Impacts and prognosis of natural resource development on water and wetlands in Canada's boreal zone

Kara L. Webster, Frederick D. Beall, Irena F. Creed and David P. Kreutzweiser

Kara L. Webster,[1] **Frederick D. Beall, and David P. Kreutzweiser.** Natural Resources Canada, Canadian Forest Service, 1219 Queen St. East, Sault Ste. Marie, ON P6A 2E5, Canada.

Irena F. Creed. University of Western Ontario, Department of Biology, 1151 Richmond St. N., London, ON N6A 5B7, Canada.

[1]Corresponding author (email: Kara.Webster@nrcan.gc.ca).

Tel: 705-541-5520

Fax: 705-541-5700

Word count (including references, figures, and tables): 47,713

Abstract

Industrial development within Canada's boreal zone has increased in recent decades. Forest management activities, pulp and paper operations, electric power generation, mining, conventional oil and gas extraction, non-conventional oil sand development, and peat mining occur throughout the boreal zone with varying impacts on water resources. We review impacts of these industries on surface water, groundwater and wetlands recognizing that heterogeneity in the dominance of different hydrologic processes (i.e., precipitation, evapotranspiration, groundwater recharge and runoff generation) across the boreal zone influences the degree of impacts on water resources. Through the application of best management practices, forest certification programs, and science-based guidelines, timber, pulp and paper, and peat industries have reduced their impacts on water resources, although uncertainties remain about long term recovery following disturbance. Hydroelectric power developments have moved toward reducing reservoir size and creating more natural flow regimes, although impacts of aging infrastructure and dam decommissioning is largely unknown. Mineral and metal mining industries have improved regulation and practices, but the legacy of abandoned mines across the boreal zone still presents an ongoing risk to water resources. Oil and gas industries, including non-conventional resources such as oil sands, is one of the largest industrial users of water and, while significant progress has been made in reducing water use, more work is needed to ensure the protection of water resources. All industries contribute to atmospheric deposition of pollutants that may eventually released to downstream waters. Although most industrial sectors strive to improve their environmental performance with regards to water resources, disruptions to natural flow regimes and risks of degraded water quality exist at local to regional scales in the boreal zone. Addressing the emerging challenge of managing the expanding, intensifying and cumulative effects of industries in conjunction with other stressors, such as climate change and atmospheric pollution, across the landscape will aid in preserving Canada's rich endowment of water resources.

Key words: natural resources, development, hydrology, biogeochemistry, cumulative effects, water quality, water quantity

1. Introduction

1.1 Water as a defining feature

Canada is a water rich nation, hosting ~5.4 % (2.9 km^3 per year) of the global renewable water supply (5.4×10^4 km^3 per year, FAO 2013). It is often reported that Canada contains 20% of the world's freshwater (Sprague 2007), but this estimate reflects the sum of water stored in lakes, reservoirs, soils and groundwater, and therefore represents the stock of freshwater rather than its renewable supply that is replenished each year from precipitation. Of the water that annually flows into Canada's oceans, approximately 50% originates in the boreal zone. Seventy-three percent of boreal rivers drain to the Arctic Ocean and Hudson Bay, 20% drain into the Atlantic Ocean, and 7% drain into the Pacific Ocean (Statistics Canada 2010). In addition to the large extent of surface waters, wetlands are a dominant feature of the boreal landscape, occupying 96.5×10^6 ha (17%) of the ecozone and representing close to 20% of the world's wetlands (NRCan 2013*c*) (Fig. 1).

1.2 Water and its ecosystem services

Canadians value their water resources. A Nanos (2009) research poll revealed 61.6% of Canadians ranked fresh water as the country's most important resource, ahead of forests, agriculture, oil and fisheries. Water is essential for all living things, and the myriad of water bodies including lakes, ponds, wetlands, rivers, and streams across Canada's boreal zone support an array of aquatic ecosystem services. These services include groundwater aquifer recharge, contaminant absorption and filtering, river flow regulation through absorbing and releasing excess water, shoreline and erosion protection, clean water supply, and habitat for waterfowl, fish and other biota (Millenium Ecosystem Assessment 2005; Woodward 2009; Wells et al. 2010; Kreutzweiser et al. 2013) (Table 1).

1.3. Role and impact of disturbance on boreal water resources

Forest disturbance followed by succession has an important natural role in boreal ecosystems (Venier et al. *in press*). Fire, insect outbreaks, blow downs, and beaver impoundments are all examples of natural disturbances operating on a broad range of spatial and temporal scales that influence boreal water resources. Anthropogenic development creates a different disturbance footprint on the landscape (Table 3). Natural disturbances may be small (blow down) or large (fire, insect) in extent, but typically their impacts are transient in time, such that the disturbance and impacts to water resources are short lived (i.e., <10 years). Anthropogenic disturbances may also be small (e.g., pulp mill) or large (e.g., clearcut, open-pit mine), but typically their geometry is different (i.e., linear *vs.* irregular; square or rectangular *vs.* polygons) and their impact times are longer causing slower or delayed ecosystem recoveries which create legacies for water resources that may be felt for decades or centuries (e.g., acid mine drainage and mine tailings) (Van Geen et al. 1997; Leblanc et al. 2000; National Research Council 2008).

Canada's boreal zone is increasingly affected by large-scale industrial activities, with estimates of total anthropogenic disturbance footprint based on interpretation of satellite imagery with approximately 24 million ha (4% of boreal) showing visible forest cutblocks accounting for more than 60% of this disturbance (Pasher et al. 2013) (Fig. 2). Industrial activities increase access for humans to once isolated lakes, rivers, and wetlands through networks of roads, seismic lines, pipelines and transmission lines and can result in removal of water, alteration of hydrologic and biogeochemical flows, increases in erosion and siltation, and increases in pollutants and contaminants in aquatic systems.

1.4 Purpose and scope of review

This paper reviews published literature describing the characteristics of boreal water resources, and reviews the status and impacts of natural resource development on the quantity and quality of water resources in the boreal zone and offers prognoses. Industries that use only raw natural resources (i.e., wood, peat, minerals, water) are considered; agriculture (crop or livestock production) and agroforestry are excluded. We review if and how water quantity and quality across the boreal zone are being affected by natural resource development. Consideration is given both to the intensity and extent of resource development impacts and the cumulative effects from combined industrial developments on boreal water resources. We focus on recent (i.e, last 20 years), peer-reviewed literature on studies in the Canadian boreal zone, but include other boreal and non-boreal examples where necessary. In a companion paper, Kreutzweiser et al. (2013) reviewed the impacts and offer prognoses of natural resource development on freshwater aquatic biodiversity in the boreal zone.

2. National perspective of boreal water resources

2.1 Hydrologic regions

Boreal basins drain into the Atlantic, Pacific, and Arctic oceans. The boreal zone can be classified into hydrologic regions that are defined by the dominance of the different components of the water budget (precipitation, evapotranspiration, discharge) that influence the distribution of wetlands, surface water and groundwater flow pathways, and biotic assemblages (Fig. 3).

Devito et al. (2005) proposed a hierarchy of five factors for classifying the dominant controls on water cycling: climate, bedrock geology, surficial geology, soil type and depth, and topography and the properties of the drainage network. Each of these factors operates over different scales (e.g., km^2 or larger for climate *vs.* m^2 for soils) and influences water cycling in different ways. An area as vast as the boreal zone exhibits significant heterogeneity in the factors (and their interactive effects) that control the cycling of water and the sediments and nutrients transported by water through boreal ecosystems.

Climate gradients occur both with latitude (primarily a temperature gradient) and longitude (primarily a moisture gradient) (Fig. 4). There is a west to east gradient in annual precipitation, with the western boreal zone receiving considerably less precipitation than the eastern boreal zone (Fig. 4A). When annual precipitation is combined with annual potential evapotranspiration (Fig. 4B), there is an even stronger gradient in the amount of water available for groundwater recharge and surface runoff (Fig. 4C). The seasonality of precipitation must also be considered. For example, in the western boreal zone most precipitation falls in the same months with the highest potential evapotranspiration, whereas, in the eastern boreal zone, precipitation is more evenly distributed throughout the year. Thus, there is a greater potential for drought in the western boreal zone. Broad-scale and re-occurring climate events (e.g., ENSO and PDO) also affect the frequency and intensity of climatic events across Canada (Shabbar and Skinner 2004; Yu and Zwiers 2007).

There is also a gradient in geology across the boreal zone. In general, sedimentary rocks dominate in the western boreal zone, whereas intrusive and metamorphic rocks of the Canadian Shield dominate in the north-central and eastern boreal zone. The surficial deposits of the boreal zone are generally dominated by glacial till blanket and veneer, with glaciolacustrine deposits distributed throughout and glaciomarine deposits surrounding the Hudsons Bay Lowlands (Fulton 1995). The soils overlying these deposits vary substantially across the boreal zone (see Maynard et al. 2014).

In the western boreal zone, the sub-humid climate, permeable bedrock, deep surficial deposits and relatively flat topography means that hydrology is dominated by vertical fluxes (high evapotranspiration and groundwater recharge), soil water storage and low runoff generation (Fig. 5). In contrast, in the eastern boreal zone, the humid climate, impermeable bedrock, shallow substrates and more complex topography means that the hydrology is dominated by lateral fluxes (high runoff), with limited groundwater recharge and high runoff generation (Fig. 5). The prevalence of different dominant hydrologic processes in different regions suggests that the potential for and severity of alterations to hydrologic regime due to impacts of natural and anthropogenic disturbances will also vary.

These same hydrologic regions also explain wetland forms and distribution across the boreal zone. Wetlands are defined as areas with 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 (Tarnocai 1980). Wetlands are classified into bog, fen, swamp, marsh or shallow water (National Wetlands Working Group 1997). Wetlands with more than 40 cm depth of peat development are further classified as peatlands (National Wetlands Working Group 1997). Wetlands form as a result of climate and landscape factors that create a positive water balance (precipitation + inflows > evapotranspiration + outflows). Climate factors that promote wetland development include high precipitation and/or low evapotranspiration. Landscape factors associated with wetland development are geomorphic positions that collect water (i.e., depression areas), that drain slowly (i.e., over confined bedrock), or are connected to regional groundwater flows. Minerotrophic wetlands such as shallow open water, fens, marshes and swamps are normally situated at positions in the landscape lower than adjacent hillslopes, such that water and mineral elements are introduced by groundwater or littoral sources in addition to atmospheric sources (Price and Waddington 2000). Thus, in the boreal zone, these minerotrophic wetlands typically form in bedrock depressions (e.g. Canadian Shield), on flat, poorly drained areas (e.g., Western Plains and Hudson Plain), and in discharge zones at the base of long, steep slopes (e.g., Cordillera). Ombrotrophic bogs rely on atmospheric sources of water. They are hydrologically isolated from lateral inflows or upward seepage and thus occur only where precipitation normally exceeds evapotranspiration during the growing season.

