
Advances in Canadian wetland hydrology, 1999–2003

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Abstract:

Wetlands form 14% of the Canadian landscape and, consequently, have considerable interaction with the hydrological resources, including water quantity and quality, both within and downstream of them. Most of these are peatlands, particularly in boreal and northern environments, and these have been well researched recently. New data also exist for mineral wetlands (e.g. prairie sloughs). Relatively little attention has been given to coastal wetlands, or the complicated systems in the Western Cordillera. This paper reviews the current studies in Canada. Copyright © 2005 John Wiley & Sons, Ltd.

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INTRODUCTION

The objective of this paper is to highlight the advances made in the hydrology (including water quality and carbon cycling) of Canadian wetlands between 1999 and 2003, following a similar report for the preceding 4 years (Price and Waddington, 2000). Canadian wetlands are categorized as bogs, fens, swamps, marshes and shallow open water (NWWG, 1997), and comprise 14% of Canada's land area (NWWG, 1988). Peatlands represent over 90% of Canadian wetlands (Tarnocai, 1998), and that is reflected herein. The purpose of this paper is to report on activities between 1999 and (early) 2003. Where necessary we have drawn upon other literature on wetlands both within and outside of Canada, to provide the context for more recent initiatives. However, this is not intended to be a comprehensive or balanced review of wetland hydrology. Its content is strongly influenced by the quantity and direction of research done between 1999 and 2003 on Canadian wetlands, mostly peatlands, but including peat deposits such as palsas and peaty deposits in tundra areas.

Wetlands are areas with the water table at, near or above the land surface for long enough to promote hydric soils, hydrophytic vegetation, and biological activities adapted to wet environments (NWWG, 1997). Wetlands may be mineral-soil wetlands or peatlands, depending on hydrological processes resulting from water exchanges dictated by climate and landscape factors. Mineral-soil wetlands, which include marsh, shallow water, and some swamps, produce little or no peat, because of climatic or edaphic conditions (Zoltai and Vitt, 1995). Peatlands are defined as wetland areas with an accumulation of organic sediments exceeding 40 cm, and include bogs, fens, and some swamps (NWWG, 1997). Fens and (some) swamps are minerotrophic peatlands, receiving water and nutrients from atmospheric and telluric sources, whereas bogs are ombrotrophic, receiving water and nutrients dominantly from direct precipitation. Wetlands exist in the landscape where the water balance ensures an adequate water supply at or near the surface. Thus, wetlands are restricted to locations where, on average, precipitation exceeds evaporation loss, or where sustained inflows from surface

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or subsurface sources alleviate the water deficit. In bogs (by definition) the source is ombrogenous (Woo, 2002), but other wetland types may have a much more complex array of sources.

Considerable progress has been made in wetland hydrology since Ingram's (1983) seminal paper, and is summarized in a recent review by Woo (2002). Within the Canadian context, regional overviews have been provided for Quebec and Labrador (Price, 2001), the McKenzie Basin (Rouse, 2000), the High Arctic (Woo and Young, 2003), and the prairies (LaBaugh *et al.*, 1998; van der Kamp and Hayashi, 1998).

ATMOSPHERIC FLUXES AND STORES

Quantification of the component fluxes and stores continues to be an important first step in understanding, managing and modelling of wetlands. Precipitation provides the source of water directly to wetlands and for recharging surface and groundwater inputs, and varies tremendously across Canada. Precipitation, particularly snow inputs and interception losses, can be difficult to quantify. This is especially true in large watersheds, because the month-to-month changes in snow cover, for example, are often much greater than the atmospheric flux terms (Strong *et al.*, 2002). The input must overcome interception losses, which in forested wetlands may represent 32% for black spruce (Van Seters and Price, 2001), or 21–25% for larger cedar and balsam fir forests in British Columbia (BC), with only 1% being stemflow. Fog interception was shown to be notable, if not significant, in a hypermaritime northwest BC wetland forest (Emili and Price, unpublished data).

