par les parties prenantes et les décideurs, le besoin de recherche sur les milieux humides basée sur des preuves ne fera que croître. Les chercheurs canadiens sont bien positionnés pour continuer à diriger la recherche en hydrologie des milieux humides au cours des 75 prochaines années.

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Introduction

Recent estimates suggest Canada hosts nearly 20% of the world's wetlands (Fluet-Chouinard et al. 2023). Wetlands comprise 14% of Canada's land area, with 90% of those wetlands being peatlands (NWWG 1997). Globally, peatlands store more carbon than all the

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world's forests combined, and Canada's peatlands account for the largest proportion of that carbon stock (UNEP 2022). Wetlands provide crucial ecosystem services, including flood risk mitigation (Pattison-Williams et al. 2018), supporting biodiversity (e.g. Vickruck et al. 2019), wildlife habitat (Markle et al. 2020,), and water quality improvement *via* nutrient retention (Cheng and Basu 2017). In this manuscript, we highlight Canadian research in wetland hydrology over the past 75 years with a focus on research conducted in Canada by Canadian researchers. From the mid-1980s onward, the exponential growth in the number of Canadian publications on wetland hydrology has provided detailed information on all aspects of hydrology and are too numerous for us to provide a comprehensive literature review. Rather, we cite examples of Canadian research that in the opinion of the authors pivotally illustrates the Canadian contribution to understanding the form and function of wetlands globally. While we include important works by non-Canadians collaborating with Canadians in Canada, we acknowledge there is relevant international wetland hydrology not presented in this manuscript.

The manuscript is organized into two sections: A) a review of landscape processes that give rise to the range of common wetland forms and how they reflect the hydrogeomorphic setting that controls their location, water stores and exchanges, and ultimately their ecology; and B) a review of specific hydrological themes relevant to wetlands in Canada and elsewhere, including i) near-surface water exchanges; ii) the role of ground ice; iii) contamination and solute transport; iv) hydrological aspects of biogeochemistry; v) disturbances and restoration/reclamation; vi) modelling approaches to wetland hydrology; and vii) Indigenous knowledge. Given that Canadian wetlands are predominantly peatlands, they have been the focus of most Canadian research, and this is reflected in this manuscript.

In Canada, wetlands are classified based primarily on hydrology (NWWG 1997) following the foundational work of Zoltai and Vitt (1995), who related wetland class and form to the strength and variability of water flow and its role as a vector for dissolved and particulate chemical constituents (Figure 1). According to the Canadian Wetland Classification System (NWWG 1997), Canadian wetlands include marshes, shallow water wetlands, mineral swamps, and peat-dominated wetlands including bogs, fens and (peat) swamps. The details of their form and function are explained through Canadian research, in Section A of this manuscript. While there is a dearth of published research focusing on Canadian wetland hydrology prior to 1970, Meyboom (1966, 1967) wrote seminal papers on groundwater relations in Prairie wetlands. There was also earlier interest in Canadian peatlands, with MacFarlane (1959) indicating that systematic research

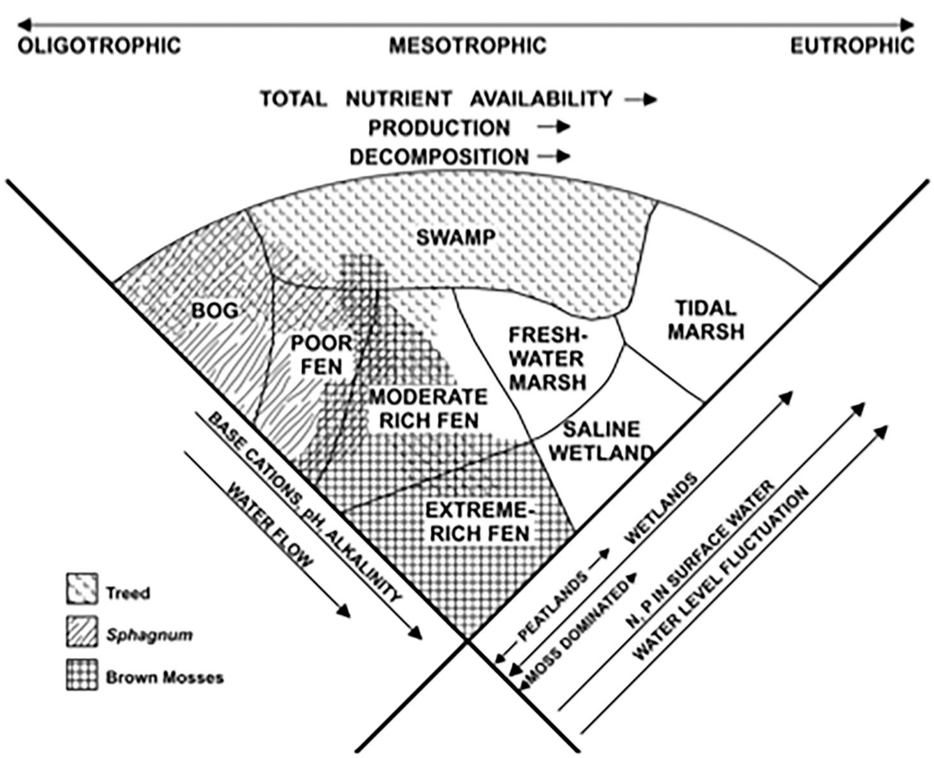

Figure 1. Bog, fen, marsh and swamp wetland classes in relation to hydrological, chemical and biotic gradients (adapted from Zoltai and Vitt 1995).

on 'muskeg' in Canada began about 1945, focusing almost entirely on engineering implications of peat-based materials and soil mechanics, in recognition of its high water content and thus high compressibility. Radforth (1965) noted environmental influences on peatland structure and development, including drainage patterns and water table, but without a comprehensive evaluation of hydrology. Sjörs (1963) provided a detailed description of peatland forms and character in the Hudson Bay Lowlands, from which he inferred aspects of their hydrological behaviour. Wetland hydrology research in Canada began more earnestly in the 1970s (e.g. Dai et al. 1974; Dai and Sparling 1973; Whiteley 1979; Woo 1979; ; 1981), and on peatlands in the early 1980s (e.g. Fitzgibbon 1982; Munro 1984). This emerging interest in part reflects the recognition of impaired wetland function caused by drainage (Whiteley 1979). As previously noted, from the mid-1980s onward, there has been an exponential increase in wetland research, indicating the growing awareness of the key ecosystem services they provide (Pattison-Williams et al. 2018; Schindler and Lee 2010). Understanding the processes that sustain wetland function are key to predicting or managing their function in the face of climate change, resource extraction, contamination and remediation. The Canadian contribution to this understanding is highlighted in Section B, below.

A) wetland characterization in Canada: a hydrological basis

It is essential that wetland scientists have a common framework and terminology to refer to ecosystems that can differ so extensively in form and function. Globally, there are many different approaches to wetland classification. Early approaches were based on botanical character, but later tended to be guided by landscape position, as in the Cowardin systems used by the US Fish and Wildlife Service, or by hydrogeomorphic setting, water source, and water dynamics, as developed for the US Army Corps of Engineers (M. Brinson). Others still allowed for the distinct regional character. A weakness of this approach to classification is that wetlands that are fundamentally similar in form and function could appear in many different landscape positions, have similar water sources, and even exhibit similar water dynamics. The Canadian system overcomes this by identifying wetland classes based on their hydrological function, then on their form (e.g. surface morphology or pattern), and type (physiognomic characteristics of the vegetation communities) (NWWG 1997). Consequently, all wetland scientists in Canada inherently understand the genetic origin and basic function of a particular wetland class, of which there are five, namely bog, fen, swamp, marsh, and shallow water. It is this designation of hydrology as the foundation for the rest of the framework that is one of the crucial innovations, and distinguishing features of the Canadian system of wetland classification. This is reflected in Zoltai and Vitt's (1995) graphical model (Figure 1) that illustrates how wetland classes relate to water level fluctuation and strength of water flow, which also drive nutrient availability, system productivity and organic matter decomposition rates. The ordination of wetlands within Zoltai and Vitt's (1995) framework also reflects landscape features and climate, thus to some extent there is a regional bias to the distribution of Canadian wetland classes (Halsey et al. 1997), as will become evident in the following characterization of Canadian wetland classes. Despite decades of scientific advancement in wetland hydrology, the Canadian Wetland Classification System has endured and remains a relevant and valuable tool for wetland hydrologists. In this first section, we highlight the hydrological character of wetland classes, as defined in Canada, using citations of relevant research performed in Canada.

Mineral wetlands: marsh and shallow water

In this manuscript, we refer to non-peatland wetlands as mineral wetlands, which include marshes, shallow water, and some swamp wetlands. Discussion of the latter is included in the review of peatland research. In Canada, marshes are defined as wetlands with shallow water that fluctuates daily, seasonally or annually, receiving water from surface runoff, stream inflow, precipitation, storm surges, groundwater discharge, or tidal action, often with marked water table drawdown that exposes sediments (NWWG 1997). Shallow water wetlands are transitional between terrestrial wetlands that are saturated or seasonally wet, and permanent deep water bodies (NWWG 1997). In the Canadian wetland classification system, marshes and shallow water wetlands are wetlands with or without emergent vegetation, respectively, and generally without the accumulation of peat. They are the least common of wetland types in Canada, but occur extensively in the Hudson Plains coastal zone, and have an important presence in the Atlantic and Maritime regions, and to a lesser extent in the Temperate and Prairie zones (Mahdianpari et al. 2020), although it is the most common wetland type in the Prairies (Halsey et al.

1997). These shallow-water wetlands may be depression, lacustrine or estuarine wetlands. In Western Canada, these are most commonly represented by Prairie slough wetlands, also referred to as 'pothole wetlands' especially in the Great Plains region, USA. Their importance in Canada is reflected in the Canadian contribution to the International Hydrological Decade (1965–1974), when 125 sloughs were monitored for water level, adjacent water table elevation, and meteorology between 1964–1975. Meyboom (1966) identified seasonal inversions of the water table, being convex in spring resulting in vertical and lateral recharge, drawdown in summer caused by evapotranspiration from the surrounding ring of phreatophytic vegetation, with a concave water table persisting in winter. These were initially described as regionally important for 'depression-focused recharge' but van der Kamp and Hayashi (1998) found recharge to the regional aquifer to be small, albeit locally important. However, Bam et al. (2020) note ephemeral systems (i.e. that dry out seasonally) have a more important recharge function than perennially flooded sloughs. Salt leaching that occurs beneath most systems affirms the concept of depression-focused recharge (Berthold et al. 2004; Parsons et al. 2004). The primary water loss from these systems is evapotranspiration (Woo and Rowsell 1993) from both the open water and fringing non-flooded wetland ring that expands and contracts inter- and intra-annually. Water table drawdown is caused primarily by evapotranspiration in these fringing wetlands, which draws water from the pond (Hayashi et al. 1998). Millar (1971) found that water loss varied directly with the ratio of shoreline length to slough area. Price (1993) quantified the probability of drying similarly, although used slough volume instead of area, and accounted for specific Prairie climatic regions. For example, only sloughs from the climatically drier southern Prairie region were dry >50% of the time, having a shoreline to volume ratio $<0.4\,\mathrm{m\ m^{-3}}$. While many sloughs have an isolated hydrological regime, water loss can include fill-and-spill generated spring runoff (Shaw et al. 2012), although it is typically a small component of the water budget (van der Kamp and Hayashi 2009). Ameli and Creed's (2017) modelling suggested that even distal wetlands contribute to water flow and quality in the North Saskatchewan River, however, surface water connectivity is more common from proximal ones. Yet, Ali et al. (2017) showed groundwater connectivity to Prairie rivers is primarily through deep groundwater.

Land conversion to agriculture profoundly affected the distribution, persistence and function of Prairie sloughs. Snowmelt is the predominant water source for sloughs (Woo and Rowsell 1993), enhanced by low frozen-ground infiltration rates that promote surface runoff on adjacent slopes (Hayashi et al. 2003). However, agricultural practices replace natural Prairie grasses that trap snow with crop-stubble that is less effective at trapping snow, resulting in snow accumulating directly in these wetland depressions (van der Kamp et al. 2003). Replacing cultivated uplands with uncultivated brome grass results in small sloughs drying out (van der Kamp et al. 1999); this condition may be more representative of the pre-agricultural landscape. Zhang et al. (2021) predict that future summers in the Prairie Pothole Region will have reduced precipitation and drier soils in the east but little change in the west. Furthermore, they predict shorter winters and thus less snow, and consequently a longer recharge season with lower recharge rates. This suggests Prairie sloughs, especially in the eastern part of the region, will be more likely to desiccate. Future function will also depend on restoration of Prairie wetlands, 70% of which have been drained since settlement (DUC, 2009). In a synthesis of literature on wetland drainage in the Prairies, Baulch et al. (2021) conclude that drainage increases annual discharge volumes, can increase runoff magnitudes and frequencies, increases nutrient export, reduces groundwater recharge, negatively affects biodiversity and habitat, and alters carbon and greenhouse gas exchange.

Coastal salt-water marshes in Canada include those of the eastern and western seaboards and the Subarctic along James and Hudson Bay. Coastal salt-water wetland water exchanges are mostly driven by tides, although Byers and Chmura (2014) noted that subsurface hydrology in a Bay of Fundy marsh was less a function of (extreme) tidal range than of soil permeability, marsh geomorphology, precipitation, and duration between inundation events. Tidal inundation of James Bay coastal wetlands in Ontario has a diminishing influence on salinity towards the south, and soil salinity reflects the maximum tidal crest (Price et al. 1992). The coastline rises isostatically at about 1.4–2 m per century (Pendea et al. 2010), resulting in a series of parallel raised beach ridges that trap water behind them, forming marshes (Price and Woo 1988a) that direct water laterally into drainage channels (Woo and diCenzo 1989). Persistent salinity in these marshes is a result of upward diffusion of relict salt (Price and Woo 1988b) from Tyrrell Sea sediments (Martini 1981). Further inland (a proxy for time since shoreline emerges due to isostasy) peat forms, raising the local surface elevation, restoring water-flow towards the coast and causing local groundwater recharge (Price and Woo 1990).