Taken together, the above-mentioned factors demonstrate the difficulty of making generalizations about hydrologic processes in the boreal zone and emphasize how the "uniqueness of place" (Beven 2000) continues to bedevil attempts at generalization and classification of hydrologic regimes. At present, the science of hydrology does not have a generally accepted classification system, unlike many other disciplines, which would permit the definitions of regions or locales where the dominant hydrologic processes are similar. There have been ongoing calls for the development of a hydrologic classification system (e.g. McDonnell and Woods 2004), and many frameworks have been proposed, built around hydrologic similarity (Wagener et al. 2007), hydrologic response units (Devito et al. 2005) or hydrologic landscapes (Winter 2001). All of the frameworks are similar in that they attempt to capture the co-evolution of climate, bedrock geology, surficial geology, soils, topography, and vegetation and how these factors contribute to the local hydrologic regime. Greater communication among groups within the hydrological community (e.g., scientists and engineers) would help in adopting a common classification.A robust hydrologic classification system would be required to extrapolate results from the spatially sparse research sites to the broader landscape.

2.2 Properties and attributes of wetlands in the boreal zone

The boreal zone encompasses both the boreal and subarctic peatland regions (Tarnocai and Stolbovoy 2006), with 67% of the boreal peatlands occurring as bogs, 32% as fens, and the remainder (1%) as swamps and marshes. Two notable extensive boreal peatland areas occur in the Mackenzie River Valley in the northwest and the Hudson Plain in the east, the latter being the single largest peatland complex in North America and the second largest in the northern hemisphere (Gorham 1991; Glooschenko et al. 1994). Wetland type is largely determined by the strength of connection to the groundwater system and the physical and chemical nature of the underlying geology (National Wetlands Working Group 1997). The natural succession of boreal wetlands is from open water to fen and eventually to bog as peat accumulation becomes sufficient to disconnect the peatland from groundwater (Zoltai et al. 1988; Tarnocai and Stolbovoy 2006). In northern parts of the boreal zone, peatlands may contain permafrost (i.e., ground that remains frozen for greater than two years) (Brown 1967). Frozen peat can develop in more southerly latitudes within peatlands where insulating moss (*Sphagnum* spp.) and black spruce (*Picea mariana*) allow frost to persist throughout the year (McLaughlin and Webster 2013).

Peat is classified as soil having 30% or more organic matter (>17% organic carbon) by weight (Soil Classification Working Group 1998) and has a relatively low bulk density (0.07 to 0.25 Mg per m^3) compared to mineral soils (Turchenek et al. 1998). Low bulk density creates high (average of 92%) porosity (volume of pore space as a proportion of total volume). This allows for greater water retention than mineral soils. These properties also affect the movement of water, with high hydraulic conductivity in the top layer of peat (~80% of water is released) and low hydraulic conductivity in deeper peat (~15% of water is released) (Turchenek et al. 1998).

Water table levels within wetlands fluctuate seasonally and annually as a result of precipitation patterns and wetland size and connectivity to surface, shallow subsurface and deeper groundwater sources (Pelster et al. 2008). During dry periods, considerable water can be stored in wetlands that provide small, sustained baseflows to streams and evapotranspiration to the atmosphere (Mitsch and Gosselink 2007). In contrast, during wet periods, the capacity of wetlands to store water is exceeded and the saturated areas act as conduits of overland flow from the uplands to the streams. In areas of permafrost (e.g., Hudson Plain, Mackenzie River Valley), groundwater-surface water interactions are restricted to shallow, localized flows in the surface active layer above permafrost during periods of thaw (Woo and Young 2003).

2.3 Land-water linkages and their influence on water resources

The interactions of the mass and energy cycles and the physical, chemical and biological processes that drive them must be considered to understand the potential impacts of natural resource development on boreal water resources.

Water enters the ecosystem as precipitation. It is first intercepted by surface vegetation, where a portion is returned to the atmosphere by evaporation, and the remainder falls to the ground as throughfall. Water evaporates from plant and soil surfaces or infiltrates and is transpired by vascular vegetation, collectively known as evapotranspiration, and any water remaining is stored in the soil. Upon soil saturation, water flows *via* surface, shallow subsurface or deeper groundwater pathways, creating areas of discharge (upward movement of water to the surface) to surface bodies of water or recharge (downward movement of water) into groundwater storage. Storm and snowmelt intensity (e.g., gentle rain or intense downpour) and the properties of the upper surface of the ground such as soil texture (percentage of sand, silt and clay), amount of organic material (e.g., porous peat materials), and depth of confining layers (e.g., impermeable bedrock or frozen permafrost layer) will determine the dominant flow paths or route that water will take. In general, surface flows predominate where there is steep relief, shallow soils and/or deeper soils with saturated conditions, and deep flows predominate where there is gentle relief, deep soils and unsaturated conditions (Dunne and Leopold 1978).

Forest ecosystems are sources of nutrients to surface water, with important downstream water quality implications (Waring and Schlesinger 1985; Chapin et al. 2011). Within forests, water flow pathways influence water chemistry by determining the materials with which water interacts. Water travelling along shallow flow pathways will interact with organic-rich compounds such as freshly fallen leaves and/or roots and forest floor materials, whereas deeper flow pathways interact with mineral-rich parent materials. The residence time of water moving along the flow path will also influence water quality (McGuire et al. 2005). Residence times are governed by presence or absence of storage structures (e.g., wetlands), the gravitational gradient (e.g., steep *vs.* gentle slope), and the physical structure of the soils (e.g., fine *vs.* coarse materials). Short residence times create oxic conditions and result in less interaction with soils that may leach into the water than longer residence times that have anoxic conditions and more interaction with soils (McGuire et al. 2005). The residence time of water in turn alters the soil chemical environment, which alters the form and thus also the solubility and mobility of nutrients (Chapin et al. 2011).

Water quantity and quality changes throughout the year reflecting changes in precipitation inputs (e.g., snowmelt, summer drought, fall storms), dominant flow pathways, and the dynamic biological and chemical environments along the flow pathways. For example, water chemistry is typically more dilute in spring due to low biological processing under the snowpack and high quantities of available water during snowmelt. In contrast, water chemistry is typically more concentrated in the summer due to high biological processing and low quantities of available water (Band et al. 2001).

In the eastern boreal zone, with shallow upland soils overlying the high relief of the Canadian Shield, water quantity and quality is predominantly determined by variable source area (VSA) regulated near-surface flows (Fig. 5). VSA refers to the concept that runoff-generating areas in the landscape vary in size and configuration over time. In VSA-controlled landscapes, as the groundwater table rises to intersect the surface soils, nutrients that have previously accumulated in the surface soils are mobilized and flushed to the stream (Creed and Sass 2011). Topography, *via* VSA control, influences the hydrologic flushing of nutrients in various ways. It affects (a) the generation of nutrient supply (e.g., nutrient-poor areas develop if soil conditions are too dry or too wet); (b) potential expansion *vs.* contraction rates of the VSAs (e.g., catchments with a greater potential for lateral expansion of source areas will have longer flushing times and higher rates of nutrient export, while catchments with less potential for lateral expansion of source areas will have shorter flushing times and lower rates of nutrient export); and (c) the transport of flushable nutrients to surface waters, which is a function of both the size and configuration of the VSA (e.g., catchments with larger, hydrologically-connected VSAs will have larger nutrient export, whereas catchments with smaller or hydrologically-disconnected VSAs will have lower nutrient export or more leaching to groundwater) (Creed and Sass 2011). This mechanism, or variations of it, has been used to explain the export of carbon (Hornberger et al. 1994; Dillon and Molot 1997; Creed et al. 2003, 2008; Richardson et al. 2009; Mengistu et al. 2013), nitrogen (Creed et al. 1996; Creed and Beall 2009; Mengistu et al. 2013), and phosphorus (Mengistu et al. 2013).

The western boreal zone is characterized by deeper upland soils and lower relief, and water quantity and quality is typically not regulated by VSA dynamics (Fig. 5). In non-VSA-dominated landscapes, the predominance of primarily vertical flows and deep subsurface pathways regulate water and nutrient transfer from land to aquatic systems (Creed and Sass 2011). Thus, consideration has to be given to local, intermediate and regional groundwater flows to understand the chemistry of surface waters. In boreal forests with these complex hydrogeological systems, predicting the chemical responses of surface waters to disturbance is extremely difficult (Devito et al. 2000). However, the size and configuration of surface saturated areas where water flowpaths created by these complex hydrogeological systems intercept with the land surface have been used to explain total phosphorus loading to boreal surface waters (e.g., Devito et al. 2000; Evans et al. 2000) and trophic status of lakes (Sass et al. 2007, 2008*a*, 2008*b*) in the western boreal zone.

Wetlands can have a large impact on water quality, even if they occupy only a small percentage of basin area. Wetlands are unique in that they act as reservoirs of water and bioreactors of nutrients during drier periods and conduits of water and nutrients during wet periods. In particular, there is a strong connection between the presence and extent of wetlands and the concentration of dissolved organic carbon in surface waters (Dillon and Molot 1997; Creed et al. 2003; 2008). Similarly, wetlands have been shown to have an important role in nitrogen and phosphorus loading to surface waters (e.g., Creed and Beall 2009; Mengistu et al. 2013) as well as contributors of mercury to aquatic ecosystems (Branfireun et al. 2005).

2.4 Climate regulation and change

Canada's water-rich boreal zone, including its glaciers, permafrost, snowpack, streams, wetlands, lakes, and groundwater, not only regulates climate, but also is sensitive to predicted changes in climate. Boreal ecosystems regulate weather and climate through both direct mechanisms such as transpiration cooling and albedo effects (Oke 1978; Barnett et al. 2005) and indirect mechanmisms such as freshwater inputs to the Arctic Ocean (Bates et al. 2008) and carbon sequestration (Webster and McLaughlin 2013; Kurz et al. 2013).