More attention has been given to evapotranspiration fluxes from wetlands. Land surface characterizations, such as CLASS (Canadian Land Surface Scheme; Verseghy, 1991), for use in global climate models, recognize the importance of wetlands. Bartlett *et al.* (2002) noted that overrepresentation of organic soils in heterogeneous environment results in the overestimation of the evaporation loss. Surfaces dominated by vascular plants are well represented, but non-vascular surfaces (e.g. *Sphagnum*) remain problematic (Comer *et al.*, 2000). Given that latent heat losses are generally the largest water sink in wetlands, research continues into the role of soil and vegetation on the energy balance. Eaton and Rouse (2001), for example, found that over a 10 year period the seasonal variability of cumulative evapotranspiration from subarctic sedge fen was controlled primarily by cumulative precipitation. Eaton *et al.* (2001) found wetland sites had high Q_E/Q^* (sensible heat/net radiation), high Priestley–Taylor α , and low Bowen ratio compared with non-wetland areas; thus, evapotranspiration losses were large in spite of a relatively low vapour-pressure deficit over the wetland surface. Petrone *et al.* (2003a) found that the presence of a higher water table, higher soil moisture and the presence of vascular plants in a cutover peatland being restored resulted in a higher evapotranspiration loss than an adjacent unrestored site. The emergence of vascular vegetation increased the surface roughness over a single growing season (Petrone *et al.*, 2003b).

SURFACE AND GROUNDWATER FLUXES

By virtue of the location of wetlands within the landscape, understanding wetland hydrology requires knowledge of groundwater and surface-water interactions. The proportion of surface and groundwater inputs, and a wetland's interaction with groundwater (i.e. recharge versus discharge function) are governed by its position within the groundwater flow system, the hydrogeologic characteristics of soil and rock material, and their climatic setting (Tóth, 1999; Winter, 1999, 2000; Sophocleous, 2002). The extent and type of groundwater interaction will influence the amplitude and duration of baseflow and water-level fluctuations, which are linked to surface saturation, runoff processes and ultimately sediment redox, biogeochemical processes, and vegetation patterns within a wetland (Tóth, 1999; Hill, 2000; Hayashi and Rosenberry, 2001).

There has been considerable research directed towards determining the role of groundwater–surface-water interactions in wetland hydrology from a variety of climatic and geologic regions throughout Canada. In permafrost-dominated areas of northern Canada, groundwater—surface-water interactions are restricted to localized flows in the surface active layer during periods of thaw (Woo and Young, 2003). In such arid

climates, patchy arctic and subarctic wetlands occupy low areas inundated during the snowmelt period, or with water storage sustained by more gradual inputs from groundwater or late-lying snowbanks (Carey and Woo, 2001a; Woo and Young, 2003). The shallow frost table limits percolation, thus facilitating the existence of these systems (Carey and Woo, 1999; 2001b; Young and Woo, 2003). Coarse sorted stones between frost mounds (Hodgson and Young, 2001), or macropores and pipes between peat hummocks (Quinton and Marsh, 1999; Carey and Woo, 2000a,b; 2002), supply water to wetlands. Wetlands such as these, having a strong dependency on water from localized and shallow sources like supra-permafrost water, ground-ice melt and late-lying snowbanks, are susceptible to climate change (Young and Woo, 2000), particularly in view of the expected deepening active layer and large evaporative losses (Young and Woo, 2003a,b).

In topographically controlled humid eastern boreal regions with shallow soils underlain by impermeable crystalline bedrock, groundwater is usually restricted to localized flow, resulting in predictable and relatively simple wetland groundwater interactions. Shallow near-surface runoff dominates, and flow rates are inextricably tied to shorter term variation in weather and, thus, susceptible to extended dry periods and climate change (Devito *et al.*, 1999a). Although groundwater is a minor input to headwater shield wetlands, the soil depth within the catchment can influence the seasonality of groundwater connectivity between uplands and wetlands by influencing water-table fluctuations and maintaining surface saturation during drought, with runoff during summer storms (Devito *et al.*, 1999a; Hill, 2000).

In regions of deeper glacial deposits, such as the Great Lakes and Laurentian regions of humid eastern Canada, larger scale groundwater interactions occur as influenced by topography and substrate grain size distribution. Groundwater interactions with wetlands located in finer grained glacial tills are limited by low permeability. Similar to Precambrian systems, soil-water storage capacity is low and near-surface runoff during storm events dominates the inputs and outputs (Hill, 2000; Vidon and Hill, 2004). These wetlands are maintained in depressions and breaks in slope with ample supply of water in a humid climate, but they are susceptible to drought periods (Hill, 2000). However, the role of larger scale topographically controlled intermediate or regional groundwater flow or of heterogeneous lithology (sand lenses) in clay-rich glacial deposits on wetlands in this region is poorly understood. In adjacent outwash landscapes with permeable surface aquifers of coarser grained material, constancy of groundwater inputs, connection, permanence of surface saturation and stability of the water table in wetlands was observed with increasing depth of aquifer (Hill, 2000; Warren *et al.*, 2001; Vidon and Hill, 2004). Aquifer lithology and deposits of lower hydraulic conductivity peat in near-stream wetland areas can also influence wetland groundwater–wetland interactions, surface saturation and runoff in glaciated outwash regions of Ontario (Devito *et al.*, 2000a; Hill *et al.*, 2000). Reversals in hydraulic gradient and groundwater flow from streams into adjacent wetlands, or wetlands back into the hillslope, appear to be influenced by riparian wetland slope and aquifer depth in outwash landscapes (Vidon and Hill, 2004). Groundwater reversals across a barrier beach also occur when hydraulic gradients controlled by lake and/or marsh water level change seasonally (Huddart *et al.*, 1999).