Coastal freshwater marshes also occur extensively around the Laurentian (Lakes Ontario, Erie, Huron and Michigan) and Manitoba (Lakes Winnipeg, Manitoba, and Winnipegosis) Great Lakes (Watchorn et al. 2012). The Great Lakes Coastal Wetlands Consortium (Albert et al. 2005) classify coastal wetlands as primarily open coast wetlands, drowned river-mouth wetlands, and barrier-protected wetlands. The open coast and drowned river-mouth wetlands have a hydrological regime dominated by lake-level that is characterized by seasonally high water in late spring, periodic seiche (wind-driven) water level changes, and long-term water level cycles separated by intervals of eight years or more (Cohn & J. E. Robinson et al. 1975). For open and drowned river-mouth wetlands, water levels are highly responsive to lake levels during the spring maxima, but diminishingly so as lake-levels decline to the wetland surface throughout the growing season (Price 1994a), thereafter responding to local weather inputs and evaporation losses (Price 1994b). When protected by barrier beaches, coastal wetlands have modulated lake water inputs and losses. With narrower barrier bars, water flow direction oscillates between lake and marsh; with broader barriers, a groundwater mound persists, and flows are bidirectional toward both the marsh and the lake (Crowe and Shikaze 2004). In Cootes Paradise, a barrier-protected wetland at the western tip of Lake Ontario, stabilization of water levels following construction of the St. Lawrence Seaway in 1959 reduced vegetation diversity (Chow-Fraser 2005) including a decline in emergent plant cover to <15% (Chow-Fraser et al. 1998). Terrestrially-sourced water inputs can be important to some barrier-protected wetlands. Schellenberg et al. (2017) found surface flow into Delta Marsh Watershed (Manitoba) was the primary input, and predicted a land-use shift to urbanization would increase this by >50%, whereas an increase in cropping (currently the dominant land-use) would have minimal effect, and reverting to natural vegetation would cause a 10% decrease in inflow.

Peatlands: swamp, bog and fen

The Canadian Wetland Classification System (NWWG 1997) defines peatlands as organic wetlands having a peat depth of >40 cm, which includes some swamps, and all fens and bogs. Swamps and fens receive base-rich groundwater and/or surface water and are said to be *minerogenous*, hence *minerotrophic*. Bogs, however, are ombrogenous (precipitation-fed), receiving no groundwater or surface water and thus are *ombrotrophic*, lacking in the base minerals that buffer acidity created by decaying organic matter (Zoltai and Vitt 1995).

Peat is an organic soil that accumulates in areas sufficiently and persistently wet such that organic matter decomposition is slower than the average annual production (Vitt 1994), and this drives peatland succession (Thormann et al. 1999). The classic peatland successional pathway is one that begins with marshes or swamps in the post-glacial environment, leading to rich fens, poor fens, and eventually bogs accompanied by paludification of adjacent upland areas (Bauer et al. 2003; Klinger and Short 1996; Lacourse et al. 2019; Price et al. 2023). As sufficient organic matter (peat) accumulates, there is a transition from rich fen (so-called because of their highly diverse vegetation community structure), which have strong mineral-rich surface and/or groundwater inputs, to moderate, then poor fen. With diminishing minerogenous water supply (Zoltai and Vitt 1995), the peatland surface rises because it is increasingly dominated by relatively decay-resistant sphagnum mosses, eventually becoming isolated from water inflow. At this stage, water input, at least to the living layer of the peatland, is exclusively provided by precipitation, and by definition this peatland is a bog. The role of water and mineral inputs to wetlands displayed by Zoltai and Vitt's (1995) graphical model (Figure 1), shows peatlands are associated with more persistently wet, relatively stable water regimes, and consequently dominated by mosses (with sedges increasingly important in fens with increasing minerotrophy), and trees present in peatlands with intermediate water-level variability.

Swamps, both mineral and peatland forms, are the least studied, least recognized, and least understood wetland class in Canada. This is because they are highly variable in form, setting, and vegetation community, thus are hard to define (Warner and Asada 2006). Swamps are wetlands dominated by woody vegetation, including shrub forms and trees, both deciduous and conifer. The dominance of woody vegetation is a consequence of their deeper and more variable water table (Jeglum 1991; Locky et al. 2005), which affects their water budget and geochemistry. Swamps can be discrete systems, sometimes extensive albeit connected to and reliant on the broader landscape for water (Devito et al. 1996) and base ion inputs (Devito et al. 1999), or transitional landscapes between upland and fen (Locky et al. 2005). In addition to direct precipitation, some swamps receive water from overbank channel flow, typically in spring (Bradford 2000), as well as groundwater, which may be small but important for determining water chemistry (e.g. Hill and Waddington 1993). Woo and Valverde (1981) found seasonal streamflow out of the swamp matched inflow. Well-defined channels experienced relatively little groundwater exchange within the swamp, but in less-defined channels there was considerable influent and effluent behaviour that had a pronounced effect on streamflow and biogeochemistry (Warren et al. 2001; Galloway and Branfireun 2004). Seasonal groundwater flow (2011–2017) through a northern Alberta margin swamp between upland and fen, ranged from 1 – 30% of rainfall (Elmes and Price 2019). However, groundwater input to a swamp situated on permeable moraine can be an order of magnitude larger than precipitation input (Roulet 1991a, 1991b), thus constituting a dominant (>70%) proportion of runoff (Waddington et al. 1993). Near-surface water in such systems promotes rapid runoff response and recession rates (Roulet 1991a, 1991b; Elmes and Price 2019). Whiteley and Irwin (1986) showed that upland and isolated swamps reduced stormflow to downstream systems. Emili and Price (2006) found swamp-forest in the oceanic climate of north-coastal British Columbia occurred on relatively steep slopes and were effective at generating runoff through small seeps. The relatively constant groundwater input from adjacent colluvial slopes maintained the water table near the ground surface in the swamp. In response to large rain events, the water table there rose quickly, and the saturated areas developed rapidly on adjacent ground surfaces (Fitzgerald et al. 2003). They found the release of surface water directly to the stream comprised up to 95% of stream discharge. However, the relatively steep gradients were associated with more rapid water table decline, which increased water detention times compared to open peatlands that had higher levels of saturation; thus slopes proportionately yielded less runoff (Emili et al. 2006). Devito et al. (2017) showed low-relief peatland-swamp wetlands in Alberta were the major source of runoff, compared to upland areas, with the lowest median annual catchment evapotranspiration and highest runoff (13 to 27% of rainfall). However, runoff from swamps is reduced by their high rainfall interception capacity. Emili and Price (2006) found seasonal interception between ∼17–22%, but with rates up to 59% for small, low-intensity rainfall. They also showed that swamp canopy intercepted fog, which was a relatively small contributor to throughfall, since it occurred mostly on days with rain that dominated the process. Duval (2019) showed interception in a mixed coniferous-deciduous cedar swamp in Southern Ontario was ∼30%.

Alberta peat-margin swamps that transition between upland and fen can have a more variable water table and downward hydraulic gradients compared to adjacent fens, albeit with similar pH, EC, and base cation concentrations (Elmes et al. 2021). Water table drawdown, which is relatively large in swamps compared to other wetland systems, affects hydraulic properties of the soil, increasing both peat particle (Redding and Devito 2006) and bulk density (Elmes et al. 2019). Moreover, soil aeration increased redox potential, resulting in a higher concentration of SO_4^{2-} that was mobilized and released to runoff (Devito and Hill 1997; Eimers et al. 2007), and increased levels of P and N in surface and pore water (Devito and Dillon 1993). Deeper water tables promote leaching of solutes in swamps, and seasonal drawdown may suppress the rise of dissolved minerals (Scarlett and Price 2013). In Alberta's Western Boreal Plain, forested uplands can draw water from wetlands and swamps at the base of hillslopes (Bauer et al. 2009), resulting in evapoconcentration of dissolved minerals and nutrients in these areas (Bauer et al. 2009; Plach et al. 2016). However, Elmes et al. (2018) found that the groundwater flow systems influencing peatlands in the Athabasca Oil Sands Region (AOSR) of Alberta are shallow and localized, with the swamp-like margins that connect upland and wetland exhibiting persistent recharge (Elmes et al. 2021). Research and modelling on similar systems in Saskatchewan confirm that regional connectivity has an important role in water supply to margin swamps, their water table dynamics, and groundwater transmission to adjacent fen peatlands (Dimitrov et al. 2014).

The regional distribution of bogs and fens in Canada varies along latitudinal and meridional energy and moisture gradients, subject to local or regional landscape. Latitude strongly dictates radiative and temperature regimes, hence plant productivity and soil decomposition rates, thus peat accumulation and peatland distribution (Primeau and Garneau 2021; Rouse 2000). Latitude also affects precipitable water in the atmosphere and potential evapotranspiration. Meridional effects relate to the degree of continentality, hence rainfall and humidity. Damman (1979) observed that in eastern North America there is a northern and southern limit to the presence of bogs because they are fed only by precipitation, which is lower at high latitudes, and potential evapotranspiration which increases towards the south, both of which limit the water supply. Fens, which have additional water sources, have a broader range (Halsey et al. 1997). Peatlands occur extensively in Canada's maritime Provinces, reflecting the wetter climate. These range from isolated systems (Damman 1978, 1986; Langlois et al. 2015a), blanket bogs that occur extensively in oceanic climates on undulating landscape (Price 1992), to peatland complexes hosting a vast array of interconnected patterned bogs and fens (Foster and Glaser 1986; Price 2009c; Price and Maloney 1994). In continental locations, peatlands occur sporadically over the Canadian Shield because of isolated drainage patterns (Branfireun and Roulet 1998), providing favourable conditions for bog and poor fen development (Moore et al. 2021; Markle et al. 2022), although fens are more common where upland connectivity is present (Devito et al. 1996). In the Hudson Bay Lowlands, bogs and fens occur extensively, primarily in peatland complexes due to the flat topography and low permeability of basal sediments (Sjörs 1959; Glaser et al. 2004), and are a primary water source for major rivers (Orlova and Branfireun 2014; Richardson et al. 2012). In the Western Boreal Plain, deep sediments facilitate groundwater exchanges (Devito et al. 2005), which along with lower precipitation favour the development of fens, although bogs also occur (Vitt et al. 1994). In High Arctic settings, peat forming wetlands can occur where groundwater or late-lying snowbanks sustain wetness (Woo and Young 2006; Young and Woo 2000), although peat accumulation is generally too thin ($<$15 cm) for them to be classified as peatlands using the Canadian definition of 40 cm (NWWG 1997). Fen peatlands occur in the low arctic where, in addition to snowmelt, rainfall can help sustain summer wetness (Roulet and Woo 1986a). In British Columbia, bogs and fens occur primarily on the coastal lowland (Howie and Van Meerveld 2013). In mountainous areas, bogs and fens generally occur in valley bottoms (Kershaw 2003, 2022), although in hyper-maritime areas they can occur at higher elevations and greater slopes (Emili and Price 2006; Fitzgerald et al. 2003).

In bogs and poor fens, sphagnum mosses are a keystone genus that dominate the plant cover (Rochefort 2000) and have a critical role in their hydrological function (Thompson and Waddington 2008). Their dominant cover, along with their resistance to decay (Glenn et al. 2006; Turetsky et al. 2008), means they are the primary constituent of bog and poor fen peat, comprising the matrix for water storage and flow. The upper layer of moss and peat has commonly been referred to as the *acrotelm*, being the variably saturated vadose zone, typically the upper ~50 cm (Price 1992; Howie and Van Meerveld 2011) but sometimes 10s of cm more (Lafleur et al. 2005). This overlies the perpetually saturated *catotelm*. The hydrological properties ascribed to the acrotelm and catotelm have commonly been used to describe many peatland functions, however, Morris et al. (2011) argue that the diplotelmic model is overly simplistic for many hydrological and biogeochemical processes, as it does not accommodate spatially relevant hotspots, nor account for the presence of biogenic gas below the water table (thus peat is not saturated). However, as disciplines mature, it is common for foundational conceptual models (like the acrotelm-catotelm model), which favoured simplicity and apparent generalizability, to be challenged in response to empirical evidence of nature's complexity. From the inception of wetland ecohydrology as a discipline, a growing body of evidence suggests that wetlands exhibit a diverse range of properties, characteristics, and functions that are most accurately expressed along a continuum, and therefore preclude easy categorization. While inconsistencies have been identified with the acrotelm-catotelm conceptual model (Morris et al. 2011), including what constitutes the boundary between the two zones, it has been a fixture of Canadian peatland literature over the past 75 years. As such it will continue to have utility in describing a characteristic pattern in hydrophysical properties that is found in many peatlands across the circumboreal region, while newer research explores concepts where the concept does not apply. This paradox is examined in detail by Morris et al. 2011). Regardless of nomenclature, the variably saturated near-surface zone of bogs exhibits strong gradients in hydraulic properties that govern water exchanges. For example, the hydraulic conductivity in this zone can decrease up to five orders of magnitude over 50 cm (Hoag and Price 1995). Drainable porosity, the property of explicitly defined layers that together comprise the aquifer's specific yield (Price et al. 2023) also decreases strongly with depth (Quinton et al. 2008). These properties have been shown to be related to bulk density (Gupta et al. 2023) which is easier to determine. The strong gradients of hydraulic conductivity result in a water table-transmissivity feedback mechanism (McCarter and Price 2017a, 2017b, 2017c; Price 1992), in which high water tables exploit the high hydraulic conductivity in the upper layer to promote runoff, and lower water tables relegate flow to the lower hydraulic conductivity layers. In patterned peatlands (i.e. peatlands with alternating ridges and pools aligned perpendicular to the dominant flow direction) this mechanism increases connectivity between peatland types and drainage channels during spring and fall, so that a fill and spill drainage mechanism operates (Quinton and Roulet 1998). The fill and spill drainage mechanism can also be observed inter-annually in Precambrian Shield headwater catchments, where in drier years wetlands decrease catchment hydrological connectivity but in wetter years increase hydrological connectivity (Lane et al. 2020). Once again challenging a foundational model, Balliston and Price (2022) suggest a three-phase connectivity regime is more appropriate in the James Bay Lowlands of northern Ontario, in which runoff activity can be characterized as disconnected (typically winter and dry summer periods), connected (typically late spring and fall), or high activity (typically during snowmelt).