Changes to global climate and hydrologic cycles have the potential to strongly impact boreal water resources (Price et al. 2013). The key impacts from climate change on the water cycle will come in the form of extreme events such as droughts and floods, seasonal shifts in flow regimes, and reduced winter ice coverage (NRTEE 2010). These hydrologic changes to the water cycle are likely to have positive feedbacks (i.e., negative outcomes). Climate models show that changes in temperature and precipitation are likely to continue affecting the partitioning of water between evapotranspiration and runoff as well as the amount of water stored in glaciers, snowpack, lakes, wetlands, soils and groundwater. Recent modelling studies suggest that the interior of North America, including the western boreal zone, will become much drier and may no longer be able to support forests (Hogg 1994; Dai 2011, 2013; Price et al. 2013). These large scale vegetation changes are likely to have large impacts on water resources (Baldocchi et al. 2000).

3. Risks from natural resource development to water resources

3.1. Introduction

Water is critical to the operation and future development of the energy, forest, and mining sectors, all of which contribute to Canada's economic wealth and social well-being (NRTEE 2010). The natural resource sectors make a significant contribution to the Canadian economy representing 12.5% of the country's gross domestic product (GDP) in 2009 (Table 2). In 2005, the natural resource sectors, which include thermal power generation, oil and gas, mining, forestry and agriculture, accounted for approximately 84% of both Canada's gross water use (the total volume withdrawn from water bodies) and total water consumed (withdrawn and not replaced) (Statistics Canada 2010). The thermal power-generating sector was responsible for the greatest gross water use, while agriculture accounted for the greatest water consumed (Statistics Canada 2010; NRTEE 2010). The current extent of the anthropogenic footprint related to natural resource development in Canada's boreal zone (Fig. 2) is expected to increase within these sectors by 50 to 60% by 2030, placing even more pressure on Canadian water resources (NRTEE 2010).

3.2. Road development

3.2.1 Introduction

Transportation infrastructure is an artefact of humans interacting with the landscape (Coffin 2007). Roads are necessary for servicing all natural resource industires and the network of roads that cut across the boreal landscape renders vast areas of land as "road-affected" (Coffin 2007). Pasher et al. (2013) estimated that linear disturbance features across the boreal totalled approximately 600 000 km, with roads and seismic exploration lines contributing to more than 80% of the linear disturbance. There is little scientific literature on impacts of roads in the boreal zone even though roads are a pervasive linear disturbance that has no natural corollary. They create gaps, fragment habitats, alter access to humans and predators, provide vectors for introduction of invasive species, change local climate, and influence local hydrology (Trombulak and Frissell 2000). The effects of roads often last well beyond their construction (or even their use), with older roads tending to have greater effects on water resources because of the places in the landscape where they were constructed and because construction practices were more intrusive than more recently constructed roads (Wemple et al. 1996, 2001; Lugo and Gucinski 2000; Wemple and Jones 2003; Dutton et al. 2005). These effects persist as long as the road remains a physical feature, continuing to alter flow routing even after abandonment and revegetation (Trombulak and Frissell 2000).

3.1.2 Water quantity. Road construction associated with natural resource development influences water quantity through the creation of road surfaces, cutbanks, ditches, and culverts (Jones and Grant 1996) that contribute to changes in the timing and routing of runoff (Trombulak and Frissell 2000). Road surfaces create increased overland flow due to an impervious surface layer that creates reduced infiltration capacity (La Marche and Lettenmaier 2001; Tague and Band 2001). Cutbanks intercept upslope subsurface flow (La Marche and Lettenmaier 2001). The depth of the road cut, which varies with local slope, and width of the road will determine the amount of subsurface runoff that is intercepted (Tague and Band 2001). Ditches and culverts provide a conduit of flows directly to the channel, increasing the flow routing efficiency by extending the drainage network (Wemple et al. 1996). Road location on the landscape, specifically position on hillslope profile, influences the effects of roads on hydrology (Jones et al. 2000). However, there are instances where, instead of concentrating flow, ditches and culverts diffuse flow if water is routed to relatively dry areas. Thus, the effect of this concentration of flow will depend upon the characteristics of the receiving area (Tague and Band 2001). Wetland habitats can be destroyed or created with alterations to surface or subsurface flow (Trombulak and Frissell 2000). For example, wetland road crossings may block drainage passages and groundwater flows, effectively raising the upslope water table and lowering the downslope water table (Forman and Alexander 1998; Partington and Gillies 2010).

Roads, in combination with forest harvesting, can have dramatic impacts on both the timing and volume of water during storms (Jones and Grant 1996). However, despite some generalizations that roads have significant effects on peak flow, the impacts will vary with underlying geology, soil texture, vegetation, type of road construction, and both seasonal precipitation and local storm events (Lugo and Gucinski 2000; Tague and Band 2001). For example, in regions with high precipitation and high relief landscapes (e.g., much of the eastern boreal zone, with the exception of the Clay Belt), the net effect of roads is typically a more rapid delivery of water to stream channels during storms, resulting in a higher and earlier peak in storm flow (Jones and Grant 1996; LaMarche and Lettenmaier 2001). Hydrologic effects are likely to persist for as long as the road remains a physical feature altering flow routing (Trombulak and Frissell 2000). Little effort has been expended in understanding the impacts of roads in low-relief landscapes.

3.1.3 Water quality. Roads affect water quality by increasing inputs of dust, generating higher erosion rates and therefore sediment loads, and increasing the turbidity of the waters (Reid and Dunne 1984; Partington and Gillies 2010) thereby impacting aquatic biota (Kreutzweiser et al. 2013). Travel intensity, road surface type, vegetation cover, climate, geologic substrate, road maintenance, and road-stream connectivity are primary factors in regulating sediment production in road systems (Lugo and Gucinski 2000). High rates of sediment production from road surfaces occur in the year immediately following road construction but diminish rapidly over time (Wemple et al. 2001). Trombulak and Frissell (2000) identified five classes of chemicals that roads contribute to the environment: heavy metals, organic molecules, ozone, nutrients, and chemicals for de-icing and dust control. This study indicates that most contamination declines within 20 m of the road but that elevated levels of contaminants often occur 200 m or more from the road (Trombulak and Frissell 2000).

3.1.4 Prognosis

Roads will continue to be a part of industrial landscapes of the boreal zone. It is likely that road networks will become denser and expand farther north in the boreal zone as natural resource exploration, development, and the associated infrastructure increase. Consequently, the influence on water resources from road construction is also likely to increase. Road construction has a lasting legacy. Even if access to roads is restricted, the road footprint never completely disappears, and hydrologic flows continue to be affected.

There is potential to offset the impacts of roads to some degree by mitigation measures. Recent technological advances enable land planners and managers to improve the location of roads, the location and installation of culverts, and the selection of appropriate road surface materials on roads near water. For example, the Government of Alberta has used a technique for mapping wet areas to generate databases available to public to optimize road design (White et al. 2012). In collaboration with Ontario Ministry of Natural Resources, FP Innovations has summarized the tools and resources available for best management practices for roads (Partington and Gillies 2010) and state-of-practice review (Gillies 2011) focussing on the unique challenges for road construction through wetlands.

3.2 Forest Management

3.2.1 Introduction

The forest sector contributed $23.6B (1.9%) to Canada's GDP, $17.2B to Canada's trade balance, and 233 900 direct jobs in 2011 (NRCan 2013*a*). Forest management activities occur in all provinces, and to a much more limited degree in the Yukon and Northwest Territories, with the boreal zone host to a significant fraction of forest management activities in Canada (Fig. 2C). Within the spectrum of forest management activities, including timber harvesting, site preparation, planting, competition management, and stand tending, only timber harvesting activities have received significant research attention with regards to their impact on water resources. Timber harvesting in the boreal zone typically employs clearcut harvesting (e.g., usually clearcuts of less than 260 ha in Ontario's boreal; OMNR undated) systems and post-harvest treatments to emulate the natural pattern of fire-origin, even-age stands of the dominant species (generally spruce, jack pine, lodgepole pine, and aspen) (Long 2009; Sibley et al. 2012). Forest management strategies in the boreal zone have evolved and large, regularly shaped cuts typical of the past have been replaced with smaller, irregularly shaped cuts containing residual material to address biodiversity concerns (Potvin et al. 1999; Venier et al. *in press*). An emerging issue in boreal forestry is biomass harvesting for bioenergy. Until relatively recently, slash (non-merchantable wood comprised of small diameter trees, tops and branches) from the harvest was either left on the ground on-site or de-limbed at the road. Now there are opportunities for use of this biomass in alternative wood products (e.g., wood pellets and chips) or in energy production (Paré et al. 2011). The removal of this material has implications for site productivity and water resources (Thiffault et al. 2010).

Our understanding of how timber harvesting affects water resources is based largely on a century of comparison studies of harvested vs. unharvested paired-basin studies (c.f. Brown et al. 2005 for a recent meta-analysis of these studies). The primary assumption underlying paired-basin studies, that changes in streamflow between a harvested and unharvested basins with similar climate, geology and vegetation isolate and quantify the impacts of harvesting, has proven to be a powerful tool in understanding the effects of timber harvesting. Canada has a long history of paired-basin studies (Buttle et al. 2000; Krezek et al. 2008; Mallik and Teichert 2009). However, not all of these studies have forest management activities as a primary focus and only a small proportion are located in the boreal zone (Fig. 3, Table 4). These paired-basin studies, combined with stand- or site-level studies, have contributed greatly to our understanding of the effects of timber harvesting of water quantity and quality.

3.2.2 Water quantity

In general, reduction of forest cover by timber harvesting (and other natural and anthropogenic disturbances) reduces canopy interception and evapotranspiration, leading to increases in soil moisture (Buttle and Metcalfe 2000; Guillemette et al. 2005; Mallik and Teichert 2009). The characteristics of the topography, soil, and surficial geology will determine how the increase in soil moisture is expressed in terms of increases in groundwater recharge *vs.* runoff. The processes connecting the changes in coverage, composition, and structure of tree canopies due to harvesting and the increase in groundwater recharge or runoff are numerous, primarily driven by the water balance and radiation balance (National Research Council 2008). There are, of course, many localized processes and the exceptions to these general principles demonstrate the importance of the particular combination of climate, geology, topography, and soils (Devito et al. 2005) on the response of a particular basin to forest management activities.

Forest canopy intercepts precipitation where a portion is subsequently returned to the atmosphere by evaporation or delivered to the ground surface as throughfall and stemflow. The amount of precipitation intercepted varies with tree species. Elliott et al. (1998) reported that interception accounted for 58%, 46%, and 16% of precipitation in mature white spruce, jack pine, and aspen stands, respectively, in north central Saskatchewan. Price et al. (1997) reported that black spruce stands in northern Manitoba intercepted between 15 and 60% (seasonal average of 23%) of precipitation, with higher rates of interception occurring during smaller precipitation events (Price et al. 1997).