In the dry prairies and boreal plain of western Canada regions, external water inputs become increasingly important to wetland maintenance. Groundwater flow is complex due to the low relief and deep glacial deposits and, depending on their position, wetlands can have a recharge, flow-through or discharge function (LaBaugh *et al.*, 1998; van der Kamp and Hayashi, 1998; Tóth, 1999; Devito *et al.*, 2000b). However, in clay-rich glacial deposits the groundwater interaction is a minor component of most prairie wetlands due to the low permeability (Conly and van der Kamp, 2001; Parsons *et al.*, 2004; van der Kamp *et al.*, 2003). Although soil-water storage exchange is low in clay-rich tills, storage is rarely exceeded during the summer, and surface runoff into ponds during this time is limited to infrequent large storm events. Thus, ponds and depressions are heavily reliant on snowdrift and snowmelt runoff over frozen soils and experience considerable seasonal water-level variability (van der Kamp *et al.*, 1999; Hayashi *et al.*, 2003). These systems are generally considered susceptible to changes in climate and land use that influences snowmelt surface-water interactions (van der Kamp *et al.*, 1999; 2003). Depressional wetlands such as these are focal points for groundwater recharge (Hayashi *et al.*, 1998, 2003), though most of the water recharge may flow to the moist margins, rather than to deep percolation (Parsons *et al.*, 2004). Such lateral recharge may be important in sustaining

water yield to local shallow wells (van der Kamp and Hayashi, 1998). The storage function of depressional wetlands can be represented mathematically with volume–depth–area relationships (Hayashi and van der Kamp, 2000; Wiens, 2001), which can be incorporated into a storage–water balance model (Su *et al.*, 2000) that can simulate long-term water-level variations (Conly and van der Kamp, 2001). However, comparison of pond systems located in coarse-grained glacial deposits in the Canadian Prairies is lacking, and regional variation in groundwater interactions on depression wetlands in coarse-grained glacial deposits is poorly understood (Winter, 1999, 2000, 2001).

Similar to the prairies systems, groundwater exchange to boreal plain pond–peatland complexes located in topographic high and low areas of fine textured, low conductivity glacial till and lacustrine deposits contribute little to the water balance (Ferone and Devito, 2004). Further, forested clay-rich hillslopes provided little snowmelt or storm runoff in most years because soil storage and transpiration demands exceed rainfall, which occurs primarily during the summer in this sub-humid environment (Kalef, 2002; Devito *et al.*, 2004). In contrast to the semi-arid prairies, runoff generation is primarily from near-surface flow from peatland areas connected to shallow pond wetlands that develop in a sub-humid climate (Wolniewicz, 2002). Topographically high ponds act as focal points for recharge, primarily providing lateral flow from the pond to peatland and adjacent hillslopes (Ferone and Devito, 2004). During larger rain events, the hydraulic gradient between peatland and pond reverses, and peatland discharges water into the pond. The low-lying pond–peatland complex functioned as a flow-through system of near-surface water originating from extensive peatlands adjacent to the pond (Ferone and Devito, 2004). The differences in shallow groundwater interaction and topography also influenced pond chemistry (Ferone, 2001) and have implications for pond response to climate or disturbance (Tóth, 1999; Winter, 2001). Further, comparative studies of ponds located in adjacent landforms dominated by coarse-grained glacial deposits (outwash sands and gravels) indicates a significant exchange of groundwater from larger scales of flow on pond water budgets, and different responses in water levels and impacts to climate and disturbance (Winter 2000).