Fens, as previously noted, are hydrologically connected to the upland landscape, receiving surface- or groundwater, the quantity and quality of which controls their character (Glaser et al. 2004), ranging from rich fens well-connected to sources of external water and dissolved minerals (Vitt and Chee 1990), to poor fens that have accumulated sufficient peat to severely reduce the hydraulic gradients that drive water and solute flow towards them (Lacourse et al. 2019). They are distinct from swamps, which are also connected to the surrounding landscape, with a less variable water table regime (see Figure 1). While groundwater inflows have often been found to be the primary source of solutes to fens (Larocque et al. 2016; Whitfield et al. 2010), this is not always the case where there is solute-rich surface water inflow (Duval and Waddington 2018). The solute inputs are variable both seasonally (Vitt et al. 1995) and annually (McLaughlin and Webster 2010). Seasonally, solutes can be flushed during snowmelt, and develop spatially distinct patterns as the system dries, especially in less connected areas (Thompson and Woo 2009). Runoff from dry periods can thus have an enhanced geochemical signature (Metcalfe and Buttle 2001). Groundwater flowing beneath these peatlands can help sustain surface wetness (Ferlatte et al. 2015), predominantly near fen margins (Elmes et al. 2021) and help generate baseflow in outlet streams (Branfireun and Roulet 1998; Buttle et al. 2004; Spence et al. 2011). This can persist in winter (Price 1987), due to contributions from adjacent bogs as well as mineral uplands (Price and FitzGibbon 1987). However, variable groundwater input can cause flow reversals within fens (Devito et al. 1997), resulting in flow from peatland to upland (Elmes and Price 2019; Ferone and Devito 2004). Reversals within peatlands that respond to radially outward hydraulic gradients can be sustained because of subsidence associated with the high compressibility of peat (MacFarlane and Radforth 1965), although the strength of flow may be mitigated because of the potential orders of magnitude decrease in hydraulic conductivity associated with seasonal subsidence (Price 2003). Alternatively, the generation of entrapped biogenic gases, mainly methane, can cause peat expansion and alter hydraulic gradients (Kellner et al. 2005; Strack et al. 2006), but also cause pore occlusion that reduces flow (Kellner et al. 2004; Kettridge et al. 2013). Peat volume change reflects patterns in water storage (Roulet 1991b) that can rival specific yield in terms of accounting for seasonal water storage changes (Price and Schlotzhauer 1999). Peat volume change has the effect of maintaining the water table closer to the surface than it would otherwise be (Whittington and Price 2006), consequently reducing peat decomposition rates and carbon release (Strack et al. 2008; Strack and Waddington 2007), and altering nutrient dynamics (Macrae et al. 2013). Persistent high water level in fens can cause aqualysis, in which peat degradation results in open water (Tardiff et al. 2009).

Compared to bogs, the additional surface or groundwater input to fens means their water tables tend to be higher and more stable (Balliston and Price 2022; Dai et al. 1974; Vitt et al. 1995), although not necessarily (Duval and Waddington 2011). Consequently, runoff from fens can be larger than from bogs (Connon et al. 2014; Price and Maloney 1994; Quinton et al. 2003). Fen peat (Elmes et al. 2021) is distinct from that in bogs due to the different plant community from which it is formed, its geochemistry, and degree of water table variability (Zoltai and Vitt 1995), hence state of decomposition. Fen saturated hydraulic conductivity (K_{sat}) decreases with depth, as in bogs, but in both peatland types its variability is high, much larger than variability arising from methodological differences between laboratory and field techniques (Rosa and Larocque 2008). While few studies have compared K_{sat} in nearby bogs and fens, Price and Maloney (1994) found K_{sat} higher in a fen water track than in the surrounding basin fen, and intermediate values in a nearby bog. Balliston and Price (2023) found fen K_{sat} to be higher, but not significantly so. Gupta et al. (2023) found that near-surface K_{sat} in three Southern Ontario peatlands was greatest for a swamp, then bog, then fen, albeit again not significantly different. In bogs and fens, K_{sat} can decrease by three or more orders of magnitude over tens of centimetres (Quinton et al. 2008). During dry periods when the water table is relatively deep, the low drainable porosity (~0.05) at depth enables the water table to rise quickly into the upper, highly conductive layer and produce a rapid hydraulic response to precipitation (Quinton and Hayashi 2004), thereby regulating water table position.

B) hydrological themes in wetland research

i) water stores and fluxes in the near surface

A key issue addressed in Canadian wetland research within the past 75 years has been to better understand the controls on soil-vegetation-atmosphere transfers (SVAT) and the near-surface hydrology that strongly influences these transfers. On vegetated wetland surfaces, plant, energy, water and carbon exchanges are closely integrated and exhibit strong feedback with the lower atmosphere (Wang et al. 2002; Yi et al. 2007). This in turn feeds back to the soil moisture conditions, biogeochemical and biophysical processes (Malhotra et al. 2016) that can modify the soil, thus feedbacks to the soil-vegetation-atmosphere fluxes (Kettridge et al. 2013, 2017; Waddington et al. 2015). This feedback is strongly governed by not only atmospheric fluxes but the structure of the variably saturated upper layer of soil between the surface and the water table (the vadose zone). Given that wetlands are defined as having high water tables, at least periodically, the vadose zone of wetlands is typically thinner than in upland settings (e.g. Price and FitzGibbon 1987; Todd et al. 2006; Wells et al. 2017), or entirely absent in the case of marshes and shallow open water wetlands. Thus, evaporation from open water that is present in many wetlands is simple in comparison to SVAT processes. Estimating and understanding open water evaporation rates, however, is complicated by markedly different atmospheric resistance than adjacent vegetated surfaces and the highly ephemeral nature of standing water levels (Price 1991). Research on vadose zone processes in wetlands, especially peatlands, is relatively recent, mostly from the 2000s onward, in part due to the inability of some instrumentation designed for mineral soils to effectively measure and parameterize key attributes of peat soils. SVAT wetland research began in earnest in the 1970s.

Progress in understanding vadose zone hydrology

MacFarlane (1957) characterized peatland ecology and the resulting peat structure based primarily on qualitative factors in developing a guide to peatland (muskeg) classification. Many of the early works were focused on characterizing peat and peatlands for their geotechnical properties (Landva and Pheeney 1980; Walmsley 1977). Yet, these works presented some of the first measurements of hyaline cell openings (vacuoles present in sphagnum mosses that affect water retention) and other peat structural components that are critical to peatland vadose zone hydrology. These early works laid the foundation for the profusion of studies by Canadian scientists over the following decades.

The upper layer of living moss and dead but poorly decomposed moss litter are rarely saturated. Lacking roots and a vascular system, mosses are reliant on the upward movement of water through capillary flow by water conducting pores (i.e. saturated and hydrologically connected at a given soil water pressure) to supply water at the evaporating surface (the growing surface of the moss, in the case of sphagnum, called the capitula) from below (Price et al. 2009; Price and Whittington 2010). As such, considerable effort has been spent describing their water retention characteristics and unsaturated hydraulic conductivity. McCarter and Price (2014) estimated upward water flow in different moss species and showed that *Sphagnum fuscum* and *S. rubellum* were better able to sustain moisture at the typical range of pressures likely to occur in moss hummocks than what is now referred to as *S. magellanicum* complex. Goetz and Price (2015) showed how different unsaturated hydraulic conductivity and water retention characteristics of sphagnum and a common fen moss, *Tomenthypnum nitens*, combined to make them equally effective at upwardly transmitting water for evaporation, despite the apparent dryness of the latter. In both cases, the mosses are the evaporative interface between the peatland and the atmosphere.

Once new growth has buried moss tissue and the plant cells subsequently die creating ‘moss litter’, decomposition processes begin, creating peat. The peat below living mosses is influenced by the peatland’s degree of minerotrophy and can result in pronounced changes in hydrophysical properties with depth and between peatlands due to differences in the rate of decomposition and consolidation (Moore and Basiliko 2006). As the degree of minerotrophy increases, decomposition typically increases, causing a shift in the pore size distribution towards smaller pores

(Balliston et al. 2018; McCarter et al. 2019; McCarter and Price 2014). These changes can be abrupt, as in the case of *Tomenthypnum nitens* (Goetz and Price 2015), but as long as the hydraulic continuity of pores is unimpacted, vertical transport of liquid water will occur (McCarter and Price 2015). This shift towards a greater proportion of smaller pores is clearly shown by the decrease of saturated hydraulic conductivity by several orders of magnitude within a ten or a few tens of centimeters from the surface in the vadose zone (Golubev and Whittington 2018; McCarter and Price 2014; Quinton et al. 2008; Taylor and Price 2015). Microscopic analysis and numerical modelling (Rezanezhad et al. 2009, 2010) confirmed that the reduction of saturated hydraulic conductivity with depth is the result of increasing pore compaction and interparticle tortuosity, and decreasing pore diameter with depth (Quinton et al. 2008; Gharedaghloo et al. 2018). Balliston and Price (2020) further described how peat structure contributed to water (and solute) rise in moss litter, and its local variability. In some sphagnum dominated systems, water vapour transport, and subsequent distillation at the capitula, have been observed to be critical for sphagnum moss desiccation avoidance (Price et al. 2009). Yet, there remains key questions on the coupled importance of liquid and vapour water fluxes to sustaining biological processes and the impacts on local ET.

While different broad morphologies of moss and peats can influence vadose zone hydrology, so can the specific species of sphagnum moss. McCarter and Price (2014) observed that sphagnum mosses that grew further above the water table were better able to retain water under greater soil water tensions and more effectively conduct water upwards to the evaporating surface (in this case the moss capitula). These trends were broadly confirmed on a variety of sphagnum species but in similar growth microforms (Goetz and Price 2015; Golubev et al. 2021; Golubev and Whittington 2018; Taylor et al. 2016; Taylor and Price 2015), suggesting that the soil hydraulic properties of a given sphagnum species may not be limited to a single ecohydrological niche or microform (Golubev et al. 2021). As such, the specific sphagnum species may be less important than the ecohydrological niche (microform) when determining their hydrological function and hydrophysical properties, but this remains an open question in the literature.

When the water table drops, causing water to preferentially drain from the larger pores, the remaining peat may not have the physical structure to support the overlying material (Price et al. 2005), resulting in a lowering of the peat surface (Lafleur and Roulet 1992; Price 2003; Price and Schlotzhauer 1999; Waddington et al. 2010, Whittington and Price 2006). When the water table rises and the drained pores refill with water, the peat surface rises to a similar degree as its decline (Price 2003; Price and Whittington 2010; Waddington et al. 2010). This is a mostly reversible volume change in undisturbed peatland and often referred to as 'mire breathing'. As the larger pores compress following consolidation, the unsaturated hydraulic conductivity and soil water retention in the unsaturated zone increase, resulting in greater hydrological connectivity between the water table and atmosphere. This mechanism represents a critical ecohydrological feedback to maintain sufficient water supply to sphagnum capitula (Golubev and Whittington 2018). However, these changes do not occur uniformly across a peatland, rather the largest changes occur in lower bulk density peat (low lawns) and the least in higher density peat (hummock microforms) (Waddington et al. 2010; Whittington and Price 2006). Changes in the vadose zone pore volume are also an important consideration for estimating water storage and surface elevation (Price et al. 2005; Schlotzhauer and Price 1999), as well as laboratory-determined hydrophysical properties (Golubev and Whittington 2018).

Scaling and partitioning of evapotranspiration

Over the past several decades Canadian wetland scientists have made significant progress towards our understanding of the scaling and partitioning of evapotranspiration (ET) in wetlands aided by the advent of more accessible high precision methodologies. Some of the first wetland energy budget and evapotranspiration work was conducted in the Hudson Bay Lowlands (Stewart and Rouse 1976a). Energy-budget calculations and equilibrium evaporation estimates from a well-drained lichen-dominated raised beach ridge and a wet sedge meadow produced a simple model, expressed in terms of incoming solar radiation and air temperature, from the comparison of actual and equilibrium evaporation, which on a daily basis are accurate to within ± 10% (Stewart and Rouse 1976b). Examples of the earliest work on fire effects on evapotranspiration were also done in this region (Rouse and Kershaw 1971). Areas of ground lichen in the Subarctic are particularly susceptible to fire either by human activity or by natural causes. Experimental work in the Hudson Bay Lowlands showed that the burning of lichen has a pronounced effect on the groundwater regime, where the lichen-dominated surfaces act as an effective mulch in preventing evaporation from the subsurface zone whereas the burned areas, which can evaporate more water into the atmosphere when moist, also develop strong resistances to evaporation as the soil surface layers become drier (Rouse and Kershaw 1971).

Work on wetlands evolved based on the principle that there is a large degree of spatial variability in wetland ET within (vegetation community scale) and among (catchment scale) landscape units, which are controlled by patterns in vegetation, soil moisture and soil physical properties. Work has been done in Canada demonstrating that spatial variability in ET within wetland and peatland landscapes can be large, and as such these systems should not be treated as homogenous units, and that they are strongly influenced by their surrounding catchment (Green et al. 2021; Petrone et al. 2008; Roulet and Woo 1986). As a result of these internal (hydrophysical properties, vegetation community) and external controls (basin shape, catchment characteristics), peatland actual ET in general, is less than potential ET (PET) (Lafleur and Roulet 1992). Further, within a peatland, sphagnum and other nonvascular wetland plant species control ET differently throughout the growing season and as such should be considered an integral part of the moisture and water balances within wetland environments at the sub-landscape unit scale (Brown et al. 2010, Scarlett et al. 2017).

Methodological developments for assessing peat soils

Canadian researchers have developed methodologies specifically designed to characterize the soil hydraulic properties of peat, which are crucial for interpreting their role in system hydrological function. Fundamental hydraulic properties of the vadose zone include its water retention behaviour and its hydraulic conductivity, which varies with the level of saturation. Thus, hydraulic conductivity depends on the morphological character of the peat matrix (Price et al. 2023). In peat, soil morphology varies spatially due to differences in the botanical origin of the plant material from which it is composed, its state of decomposition and its position in the soil profile that affects its degree of consolidation (McCarter et al. 2020). By comparison to mineral soils, peat is delicate, especially the living and poorly-decomposed moss and litter that dominates the upper layer in bogs and many fens. This, along with its high compressibility, render methods developed for fixed volumes of mineral soil unsuitable for measuring its water retention characteristics and its unsaturated hydraulic conductivity function (Caron et al. 2015; Price et al. 2008). To address this, Price et al. (2008) developed the floating tension disc method for laboratory use that concurrently measures soil water retention and hydraulic conductivity curves below saturation in the highly compressible and low bulk density organic soil (specifically mosses and sphagnum litter). With this method, a hydraulic and pressure gradient is induced across a peat core to measure the steady-state flow of water and soil moisture content. However, by inducing a pressure gradient across the core, the recorded pressure head does not necessarily agree with the average internal soil water pressure, so McCarter et al. (2017a, 2017b, 2017c) updated this method by reversing the direction of flow, thus eliminating the pressure gradient across the core.

Canadian scientists have also focused methodological developments in peat-based growing substrates (Allaire-Leung et al. 1999; Nemati et al. 2002). Nemati et al. (2002) used tension tables to induce a constant pressure potential on a sample and measure the air-entry pressure using a mini-tensiometer and a pressure transducer. Allaire-Leung et al. (1999) used time domain reflectometry to show how plant growth impacts peat hydrophysical properties. However, these techniques have not been adopted by the wider peatland research community.