In the boreal zone, snowfall is a significant contributer to the water balance of most basins. Pomeroy et al. (1998) found that snow interception also differs among forest types, with largest interception in black spruce stands, intermediate in jack pine stands, and smallest in mixed wood (aspen and white spruce) stands. Sublimation of the intercepted snow was approximately 38 to 45%, 30 to 32%, and 10 to 15% for black spruce, jack pine, and mixed wood stands, respectively (Pomeroy et al. 1998). Comparing mature, clearcut, and regenerating stands in northern Saskatchewan, Pomeroy and Granger (1997) found that clearcuts accumulated the most snow, followed by mature mixed wood stands, mature jack pine stands, and regenerating stands (15 years old) had the least accumulation. Similar results were found in other regions, including conifer stands in British Columbia (Winkler et al. 2005) and Alberta (Golding and Swanson 1986), where the effect of forest removal was to increase snow accumulation, resulting in greater delivery of water to the soil substrates.

The size of canopy openings in relation to the surrounding forest influences the radiation balance within forest stands. Increases in solar radiation have pronounced effects on evapotranspiration and snowmelt (Winkler 1999). In a study of a variety of forest types in northern Manitoba, Metcalfe and Buttle (1999) showed that increasing canopy density, measured as gap fractions, reduced incoming short-wave radiation and wind speeds, reducing latent and sensible heat fluxes and subsequent melt rates. Pomeroy and Granger (1997) found that clearcuts underwent an earlier and more rapid melt. Faria et al. (2000), in sites in central Saskatchewan, found that the rate of snow melt was inversely correlated with snow water equivalent (higher melt rates with lower snow water equivalents), and covariance between melt rate and snow water equivalent led to the largest acceleration of snow cover depletion in medium density stands and the smallest acceleration in high density stands.

The impacts of timber harvesting on runoff related to the water and radiation balances, together with site-specific changes to water flow pathways associated with land surface erosion, compaction, rutting, and loss of organic matter, have been the subject of many general reviews (e.g., Bosch and Hewlett 1982; Stednick 1996; Brown et al. 2005), including several examining the Canadian experience (Buttle et al. 2000, 2005, 2009; Moore and Wondzell 2005; Mallik and Teichert 2009). The majority of these studies focused on how timber harvesting affects peak flows because of concerns for flooding and associated increases in stream scouring and bank undercutting, which in turn can affect water quality and aquatic habitats downstream through transport and subsequent deposition of sediment (Alila and Beckers 2001). Studies examining the effects of timber harvesting on base flows and long-term (> five years) recovery are largely absent.

In the eastern boreal zone, dominated by a hydrologic regime characterized by high precipitation and relatively high relief, runoff responses to forest management are extremely variable, ranging from negligible to moderate depending on the amount of basin disturbed and the prevalence of skid trails and road ditches. The "Ruisseau des Eaux-Volées" Experimental Watershed (REVEW) in boreal Quebec, forested primarily by balsam fir but with some white spruce, white birch, and black spruce, has been the site of a number of studies on timber harvesting impacts of streamflow. Plamondon and Ouellet (1980) reported on the effects of harvesting 31% of a 394 ha basin and found no changes in annual, seasonal, and monthly runoff, and no changes in the timing of peak and base flows. A subsequent study demonstrated that harvesting 85% of a 122 ha basin increased peakflow by 63% and reduced lag times in the storm hydrograph (Guillemette et al. 2005). The relatively large peak flow response, when compared to 50 other similar studies globally, was attributed to increased channelized flow to the basin outlet due to skid trails and road ditches (Guillemette at al. 2005). Another REVEW study compared harvesting 50% of four small basins (< 50 ha) with harvested areas distributed at different distances from the stream and found that the configuration of the harvests had little effect on peak flows (Tremblay et al. 2008). A study of larger basins (400 to 11 900 ha) in the boreal forest of northeastern Ontario, with uplands dominated by black and white spruce with minor occurrence of balsam fir, jack pine, white birch, and trembling aspen, and lowland areas dominated by tamarack and black spruce, where the proportion of the basins harvested varied from 5 to 25%, reported no definitive effects on annual runoff or peak flow timing or magnitude (Buttle and Metcalfe 2000). They did, however, report that there was some augmentation of base flows in summer months and suggested the muted responses reflected the ability of large basins to buffer the response to timber harvesting. Metcalfe and Buttle (1999) noted that water storage and evaporation in small wetlands and ephemeral surface depressions is a fundamental component of the basin water balance that may influence runoff response.

In the western boreal zone, dominated by a hydrologic regime characterized by low precipitation and relief, runoff responses were more difficult to discern. In the aspen dominated mixedwood forest of northeastern Alberta, Devito et al. (2005) found that harvesting effects were largely obscured by interannual variation in precipitation and soil water storage potential. In these areas, where deep sedimentary deposits are interspersed with clay lenses, greater infiltration and increased groundwater recharge lead to raised water tables following harvest (Smerdon et al. 2009) rather than creating runoff. Runoff generation only occurred under exceptional circumstances (one in 20 years) and within specific landscape units (ephemeral draws and wetlands) that were only rarely hydrologically-connected to surface waters.

Swanson and Hillman (1977) examined the effects of timber harvesting in nine control and nine harvested basins in the foothills of the Rocky Mountains in west-central Alberta, where precipitation and relief are much greater than the boreal plains to the east. In this lodgepole pine – white spruce forest, they reported that snowmelt runoff increased 59%, growing season runoff increased 27%, and peak flow response to summer storms increased 1.5 to 2 times. In another foothills basin (Streeter Creek), a 50% harvest distributed as small (<1 ha) cutblocks increased growing season runoff by 175% (Swanson et al. 1986).

3.2.3 Water quality

Timber harvesting causes many physical, geochemical, and biological changes to a forest ecosystem. Alterations to the soil's temperature, structure and nutrient cycling (Table 5) can affect the character of soil pore water, which is subsequently reflected in surface water chemistry as water is transported through the landscape by hydrologic flows, impacting downstream energy and nutrient cycling (Buttle et al. 2000) and aquatic biodiversity (Kreutzweiser et al. 2013).

3.2.3.1 Water temperature. The impacts of timber harvesting on stream water temperatures are variable. Timber harvesting results in higher incident radiation on the soil surface that results in an increase in soil temperatures. Timber harvesting also results in changes in wind patterns that influences soil temperatures. Changes in soil temperature influence temperatures of saturated and unsaturated water flow and, thus, the temperatures in downstream aquatic ecosystems (Johnson and Jones 2000). Temperatures in small headwater streams in sub-boreal Engelmann spruce and subalpine fir forest ecosystems of British Columbia remained four to six degrees warmer and diurnal temperature variation remained higher than in the control streams regardless of riparian buffer retentions five years after the completion of timber harvesting treatments (MacDonald et al. 2003a). Although initially the high-retention treatment acted to mitigate changes in temperature, successive years of wind throw reduced forest canopy density creating temperature impacts equivalent to a clearcut (MacDonald et al. 2003a). In contrast, in more eastern parts of the boreal zone, Tremblay et al. 2009 found that summer daily maximum and minimum stream temperatures remained within ± 1°C in balsam fir dominated forests that were clearcut but had a 20 m riparian buffer. Similarly, Kreutzweiser et al. (2009*b*) found that only at the most intensively harvested riparian buffers were there significant (~4°C) increases in daily maximum stream temperatures for about six weeks in the first summer in boreal mixedwood stands after logging than in pre-logging years or reference sites, after which temperatures returned to normal. Steedman et al. (2001) found that it was only early summer littoral water temperatures of small lakes in clearcut shorelines of northwestern Ontario that were associated with increases of 1 to 2°C in maximum temperatures and increases of 0.3 to 0.6°C in average diurnal temperature range, compared with undisturbed shorelines or shorelines with 30 m riparian buffer strips three years after harvesting. Timber harvesting has been shown to have variable effects on water temperatures, with the largest impacts usually in early season of the year following canopy openings when the riparian canopy is drastically reduced either by harvesting or post-harvest wind throw.

3.2.3.2 Suspended sediments. Erosion occurs when there has been physical disruption to the soil surface (e.g., from rutting or mechanical site preparation) and where local topography results in increased soil moisture or channelized flows that cause slope failure and creates delivery channels to receiving waters (Hutchinson and Moore 2000; Lewis et al. 2001; but see MacDonald et al. 2003*b*). Erosion results in the transport of sediment into adjacent streams, wetlands and lakes. This material takes with it nutrients that are embedded within or sorbed (*i.e.,* adhered) onto it (Prepas et al. 2006). MacDonald et al. (2003*b*) found modest increases in suspended sediments during snowmelt in headwater streams in two harvested basins immediately post-harvest, but declined rapidly within two or three years. Tremblay et al. (2009) also observed increased suspended sediment concentrations post-harvest, although there was not enough pre-harvest data to statistically confirm this result. Kreutzweiser et al. (2009*a*) measured fine sediments in three harvested boreal catchments that included partially-harvest riparian buffers, and detected significant post-harvest increases in fine sediments at only one of three sites, and only in the first year after harvest. In addition to the sediments from erosion, Steedman and France (2000) found that wind-blown sediment from black spruce - jack pine clearcuts, roads, and skid trails, where soil disturbance is large and able to reach the lake, may have led to elevated levels of littoral sedimentation, although it was thought this mechanism would not cause important changes in water quality. Although some studies do not examine sedimentation directly, many have found that water clarity is reduced by timber harvesting over a variable range in time (one to four years) following harvest (Carignan et al. 2000; France et al. 2000; Steedman 2000; Steedman and Kushneriuk 2000; Knapp et al. 2003; Garcia et al. 2007; Bertolo and Magnan 2007; Winkler et al. 2009) although one study found no effect on water clarity (Prepas et al. 2001*a*).