The boreal and temperate mountain and plains areas within in the Western Cordillera provide excellent locations for conducting field studies to increase our understanding of processes influencing groundwater–surface-water interactions in wetlands from a range of topographic, geologic and climatic conditions. Considerable work has been conducted on groundwater interactions in forested and montane uplands and streams (Carey and Woo, 1999; also see Buttle *et al.* (2005)). Also, wetlands and lakes have been shown to moderate the thermal regimes of groundwater-fed streams draining cut and forested catchments in the interior of BC (Mellina *et al.*, 2002). Fitzgerald *et al.* (2003a) showed that small headwater swamps in north coastal BC are critical interfaces between steep, well-drained forested upland slopes and runoff, and they should be avoided during timber harvesting. Surface runoff from sloping forested swamps on moderate slopes organizes itself quickly into ‘seeps’ (Emili and Price, unpublished data), which diverts this channellized water away from more gently sloping blanket bogs (Fitzgerald *et al.*, 2003b). Runoff from these blanket bogs thus has a much lower runoff ratio than micro-catchments containing these seeps (Emili and Price, unpublished data).

Internal water flow through peat wetland systems has received relatively limited attention recently. Reeve *et al.* (2000) suggested that the extent of vertical flow in peatlands, such as in the Hudson Bay Lowland, is limited by low-permeability mineral substrate (i.e. rather than the humified peat layers), promoting lateral flow through the acrotelm. However, Fraser *et al.* (2001a) noted reversals between recharge and discharge to and from the deeper peat layers during wetter and dryer periods respectively. Transfer of water from deeper peat layers to the surface is important in sustaining soil moisture in the surface layer, and changes of soil moisture in this layer should be accounted for in the water balance (Lapen *et al.*, 2000). Vertical redistribution of peat pore-water has also been shown to occur from the saturated zone to the soil-moisture zone (Kennedy, 2002) during peat consolidation (Lang, 2002), and may be enhanced by pressure caused by methane generation (Price, 2003). These exchanges have implications for water quality (Fraser *et al.*, 2001b).

Runoff from wetlands is controlled by the rate and magnitude of inputs, and the efficacy of storage, usually involving an interaction of surface and shallow groundwater (Gibson *et al.*, 2000). Although the presence of surface water may suggest its importance as a delivery mechanism, mixing model studies demonstrate the

predominance of 'old' water in storm runoff from headwater swamps, for example (Brassard *et al.*, 2000; Fitzgerald *et al.*, 2003b). The efficiency of these mechanisms was reflected by higher runoff to northern Alberta lakes in wetland-dominated catchments, compared with upland systems (Gibson *et al.*, 2002). The expansion and connectivity of saturated surfaces associated with peatland areas in the boreal plains region of Alberta are directly related to regional runoff regimes (Wolniewicz, 2002). Where deep seasonal frost or permafrost is present, the rate and spatial pattern of thaw control the runoff pathways (Quinton and Marsh, 1999; Carey and Woo, 2001b) and 'new' water (snowmelt) may dominate (Metcalf and Buttle, 2001). Surface retention by wetlands, however, may delay surface-water outflows (Metcalf and Buttle, 1999). In contrast, wetlands with deeper flow systems may experience a notable baseflow component (Beckers and Frind, 2001). In spite of the efficiency of wetland runoff processes, the flood mitigation role of wetlands is often overstated (Simonovic and Juliano, 2001). Modelling of wetland runoff response (McKillop *et al.*, 1999) requires careful parameterization of hydraulic parameters at a variety of scales (e.g. Letts *et al.*, 2000).

WATER FLOWS IN DISTURBED WETLANDS

The health of wetland systems is threatened by various direct and indirect anthropogenic changes (Detenbeck *et al.*, 1999). Anticipated changes in the global climate are expected to increase the soil-water deficit in North America, which will alter the water balance of wetlands. Monitoring networks often have insufficient spatial and temporal sampling to establish patterns of variability or provide insight into processes related to climate change (Conly and van der Kamp, 2001). At the local scale, wetland disturbances may be direct and intentional, such as for agriculture (van der Kamp and Hayashi, 1998), drainage of forestry (Prévost *et al.*, 1999), or peat mining (Price *et al.*, 2003).

Forest drainage following harvesting of a treed peatland has been shown to improve growth of naturally regenerated black spruce (Jutras *et al.*, 2002; Pepin *et al.*, 2002). Although short-term studies concerning planted seedlings cannot effectively corroborate this (Roy *et al.*, 1999, 2000b), there are some limitations to the effective lowering of the water table. This arises because of changes to the peat substrate following drainage, including subsidence and changes in aeration dynamics (Silins and Rothwell, 1999), accelerated peat decomposition (Prévost and Plamondon, 1999), and a decrease in the effective hydraulic conductivity as the water table drops (Belair *et al.* 2003). There is also concern that 'wetting up', caused by decreased interception (Roy *et al.*, 2000a), may impair root aeration and accelerate *Sphagnum* growth sufficiently to overtake seedlings (Roy *et al.*, 2000b). Drainage at these sites increased baseflow, caused a 25% increase in total runoff, increased dissolved and suspended solids, and increased temperature variability in outflow (Prévost and Plamondon, 1999).