Golubev et al. (2021) questioned the appropriateness of the standard 5 cm core, comparing the derived soil hydraulic properties of a 15 cm core to its three 5 cm constituent cores. They noted that the minimum pressure step was equal to the core height (thus 15 cm in the larger core) such that use of a larger core is fundamentally unable to characterize the critical soil water retention and unsaturated hydraulic conductivity at pressures close to zero (saturation). In other developments, computed tomography scans have been used to estimate hydraulic conductivity, pore network geometry, and solute transport behaviour (Gharedaghloo et al. 2018; Quinton et al. 2009; Rezanezhad et al. 2009, 2010. While providing a meaningful contribution to the understanding of peat properties, this method is not commonly available to most researchers.

ii) solute transport

The transport, transformation, and fate of nutrients and other solutes is integrally linked to the form and function of wetlands, and the role they perform in the broader landscape. Moreover, incidental introduction of contaminants into wetlands from atmospheric pollutants (Branfireun et al. 1999; McCarter et al. 2022), accidents like train derailments or pipeline rupture

(Zoltai and Kershaw 1995), and industrial or agricultural effluent (Mudroch and Capobianco 1979); and the deliberate release for the purpose of treatment (Viraraghavan and Ayyaswami 1987) has established the value of solute transport research in wetlands. Many studies have been conducted to demonstrate the distinct biogeochemical processes that occur in wetlands and highly organic soils, such as peat. Yet, these studies should be viewed within the broader context of scientific developments in general solute transport theory in conventional mineral porous media (many of which were also conducted by Canadians, see Hayashi and van der Kamp (2023) for a review). Early work on solute transport in wetland soils focused on the use of peat as an adsorptive medium, due to the large specific surface area (Tinh 1970) and high organic carbon content (Smith et al. 1958). Laboratory experiments demonstrated the ability of peat to effectively adsorb, and therefore retard, heavy metals (Coupal and Lalancette 1976), hydrocarbons (Gharedaghloo and Price 2021; Zytner 1994), volatile organics (Zytner et al. 1989), naphthenic acids (Janfada et al. 2006), landfill leachate (Cameron 1978), and nutrients found in wastewater (Rana and Viraraghavan 1987).

The unique characteristics of wetlands have led them to be used to treat contaminated wastewater, not only due to the adsorptive properties of organic soils but also the high rate of biogeochemical cycling (Kennedy and Mayer 2002). Natural and artificial treatment wetland pilot projects were established across Canada to assess the efficacy of contaminant retardation, transformation, and removal, such as those near James Bay as described by (Dubuc et al. 1986), eastern Ontario (Fernandes et al. 1996), south-central Ontario (Rochfort et al. 1997), and the Kivalliq Region of Nunavut (Yates et al. 2012). Many of these projects placed an emphasis on understanding function, as opposed to process. Yet, elucidating these complex interactions between surface chemistry, pore geometry, and solute reactivity in highly organic soils have also been a significant contribution of Canadian scientists. Initial forays into these interactions involved core-scale laboratory experiments, like those of Price and Woo 1988c; Hoag and Price (1997), and later authors (Caron et al. 2015; Kleimeier et al. 2017; McCarter et al. 2018, 2019; Rezanezhad et al. 2012). These experiments found earlier solute arrival and a more prolonged tailing than would be expected in a conventional porous media, which was attributed to the dual porosity structure of peat. This structure, which is particularly associated with sphagnum peat, is a consequence of the large proportion of dead-end pores (Hoag and Price 1997; Price and Woo 1988c) that act as water and solute reservoirs. Although the geometry of these pores precludes water flow, the process of molecular diffusion attempts to equilibrate concentrations between the mobile and immobile regions, causing an exchange of solutes proportional to the concentration gradient. However, Simhayov et al. (2018) and McCarter et al. (2019) noted that in some peats the exchange rates were essentially instantaneous, resulting in solute transport not being significantly retarded, and therefore capable of being represented by the conventional advection-dispersion equation.

Gharedaghloo et al. (2018) used pore network modelling of a high-resolution peat sample collected from Scotty Creek, NWT, demonstrating the locally isotropic nature of peat and increase in tortuosity with decomposition. These physiochemical characteristics of peat have been found to have profound implications for ecohydrological and biogeochemical function of peatlands. Rezanezhad et al. (2017) showed that the dual porosity structure of peat has implications on the biogeochemical cycling of nitrogen, specifically that the larger interfacial area between the mobile and immobile pore space found in deeper, more decomposed peat enhances nitrate reduction. The attenuation of contaminants due to the adsorptive capacity of peat and diffusion into dead-end pores contributed to the resilience of graminoids and mosses exposed to the high salinity of oil sands process-affected water (Rezanezhad et al. 2012b). Rezanezhad et al. (2012a) showed that the presence of vascular plants increased upward solute transport, due to hydraulic lift and redistribution, lowering the time for a contaminant to potentially impact the surface vegetation. Boudreau et al. (2009) observed adsorption of potassium and copper in a peat-based growing media, significantly limiting upward transport of copper due to its higher adsorption affinity. Recent research has also examined the fate and transport of non-aqueous phase liquids (NAPL) in peat, characterizing the multiphase flow behaviour of diesel (Gharedaghloo and Price 2019; Gupta et al. 2023) and the response of peatland microbial communities to NAPL contamination (Gupta et al. 2020). This work suggested that minimally invasive methods of remediation, such as ditching, artificial water table manipulation, and biodegradation should be explored as viable techniques for reducing the volume of immiscible contaminants in peatlands (Gharedaghloo and Price 2019; Gupta et al. 2023).

One of the first attempts globally to explore solute transport in peatlands at the field scale was the experimental work of Hoag and Price (1995), which described the results of a natural gradient tracer test in a Newfoundland blanket bog. This study affirmed that the axiom of macroscopic heterogeneity in the hydraulic conductivity field exerting a dominant influence on the transport of solutes in groundwater remains true in peatlands. Due to the typical pattern found in peatlands in which hydraulic conductivity decreases several orders of magnitude within decimeters of depth, the tracer movement was largely constrained to the high permeability near-surface (Hoag and Price 1995). However, it also demonstrated that the physical structure of peat, specifically the presence of the immobile porosity, could considerably slow the advance of a solute front due to diffusion into dead-end pores, even when the average groundwater velocity is high and the solute is ostensibly non-reactive. This latter notion, the presumption that the widely-used tracer, chloride, is conservative, has been challenged decades later by McCarter et al. (2019) who proposed that the negatively-charged peat surface can push anions into the highest velocity regions of the pore space in a process known as anion exclusion, while at higher chloride concentrations ($>100\,\mathrm{mg\ L^{-1}}$) anion adsorption retards chloride transport (Caron et al. 2015; McCarter et al. 2018, 2019).

The exchange of solutes between wetlands and the surrounding landscape has also been an area of active Canadian scientific inquiry. At a prairie slough in Saskatchewan, Hayashi et al. (1998) revealed the seasonal cyclical movement of chloride between the wetland and adjacent upland. Salinity cycles operating on multiannual timescales, related to wet and dry climatic periods, were described by Heagle et al. (2013) at a similar Saskatchewan wetland. Dry periods also impacted sulphate export at two headwater swamps in central Ontario, following a prolonged water table decline in which oxic conditions resulted in sulphur oxidation, subsequent high-intensity precipitation events resulted in disproportionately high solute export at the wetland with limited upland connectivity (Devito and Hill 1997). At a sloped wetland complex consisting of swamp, wooded bog and open bog, in coastal British Columbia, Emili and Price (2013) observed how slope position interacted with event and seasonal water availability, influencing the concentration and composition of major ions.

Building on the field experiment of Hoag and Price (1995), McCarter and Price (2017b) performed a field-scale solute transport experiment in a ladder fen by releasing a simulated chemical load of wastewater treatment plant effluent. At the study fen there was preferential water and solute transport through the low-lying regions in peat ridges that bisect pools in ladder and ribbed fens. Over 90% of the solutes were transported through the low-lying regions and the high hydraulic conductivity near-surface peat (McCarter and Price 2017a, 2017b). Balliston et al. (2018) explicitly demonstrated the spatial correlation between solute movement and peatland microtopography, with preferential transport occurring along hollows at a domed bog in northern Ontario. These preferential flow paths significantly increase the downgradient transport of reactive solutes (McCarter et al. 2017b). In the same experiment as McCarter and Price (2017b), McCarter et al. (2017b) observed a rapid removal of nitrogen, ammonium, and phosphate from the pore-water but sulphate was exceptionally mobile (similar to that of sodium). Despite these advances in understanding reactive solute transport, there are still research gaps that require further investigation, particularly integrating the mechanisms of solute transport in peat, which have been explored at the pore- and core-scale, with processes that have relevance at the field- and landscape-scale. Given the complexity of peat as a medium that exhibits compressibility, a dual-porosity structure, mixed-wettability, and is frequently unsaturated (and frozen), this is not a trivial task.

Iii) the role of Ground-Ice in wetland hydrology

Frozen ground is a key factor influencing soil temperature and moisture, subsurface hydrology, rooting zones and nutrient cycling of wetlands (Smerdon and et al. 2005; Tarnocai 2009). Frozen ground is ubiquitous in Canada's Arctic, Subarctic and northern Temperate zones where it occurs seasonally or in the form of permafrost. Arctic wetlands occur in the zone of continuous permafrost, and seasonal water exchanges are constrained to a relatively thin supra-permafrost layer. In the Subarctic, discontinuous permafrost results in more complicated patterns of groundwater exchange (Hayashi et al. 2004), while in temperate wetlands, where frost is seasonal, the hydrological impact of frost is limited to the annual snowmelt runoff event (Woo 1986). Arctic wetlands often contain peat, but the depth of accumulation is typically too thin to meet the definition of a peatland (i.e. $>40\,\mathrm{cm}$ depth; NWWG 1997). Elsewhere in Canada, research on the role of ground ice in wetlands have focused on peatlands, in part because of their high hydrological sensitivity to the presence of ground ice, and in part because of the role of peatlands in promoting ground frost, especially permafrost (Brown 1963).

In the Canadian Arctic, wetlands cover about 3–5% of the land area (NWWG 1988). Woo and Young (2006) noted that Arctic wetlands range in size from small, isolated patches to regional ($>100\,km^2$) wetlands. Patchy wetlands are more common in the High Arctic and include wetlands below snowbanks (Lewkowicz and Young 1990), groundwater-fed wetlands (Young and Woo 2000), valley bottom wetlands (Rydén 1977), riparian wetlands (Young 2008), tundra ponds (Woo and Guan 2006), and wetlands associated with lakes (Winter and Woo 1990). Extensive wetlands are found in low-lying and perennially wet terrain including sedge-moss meadows (Abnizova and Young 2010), ice-wedge polygon fields (Young et al. 2010), and deltas (Marsh and Hey 1989).

In contrast to the wetlands of other regions, Arctic wetlands are typically underlain by permafrost (NWWG 1988) and, for most of the year, are covered by snow (Church 1974). Where permafrost is present, it can prevent or restrict hydrological interaction between surface and groundwater systems (Brandon 1962). Moreover, the terrain above permafrost has a unique suite of geomorphic features that influence the hydrology of Arctic wetlands. These include frost cracks, ice-wedge polygons, ice-cored mounds and other types of cryogenic patterned ground, as well as thermokarst terrain caused by permafrost thaw-induced ground surface subsidence (Woo and Young 2006). At the end of the Arctic winter, snowmelt releases at least half the annual precipitation (Woo 1986). Because the water table is typically close to the ground surface when wetland surfaces begin to freeze at the end of summer (Landals and Gill 1973), when meltwater is released at the end of winter it is largely restricted from infiltrating the ground (Woo 1986). Instead, the meltwater refreezes at the base of the snowpack, forming basal ice. Following the removal of the snow and ice cover from wetlands, the upper surface of the frozen, saturated layer of ground defines the relatively impermeable frost table which descends through the active layer as it thaws. The water table and frost table represent the upper and lower boundaries of the thawed, saturated portion of the active layer which conducts the large majority of subsurface runoff through wetlands (Quinton and Hayashi 2004). Since the horizontal hydraulic conductivity in the active layer decreases by several orders of magnitude with increasing depth (Quinton et al. 2000), subsurface flow rates decrease as the depth of thaw increases. However, as noted above, even when the active layer is fully thawed, hydrological interaction between active layer (i.e. supra-permafrost) water and sub-permafrost groundwater is precluded or restricted, with some exceptions.

Runoff over wetland ground surfaces is most prevalent at the end of winter when the snowmelt water supply is greatest, and the relatively impermeable frost table is close to the ground surface. The duration of overland flow is governed by the snowmelt water supply, rate of ground thaw, and degree of hydrological connection with upslope water sources (Woo 2012). Unlike the wetlands of other regions, those of the Arctic do not normally receive significant precipitation following the spring freshet. In the high Arctic, frequent fog and low cloud deposit trace rainfall in summer, while at lower latitudes, summer rain events can provide greater input (Prowse 1990).

At the hillslope scale, the distribution of surface and subsurface runoff in Arctic wetlands can be strongly affected by the unique geomorphic features of permafrost referred to above, and their associated type and distribution of vegetation. Examples include preferential flow around frost mounds (Hodgson and Young 2001), along frost cracks and ice-wedges (Harp et al. 2020), 'inter-tussock' and 'inter-hummock' (Quinton and Marsh 1998) channels, and along 'water tracks' (Price and Maloney 1994). At the basin scale, Obradovic and Sklash (1987) reported that saturated source areas for overland flow shrink soon after the spring freshet, leaving subsurface flow as the dominant runoff mechanism during summer. However, Lewkowicz and Young (1990) found that many wetlands continue to function as runoff source areas long after adjacent terrains stopped producing runoff. Such wetlands are often located downslope of late-lying snowpacks which continue to supply water well into the summer period (Lewkowicz and Young 1990). Roulet and Woo (1988) found that the spatial integration of runoff producing areas was achieved not gradually but abruptly once the water storage threshold of basin hydrological units were exceeded.

Although evapotranspiration rates decrease with increasing latitude due to lower energy availability and a higher proportion of non-vascular plants, Prowse (1990) reported that the relative importance of ET to the annual water balance increases since precipitation rates decrease more rapidly with latitude. Brown et al. (1968) found that approximately 50% of rainfall on Arctic regions evaporates, and Roulet and Woo (1986) reported that 66% of the total annual precipitation (rain and snow) evaporated at Baker

Lake, N.W.T. During summer, the deeper frost and water tables, and the increased energy available for ET decreases the amount of water available for surface or subsurface runoff.