3.2.3.3 Dissolved organic carbon and nutrients. Changes occur to soil nutrient cycles post-harvest (see review by Kreutzweiser et al. 2008). Nutrient cycles such as carbon, nitrogen and phosphorus are driven primarily by microbial activities. Timber harvesting affects both the environmental constraints on and the sources of organic material for decomposition (Kreutzweiser et al. 2008). The warm, mesic and aerated conditions created from timber harvesting operations are ideal for microbial metabolism and their mineralization of organic matter. The rate and timing of decomposition is also constrained by the amount of organic material left following harvest and its quality (i.e., more easily degradable leaf or needle litter and fine roots *vs.* less easily degradable branches and stems) (Hazlett et al. 2007). The amount and type of residue (slash) retained on the site will differ depending on timber harvest approach and whether whole tree (de-limbing at road, less residue in plot) or tree-length (de-limbed where cut, more residue on plot) methods were used. Following timber harvest, the demand for nutrients from vegetation is low because of the removal of plants and the lag time in regeneration (Kreutzweiser et al. 2008). Thus, high concentrations of nutrients can accumulate in the soil and soil pore water, creating the potential for leaching into groundwater or loading to downstream ecosystems (Kreutzweiser et al. 2008).

Dissolved Organic Carbon (DOC): Disruption of the forest floor containing easily decomposed residues (leaves and bark) or damage to wetlands where there is deeper organic layers in combination with changes to soil environmental conditions that enhance microbial activity following harvesting can create hydrologically mobile sources of dissolved organic carbon (DOC). A rise in groundwater level following timber harvesting can mobilize DOC both during peak and base flow conditions (Laudon et al. 2009). Most studies reviewed found an increase in surface water DOC concentrations following harvest that were maintained for a few years followed by a decline towards pre-harvest conditions in lakes (Carignan et al. 2000; France et al. 2000; Garcia et al. 2007; Lamontagne et al. 2000; Steedman 2000; Bertolo and Magnan 2007; Hausmann and Pienitz 2009; Winkler et al. 2009). A few studies reviewed showed no effect in streams (Hillman et al. 1997) or lakes (Knapp et al. 2003; Desrosiers et al. 2006), and one study showed a negative effect in lakes (France et al. 1996) of harvesting on surface water DOC concentrations. In a survey of lakes in the eastern boreal zone, the average increase in concentration of DOC attributable to harvesting was 2 mg/L, with a range of -2 to +5 mg/L (France et al. 2000).-The specific silvicultural practices applied have an impact on how long surface water DOC concentrations remained elevated. For example, Schelker et al. (2012) determined that while clearcutting increased DOC concentrations, site preparation (intentional ground disturbance in preparation for planting) had an even more profound effect on DOC concentration. However, since DOC concentrations in undisturbed basins are closely linked to presence and extent of permanently or transiently saturated soils (Dillon and Molot 1997; Creed et al. 2003, 2008), the presence of these features may confound harvesting effects (Hillman et al. 1997).

Nitrogen: Timeber harvesting effects on nitrogen mobility and export are mediated by microbial processes affecting mineralization and nitrification (Mallik and Teichert 2009). Studies examining the impacts of timber harvesting on nitrogen in surface waters have observed a variety of responses in boreal lakes. These include increases in total nitrogen (Lamontagne et al. 2000; Steedman 2000; Garcia et al. 2007; Hausmann and Pienitz 2009), organic nitrogen (Carignan et al. 2000) and inorganic nitrate-nitrogen (Tremblay et al. 2009). Other studies have shown no effect on total nitrogen (Garcia and Carignan 2000; Prepas et al. 2001*a*; Bertolo and Magnan 2007; Winkler et al. 2009) or nitrate-nitrogen (Carignan et al. 2000; Lamontagne et al. 2000). Variation in nitrogen response to harvesting is due to local site factors such as vegetation type controlling carbon to nitrogen ratio, local environmental conditions such as temperature and moisture controlling rates of mineralization, and local topography affecting its aquatic *vs.* atmospheric fate (Holmes and Zak 1999; Lamontagne 2000).

Phosphorus: Timber harvesting effects on phosphorus export is related to soil erosion and soil properties that influence biotically controlled mineralization processes or abiotically controlled adsorption and desorption processes. Landscape position affects these processes (Mallik and Teichert 2009). Studies examining the impacts of forestry on phosphorus loading to surface waters reveal a variable response. Several studies observed increased concentrations of total phosphorus in lakes (Carignan et al. 2000; Lamontagne et al. 2000; Prepas et al. 2001*a*, Garcia et al. 2007; Hausmann and Pienitz 2009; Winkler et al. 2009). Other studies observed no change in concentrations of total phosphorus concentrations (Evans et al. 2000; Steedman 2000; Bertolo and Magnan 2007) or a decrease in total phosphorus concentrations (France et al. 1996) or phosphate (Tremblay et al. 2009) in lakes.

Base cations and anions: Base cation (calcium, magnesium, sodium, potassium) and anion (chloride, sulphate) exports are controlled primarily by soils, geology, pH, and organic matter associations rather than decomposition processes. Base cations in water were generally observed to increase following timber harvesting. Increases were observed for calcium (Carignan et al. 2000; Lamontagne et al. 2000), magnesium (Tremblay et al. 2009), sodium (Lamontagne et al. 2000), and potassium (Carignan et al. 2000; Lamontagne et al. 2000; Tremblay et al. 2009). However, Steedman (2000) observed decreases in calcium and magnesium. Trends in acid anions are less well studied, and include sulphate increases (Carignan et al. 2000) or no sulphate change (Garcia and Carignan 2000; Prepas et al. 2001*a*; Hausmann and Pienitz 2009), and chloride increases (Carignan et al. 2000; Lamontagne et al. 2000; Steedman 2000) or no chloride change (Prepas et al. 2001*a*; Hausmann and Pienitz 2009). Water pH is primarily driven by balances in cations and anions and, in general, timber harvesting appeared to have little effect on pH (Hillman et al. 1997; Prepas et al. 2001*a*; Steedman 2000; Garcia et al. 2007; Hausmann and Pienitz 2009), although Tremblay et al. (2009) found a slight decline in pH following timber harvesting.

Studies investigating the impacts of forestry on water quality parameters within the boreal zone have tended to focus more on lakes than streams, with more studies from the eastern boreal zone compared to the western boreal zone. Although the majority of studies have shown short-term increases in many of the nutrients, it is not well known what the longer-term changes are following timber harvesting and stand regrowth. Nutrients may differ in their responses, with mobile nutrients (potassium, chloride, sulphate, nitrate) that are released following a harvest being rapidly flushed out of the basin (e.g., 50% decrease in three years), while other nutrients show little change or continued increases after three years (Carignan et al. 2000). The rate of decline in nutrient export will be determined by microbial demand for nutrients during decomposition, vegetation demand for nutrient uptake, and water availability for solubilising and mobilizing nutrients.

3.2.3.5 Mercury. Forestry operations have been shown to increase mercury and methyl mercury output from boreal catchments (Porvari et al. 2003). Clearcutting and/or site preparation significantly increases the mobility of total mercury and methyl mercury accumulated in forest soil and may be an important factor for the total input of mercury to boreal freshwater ecosystems (Porvari et al. 2003). Garcia et al.'s (2007) study in the Haute-Mauricie region of northwest Quebec found that the concentration of methyl mercury in lakes where forests were harvested were significantly higher than in reference lakes, and these changes paralleled those in DOC concentrations. Similar to what was observed with DOC, Sørensen et al. (2009) suggested that the majority of mercury outputs are primarily from site preparation, and not from timber harvesting, and therefore mitigative measures can be taken to reduce contamination of surface waters by mercury caused by site preparation.

3.2.4 Prognosis

More than a century of research into interactions between forestry and water resources has led to significant improvements in forest management practices to protect water resources (Ice et al. 2010). Significant improvements have been made in forest management planning, beneficial management practices, and certification standards. Improving road placement (White et al. 2012), restricting machine activity in wet areas (e.g., OSIC 2011), and optimizing the design of riparian buffers (Creed et al. 2008) are examples of changes that have led (or could lead) to improvements in water quality particularly in reducing sediment and nutrient runoff.

The role that riparian buffers play in mitigating forest management impacts on water resources has been evaluated (Buttle 2002). Fixed-width riparian buffers have become the norm for streamside protection from forest management activities, due to their relatively easy implementation in forest planning, yet few experiments have been done to test the efficacy of riparian buffers of a particular width or explore site or landscape-specific modifications (Richardson et al. 2012). A couple of examples from the boreal zone suggest that the uncut “donuts” are not essential for maintaining water quality. Tremblay et al. (2009) found that the proximity of the cutblock to the stream network and logging within riparian buffers did not appear to affect water quality, however the harvest was only of moderate intensity (50% cut). Prepas et al.'s (2001*b*) study within the western boreal forest determined that where climatic and physiographic variability produces complex hydrologic pathways, standard width riparian buffers are not adequate for protecting aquatic systems. In the same region, Creed et al. (2008) developed a remote sensing technique where a probability map of wet area formation was calculated from a time series of remotely sensed images and related to the return period of discharges from the basin. The relationship between wet area and return period was proposed as an approach for estimating risk associated with harvesting in these critical areas, enabling land managers to design variable width riparian buffers based on the level of risk they are willing to accept with respect to increasing nutrient loading to surface waters following harvesting (Creed et al. 2008). A move away from a "one size fits all" approach to the design of riparian buffers towards a customization of buffer widths that considers relevant landscape-specific factors based on easily derived terrain information could further strengthen forest management planning.

Modern forest management strategies aim to emulate natural disturbance as much as possible in its methods (Hunter 1993). This has implications for the protection of water resources, because natural disturbances often occur in riparian buffers, and emulating those disturbances will require timber harvesting closer to water than previously allowed under conventional riparian buffers (Kreutzweiser et al. 2012). This emphasis on emulation of natural disturbance is providing impetus for new directions in basin and riparian forest management (Naylor et al. 2012), but implementation across the boreal zone will need to consider and adjust for landscape contexts including regional hydrologic regimes and disturbance patterns (Sibley et al. 2012). Implementation will also need to consider conditions under which riparian buffer harvesting does not mimic natural disturbances. Although achieving harvest patterns that are similar to fire patterns near water can be successful, creating similar response in processes is more difficult. For example, fires in riparian buffers result in standing dead trees that can provide shade (temperature control) and pulsed inputs of large wood (organic matter input and retention) to streams, whereas harvesting in riparian buffers generally does not (Moore and Richardson 2012). A similar problem is encountered with emulating the impacts of fire on water quality. In a series of syntheses, Buttle et al. (2000), Carignan and Steedman (2000), Pinel-Alloul et al. (2002) and Nitschke (2005) showed that there were substantial differences between fire and harvesting. For example, DOC, mercury, sodium, and potassium responded more strongly to harvesting whereas nitrate-nitrogen, phosphorus, calcium, and magnesium responded more strongly to fire, with other parameters responding similarly.