Sites that are mined for peat have additional stresses. Removal of the living layer, or acrotelm, severely impairs hydrological function (Price *et al.*, 2003). Following abandonment, drainage ditches can remain effective for decades (Van Seters and Price, 2001), and the irreversible changes to morphology and peat structure alter flow patterns (Van Seters and Price, 2002). Spontaneous regeneration of the dominant peat-forming moss, *Sphagnum*, is curtailed because of the unstable surface caused by needle-ice formation (Groeneveld and Rochefort, 2002), the strong capillary retention of the cutover substrate, and development of a litter layer of ericaceous leaves (Price and Whitehead, 2004), which restrict water flow to the mosses. Regeneration in wetter areas of old manually block-cut peatlands can occur spontaneously (Price and Whitehead, 2001). Whereas average wetness conditions on abandoned vacuum-harvested sites (the modern extraction technique common in Canada) are little different than block-cut sites, the extreme spatial variability of wetness in the latter sites ensures that at least some loci for regeneration exist (Price *et al.*, 2003). Restoration plans must account for the cumulative human impacts on wetlands, including broader landscape effects (Bedford, 1999). Managed restoration requires rewetting of the site, often by blocking ditches. More aggressive restoration management has demonstrated the effectiveness of artificial terraces (von Waldow, 2002) or shallow basins (Price *et al.*, 2002) to retain snowmelt water. Application of straw mulch reduces

evaporative losses (Petroni *et al.*, 2001, 2003a) and reduces the scale of soil moisture variability (Petroni *et al.*, 2003b). Important changes to the hydraulic character of the remaining peat occur with drainage and peat extraction (Schlotzhauer and Price, 1999) that profoundly affect storage exchanges (Price and Schlotzhauer, 1999). Compression of the peat caused by seasonal declines in the water table (Lang, 2002) can cause up to a three orders of magnitude decline in hydraulic conductivity (Price, 2003), although this would be considerably less in locations with thinner peat, such as in arctic systems. Incorporation of these peat volume changes and its effect on hydraulic parameters, into a vertical one-dimensional numerical flow model (Kennedy, 2002), allowed the prediction of soil-water pressure, soil moisture, water table and peat surface elevation, and prediction of various abandonment and restoration scenarios.

At the local to regional scale, indirect disturbance can impact or interact with hydrological cycle and surface and groundwater linkages between adjacent uplands and wetlands. These linkages are important for maintaining the hydrological and ecological integrity of wetlands (e.g. Hayashi and Rosenberry, 2001). Such interactions fall within the realm of forest or upland hydrology, and the reader is referred to Buttle *et al.* (2005). Clearly, a broader catchment/landscape approach is required to understand upland–wetland linkages and the potential impact of individual and cumulative catchment disturbance to the receiving wetland (Bedford, 1999; Hill, 2000). Examination of wetland position within the surface and groundwater flow system can indicate the relative importance of regional- and local-scale disturbance impacts on wetlands and other aquatic systems (Devito *et al.*, 2000b; Hill, 2000). Limited runoff from both forested and cut portions of a sub-humid boreal catchment, due to the low rainfall relative to soil storage and evapotranspiration, resulted in little influence of logging aspen uplands on the hydrology of a groundwater-fed valley wetland (Kalef, 2002). The conversion of cultivated fields to permanent grass and shrub resulted in the drying out of adjacent depression wetlands (sloughs) by severely reducing localized snow drift and spring melt runoff inputs (van der Kamp *et al.*, 1999, 2003).

CARBON CYCLING

Undisturbed peatlands are presently a relatively small sink for CO₂ and a large source of CH₄. When the 'global warming potential' of CH₄ is factored in, many peatlands are neither sources nor sinks of greenhouse gases (Waddington and Roulet, 2000). However, land-use change significantly alters greenhouse-gas emissions (Roulet, 2000). Considerable progress has been made on understanding carbon cycling processes in Canadian mined, cutover and restored peatlands.

The exploitation of peatlands for *Sphagnum* is widespread in North America and Europe. Peat mining, through the combination of drainage, peat removal and subsequent abandonment, alters the environment so severely that *Sphagnum* spp. are unable to colonize. Cutover peatlands represent a persistent source of atmospheric CO₂, losing carbon of 300 to 400 g m⁻² year⁻¹ (Waddington *et al.*, 2002; Waddington and McNeil, 2002; Petroni *et al.*, 2003, 2004a,b). Peat oxidation is more dependent on peat moisture content and peat temperature (Waddington *et al.*, 2002) than peat carbon quality (Waddington *et al.*, 2001), suggesting that water management (e.g. Price *et al.*, 2002) may reduce CO₂ losses from mined peatlands (Waddington and Price, 2000).