Subarctic and high boreal regions of Canada have extensive areas of peatland-dominated terrain that is closely associated with the occurrence of discontinuous permafrost (Brown 1964). While frozen soils can promote wetland occurrence by keeping the water table close to the surface (Roulet and Woo 1986b), surficial peat can also initiate permafrost development and help preserve it. Peat efficiently conducts energy towards the atmosphere during winter when it is saturated with ice, while thermally insulating the ground from gaining energy during summer when the surficial peat is relatively dry (Brown 1963). In the zone of sporadic or discontinuous permafrost, seasonal frost is ubiquitous, thus many of the peatlands of this region demonstrate similar responses to the annual snowmelt event since the snowpack melts well before the seasonal ground frost is thawed (Woo and Winter 1993). However, as the seasonal frost and snowmelt moisture supply diminish, the contrast of hydrological functioning among organic terrain types increases and is particularly evident between permafrost and permafrost-free terrains.

In continental western Canada, discontinuous permafrost is generally restricted to bog peatland areas (Vitt et al., 2018), especially peat plateaus. Permafrost-free peatlands, typically include collapse scar wetlands and channel fens (Holloway and Lewkowicz 2020). Peat plateaus rise 1-2 m above their adjacent permafrost-free peatlands (Vitt et al. 1994) and are therefore relatively well-drained and possess well-developed unsaturated layers, while the water table of adjacent peatlands remains at or near the ground surface. The relatively dry peat plateaus may not have met the early definition of "wetland" (Tarnocai 1980), but the current definition using the Canadian Wetland Classification System (NWWG 1997) simply requires the surface be sufficiently wet to induce hydric soils and vegetation adapted to a wet environment. The presence of peat attests to this. The relatively dry peat plateaus support a tree cover, while adjacent wetlands are largely treeless (NWWG 1988). The unsaturated layer of peat plateaus acts as an effective thermal insulator that promotes permafrost even where the mean annual air temperature is above 0° C (Camill 2005). The elevation of the upper surface of permafrost is greater than the water table of the adjacent wetlands and for this reason, peat plateaus are often referred to as '*permafrost dams*' since they can effectively impound wetlands (Kurylyk et al. 2016; Quinton et al. 2019).

Peat plateaus and collapse scar wetlands (hereafter 'collapse scars') are typically arranged into distinct plateau-wetland complexes separated by channel fens (Aylsworth et al. 2000). Peat plateaus follow a continuous cycle of development initiated by ice bulb formation and displacement of the wetland ground surface, and eventually, decay driven by thermokarst processes (Zoltai 1993). In a stable climate, this cycle transforms tree-covered plateaus into treeless, permafrost-free collapse scars, and back into plateaus over a period of centuries (Treat and Jones 2018). Permafrost below mature plateaus is on the order of 10 m thick (McClymont et al. 2013) with nearly vertical edges (Hayashi et al. 2004). Plateaus function primarily as runoff generators, with water conveyed mainly through the thawed, saturated layer separating the water table from the relatively impermeable, sloping frost table (Wright et al. 2009). Peat plateaus have two distinct runoff source areas (Connon et al. 2014). Primary runoff drains the sloped edges of plateaus directly into the basin drainage network (i.e. channel fens or stream channels) throughout the thaw season. Secondary runoff is neither direct nor continuous (Connon et al. 2015) as it enters the drainage network indirectly through an intervening wetland or wetlands. Since collapse scars are surrounded by raised permafrost, their hydrological function is typically considered to be one of water storage (Quinton et al. 2003). Fens collect water from their adjacent plateau-wetland complexes and route it along their broad, hydraulically rough channels (Hayashi et al. 2004) in a manner consistent with roughness-based algorithms (Kurylyk et al. 2016). Because each of the major peatland types have characteristic hydrological functions, basins with different proportions of these peatland types produce different hydrograph responses (Connon et al. 2015), and likewise, widespread transformations of one peatland type to another within a single basin has the potential to alter the basin hydrograph.

The majority (>85%) of the energy flux conducted vertically into the plateau peat profile is partitioned toward melting ice by lowering the frost table (Hayashi et al. 2007). During ground thawing, the frost table separates the thawed and frozen portions of the active layer, and closely approximates the zero-degree isotherm (Woo 1986). The depth dependency of hydraulic conductivity and the nearly impermeable nature of the frost table makes the degree of active layer thaw a primary factor controlling the rate of subsurface flow. However, spatial variations of thaw depth result in a frost table 'topography' that varies over time and space, and as a result, the rates and directions of subsurface flow also change over time and space (Wright et a., 2009). Water drains toward frost table depressions, leading to local areas of increased wetness. Since wet peat conducts energy from the ground surface to the frost table with greater efficiency than drier peat (Hayashi et al. 2007), localized wet areas are also areas of preferential ground thaw, and such wet areas tend to expand and coalesce over the summer period as thaw depths and therefore local hydraulic gradients increase (Ackley et al. 2021). The rates and directions of subsurface drainage vary over time as the active layer thaws, a process which re-distributes soil moisture and nutrients, while influencing vegetation cover (Van Huizen and Petrone 2020; Waddington et al. 2015). Further, work by Van Huizen et al. (2020) has shown that in non-permafrost peatlands, seasonal ground ice has significant energy and drainage implications, which affect ET during the snowmelt and snow-free periods.

iv) hydrological drivers of wetland biogeochemistry

Wetland biogeochemistry research in Canada over the past 75 years has been extensive and worthy of a lengthy review on its own. Therefore, here we focus on hydrological drivers of wetland nutrient retention and dissolved organic carbon export, and the response of carbon and nutrient cycling to water table manipulation. Carbon storage and greenhouse gas exchange are important ecosystem functions performed by wetlands, with the large stocks of carbon accumulated in wetland soils, mainly peatlands, providing climate-regulating services (Helbig et al. 2020a; Roulet 2000). Further, wetlands act as hotspots for biogeochemical cycling (Cheng and Basu 2017), often resulting in nutrient retention (e.g. Devito and Dillon 1993) and hydrological export of dissolved organic carbon (DOC) (e.g. Fraser et al. 2001a; Moore 2009; Strack et al. 2008; Waddington and Roulet 1997). Hydrological conditions, including hydroperiod, mean and variability in water table position, and soil moisture content, control the rate of wetland biogeochemical cycling, both as drivers of plant productivity and microbial activity. Wetland connectivity and total discharge are also key drivers of nutrient and carbon exports. It is then not surprising that many studies of wetland biogeochemistry include hydrological measurements and vice versa (Price and Waddington 2000).

Wetland nutrient retention and DOC export

By the 1970s wetlands were identified as potential sites for nutrient retention; however, few nutrient budgets had been conducted. Once measurements of both inorganic and organic forms of nutrients were considered, it was noted that the role of wetlands in nutrient retention was more complicated. Wetlands may retain nutrients in some forms while exporting others (Rutledge and Chow-Fraser 2019), indicating their role as sites of nutrient transformation. The pathway of water flow through the wetland also affects retention (Hill 1993). For example, Gehrels and Mulamoottil (1989) observed net retention of total phosphorus (TP) in a Typha marsh in south-western Ontario, but groundwater exported TP while surface flow resulted in retention; orthophosphate was exported from the site. Similarly, although wetlands are generally sinks for inorganic N, they can be sources of organic N with variation in space and time depending on hydrological connections within the wetland and with the catchment (Devito et al. 1989). Further, wetland connectivity to the stream may be important for predicting nutrient export. Casson et al. (2019) observed that near-stream wetland area was a better predictor of NO_3 export than total catchment wetland area across 10 forested catchments on the Canadian Shield, suggesting that connection to the stream was important; however, total phosphorus and DOC export were better predicted by total wetland area. Overall, during low flow periods wetlands tend to act as sites for nutrient retention while export is more likely to occur as discharge increases (Devito et al. 1989; Devito and Dillon 1993). Therefore, snowmelt is generally a period of nutrient export from wetlands (Burd et al. 2018; Eimers et al. 2009). That said, the presence of wetlands (or other storage features) can reduce total catchment nutrient export during high flow periods by reducing peak flow (D'Amario et al. 2021).

The role of wetlands as hotspots of biogeochemical transformations as mediated by hydrology (Lam et al. 2022) is further illustrated by work on sulphur and mercury cycling. The transformation of inorganic mercury into methylmercury (MeHg), the far more mobile and bioavailable form, is conducted by sulphate-reducing bacteria and is thus closely linked to redox conditions that will vary with water table fluctuations and inputs of organic substrates and ions (Branfireun et al. 1999; Branfireun and Roulet 2002; Mitchell et al. 2008). Groundwater discharge areas in wetlands can provide a source of sulphate; thus, higher methylmercury porewater concentrations have been observed where groundwater recharge predominates in both temperate Ontario wetlands and boreal sites in the Northwest Territories (Branfireun et al. 1996; Branfireun and Roulet 2002; Gordon et al. 2016). However, hydrological flowpaths will also affect interactions of sulphate-rich water from upland (Devito and Hill 1997; Mitchell et al. 2008) or anthropogenic (McCarter et al. 2017a) sources with organic rich peatland soils. Devito and Hill (1997) observed minimal sulphate reduction during high flow periods in swamps on the Canadian Shield as surface flow predominated and interactions with anoxic peat were limited, while McCarter et al. (2017a) observed higher MeHg concentrations within a point-source sulphate plume associated with flow through peat and lower concentrations within pools of a ladder fen in the James Bay Lowland. In addition to water source, water table fluctuation within wetlands contributes to sulphur and mercury cycling and subsequent export. Studies across temperate wetlands in Ontario have shown that water table drawdown results in increased sulphate availability through oxidation (Eimers et al. 2008; McLaughlin and Webster 2010) and this can contribute to higher rates of mercury methylation and thus export of both sulphate (Devito and Hill 1997; Eimers et al. 2007; Schiff et al. 2005; Szkokan-Emilson et al. 2013; Warren et al. 2001) and MeHg upon rewetting (Galloway and Branfireun 2004). McCarter et al. (2022) determined that hydrological flushing of a catchment-scale experimental atmospheric sulphate addition decreased surface water MeHg concentrations by ~25% once sulphate additions ceased, with the other 75% due to changes in net demethylation and MeHg sorption.

Several studies across Canada, from Temperate to Subarctic regions, have reported the strong predictive power of the proportion of wetland in the catchment for DOC export (Casson et al. 2019; Dillon and Molot 1997; Eckhardt and Moore 1990; Koprivnjak and Moore 1992; Li et al. 2015; Richardson 2012). Therefore, improving identification of wetlands within forested areas (i.e. ‘cryptic wetlands’ that may have similar vegetation to the surrounding uplands), can improve estimates of catchment DOC export (Creed et al. 2003). This clearly indicates the importance of wetlands as a source of DOC to streams and receiving water bodies (Hillman et al. 2004).

As with nutrient and metal export, hydrological conditions are also key drivers of DOC export from wetlands by controlling source areas for streamflow and total discharge. Discharge may be either positively (Fitzgerald et al. 2003; Hinton et al. 1998), negatively or not correlated (Hinton et al. 1997) to DOC concentration, with the difference linked to whether high flow conditions connect water sources from organic rich wetlands to the stream or increase contributions from mineral soil horizons (Schiff et al. 1997); rainfall and snowmelt events also dilute DOC concentrations (Emili and Price 2013). Regardless of changes to concentration in response to flow conditions, DOC export is dominated by high flow periods including snowmelt (Shatilla and Carey 2019; Waddington et al. 2008) and storm events (Hinton et al. 1997). Thus, drought conditions tend to limit DOC export (Eimers et al. 2008) and total discharge is an important control on total DOC export from wetland ecosystems (Fraser et al. 2001a; Moore 2003; Waddington et al. 2008)

Response of carbon and nutrient cycling to water table manipulation

Carbon accumulation in wetlands occurs largely due to the slow decomposition associated with the anoxic conditions that arise in saturated sediments or soils. Therefore, water table position is a strong predictor of net carbon exchange and methane emissions (e.g. Moore and Roulet 1993; Price and Waddington 2000). Studies using peat cores have shown strong relationships between water table position and C cycling with drier conditions resulting in higher CO_2 emissions, lower CH_4 emissions and higher pore water DOC concentrations (Blodau et al. 2004; Moore and Dalva 1993; Moore and Knowles 1989). Comparable results have been reported from sites affected by drainage in southern Québec and northern Ontario (Glenn et al. 1993; Roulet et al. 1993).

As non-permafrost wetlands in Canada are predicted to experience drier conditions under climate change (Helbig et al. 2020b; Roulet et al. 1992), field scale water table drawdown experiments have also been used to predict wetland responses to climate warming. Shifts in carbon cycling driven by drier soil conditions that enhance organic matter decomposition and methane oxidation are partially offset by changes to the wetland plant community (Munir et al. 2015; Strack et al. 2006; Strack and Waddington 2007; Waddington et al. 1998). These plant community shifts are also dependent on small-scale variations in initial moisture conditions within the wetland arising from microtopography; initially wet areas in moist temperate peatlands in southern Québec experienced an increase in plant productivity that maintained or enhanced both net C uptake and CH_4 emissions following water table drawdown (Strack et al. 2004;

Strack and Waddington 2007). In contrast, forested continental bogs in northern Alberta experienced an increase in tree and shrub growth in response to a lowered water table and resulting increase in nutrient availability (Munir et al. 2014, 2017), but this was not enough to offset soil carbon losses unless sites also experienced soil warming treatments (Munir et al. 2015).

v) wetland disturbance

Disturbance to Canadian wetlands occurs extensively through climate-mediated disturbances such as wildfire and permafrost thaw (Wilkinson et al. 2023), regionally through resource extraction activities such as oil and gas operations (Volik et al. 2020), and locally through drainage for forestry (Silins and Rothwell 1998), agriculture (Walters and Shrubsole 2003), urbanization (Birch et al. 2022), peat extraction (Price et al. 2003), and road construction (Bocking et al. 2017). Intensity of these disturbances range from complete loss of wetlands removed for open pit mining (Rooney et al. 2012), to loss or change in hydrological function caused by disturbance (Webster et al. 2015). The extent and impact of the change on ecosystem function depends on the nature of the disturbance and of the resilience of the peatland to change (Harris et al. 2020). Given that the vast carbon store in Canadian peatlands provide essential ecosystem services for global climate (Harris et al. 2022), the ecohydrological resilience of peatlands to disturbance thus impacts a range of spatial and temporal scales, often in complex manners (Morris et al. 2011). As such, the magnitude and direction of the response of peatlands to land-use and climate change are difficult to assess with confidence (Harris et al. 2022; Moore et al. 1998). Nevertheless, with respect to peatlands, the general persistence of many ecosystem functions, for example their ability to continue as a net carbon sink in response to disturbance, has generally resulted from the ability of peatlands to regulate their water content (McCarter et al. 2020; Waddington et al. 2015). This self-regulation allows peatlands to sustain a high degree of wetness that dampens system instabilities that could result in ecohydrological collapse (Kettridge et al. 2015), and release of globally important carbon stocks (Harris et al. 2022; Wilkinson et al. 2018). We refer to the interaction of peat hydrophysical properties (McCarter et al. 2020; Rezanezhad et al. 2016) and peatland ecohydrological feedbacks (e.g. Waddington et al. 2015) in resisting a change in peatland form and function (e.g. long-term carbon storage) as peatland ecohydrological resilience to disturbance (see Waddington et al. 2015).