Given the importance of forestry operations in the boreal zone, a large knowledge gap remains in understanding the persistence of impacts of timber harvesting activities. Most studies do not continue beyond two or three years post-harvest. While studies of this duration may capture recovery of rapidly responding effects, it may not capture some of the longer-term impacts or the time scales of recovery in runoff and water quality that have been observed in some forest ecosystems (Carignan and Steedman 2000; Nitschke 2005). Furthermore, studies on the impacts of forestry on water resources have been predominantly conducted outside of the boreal zone or within the eastern boreal zone and the applicability of these results to the entire boreal zone is unknown. Also, studies on the impacts of forestry on water quality within the boreal zone have focused primarily on lakes, not streams. Although some general trends in water quality parameters were observed, there was considerable variability and inconsistency in water quality observations, reflecting the substantial spatial heterogeneity and temporal variability of drivers of hydrologic and biogeochemical processes within the boreal zone (*sensu* Devito et al. 2005), preventing a predictive understanding of the impacts of forest management on water quality (Carignan and Steedman 2000; Pinel-Alloul et al. 2002; Kreutzweiser et al. 2008).

Several studies have considered the impacts of harvesting intensity and typically the impacts became more pronounced with increasing intensity of harvest, when normalized by the lake surface area or volume (Carignan et al. 2000; Pinel-Alloul et al. 2002). However, forest management studies have typically focused on low order, headwater basins, and the spatially cumulative impacts of timber harvesting at larger scales has rarely been evaluated (Buttle and Metcalfe 2000), or in combination with other landscape disturbances (Buttle et al. 2005). Furthermore, scaling impacts from the stand level to the basin or landscape scale still remains a challenge.

.

3.2.5 Unique considerations for timber harvesting within peatlands

3.2.5.1 Introduction. Timber harvesting in treed peatlands presents unique challenges for forestry. Timber harvesting within peatlands is not a widespread practise in the Canadian boreal zone as it is in the boreal zone of northern Europe (Poulin and Pellerin 2001), although it occurs in certain areas, such as the Clay Belt in Ontario and Quebec and in north central Alberta. Tree productivity in peatlands is generally low because of a high water table, poor aeration (Campbell 1980), low substrate temperature (Lieffers and Rothwell 1987), and inadequate nutrient availability (Tilton 1978). However, treed peatlands have marketable black spruce (*Picea mariana)* and tamarak (*Larix laricina)* that are harvested primarily as high-quality pulpwood (Locky and Bayley 2007).

3.2.5.2 Water quantity. In treed peatlands, water table increases after harvesting is referred to as "watering-up" (Dubé et al. 1995). Watering-up is mainly due to a reduction in interception and evaporation of precipitation and reduction in transpiration following forest cover removal (Dubé et al. 1995) as well as flow path disturbances during forestry operations (Aust et al. 1997). Immediately following timber harvest, water table levels can increase by 4 to 22 cm (Berry and Jeglum 1991; Dubé et al. 1995; Roy et al. 1997, 2000). These elevated water tables can persist for many years. Pothier et al. (2003) did not detect water table recovery five years after harvesting, and Marcotte et al. (2008) found that water table levels in undrained plots were still 5 to 7 cm higher than the pre-cut levels 10 years after clearcutting, which was attributed to precipitation interception by vegetation being only 50% of the pre-harvest value. Watering-up creates conditions conducive for paludification (i.e., peat accumulation) and is considered a threat to future forest productivity (Lavoie et al. 2005). Silvicultural practices within peatlands commonly include draining to improve forest productivity prior to harvest (Poulin et al. 2004) and after harvesting (Roy et al. 2000) to improve soil conditions for future tree regrowth by increasing the depth of aeration and allowing previously shallow rooting systems to expand deeper into the soil (Lieffers and Rothwell 1987; Prévost et al. 1997;

Silins and Rothwell 1999; Roy et al. 2000), although this practice would have negative implications for atmospheric carbon exchange (Kurz et al. 2013). Watering up can alter local downstream flows, although there has been no recent work to quantify the potential magnitude of impact. Drainage also impacts downstream flows, with one study showing drainage increased and sustained summer low flows by 25% (Prévost et al. 1999).

3.2.5.3 Water quality. The increase in water table following timber harvest in treed peatlands creates an anoxic environment (Aust et al. 1993), root growth (Kozlowski 1984; Lieffers and Rothwell 1986), and nitrate assimilation (Morris 1997). Locky and Bayley (2007) and Prévost et al. (1999) found that surface waters in young clearcuts had significantly larger nutrient concentrations compared with controls, likely due to soil warming. Peatland draining prior to timber harvest causes changes in water chemistry, opposite to that of watering up, as a result of increased oxygen transport into the peat (Silins and Rothwell 1998). Numerous negative effects of peatland draining have been observed, including increases in suspended sediments, nutrients, specific conductivity, and pH of peat soil water (Lieffers and Rothwell 1987; Lieffers 1988; Rothwell et al. 1993; Sheehy 1993; Paavilainen and Païvänen 1995; Prévost et al. 1997, 1999; Lavoie et al. 2005). These water chemistry changes have the potential to influence downstream water quality during periods when the peatlands are hydrologically connected in the regional flow network (e.g., during ditching and in high flows in following weeks, to months or even years [Prévost et al. 1999]).

3.2.5.4 Prognosis. Changes in demand for different wood products will determine if forest management activities within treed peatlands are likely to expand or decline.

Extensive research that occurred during eras when demand for forest products from peatlands was high and from the European experience has identified many beneficial management practices to minimize impacts on water resources. Although peatland draining following clearcutting or pre-commercial harvesting helps in the recovery from “watering up”, it is considered a costly and remedial approach to the problem (Marcotte et al. 2008). For example, drainage should be limited to the first cohort stand following harvesting (Lavoie et al. 2005) because the improvement in tree growth may be relatively small without additional fertilization (Jutras et al. 2002). Preventive measures, including silvicultural treatments that promote regeneration and evapotranspiration, along with protecting understory vegetation, should be employed to limit water table rise (Lavoie et al. 2005). Partial and shelterwood cutting methods are among the best options, since watering-up was found to be roughly proportional to cutting intensity (Päivänen 1980; Pothier et al. 2003). Partial cutting scenarios also conserve the vertical complexity of the stand that is essential to the recovery process (Marcotte et al. 2008). This requires the use of appropriate equipment to minimize site disturbances while the ground is frozen (Locky and Bayley 2007). Applying these practices will help in minimizing negative impacts to water resources within commercially viable forested peatlands.

3.3 Pulp and paper operations

3.3.1 Introduction

Pulp and paper is an important component of Canada's forest product industry and in 2010 contributed to 0.7% of the GDP (NRCan 2013*a*) and $9.8 B per year in exports from newsprint and wood pulp (NRCan 2011). During the same year, Canada ranked first in world production of newsprint (13.8%) and second in world production of wood pulp (11%). Boreal forests are the dominant contributor to the pulp and paper industry. As of 2011, there were 17 active and 7 closed or decommissioned pulp and paper mills in the boreal portion of the commercial forest (Fig 3. in Brandt et al. 2013).

The pulp and paper industry is capital-intensive, characterized by complex system processes to convert cellulose fiber from trees into a wide variety of traditional products such as pulps, papers, and paperboards (Environment Canada 2012*a*) and emerging products such as fibre-bioplastic composites (NRCan 2013*b*). Wood is reduced to fiber either by cooking in chemicals or by mechanical means (Environment Canada 2012*a*). The fibers are then mixed with water, adhering to one another as the water is removed by pressure and heat. Chemical pulping (kraft and sulfite process) uses sulphur to extract fiber that is exceptionally strong and is used for magazines, printing and graphics papers, grocery bags, and corrugated packaging (Biermann 1993). Mechanical pulping mills physically shred wood into pulp with grindstones and/or heat to produce pulp that has weaker fibers and is commonly used for newspapers (Biermann 1993).

3.3.2 Water quantity

Large volumes of water are required for processing and cooling during pulp and paper production, however most of the water is not consumed. The majority of water is taken from surface waters (98%), primarily rivers (76%) and lakes (19%) (NCASI 2010). About 92% of the water used by pulp and paper processes is returned directly to the surface waters (river 74%, lake 13%) following treatment. 8% is evaporated during manufacturing and wastewater treatment, and about 0.4% is imparted to products or solid residuals (NCASI 2010). The large amounts of water used in processing and cooling reappear as effluent (Pokhrel and Viraraghavan 2004). It is estimated that mills release between 50 000 to 150 000 m^3 of effluent per day (Chambers et al. 2001; Hewitt et al. 2006) and in 2007 total national effluents flows were estimated at 1 560 × 10^6 m^3 with an additional 229 × 10^6 m^3 in non-contact cooling water flow (NCASI 2010). Over a period of 17 years (1992 and 2009), water use intensity within the industry has declined from ~85 m^3 per tonne to ~55 m^3 per tonne (FPAC 2011). There is no evidence to suggest that boreal water quantity is significantly affected by water use in the pulp and paper industry.

3.3.3 Water quality

Pulp mill effluent is a known pollutant of surface waters (Walden 1976). Pollutants are generated at various stages of pulping and paper making process (Pokhrel and Viraraghavan 2004). Effluents are a complex combination of waste streams produced in debarking, pulp washing, bleaching, and regeneration of cooking chemicals (Pokhrel and Viraraghavan 2004; Hewitt et al. 2006). Chemicals in effluents may be from the wood itself or from chemicals added during the pulping and bleacing process and include suspended solids, lignins, resins, fatty acids, volatile organic carbon (e.g., terpenes, alcohols, phenols, methanol, acetone, chloroform, etc.), mercury and inorganic chlorine and organochlorine compounds.

During the 1980s, pulp and paper production worldwide became an area of increased environmental scrutiny by the public as dioxins and furans in effluents and paper products were found to have toxic effects on aquatic organisms (McMaster et al. 2006*b*; Kreutzweiser et al. 2013). As a result of studies conducted internationally and in Canada, new regulations came into force in 1992 to set revised limits for biochemical oxygen demand (BOD; the amount of oxygen needed to decompose organic matter), total suspended solids (TSS), and dioxins and furans (Environment Canada 2005*b*). To meet the new regulatory limits, the industry was required to make process changes (e.g., switching from elemental chlorine to chlorine dioxide in the bleaching process) and treatment changes to reduce pollution and environmental impact (McMaster et al. 2006*b*). Effluents are now treated with both primary treatment to remove solids in settling basins, and secondary treatment to adsorb, settle, and promote microbial breakdown of biodegradable material in order to reduce BOD and levels of toxic organic compounds (Pokhrel and Viraraghavan 2004; Hewitt et al. 2006).