The restoration of *Sphagnum* mosses on cutover sites has the potential to sequester atmospheric CO₂, thereby returning the peatland to a peat accumulating system (Waddington *et al.*, 2003a,b). Restoration not only increases plant production, but also decreases total respiration (Waddington and Warner, 2001). *Sphagnum* production varies between species (e.g. *S. fuscum* > *S. capillifolium*) according to their ability to withstand harsh conditions on restored peat surfaces (Waddington *et al.*, 2003a). A stable moisture supply is more beneficial to *Sphagnum* growth (Rocheffort *et al.*, 2002), compared with repeated wetting and drying events (McNeil and Waddington, 2003). The application of a mulch surface improves moisture conditions near the peat surface, but mulch decomposition represents a short-term source of atmospheric CO₂ (Waddington *et al.*,

2003b). Petrone *et al.* (2001) determined that a recently restored peatland was a larger source of CO₂ than an adjacent cutover site, in part because the mulch decomposition exceeded the new production of mosses and vascular plants.

As vascular plants colonize restored peatlands, CH₄ flux increases due to both the supply of labile carbon and the enhanced CH₄ transport (Day and Waddington, unpublished data). CH₄ flux from cutover peatlands is greatest in drainage ditches (Waddington and Price, 2000) owing to their permanently flooded conditions and supply of highly labile dissolved organic carbon (DOC; Tóth, 2003). Belissario *et al.* (1999) found that sites of high CH₄ emissions in undisturbed peatlands had enriched δ¹³CH₄ signatures, suggesting the importance of the acetate fermentation pathway on methanogenesis. DOC has also been correlated to CO₂ exchange in cutover peatlands (Glatzel *et al.*, 2003).

Many landscape-scale studies of carbon exchange from undisturbed peatlands have taken place in Canada in the last few years. These, and many carbon cycling studies, have been constrained to growing-season measurements. However, Lafleur *et al.* (2001a) determined that the non-growing-season CO₂ loss from an ombrotrophic peatland is not small (183 g m⁻²). On an annual basis, however, the peatland was an annual net CO₂ sink (248 g m⁻² year⁻¹). Joiner *et al.* (1999) determined that a boreal fen was a net source of CO₂, losing 31 g m⁻² of carbon in 1994 but was a net carbon sink in 1996 (−92 g m⁻²). The interannual difference was linked to an earlier snowmelt and thaw of the fen surface, leading to drier summer conditions. Griffis *et al.* (2000a) suggest that an early snowmelt combined with wet and warm conditions during the spring period leads to large carbon acquisition even when drier conditions prevail over the majority of the growing season. CO₂ exchange in an adjacent wetland forest, however, was related to timing of snowmelt and heat content prior to leaf out (Lafleur *et al.*, 2001b; Rouse *et al.*, 2002).

Peatland surface topography leads to differences in carbon exchange processes and vegetation assemblage (Waddington and Roulet, 2000). Griffis *et al.* (2000b) scaled community-level CO₂ measurements from hummocks and hollows to tower measurements at the landscape scale. Variability in peatland surface topography (hummock, hollow, peat plateau, etc.) leads to differences in soil moisture, temperature, vegetation type and biomass (Moore *et al.*, 2002). This creates differences in peat CO₂ production (Scanlon and Moore, 2000), the fluxes of CH₄ and CO₂ (Dalva *et al.*, 2001), and peat accumulation (Robinson and Moore, 1999). It is expected that different communities will respond differently to climate change (Griffis *et al.*, 2000b). Clair *et al.* (2002) suggest that, under a 2 × CO₂ climate-change scenario, carbon loss from a small temperate wetland will almost double from 0.6% to 1.1% of total biomass.

Differences in vascular and non-vascular vegetation–carbon–water dynamics were incorporated in a dynamic model of long-term peat accumulation (Frolking *et al.*, 2001). The model suggests that bogs are more sensitive than fens to climate conditions. Moreover, warmer and wetter conditions were found to be more conducive to peatland development (Frolking *et al.*, 2001). In a simpler model, focusing on the non-linear interactions among peat production, decomposition and hydrology, Hilbert *et al.* (2000) also demonstrate the sensitivity of peat accumulation to peatland water balance.