Canadian researchers have made important scientific contributions to understanding the interaction of moss traits, peat hydrophysical properties, and ecohydrological feedbacks that maintain near-surface wetness of northern peatlands (e.g. McCarter et al. 2020; Rezanezhad et al. 2016; Waddington et al. 2015). Many of the water table depth (WTD)-ecohydrological feedbacks reviewed by Waddington et al. (2015) are connected to foundational Canadian hydrological research including those with: i) afforestation and/or shrubification feedback (Farrick and Price 2009; Landhäusser et al. 2003; Lieffers and Rothwell 1987; Moore et al. 2022), ii) moss surface resistance and albedo feedback (Kettridge and Waddington 2014; Lafleur et al. 2005; Price et al. 2009), iii) transmissivity feedback (Fraser et al. 2001b; McCarter et al. 2020), iv) peat deformation feedback (Kellner et al. 2003; Price 2003; Price and Schlotzhauer 1999), v) specific yield feedback (Price 1992), vi) peat decomposition feedback (Morris and Waddington 2011; Strack et al. 2005), and vii) moss productivity feedback (McCarter and Price 2014; Thompson and Waddington 2008). Canadian research has also highlighted that the negative feedbacks (which act to moderate water table changes) outnumber the positive feedbacks (which act to amplify water table changes) and thereby generally contribute to peatland ecohydrological resilience to disturbance. While the strength of the ecohydrological feedbacks varies by peatland type and climate (see Waddington et al. 2015 for review), recent research suggests that deeper peatlands may have stronger negative and autogenic ecohydrological feedbacks (e.g. Moore et al. 2021; Wilkinson et al. 2020a, 2020b) and by extension greater ecohydrological resilience to disturbance (Morison et al. 2020). This 'survival of the deepest' resilience depth (or range of depths) threshold concept where peatland depth may be a key indicator of peatland vulnerability to disturbance (e.g. Hilbert et al. 2000) is likely to vary with hydrogeological and hydroclimatic setting and disturbance type. With this ecohydrological resilience lens in mind we review Canadian peatland research on the resistance, resilience, and vulnerability of northern peatlands to the major disturbances, especially as caused by climate change, wildfire and peat extraction.

Permafrost thaw-induced disturbance

In Canada's Arctic and Subarctic regions, permafrost thaw is a leading cause of wetland disturbance owing to rising air temperatures since the 1970s (Biskaborn et al. 2019a, 2019b). The rates and patterns (Beilman and Robinson 2003; Nitze et al. 2018; Thie 1974) and mechanisms (Devoie et al. 2021; Gibson et al. 2018) of permafrost thaw-induced change to peatlands have been quantified, and new methods of detecting permafrost degradation using a combination of aerial photographs, satellite remote sensing and LiDAR were developed (Chasmer and Hopkinson 2017). Permafrost is particularly susceptible to rapid thaw and disappearance in the Subarctic (Richter-Menge et al. 2017) where it is relatively thin, and its temperature is already near the melting point. Thaw-induced fragmentation of plateaus accelerates thaw rates of the underlying permafrost (Chasmer and Hopkinson 2017), which involves simultaneous lateral recession of permafrost edges and lowering of the permafrost table (Devoie and Craig 2019). As permafrost thaws, the overlying plateau ground surface subsides (Smith et al. 2008), a process that ultimately removes the topographic gradient between plateaus and adjacent wetlands, inundates the plateau ground surface, and transforms it into a permafrost-free, treeless wetland (Camill 1999) or thermokarst lake (Sannel and Kuhry 2011). Since peatland form exerts a primary control on peatland hydrological function, such a transformation can affect water flow and storage processes within the transformed peatland, between it and adjacent peatlands, and potentially affect the basin hydrograph (Connon et al. 2018). Since permafrost thaw occurs to varying degrees throughout the circumpolar region (Kwong and Gan 1994; Romanovsky et al. 2010; Biskaborn et al. 2019a, 2019b), the resulting peatland transformations and hydrological impacts are potentially widespread (Hinzman et al. 2013; St. Jacques and Sauchyn 2009; Walvoord and Kurylyk 2016).

In Canada, the transition from plateau forest to permafrost-free forest has occurred in less than half a century, far faster than the process of forest re-establishment described by Zoltai (1993), which depends on permafrost regrowth. Canadian researchers have shown that the former involves several distinct stages that alter the ecohydrological environment and introduce new processes and feedbacks (Carpino et al. 2021). These include wetland capture (Connon et al. 2014), talik development (Devoie et al. 2019b), partial drainage of captured wetlands (Haynes et al. 2018), development of sphagnum hummocks and their colonization by tree saplings (Haynes et al. 2021) and coalescence of treed hummocks to form a continuous tree cover (Disher et al. 2021). However, each of these processes are subject to disturbances such as wildfires, which can alter ground thermal regimes (Smith et al. 2015) and accelerate permafrost thaw (Gibson et al. 2018).

Research in the southern Northwest Territories has shown that as the permafrost separating wetlands thaws, ephemeral channels form over the subsiding ground surface (Connon et al. 2014) enabling water to cascade from one wetland to the next as storage thresholds are exceeded. This wetland capture process effectively taps water stored in the interior of plateau-wetland complexes, thereby increasing the contribution of secondary runoff to the basin drainage network. In the Northwest Territories it was shown that wetland cascades often function as ecotones with ombrotrophic wetlands at their headwaters in the plateau-wetland complex interior, and minerotrophic wetlands lower in the cascade sequence (Gordon et al. 2016). Permafrost monitoring demonstrated a transition from stable to thawing permafrost after a threshold summer thaw depth of between 0.6 and 0.8 m is exceeded (Connon et al. 2018). It was also found that once a talik forms, the permafrost thaw rate increases five-fold, suggesting that talik formation is a 'tipping point' that accelerates thaw (Devoie et al. 2019b). The occurrence of taliks in Canadian peatlands has increased over the last decade (O'Neill et al. 2020) and they are now a common feature of high-boreal peat plateaus and particularly prevalent below linear (i.e. seismic) disturbances (Braverman and Quinton 2016; Smith and Riseborough 2010; Williams et al. 2013) and peatlands disturbed by wildfire in recent decades (Gibson et al. 2018). Unlike other hydrological pathways, taliks conduct water throughout the year. Permafrost thaw induced peatland transformation therefore introduces new hydrological pathways that connect to new runoff source areas and offers a plausible explanation for trends of increasing annual basin discharge initially reported by St. Jacques and Sauchyn (2009).

Effects of disturbance on evapotranspiration

Work on Canadian wetlands, especially in the Boreal zone, has shown that the effects of disturbance on ET, soil moisture distribution, lateral hydrological fluxes between landscape units, and feedbacks are complex and often extend beyond the visual boundaries of the wetland (Plach et al. 2016; Waddington et al. 2015). Early on, Woo (1992) demonstrated that long-term climatic warming will alter many physical attributes of wetlands, leading to earlier snowmelt, higher evapotranspiration, and lowering of the water table. However, the impacts will vary for wetlands in different parts of Canada (Woo 1992). For example, in subhumid regions, it is likely that peatland-upland forest hydrological interactions have allowed a peatland-dominated landscape to overcome available moisture limitations and develop and persist in a region of frequent and prolonged (multi-annual) drought periods (Devito et al. 2005)

In general, work has shown that the relationship between transpiration, soil moisture, and water table are controlled by vegetation and canopy conditions (van der Kamp and Hayashi 2009; Warren et al. 2018). These relationships form the basis of within-peatland (autogenic) feedbacks that control ecohydrological – climatic interactions (Blanken and Rouse 1996; Waddington et al. 2015) and are important to larger-scale climatic interactions and responses to external disturbance. Understanding this interaction is especially important, as hydrological conditions in peatlands change in response to climate or disturbance, where peatland tree cover can change in density and composition (increased productivity, encroachment, etc.) (Pellerin and Lavoie 2003; Ropars and Boudreau 2012). Tree density within a peatland influences surface roughness, determined by the structure of the forest canopy and individual trees (Strilesky and Humphreys 2012). Thus, stands with differing tree compositions (i.e. deciduous, coniferous) but the same height will interact with the wind field differently, because the level of active exchange with the atmosphere for each type of tree is different (Green et al. 2021, 2022). As peatland tree growth increases, surface roughness increases and aerodynamic resistance decreases (Moore et al. 2013), until the canopy becomes so dense that it becomes aerodynamically smoother (Waddington et al. 2015). As such, strong links are observed among water use efficiency, tree cover density, composition and age and water table dynamics, which creates the potential for drying due to the water table depth-afforestation feedback (Humphreys et al. 2006).

Understanding internal and external feedbacks and processes that permit wetland/peatland ecosystems to efficiently use available moisture while maintaining maximum productivity is important for managing these systems considering climate and land-use change (Rooney et al. 2015) and reclamation (Ketcheson et al. 2016). Literature suggests that mass and energy exchange are strongly influenced by local boundary layer conditions and biophysical controls (Plach et al. 2016; Solondz et al. 2008). Further, the landscape mosaic of peatlands and forests in much of the Boreal suggests that not only is there a hydrologic synergy between these landscape units, but that uplands provide a mechanism to moderate evapotranspiration losses in the high atmospheric demand climate (Brown et al. 2010). Landscape heterogeneity, ground surface morphometry, tree density and height, and orientation of forested hummocks (with respect to aspect and dominant wind direction) shelter adjacent peatlands limiting their evapotranspiration (Green et al. 2021, 2022).

Wildfire

Wildfire represents the largest areal direct disturbance of Canadian peatlands (Turetsky et al. 2002) and Canadian wildfire and wetland scientists have made significant contributions to the understanding of the ignition, combustion vulnerability (Waddington et al. 2012) and ecosystem recovery of peatlands. While much of this research has been carried out in the subhumid Boreal Plains of Alberta, all these processes are directly and indirectly controlled by ecohydrological processes (Lukenbach et al. 2016; Wilkinson et al. 2019) and the peatland water balance (Elmes et al. 2018). Peat fires are dominated by smouldering combustion (Frandsen 1987; Thompson et al. 2015a) and the propagation of smouldering combustion is controlled by the ratio of energy sink to fuel source, which can be approximated with gravimetric water content (GWC) (Benscoter et al. 2011; Lukenbach et al. 2015a). As such, peat wildfire combustion is controlled by the cross-scale variability in peatland water balance (Elmes et al. 2019), hydrological connectivity (Hokanson et al. 2016, 2018), peat hydrophysical properties (Thompson and Waddington 2013a) and moss species (Wilkinson et al. 2019). In general, natural peatlands have low ignition and peat combustion potential given their ability to maintain a wet near-surface and the presence of generally low-density surface mosses, litter, and peat. (e.g. Benscoter et al. 2011; Lukenbach et al. 2015a). However, peat combustion in some peatland types (e.g. peat swamps, peatland margins and drained peatlands) can be very high to extreme (e.g. Granath et al. 2016) as these peatlands are not only denser (greater fuel source) but are also drier during periods of drought as manifested through the WTD-specific yield feedback (Elmes et al. 2018; Nelson et al. 2021; Wilkinson et al. 2018). Hokanson et al. (2016) found that peat burn severity was higher in Boreal Plains peatlands with lower groundwater connectivity suggesting that peatland hydrogeological setting can be used to identify peat smouldering hotspots (Hokanson et al. 2018; Wilkinson et al. 2019).

The resilience of peatland form and function to wildfire is ultimately determined by the ability of burned peatlands to recover the carbon lost from combustion within the fire return interval (Ingram et al. 2019). While deep peat burning has generally been shown to be counterbalanced by rapid moss recolonization in Alberta peatlands, as the post-fire peat surface is closer to the water table and has higher moisture content and lower soil water tension (Lukenbach et al. 2015b). Kettridge et al. (2015) found that extreme burn severity in the Salteaux peatland near Slave Lake, Alberta can result in an ecohydrological regime shift. As such, Canadian researchers have determined that peatland ecohydrological resilience to wildfire is non-linearly controlled by the relative change in peatland water balance and peat hydrophysical properties (Lukenbach et al. 2017; Nelson et al. 2021) due to changes in peat burn severity (e.g. Sherwood et al. 2013). For example, while the net change in water storage can be largely unchanged by wildfire (Thompson et al. 2014) the variability in peatland water table generally increases following wildfire due to the burning of low-density, high specific yield surface peat (Thompson and Waddington 2013b). Evapotranspiration tends to dominate water loss from northern peatlands (Lafleur et al. 2005; Petrone et al. 2007), and although the available energy for evapotranspiration increases post-fire (Thompson et al. 2015b) the removal of the canopy and understory restricts transpiration causing evaporation to dominate post-fire ET. Nevertheless, evapotranspiration increases only marginally following wildfire (Thompson et al. 2014) due to the capability of a burned peatland to regulate ET following wildfire - a key control on peatland ecohydrological resilience.

Following wildfire, moss and near-surface peat often develop fire-induced hydrophobicity (Elmes et al. 2019; Kettridge et al. 2014; Moore et al. 2017). Post-fire hydrophobicity has been found to increase with decreasing moisture content, highlighting the importance of moss moisture retention properties (Moore et al. 2017). Sphagnum moss moisture content, especially hummock-forming species such as *Sphagnum fuscum* (McCarter and Price 2014), tends to be comparatively high, leading to lower burn severity (Hokanson et al. 2016), very low to no water repellency (Moore et al. 2017) and fast post-fire recovery (i.e. ecohydrologically resilient) (Lukenbach et al. 2017) relative to feather mosses (Kettridge et al. 2014). In peat and moss, where moisture content is low following wildfire, the post-fire hydrophobic layer can act as a barrier to the upward transfer of water for evaporation (Kettridge et al. 2017) with surface resistance to post-fire evaporation and at the peatland-scale correlated non-linearly with near-surface tension (Kettridge et al. 2021). This hydrophobicity–evaporation feedback (Wilkinson et al. 2020a, 2020b) promotes moss recovery (Lukenbach et al. 2017) and ecohydrological resilience (Kettridge et al. 2017, 2019). However, in peatlands with high burn severity where hydrophobicity is low (Wilkinson et al. 2020) or even decreases in the uppermost layer due to wildfire (Elmes et al. 2019), this water conservation feedback breaks down (Wilkinson et al. 2020). As such, extreme peat burn severity due to drainage (Granath et al. 2016; Wilkinson et al. 2018) or to future peatland drying (Helbig et al. 2020b) enhances post-fire drying and reduces ecohydrological resilience (Kettridge et al. 2015) to wildfire. McCarter et al. (2021) found that the depth of peat extraction (comparable to different depths of burn, i.e. the removal of peat from the profile from extraction or burning) led to divergent post-fire soil water conditions in the Wainfleet Bog in southern Ontario. Therefore, on abandoned extracted peatlands sphagnum recovery and peat re-ignition potential is altered. This highlights the complex interactions multiple disturbances (drainage, extraction and wildfire) have on peatland ecohydrology.