Since the Pulp and Paper Effluent Regulations under the Fisheries Act came into force in 1992, there have been declining trends in discharge loads of total suspended solids, adsorbable organic halides, dioxins, and furans and declines in BOD (Chambers et al. 2000; NCASI 2010). Dioxins have been virtually eliminated through process changes in mills and decreased toxicity of effluents through biological effluent treatment, reducing BOD substances by 90% and suspended solids by 70% (FPAC 2011).

Changes to effluent treatment to reduce contaminants have not reduced nutrient load. Eutrophication that results from elevated nutrient concentrations in water remains a predominant environmental issue for pulp mill effluents (Bothwell 1992; Biermann 1996, Chambers et al. 2000, 2006). This is particularly a concern in northern boreal rivers where they are naturally nutrient deplete (Chambers et al. 2000). The increased nutrient concentrations result in increased production of phytoplankton and aquatic plants, cause changes in the abundance and composition of consumers, and contribute to declines in dissolved oxygen (Smith et al. 1999). Research conducted as part of the Northern River Basins Study (NRBS) in the western boreal zone (see Gummer et al. 2000) found elevated levels of total nitrogen and total phosphorus in the Athabasca and Wapiti rivers (Chambers et al. 2006). Pulp mills contributed 22% of the P and 20% of the N load discharged from the Wapiti to the Smoky river, and 6 to 16% of the P load and 4 to 10% of the N load in the Athabasca River (Chambers et al. 2000). These nutrient concentrations were elevated downstream of the pulp mills, particularly during the low discharge periods of fall and winter (September to April; Chambers et al. 2000). These elevated nutrient concentrations extended 4 to 120 km downstream of the point of effluent discharge (Chambers et al. 2000).

3.3.4 Prognosis

The GDP attributed to the Pulp, Paper and Paperboard Mills industry has decreased from $8.1 billion in 2002 to $5.7 billion in 2011. Global market trends, including increased demand for paper for small printers but declines in newsprint coupled with changing markets in which there is greater use of recycled fibre and fibre from plantations (Whitman 2005), has affected the viability of the Canadian pulp and paper industry (FPAC 2011). Changes or recovery in traditional markets and acceptance and application of new technologies will impact how this sector develops into the future.

Declining newsprint demand and improved effluent treatments indicate that effects of toxic compounds in pulp mill effluents on water quality are likely to decline in the boreal zone. Furthermore, compliance with toxic substance control regulations over the past couple of decades has resulted in improved water quality of pulp mill effluents (Environment Canada 2005*b*). However, studies from the western boreal forest indicate that nutrient additions from mill effluents continue to be a persistent challenge and will pose risks of eutrophication effects in downstream water bodies. Schindler and Lee (2010) point out that eutrophication in riverine boreal lakes is an increasing problem, and they suggest that the contribution of pulp mill effluents has to be considered along with cumulative effects of municipal sewage, agricultural, and shoreline developments. As a result of research from the Northern River Basin Study, nutrient loading to the Athabasca and Wapiti rivers has improved and pulp mills are required to develop nutrient minimization plans and investigate new methods to reduce nutrient discharge (Chambers et al. 2000). These examples from the western boreal suggest that continued improvements in effluent processing technologies are warranted to further reduce nutrient and suspended sediment loads.

3.4 Electric power development

3.4.1 Introduction

Canada is the third largest producer of hydroelectricity in the world (behind China and Brazil) accounting for approximately 10.5% of the world's total production (Observ'ER 2011). In 2010, electricity generation in Canada amounted to 588.9 terawatt hours (NRCan 2011), with hydroelectric generation representing 59.1% of total generation, and thermal sources (e.g., coal, nuclear, natural gas and petroleum products and waste) contributing 23.4% (NRCan 2011). In the same year, Canada exported $2.2B in electricity, exclusively to the United States (NRCan 2011).

In 2000, there were 713 large dams greater than 5 m in crest height across Canada in the boreal zone (Brandt et al. 2013). These dams produce a large portion of Canada's hydroelectric power, with 39% percent of Canada's hydroelectric capacity generated from rivers arising in or flowing through the Boreal Shield ecozone alone (as defined by ESWG 1996) (Urquizo et al. 2000) (Fig. 9 in Brandt et al. 2013). Two hydroelectric power projects within the boreal zone, Le Grande complex (Quebec) and Churchill Falls (Labrador), are ranked among the largest projects in the world in terms of capacity and reservoir size (International Hydropower Association 2007; Lewis 2012). Other large dams in the boreal zone can be found on the Moose River in Ontario, Nelson River in Manitoba, Saskatchewan River in Saskatchewan, and Peace River in British Columbia.

Canada has diverted more water by damming rivers than any other country (Dynesius and Nilsson 1994; Ghassemi and White 2007). Almost all of the diverted water volume in Canada (97%) is used for hydroelectric power generation (Lasserre 2007). Although the distribution of dams is considerable (Fig. 7), the boreal zone remains an area with the greatest number of large, undammed, free-flowing river systems in North America (Fig. 8, Dynesius and Nilsson 1994).

The impacts of hydroelectric power generation on water resources differ depending on the mode of generation, whether it is a conventional hydroelectric, thermal, or run-of-the-river installation (Baxter 1977; Renöfält et al. 2010). Conventional hydroelectric power requires construction of dams to impound water to generate electricity and is the most common type of power generation in the boreal zone. Thermal power generation uses heat energy from fossil fuels or uranium to produce steam to drive turbines. Run-of-the-river installations are small, low capacity hydroelectric power installations where natural flow of the river itself is used, requiring little to no water retention within reservoirs. In addition to generating stations and impoundments, hydroelectric power developments also bring with them thousands of kilometres of transmission lines along linear corridors that are cleared of trees and maintained by herbicides and cutting (Urquizo et al. 2000; Wells et al. 2010), creating impacts, at least initially, that are similar to forest management activities (see Section 3.2).

3.4.2 Water quantity. Of all the modes of hydroelectric power generation, conventional hydroelectric dams have the largest impact on boreal water resources, with impacts both upstream and downstream of the dam (Baxter 1977; Rosenberg et al. 1995, 1997; Environment Canada 2004; Woo and Thorne 2009; Renöfält et al. 2010). Direct and often obvious *upstream* impacts include flooding of riparian buffers, wetland and upland habitats, conversion of lotic (flowing) environments to lentic (standing water) systems, shoreline erosion with water level fluctuations (including thermokarst slumping where permafrost exists), and altered groundwater recharge patterns. The amount of water impounded upstream of a dam differs greatly from one site to another depending on the local topographic relief (Baxter 1977). Upstream flooding associated with conventional hydroelectric dams is also the main cause of peatland loss in the country (Rubec 1991). Although recent data were not available, in the early 1990s, approximately 900 000 ha of peatlands were flooded, mostly in Québec, Manitoba and Alberta (Rubec 1991; Urquizo et al. 2000).

Downstream impacts of hydroelectric dams include decreased groundwater recharge, streambank erosion, and associated change in stream morphology (Baxter 1977; Rosenberg et al. 1995, 1997; Environment Canada 2004; Woo and Thorne 2009). Conventional hydroelectric dams do not diminish the magnitude of downstream water flow, unless it involves large reservoirs that lead to increased evaporation losses. However, the timing of water flow is significantly altered to meet electricity demands (Woo and Thorne 2009). Changes to downstream flow occur immediately after the dam construction is complete; flows are dramatically reduced during the time required to fill the reservoir, sometimes for up to several years (Rosenberg et al. 1995; Déry and Wood 2005). Once the dam is in operation, the natural river flow regime is altered, including changes to the magnitude, frequency, duration, timing, and rate of change of flows (Magilligan and Nislow 2005). Alteration to the natural river flow regime has consequences not only for water resources, but also for the entire ecological integrity of the river system (Poff et al. 1997).

Hydroelectric power developments characteristically trap high spring flows for storage in reservoirs and release higher-than-normal flows in winter when the power is needed (Woo and Thorne 2009). This results in an attenuation of the normal hydrograph in spring and enhancement in winter (Rosenberg et al. 1997). This leads to a decrease in the amplitude of annual variations in water levels, although a higher frequency of small-amplitude, short-period variations may result as the discharge is varied with electricity demand (Baxter 1977). For example, Peters and Prowse (2001) found in the Peace River that even at 1 100 km downstream of the reservoir average winter flows were 250% higher and annual peaks (1, 15 and 30 day peaks) were 35 to 39% lower.

The regulation of flows that result in little to no annual spring flooding has been identified as an important impact on delta ecosystems that depend upon flooding for replenishment of water and nutrients (Woo and Thorne 2009). The Peace-Athabasca Delta, the largest boreal freshwater ecosystem in the world that has been recognized as both a Ramsar and UNESCO World Heritage Site, has been drying since impoundment of the Peace River at the Bennett Dam. Weirs were built to recreate the hydraulic damming effect of the pre-impoundment Peace River to restore flooding to the delta (Rosenberg et al. 1995), but data suggest the delta continues to dry, a condition exacerbated by climatic variability and climate change (Culp et al. 2000; Prowse and Conly 2000; Beltaos et al. 2006; Peters et al. 2006, Timoney 2006; Peters and Buttle 2010) and other socioeconomic changes (Timoney and Lee 2009). However, paleolimnological evidence indicates that recently observed dryness is part of a longer trend, which began some 20-40 years prior to Peace River regulation and not out of the bounds of historical variability (Wolfe et al. 2005).

Many large hydroelectric power projects divert water from the natural basin into a neighbouring basin (Ghasseni and White 2007). For example, water from the Churchill

River is diverted to the Nelson River in Manitoba, and the Eastmain and Caniapiscau rivers are diverted into the LaGrande River in Quebec. Such diversions further exacerbate downstream flow patterns, by decreasing flows in the contributing river and increasing flows in the receiving river (Newbury et al. 1984; Woo and Thorne 2009). Disruptions of freshwater flows, particularly for the many boreal rivers flowing northward to the Arctic Ocean, affect the timing of the spring breakup of ice in the lower reaches of the river, hastened by the increased hydrostatic pressure of water flowing from the south to the north (Baxter 1977). Any or all of these changes in upstream and downstream flow regimes have implications for the aquatic organisms that live in those habitats (Rosenberg et al. 1997; Kreutzweiser et al. 2013).