The peatland carbon simulator (PCARS), developed by Frolking *et al.* (2002), is a process-oriented model of the contemporary carbon balance of northern peatlands. Seasonal patterns and the general magnitude of net ecosystem exchange of CO₂ were similar to measured tower data (see Lafleur *et al.* (2001a)). PCARS was designed to link to the CLASS model and will prove valuable in examining climate–peatland carbon feedbacks in future research. The model incorporates the exchange and interaction of CH₄, CO₂, and DOC.

Fraser *et al.* (2001b) determined that DOC export from an ombrotrophic bog was 12% of the magnitude of the carbon sink measured at the same peatland (Lafleur *et al.*, 2001a). DOC concentration in the acrotelm was variable, with a higher aromaticity and fulvic fraction ('allocthonous-like') than catotelmic waters. The influence of hydrology on the patterns of supply and quality of DOC has also been shown to have a major influence on the cycling of nitrogen (Hill *et al.*, 2000) and mercury.

WATER QUALITY

Process-level investigation of the hydrological and biogeochemical controls on the mechanisms and rates of element transformation (Devito *et al.*, 2000a; Hill *et al.*, 2000) are some of the most important advancements in the understanding of water quality in wetlands. For example, the spatial discontinuities in denitrification zones controlled by hydrology and lithology can confound the understanding of denitrification mechanisms (Hill, 2000). In a forested riparian wetland that received groundwater with elevated nitrogen concentration ($10\text{--}30\text{ mg l}^{-1}$), denitrification was carbon limited and only became nitrogen limited in narrow zones of strong denitrification in pockets of buried peat (Hill *et al.*, 2000). These findings were supported through nitrogen isotope analyses (Devito *et al.*, 2000a), confirming that groundwater flow paths, and the adequate supply of terminal electron donors and acceptors, control microbial denitrification in riparian wetlands. During snowmelt periods, when surface flow occurs, runoff can bypass riparian 'buffer' zones and directly enter streams (von Waldow *et al.*, 2002). Isotopic methods combined with *in situ* tracer experiments can demonstrate microbial denitrification activity and rates (Mengis *et al.*, 1999). Catchment geomorphology and hydrological connectivity must be considered when assessing catchment nitrogen dynamics (Devito *et al.*, 1999b; Schiff *et al.*, 2002).

Phosphorus dynamics remain poorly understood (Devito *et al.*, 2000b). Carlyle and Hill (2001) determined that groundwater flow governed redox conditions, and geochemistry strongly influences the solubility and mobility of phosphorus. A detailed three-dimensional analysis of groundwater flow, dissolved oxygen and iron species (Carlyle and Hill, 2001) revealed anaerobic zones where Fe^{3+} was reduced to Fe^{2+} . This influenced groundwater soluble reactive phosphorus (SRP) concentrations. As with denitrification (Hill *et al.*, 2000), patterns of SRP concentration reflect local interaction of flowpaths with different redox states.

Microbially catalysed biogeochemical reactions in wetlands have recently been investigated in more detail. Laboratory incubations by Blodau *et al.* (2002) showed that iron cycling in northern peatlands had little influence on carbon flow, but sulphate reduction had the potential to limit methane production by 48 to 86%. Interest in the use of natural and constructed wetlands for the remediation of runoff, wastewater, and industrial effluent continues to garner interest in Canada. This discourse is beyond the scope of this work; however, the review by Kennedy and Mayer (2002) examines this area in detail. Importantly, they conclude that cold-weather performance must be better understood before water-treatment wetlands can gain widespread acceptance in Canada. More effective translation of our understanding of the hydrology and biogeochemistry of natural cold-region wetlands will undoubtedly inform this research in the coming years.

Wetlands remain a focus of research in mercury cycling, both in terms of the mechanisms governing the production of methylmercury *in situ* (Branfireun *et al.*, 1999; Heyes *et al.*, 2000) and their controls on the fate and transport of both inorganic mercury and methylmercury (Branfireun and Roulet, 2002; Rencz *et al.*, 2003). At the mesocosm scale, Branfireun *et al.* (1999) investigated the link between atmospheric sulphate deposition and the production of methylmercury in peat. They found a clear increase in the amount of methylmercury, a potent neurotoxin, with sulphate addition, making the first direct link between sulphate deposition in precipitation and the mercury cycle in peatlands. Most research on mercury cycling in wetlands has focused on acidic, nutrient-limited *Sphagnum*-dominated peatlands. However, Galloway and Branfireun (2004) found that the transport of total mercury in a temperate swamp was greatest when the wetland and surface streams were hydrologically connected. Methylmercury was released downstream, with yields similar to that of more acidic peatland systems. An inverse relationship between methylmercury and sulphate concentrations was observed, with the highest and lowest concentrations respectively found during the periods of persistent inundation and the onset of anaerobic conditions in the wetland sediments.