Peat extraction and restoration

Canadian research has highlighted that while peatlands are generally ecohydrologically resilient to single disturbances, multiple compounding disturbances (e.g. drainage and wildfire, drainage and harvesting) often lead to a loss of peatland function and regime shift (Kettridge et al. 2015; McCarter et al. 2021; Sherwood et al. 2013; Waddington et al. 2002). Following these ecohydrological tipping points, active peatland restoration or reclamation is often required (Poulin et al. 2005). Restoration of peatland functions following peat extraction can return them to carbon sinks (Nugent et al. 2018) and has the potential to mitigate carbon loss from wildfire combustion (Granath et al. 2016; McCarter et al. 2021).

Peat extraction began in Canada in the 1940s (Daigle et al. 2001), but prior to a publication by Keys (1992) there was little or no documented recognition of the need for restoration, nor any systematic attempt or method for doing so. In Canada, peat was initially extracted for use as a fuel, but from the 1900s onward has primarily been for its use as a horticultural amendment (Warner and Buteau 2000). A major challenge for restoration practitioners is that extracted peatlands present a paradoxical scenario where the peatlands, in which peat was extracted to be used as a growing medium, are inhospitable to the reestablishment of peat-forming vegetation (Price et al. 2003), notably sphagnum mosses (Rochefort 2000). The switch from manual block-cutting to mechanical harvesting in the 1970s increased the hostility of the extracted surface to plant re-establishment because of the harsher hydrological environment with respect to plant water-supply, distance from propagules, deeper drainage ditches and fewer ecological niches (Price et al. 2003). Very little work was completed on peatland restoration in Canada until the 1990s when the Peatland Ecology Research Group was formed, based out of Université Laval, which comprised an interdisciplinary team of scientists (notably Line Rochefort, Jonathan Price, and Mike Waddington). Over the next decade their research grew in scale from plot and lab experiments, culminating in the ecosystem-scale restoration of the now well-documented ~8 ha Bois de Bel (BDB) peatland in Québec, and with it the development of the Moss Layer Transfer Technique (MLTT) (Rochefort et al. 2003). It was reasonable to assume before trials began that maintaining adequate soil moisture (and water table) was important, thus various efforts to reduce hydrological stress at the site, such as ditch-blocking (Price 1997), pond creation (Price et al. 2002), field reprofiling (Bugnon et al. 1997), and bund creation (Shantz and Price 2006), were employed with varying degrees of success. The key hydrological discovery was that rewetting (the water retention strategies) alone was not enough. This is because despite blockage of ditches, the frozen cutover surface quickly shed snowmelt water because surface water detention was low (Shantz and Price 2006) and late spring and summer evaporative demands were high (Petrone et al. 2001; Price 1997). To address this, straw mulch was used to cover donor moss, which helped increase the surface albedo (to reduce solar heating) and relative humidity (to lower the vapour pressure deficit) at the boundary layer (Price et al. 1998), making the surface more hospitable for moss regeneration. At the same time, Price and Whitehead (2001), using a nearby block-cut peatland in Québec that had some natural peatland revegetation present, established that sphagnum mosses occupied areas only where soil-water pressures remained higher than −100 cm, which is now a key metric for restoration practitioners. Van Seters and Price (2001) found that runoff was significantly higher than a nearby natural peatland, highlighting the need for active restoration efforts (e.g. ditch blocking) to restore the hydrological function of the site.

In the intervening two decades since its restoration, researchers continue to study BDB, and other sites, to refine the MLTT, which is now used broadly in North America (see Gonzalez and Rochefort 2019) to restore bog peatlands. Recent work at BDB has discovered a hydraulic disconnect between the newly established moss carpet and the underlying remnant peat, such that a capillary barrier restricts water from reaching the moss capitula at the surface (McCarter and Price 2015). Recent studies have investigated squishing or compressing the moss layer to reduce pore sizes to weaken the capillary barrier effect (Gauthier et al. 2022). Golubev and Whittington (2018) quantified how artificial compression increased the unsaturated hydraulic conductivity, while simultaneously decreasing the saturated hydraulic conductivity. Given the broad effectiveness of MLTT for returning peatland function, its application as soon as possible after peat extraction operations cease can drastically decrease the time for the restoration site to become a net greenhouse gas sink (Nugent et al. 2019). MLTT has now also been applied for the restoration of other disturbances such as well-pads in the Alberta oil sands region (Engering et al. 2022) and roads in Québec (Pouliot et al. 2021) following appropriate site preparation.

While the MLTT has had success for bog restoration, attempts to modify the methodologies for fen restoration are underway (Hawes 2018; LeBlanc et al. 2012; Lobreau n.d.; Malloy and Price 2014). As bogs succeed from fens, peat companies will often stop extraction at a site when the depth to fen (sedge) peat is reached; as such, restoration towards fen might be more appropriate. As fens also have a more complicated hydrology than bogs, one such consideration being used is to create artificial ecotones between the peat extraction site and the surrounding natural peatland or upland forest to encourage the flow of water and nutrients from the surrounding area to support the restoration site (Yamoah 2023). The importance of natural ecotones, also called laggs, has received increased study recently as well (Howie and Van Meerveld 2011; Paradis et al. 2015) and highlight how these zones have distinct hydrological and hydrochemical gradients (Langlois et al. 2015a, 2015b, 2017) that support specific plant communities.

Not all peatland restoration projects in Canada have used the MLTT, since various other land uses (such as for agriculture or forestry) have occurred. One such site, the ~3000 ha Burns Bog in lower mainland BC (Howie et al. 2009a, 2009b; Howie and Van Meerveld 2011), has a rich history of disturbances and has made notable progress in its restoration, which has largely focused on rewetting from hand-built dams (nearly 500), a 100 m long underground wall (sheet piling) at the bog edge, and removing tree seedlings after an area of the bog burned. Peatlands are also impacted by 'natural' disturbances, such as beaver activity, which have required researchers to consider fen restoration as part of landscape management (Westbrook et al. 2017).

While much of the hydrological restoration literature in Canada has been focused on peatlands, important advances in the hydrology of mineral wetland restoration, focusing mostly on Prairie sloughs, also exists. The mentality that 'a drained wetland is a good wetland' was prevalent across much of western Canada as settlement/agriculture moved westward with estimates that 70% have been lost (Lands Directorate 1986), the most impacted wetland being the iconic prairie pothole or slough. These wetlands, that cover the southern parts of the provinces of Alberta, Saskatchewan and Manitoba (and USA states of North and South Dakota and Iowa), are home to ~50-70% of North American waterfowl (Smith et al. 1964) and represent an important carbon store (Euliss et al. 2006). Despite this, these systems are still undervalued by society, and as such, understudied (Badiou et al. 2011). Therefore, the need for pothole restoration has gained traction in recent years; in particular, for their role in both flood and drought protection and prevention (Goyette et al. 2023). Spence et al. (2022) note that the drainage of these wetlands increases their hydrological connectivity in the landscape, and thus how they contribute to runoff during rain and snowmelt events. Goyette et al. (2023) used HYDROTEL (a semi-distributed hydrological model) to test wetland restoration scenarios on peak and low flows. While they note that their findings varied widely among sub-watersheds, they conclude that increasing wetland coverage between 20 and 150% is needed to combat climate change impacts. Bortolotti et al. (2016) found that within a decade following restoration, water chemistry, macroinvertebrate and submersed aquatic vegetation communities closely resembled nearby natural wetlands in Southern Saskatchewan but note that protecting existing intact wetlands would be a better strategy. Similarly, Goyette et al. (2023) state that the 'no-net-loss' policy is not sufficient, and to maintain current hydrological cycles, a 'net gain' is needed.

Mining and peatland reclamation

Unlike wetland restoration, which seeks to return original functions to a degraded system, reclamation involves the creation of a completely designed and constructed landscape that nevertheless must account for landscape features and climate, that ideally achieve a suite of natural wetland functions common to the targeted wetland class (Daly et al. 2012). This is distinct from 'treatment wetlands' that are designed to treat non-point source pollution or low-flow wastewater, which are less-common in Canada compared to USA because of challenges associated with cold-weather performance (Kennedy and Mayer 2002). Prior to 2010, there was a dearth of well-documented wetland reclamation in Canada, although Whitelaw et al. (1989) developed guidelines for swamp reclamation on abandoned farmland based on a critical analysis of literature. They suggested that manipulating hydrology through removal or disabling of tile drains is preferable to engineered water-level control structures. More recently, wetland reclamation in Canada has become a prominent discipline, primarily because of initiatives directed at achieving 'equivalent land capability' for wetland-dominated terrain (OSWWG 2000) on post-mined oil sands landscapes.

Rooney et al. (2012) identified the loss of peatland from the Athabasca Oil Sands Region (AOSR) in Alberta, suggesting their reclamation is severely hindered by the altered topography of the post-mined landscape. While open water wetlands or marshes are easier to build (Alberta Environment 2008), their hydrological, biogeochemical and ecological function is entirely different than peatlands that comprised the pre-disturbance landscape (Rooney et al. 2012). Moreover, their natural occurrence in AOSR is only about 8% of wetland area (Ridge et al. 2021). Given that the majority of the AOSR is covered by peatlands, predominantly groundwater-fed fens (Vitt et al. 1996), these became the focus of reclamation research. The first conceptual design was proposed by Price et al. (2010), who used a numerical groundwater model to determine the optimal geometry and requisite material hydraulic properties of an upland to sustain fen hydrology under extreme drought conditions. This conceptual model demonstrated the theoretical viability of fen construction, resulting in a mandate to test the design on a pilot scale. The relatively simplistic design of Price et al. (2010) was adopted by Suncor Energy who completed the construction of

Nikanotee Fen Watershed (NFW) in January 2013, using salvaged and mine waste materials including peat and fine forest-floor soil stripped from areas designated for expansion of the mine, and coarse tailings sand. Simultaneously, Syncrude Canada Ltd. built the Sandhill Fen Watershed (SFW), which attempted to mimic the complex synergistic water sharing between upland and peatland landforms as observed in the Utikuma Lake region ~ 250 km southwest of AOSR (e.g. Ferone and Devito 2004; Petrone et al. 2008; Smerdon et al. 2005). Ketcheson et al. (2016) provide a comparative overview of NFW and SFW and note that although replicating the complex landscape interaction that inspired SFW should be a reclamation goal, the simplicity of NFW aided monitoring and modelling of fundamental hydrological processes, which was advantageous given the experimental nature of these watersheds. Ultimately, the desire is to develop the knowledge to allow for the consistent construction of carbon-accumulating ecosystems that are resilient to normal climatic stress, and support vegetation assemblages similar to the undisturbed boreal landscape.

The NFW relied on groundwater from an upland aquifer to support a high and stable water table in the downgradient fen. Due to this design, where water was shared in a largely unidirectional manner, the trajectory of the fen was intrinsically tied to the development of the upland and so could not be examined in isolation. Ketcheson et al. (2017) concluded that NFW design could sustain appropriately wet conditions in the fen despite lower than anticipated upland recharge in the early years post-construction. Upland recharge increased over a 4-year period following construction owing to weathering of the fine cover soil that overlies tailings sand (Sutton and Price 2020a, 2020b), one of several time-dependent processes that steered system development in the short-term. However, over the long-term, the recharge function of the upland was reduced by the growth of transpiring vegetation, although recharge was predicted to remain sufficient to drive fen function as the upland approaches a climax vegetation cover (Sutton and Price 2020a, 2020b). Fen evapotranspiration was only weakly tied to WTD (Scarlett et al. 2017), either because the water table was at or slightly above the surface, or because when drier, peat hydraulic properties retained water against drainage to the water table (Scarlett and Price 2019), making it available for ET. At SFW, Nicholls et al. (2016) also showed that wet conditions in the fen limited ET variability and was the dominant flux when the discharge control structures were not being artificially manipulated. The more complex design, active water table management, and greater watershed relief of SFW led Biagi et al. (2021) to conclude that the conditions were not favourable for fen peatland development as marsh-like conditions prevailed in the lowland area that were not favourable for peat accumulation and water conservation.

Water availability is not the only challenge that wetland reclamation in the AOSR must contend with, because residual solute pools associated with tailings sand (Simhayov et al. 2017; Biagi and Carey 2022), used for constructing uplands introduces water quality concerns. Kessel et al. (2018) estimated the arrival time of solutes (notably Na^{+}) to the fen was 4 to 11 years, with the greatest flux in summer driven by fen ET. Kessel et al. (2021) noted the dilution of salinity in the upland aquifer and fen caused by enhanced recharge in upslope recharge basins that provided a disproportionate contribution to total upland recharge. Biagi et al. (2019) found fen salinity to be notably elevated by evapotranspiration, which in summer is greater than precipitation (Scarlett et al. 2017), and has a particularly pronounced effect directly at the surface (Yang et al. 2022). Sutton and Price (2020a, 2020b) performed Monte Carlo simulations of future weather scenarios, which demonstrated the water balance of Nikanotee Fen generated sufficient flushing of salts to keep fen salinity below the stress-threshold of *Carex aquatilis*, currently the dominant plant species (Borkenhagen and Cooper 2019). Although future climate conditions introduced considerable uncertainty, the modelling indicated that fen salinity was likely to peak 20 years post-construction and decline thereafter. At SFW, the engineered control structures designed to suppress salinity were effective when used (but compromised the wetland hydrology); after the initial year they were used minimally, and electrical conductivity in the fen doubled in the following year, with enrichment being most profound at fen margins (Biagi et al. 2019).

Reclamation of upland landscapes can produce conditions suitable for the formation of unplanned 'opportunistic' wetlands, with much of the area having a water table within 50 cm of the surface (Wytrykush et al. 2012) and occurring on as much as 17% of a reclaimed upland (Hawkes et al. 2020). Little-Devito et al. (2019) found the hydrology suitable for the establishment of woody, swamp vegetation. The soil texture of uplands was found to influence the characteristics of these opportunistic wetlands. At sites with coarser deposits, groundwater discharge areas at the toe-slopes were common, whereas on finer deposits, wetlands were more isolated with little input from adjacent areas (Little-Devito et al. 2019). While to differing degrees both peatland and wetland reclamation produce novel systems (Nwaishi et al. 2015), it should be noted that thus far opportunistic wetlands lack many of the ecohydrological functions associated with pre-disturbance peatlands. Yet these systems are still in their infancy, as wetland development is a slow process that occurs on timescales of millennia. Although projections of constructed wetland trajectory are largely optimistic (Sutton and Price 2022), ultimately only patience and diligent monitoring will reveal the true development of these ecosystems.