The observation of an inverse relationship between inundation and sulphate concentrations is consistent with the findings of Warren *et al.* (2001). They found a strong relationship between hydrological connection, reducing conditions in temperate swamp sediments and stream sulphate concentrations (Devito *et al.*, 1999a; Eimers and Dillon, 2002). This clarified the inverse relationship between water-level fluctuations and sulphate reduction and oxidation (Devito and Hill, 1999).

Studies of the effect of anthropogenic impacts on wetland water quality are relatively sparse. Heyes *et al.* (2000) found that elevated methylmercury concentration was the consequence of mercury methylation in an experimentally flooded anaerobic wetland surface. The flooding facilitated the exchange of nutrients (importantly sulphate) between the peat surface and the surface water. Prévost and Plamondon (1999) found that surface water from drained sites had significantly higher electrical conductivity than natural peatlands because of leaching of N, Na, S, Ca, and Mg. Water quality did not return to pretreatment levels even after 5 years. The effects of atmospheric contaminant loading to wetlands via precipitation may significantly influence biogeochemical cycles (Branfireun *et al.*, 1999) or ecosystem function (Donald *et al.*, 1999, 2001). Donald *et al.* (1999) found that the concentrations of agricultural pesticides in wetlands in the prairies were positively related to precipitation amount, and often exceeded the limits recommended for the protection of aquatic biota.

SCALING ISSUES

Generalizing site-specific studies and scaling up from the upland–wetland boundary to the regional scale is a major research challenge, not only for wetland hydrology (Sophocleous, 2002). Devito *et al.* (2000a,b) developed a hierarchy of landscape factors predicting surface water and groundwater connectivity of wetlands and lakes, and the potential susceptibility of surface waters to land-cover changes. Such qualitative landscape approaches may need to precede more quantitative modelling methods (e.g. Woo and Young, 2003). Currently, regional three-dimensional surface water and groundwater hydrological models are not well equipped to represent more local phenomena at the interface of groundwater–surface-water interactions in wetlands (Sophocleous, 2002). Use of readily available surficial geology data, topography and climate as indices of hydraulic gradient and hydraulic conductivity provides a reasonable basis for prediction of the dominant components of the water balance and the scale of linkages of the wetland to the surrounding a region (Winter 2000).

Advances in remote sensing of wetlands have provided a method for generalizing site-specific to regional studies and assessing dominant hydrological processes at different scales. For example, RADARSAT, LANDSAT and SPOT images provide data for assessing historical and current patterns of flooding in the Peace–Athabasca Delta (Töyrä *et al.*, 2001, 2002), and for parameterization of hydraulic flow models (Pietroniro *et al.*, 1999). Such methods are suited to evaluation of aggradation or plant succession in these systems, and may clarify or challenge the paradigm of delta degradation, such as that attributed to the effect of the WAC Bennett Dam on the Peace–Athabasca Delta, for example (Timoney, 2002). LANDSAT and RADARSAT images can also be used to predict the dominant runoff-generating areas, and the dynamic nature of small localized depressions and coarse-scale catchment boundaries in influencing regional runoff (Wolniewicz, 2002).

CONCLUSIONS

Considerable challenges face wetland hydrology and water-quality research. Data are often spatially and temporally too sparse to characterize the dynamics of wetland hydrology adequately, and especially the biogeochemical processes. Biogeochemical and hydrological studies are often not satisfactorily convergent, and there are profound uncertainties surrounding the impact of environmental change on wetland biogeochemical function (Hill, 2000). Many of these problems are intractable in either the field or the laboratory, but a promising convergence of scales of investigation is emerging that will couple these two approaches, leading to significant advances in the coming years.

The range of studies reported herein attests to the vibrant community of Canadian wetland research. However, it is evident that certain geographical and topical deficiencies exist. Little research has been done in the complicated wetlands systems of BC, where wetland systems do not fall easily into wetland categorizations

developed in other parts of the country. Relatively little work is being done in the Maritime Provinces, and especially on coastal wetlands. Recent work on Great Lakes coastal systems (Crowe *et al.*, 2004) may partially address the latter point. As many of the fundamental hydrological processes become better understood, future research on linkages to water quality and carbon dynamics, for example, becomes more feasible, and more amenable to modelling.

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