Peatland reclamation of 'temporary' features present in the AOSR, such as seismic lines created for geologic exploration is also ongoing. Although these features retain peatlands on the landscape, the removal of the forest cover and compression of soils (Davidson et al. 2020) alter wetland function and these disturbances are widespread, affecting at least 1900 km^2 of bogs, fens and swamps in the province of Alberta (Strack et al. 2019). Widespread landscape disturbance has been identified as a key driver of the decline in woodland caribou populations, a species at risk in Canada (Alberta Environment and Parks 2017). Thus, actions to accelerate the recovery of seismic lines are being actively applied, particularly in peatland areas where trees are slow to regenerate (van Rensen et al. 2015). To date, reclamation actions are mainly applied to seismic lines crossing peatlands and largely involve silvicultural mounding treatments followed by tree planting (Dabros et al. 2018). Disturbance and inversion of the peat profile during these activities results in significant changes to soil hydrophysical properties, increasing bulk density and soil water retention (Davidson et al. 2020; Kleinke et al. 2022), with consequences for tree growth under investigation. However, testing of new mounding methods that retain the structure of the surface peat show promise for retaining hydrological function, peatland plant communities and accelerating the return of canopy cover (Kleinke et al. 2022).

vi) modelling: understanding processes and prediction

Canadian hydrologists have employed and developed increasingly sophisticated numerical models to identify the crucial role wetlands have within the broader landscape, characterize important processes, and quantify sensitivity to disturbance. Studies have explored a range of spatial scales, from the large wetland-dominated drainage basins of Canada down to a collection of individual pores. As computing power has become less of a limitation and recognition of the significance of peatlands permeates across disciplines, the breadth, depth, and number of modelling studies has, and will continue to be, on an upward trajectory.

A common theme in large-scale models of Canadian watersheds is the recognition of the essential role that wetlands perform in regulating the patterns of streamflow, ultimately emphasizing that they cannot be ignored. In the Liard River watershed (277,100 km^2), which originates in the Yukon, wetland-dominated catchments required a separate model structure to represent the 'fill-and-spill' behaviour that promotes a characteristically flashy hydrograph response to snowmelt and high-magnitude precipitation events (Brown and Craig 2020). This was accomplished by implicitly mimicking the surface detention capacity and marked decline in hydraulic conductivity with depth, which is typical of peat (Quinton et al. 2008). Wright et al. (2009) demonstrated this process more explicitly at Scotty Creek, NWT (also within the Liard River watershed); however, in their model the detention storage that needed to be exceeded to produce flow was located in subsurface depressions in the frost table. Peatlands also needed to be explicitly included in a model of the Athabasca River basin (95,300 km^2) of Alberta to achieve adequate matches between the simulated and observed water balance due to their strong internal water conservation mechanisms that increased downstream water availability through a reduction in evapotranspiration (Hwang et al. 2018). Similarly, the pronounced reduction in evapotranspiration during dry (low water table) conditions due to peatland self-regulation was a crucial feature to represent in a groundwater flow and solute transport model of a reclaimed fen watershed near Fort McMurray, Alberta (Sutton and Price 2022), and a comprehensive ecosystem dynamics model of an undisturbed moderate-rich fen near Lac La Biche, Alberta (Mezbahuddin et al. 2016).

Many of the first field-scale models of Canadian wetlands pursued a deeper understanding of the processes that influence wetland development. Price and Woo (1990) developed an advective-dispersive solute transport model of a coastal salt marsh along James Bay, Ontario that demonstrated how freshwater recharge suppressed near-surface salt concentrations, likely contributing to shifts in botanical composition. Fraser et al. (2001b) performed diffusive modelling at Mer Bleue bog, ON, which indicated that diffusion alone could not replicate the observed geochemical profiles, rather, transient flow-reversals were found to enhance the upward mobility of nutrients and ions. Groundwater flow modelling of two Newfoundland blanket bogs suggested that low peat hydraulic conductivity along the system margins contributed to bog development by impeding drainage, thereby enhancing peat accumulation (Lapen et al. 2005). Research on Canadian wetlands has also contributed to the progression of modelling platforms specifically intended to simulate wetland development on the millennial scale in which they evolve. These include the DigiBog platform developed by Morris et al. (2012), which explicitly simulates litter accumulation, peat decomposition, and the subsequent impact of decomposition on hydraulic conductivity; the updated version, DigiBog_Boreal, which can account for snow processes and short growing seasons (Ramirez et al. 2023); and the carbon balance approach of the Holocene Peat Model (Frolking et al. 2010).

Incorporating ecohydrological feedbacks into hydrological and earth system models has been particularly important when considering the implications of projected climate change on the behaviour and resilience of wetlands (Frolking et al. 2009; Helbig et al. 2020b). Yet, given the uncertainties inherent in projected climate change and those introduced through spatial and temporal downscaling (Zhang et al. 2011), many studies that address the future of Canadian wetlands perform sensitivity analyses rather than selecting a single scenario. Bourgault et al. (2014) prescribed reduced recharge to a model of the Lanoraie peatland complex in Québec, which indicated a pronounced decline in peatland water tables and diminished contributions from the peatlands to downgradient rivers, resulting in markedly reduced base flow. Levison et al. (2013) applied a variety of recharge anomalies to a headwater aquifer and peatland system in Southern Québec depending on the relative change in precipitation the peatland could receive greater groundwater inflow or become perched and disconnected from the aquifer – both of which imply the possibility of altered vegetation communities. In combination with a soil moisture dynamics model, Moore and Waddington (2015) modified the vascular water-stress formulation of Rodriguez-Iturbe and Porporato (2005) to sphagnum mosses to assess how peatlands may respond to climate change. This work indicated an increased frequency of moss desiccation under projected climate conditions, potentially instigating shifts in sphagnum community composition.

Given that the function and behaviour of wetland ecosystems is intimately tied to exchanges of moisture and energy in the unsaturated zone, many modelling studies have focused on the highly dynamic near-surface region. Kennedy and Price (2004) modified the FLOCR model to account for the multifaceted effects of compression in a cutover peatland near Sainte-Marguerite-Marie, Québec. Gauthier et al. (2018) performed unsaturated modelling of a variety of compressed moss profiles, indicating that artificial compression of regenerated mosses could hasten the recovery of cutover peatlands. In nearby Shippagan, New Brunswick, Elliott and Price (2020) performed soil moisture modelling by implementing the widely used Richard's equation at an experimental cutover peatland, which highlighted the often-considerable differences between estimates of soil hydraulic parameters based on modelled field behaviour and those estimated in the laboratory. Also using Richard's equation, Kettridge et al. (2016), identified that sphagnum moss and peat properties tend towards maximizing water-use efficiency, meaning the hydrophysical properties of an Alberta peatland reflected the ideal balance between carbon accumulation and water storage. In certain situations, however, the Richards equation may be inadequate. Dimitrov et al. (2010) demonstrated at Mer Bleue bog, Ontario, that the inclusion of macropore flow was instrumental to adequately simulating the patterns of soil water movement.

Early wetland modelling of evapotranspiration began as a comparison of energy budgets with equilibrium estimates to demonstrate that the latent heat flux can be accurately determined by the Priestley and Taylor (1972) model (Stewart and Rouse 1976a, b; Roulet and Woo 1986b). This soon evolved to examining how surface cover can influence climatological resistance and interactions with the local climate of wetland classes representative of the Boreal/Subarctic (Lafleur and Rouse 1988) and from temperate wooded swamps (Munro 1987). Significant advances to wetland modelling capability began with the validation of the Canadian Land Surface Scheme (CLASS) for several wetland types, incorporating organic soil parameters (Letts et al. 2000), which resulted in improvements in turbulent flux estimates for the vascular plant-dominated fen and marsh wetlands (Comer et al. 2000). However, bogs remained a challenge, as the dominance of non-vascular plant cover results in surface evaporation from soil, open water, and especially moss are the primary components of peatland ET (Malhotra et al. 2016). Consequently, surface resistance will largely be a function of surface characteristics, non-vascular plant cover notwithstanding. For example, variations in surface resistance were found to be a result of diurnal wetting and drying of the near surface layer, impacting the daytime variation of resistance to evaporation from the peat surface (Price 1991; Raddatz et al. 2009; Wessel & Rouse 1994). Recently, further research has been conducted to address the explicit treatment of moss surface resistance in models and how this surface interacts, and feeds back with the vascular canopy (Waddington et al. 2015). Modelling studies suggest that evaporation from mosses, which decreases with increasing tree cover due to a reduction in radiation at the surface, counteracts the potential drying effects of increased tree transpiration (Kettridge et al. 2013). However, decreased light at the surface with increased tree cover can produce a shift in moss communities to more shade tolerant species (Van Huizen et al. 2022), but the strength and ultimate direction of this potential feedback remains unclear (Waddington et al. 2015).

Overall, wetland hydrological modelling to date has provided a critical test of the state of our understanding Canadian wetland function and development of testable hypotheses. Ongoing improvement in the representation of processed-based knowledge of Canadian wetland hydrology within numerical models will assist future decision-making about these ecosystems in response to disturbances such as climate and land-use change.

vii) Indigenous knowledge

Indigenous groups in Canada often describe their language as having 'come from the land'. It is therefore not surprising that Indigenous languages across Canada have an extraordinarily rich vocabulary of words defining and describing the biophysical attributes of the natural environment. Historically, there has been very little synergy between university-based wetland researchers and Indigenous knowledge holders, and for this reason, very few Indigenous words have entered the scientific nomenclature. A noteworthy exception is the word 'muskeg', which has roots in the Cree and Ojibwa languages, and although it lacks precise scientific definition, is often used generally to mean organic terrain, including peatlands as defined in the present paper. Recent challenges arising from climate warming have catalysed new collaborations between researchers and Indigenous communities that focus on the sharing of knowledge and experiences on subjects of common concern. Indigenous knowledge provides valuable insights into how wetland environments are affected by climate change, and how Indigenous uses of wetlands have adapted as a result. Examples include documented changes to travel routes attributed to geomorphic changes in the wetland-dominated coastal landscapes of Hudson Bay (Lemelin et al. 2010); and the wetting of perched basins in the wetland landscapes of the Peace-Athabasca Delta caused by ice-jams (Beltaos 2023). Indigenous knowledge also helps to direct and validate scientific investigations. This process often termed 'two-eyed seeing' (Bartlett et al. 2012) fuses Indigenous and scientific knowledge (Quinton et al. 2022) and produces outcomes and solutions that can be more confidently applied by community decision makers (Woo et al. 2009). Scientific and Indigenous knowledges draw on observations made over vastly different time scales, and for that reason, the two knowledge types are complementary. The notion that journal articles published more than 10 years ago are old, or that the 75-year period considered in this paper includes most or all wetland knowledge, contrasts sharply with the reverence that Indigenous communities hold for the knowledge passed down to them by their elders. Like their language, the knowledge that developed with it is not seen as old but as 'enduring'. The success of such fusions continues to inspire further collaborations. For example, the Mushkegowuk Council, representing Cree First Nations in the Hudson and James Bay Lowlands recently hosted a conference to identify wetland knowledge gaps, and to highlight research priorities to help resource managers and policy makers make better decisions in the Mushkegowuk Territory (Middleton 2017). Similar meetings have been organised by Indigenous community leaders in Dene traditional lands of the Northwest Territories (Quinton et al. 2022).

Conclusion

In this summary of 75 years of Canadian wetland hydrology research, the breadth of studies reported herein attests to the large, vibrant, and productive community of Canadian wetland science. Canadian wetlands have enormous ecological value, with boreal wetlands alone estimated to provide $435.6 billion annually for ecosystem services including carbon storage, flood control, and water quality improvement (Anielski and Wilson 2009). However, Canadian wetlands are undergoing extraordinary transformative change that has implications for ecosystem function, ecological integrity, and source water protection. Natural resource development, which is important to Canada's national economy, is expanding at the same time as an intensification in climate change driven natural disasters such as wildfire, flood and drought.

Research needs to continue to investigate how these disturbances interact to fully describe the threats facing Canadian wetlands and the regional and global ecosystem services they provide.

Evaluating changes necessitates a good understanding of wetland functions against which change can be assessed. This review demonstrates that for many functions in undisturbed wetlands, such as water table behaviour, runoff response, evapotranspiration processes, groundwater interactions, vadose zone hydrology, freeze-thaw processes and carbon sequestration and decomposition mechanisms there is a rich base of knowledge and established methodologies. However, as with most empirical field-based research there remains a need to build the database of spatial and temporal character and variability. There are other aspects of wetland hydrological function that remain inadequately understood, such as nutrient regimes and processes, solute and non-aqueous phase liquid transport behaviour in peatlands, and ecohydrological feedbacks, thresholds, and resilience. Identifying and characterizing feedbacks within and among most of these processes is an ongoing challenge, even in pristine systems. The challenge is compounded in systems encountering multiple disturbances. While the numerous publications cited here and elsewhere strive to provide a simple interpretation of hydrological interactions in wetlands, this belies the inherent complexity of these systems which we must continue to work to understand and document. Further, predicting the response of a wetland to disturbance, requires an understanding of its hydrologic context. That is, the response of a wetland will often be a unique function of wetland class/form/type, external drivers such as weather, and internal changes to vegetation and soil hydrophysical properties that will modify the response to water inputs, losses and stores. Documentation of the response to change in specific contexts will remain valuable, but wetland hydrologists also need to develop indicators of change and action thresholds to trigger intervention, and build hydrological insight to anticipate and minimize adverse outcomes before they happen.

The Canadian wetland hydrology research we have summarized here highlights Canada's leadership in the globally-relevant domains of wetland protection, conservation, reclamation and restoration. Our summary has demonstrated that Canadian research has made major advances in all aspects of wetland hydrology, and in particular, identifying disturbance impacts and developing strategies to restore wetlands following land-use change. Given that Canada contains at least a fifth of the world's wetlands, and the world's second largest peatland complex, it is essential that wetland hydrology, and the processes it drives, remain a priority of government, industry, academia and practitioners. Accomplishing this will not only require wetland scientists to continue the bold, compelling work of the past 75 years, but invest in cross-disciplinary collaboration, effective scientific communication, and partnership with land-managers, local communities, and Indigenous organizations.

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