Other forms of hydroelectric power generation have less impact on water quantity. Power generation by run-of-the-river installations is very small (<10 Megawatt) in comparison to conventional hydroelectric dams (> 2 000 Megawatt) (Paish 2002; Egré and Milewski 2002). Run-of-the-river hydroelectric installations create minimal changes in flow, since they have little to no reservoir retention. The purpose of the dam in a run-of-the-river installation is to direct and control the flow of the stream and little water is impounded (Baxter 1977). However, if there is an increase in the popularity of run-of-the-river power generation, the cumulative impacts of multiple installations along a river section will need to be considered (Douglas 2007). Thermoelectric power is one of the largest water users (but not consumers) among natural resource sectors in Canada (Statistics Canada 2010). Oil and coal-fired thermal power generating plants generally use and discharge more water than natural gas-fired plants, and gas-fired plants are becoming more prevalent. The main impact on water quantity with thermal power generating stations is the loss of water through evaporation during steam generation or during cooling, if cooling towers are parts of the operation (Smith and Tirpak 1989; Statistics Canada 2010).

3.4.3 Water quality. Conventional hydroelectric power generation, through changes in upstream and downstream flow regimes, has large impacts on water quality. Upstream changes to water quality include sedimentation from bank erosion, nutrient enrichment from leaching of flooded material, and changes in thermal regime due to shorter water residence times and less mixing (Baxter 1977). Perhaps one of the most significant effects of forming upstream reservoirs is the release of methyl mercury, a strong vertebrate neurotoxin, into water (Tremblay et al. 2004; Rosenberg et al. 1997). Methyl mercury is of particular concern in the boreal zone because of the high density of organic matter deposits found in wetlands, and riparian areas, which when combined with naturally occurring or anthropogenically-deposited mercury and flooding induced anoxic conditions result in ideal conditions for mercury methylation (Kelly et al. 1997; Heyes et al. 2000). The decomposition of organic matter in flooded lands, particularly wetland soils in reservoirs, fuels the microbial methylation of inorganic mercury to methyl mercury (Hall et al. 2005). Impoundments increase the surface area of potential mercury methylation by imposing anoxia over the entire flooded area and by facilitating the exchange of nutrients and methyl mercury between the ground surface and the surface water (Heyes et al. 2000). Ecosystem-scale experiments from the Experimental Lakes Area have improved our understanding of mercury dynamics in reservoirs (e.g., Bodaly et al. 2004; St. Louis et al. 2004; Hall et al. 2005). Flooding of newly created reservoirs creates more methyl mercury than flooding of existing lakes (Rosenberg et al. 1995). Flooding of wetland areas is more detrimental than flooding of upland areas since the higher concentration of organic matter produces methyl mercury for longer periods than flooding of upland areas (St. Louis et al. 2004; Hall et al. 2005). In a flooding experiment in boreal upland forests, Hall et al. (2005) found that within five weeks of flooding, methyl mercury concentrations in the reservoir outflows exceeded those in reservoir inflows. The initial pulse of methyl mercury production in all experimental reservoirs lasted for two years, after which net demethylation began to reduce the pools of methyl mercury in the reservoirs, but not back to levels found prior to flooding (Hall et al. 2005).

Many of the impacts caused by conventional hydroelectric dams on the streams below them are the reverse of those produced on the reservoirs above them (Baxter 1977). Attenuated spring flows reduce organic matter and nutrient loading to downstream waters during biologically active periods, ultimately influencing coastal productivity (Rosenberg et al. 1997). Other downstream impacts include altered water temperature regimes (cooler in the summer and warmer in the winter) and increased turbidity due to sediment inputs from erosion (Baxter 1977).

As for other types of hydroelectric power generation, run-of-the-river has few impacts on water quality, allowing sediment to pass through relatively unimpeded (Kondolf 1997). Thermal power generating stations have impacts on water quality, with the primary concerns related to elevated temperatures of discharge waters and potential pollution from corrosion-control products used in cooling waters (Donahue et al. 2006; NRTEE 2010). Madden et al. (2013) observed discharge water temperatures could be 9.5 to 10°C higher than intake water during summer. Although no boreal specific values could be found, this effect could be larger in northern ecosystems where intake temperatures would be lower, particularly in the winter. These thermal effects may be quite localized, but may impact lake stratification and other chemical and biological processes (Baxter 1977; Environment Canada 2004). In the United States, thermal regulations have caused a shift in the design of new cooling systems, from once-through cooling systems that discharge heat back into water sources, to cooling systems that evaporate water with towers and ponds that alleviate the thermal pollution but result in increased water consumption due to evaporation (Smith and Tirpak 1989).

Thermal power generation, particularly with coal-fired generating stations, also result in atmospheric emissions of pollutants that may be deposited on local water bodies, or enter the airshed to be subsequenty deposited on downwind water bodies. The impact of the pollutant emission and deposition is similar to that described for other resource industries (see section 3.7). Donahue et al. (2006) found using paleolimnological reconstructions within a 35 km radius of coal-fired power plants at Wabamun Lake in central Alberta that sediment concentrations of mercury, copper, lead, arsenic and selenium had increased by 1.2 to 4 times since the four power plants had been built in 1950. Pollution abatement technologies will become increasingly important, particularly if increased thermal power generating capacity in Alberta is met by expansion of the coal-burning industry (Donahue et al. 2006).

3.4.4 Prognosis. Electric power generation has differential impacts across the boreal zone. With high topographic relief and wetter climates, the eastern and cordilleran parts of the boreal zone are ideal for hydroelectric power generation, while flatter and drier areas of west-central part of the boreal zone are dependent primarily on thermal power generation. Thus, the specific impacts of these different technologies will be manifested in the areas in which they occur.

There are currently approximately 55 thermal power generation stations across the boreal zone, mostly in Alberta and the Northwest Territories. An increasing demand for cleaner power will likely lead to a decline in fossil fuel based thermal generating stations creating an energy gap that will likely to be filled by hydroelectric power generation based on global trends (Kumar et al. 2011). Northern rivers in the boreal zone hold the most remaining potential for large-scale hydroelectric power development in Canada (Fig. 11; Environment Canada 2004), and technology exists to more than double the existing hydroelectric power capacity in Canada (Canadian Hydropower Association 2008). From 2011 to 2030, there is a potential for 158 hydropower projects totalling 29 000 Megawatts of new capacity in Canada (HEC Montréal 2011). A large number of new conventional hydroelectric power developments have already been identified across the boreal zone (Environment Canada 2004; Fortin and Collu 2008). Although run-of-the-river installations are more environmentally friendly, their low capacity output is likely to limit their use to meeting specialized local power demands. However, efforts are being made to improve technologies related to small hydropower (1-50 Megawatt) capacity (e.g., NRCan 2009). Increased use of other alternative energies for electricity generation, particularly biomass and geothermal, may bring new impacts on surface and ground water resources.

Many of the impacts of conventional hydroelectric power generation can be mitigated through avoidance of diversions, careful site planning, and use of power storage technologies. For example, upstream impacts can be minimized by reducing the size of reservoirs. This can be achieved by careful site selection to ensure minimal land is flooded (Baxter 1977; Mailman et al. 2006). Downstream impacts can be minimized by allowing for natural flow regimes. Small dams can be converted to run-of-the-river installations and larger dams can use pump storage or other means to store potential or electrical energy (Baxter 1977; Egré and Milewski 2002; Mailman et al. 2006; Renöfält et al. 2010). Most of these environmental mitigation measures require flow modifications that could reduce power production. An important challenge for river management will be to identify situations where measures involving relatively small production losses can have major ecological advantages (Renöfält et al. 2010).

Mitigation strategies have been proposed that may reduce the impacts of methyl mercury although not all strategies have been adequately developed or tested. Some of these mitigation strategies are theoretical and not feasible at large scales and often shift the mercury to a different fate, whether it be volatilized back to the atmosphere *via* burning or transported downstream *via* regulated water flows (Mailman et al. 2006). The best mitigation strategy is to minimize the area flooded (i.e., deepening channels or avoiding flat topography) and ensure that wetland areas with high organic matter are avoided (Rosenberg et al. 1995, 1997; Bodaly et al. 2004). Other mitigation strategies include pre-flooding activities to reduce methyl mercury inputs, including removal of standing trees or burning before flooding (e.g., Rosenberg et al. 1995; Mailman and Bodaly 2005, 2006), and post -flooding activities that involve flow regulation to release methyl mercury during periods of peak flow production, intensive fishing to remove methyl mercury bioaccumulated in fish (e.g., Verta 1990; Surette et al. 2003), adding selenium to reduce methyl mercury bioaccumulation (e.g., Turner and Rudd 1983), adding lime to acidic systems to reduce methyl mercury bioaccumulation (e.g., Wiener et al. 1990), adding phosphorus to cause growth dilution by stimulating productivity (e.g., Kidd et al. 1999), demethylation of methyl mercury by ultraviolet light (e.g., Sellers et al. 1996), capping and dredging bottom sediments (e.g., Hecky et al. 1987), and aerating anoxic bottom sediments and waters (e.g., Matilainen et al. 1991). The optimal mitigation strategy will depend on the system and situation. Mitigation strateigies reviewed by Kreutzweiser et al. (2013) for reducing impacts of hydroelectric power generation on aquatic biodiversity are equally relevant to reducing effects on water quantity and quality.

With many hydroelectric dams constructed 40 to 50 years ago, issues related to maintaining, upgrading, and decommissioning aging hydroelectric installations are increasing (Environment Canada 2004). Long-term planning is necessary to identify the needs for these facilities. Facility improvements can be viewed as opportunities to maximize hydroelectric power generation through implementation of technological advances and to minimize environmental impacts. Facility decommissioning can be viewed as opportunities to avoid further adverse effects on water resources both in reservoirs and downstream areas (Hart et al. 2002; Stanley and Doyle 2003; Environment Canada 2004). Bednarek (2001) reviews the ecological impacts of dam removal, demonstrating that restoration of unregulated flow regimes increases biotic diversity by returning riverine conditions and sediment transport to formerly impounded areas; however, there are increased sediment loads (and the contaminants they contain) over the short-term. There is a scarcity of empirical knowledge on environmental responses to dam removal (Hart et al. 2002), particularly how dam characteristics may influence their decommissioned effects (Poff and Hart 2002).

Comprehensive cumulative effects assessments of hydroelectric mega-projects is challenging because downstream areas often are out of the jurisdiction of the agency responsible for the upstream water development project and for studying its potential impacts (i.e., the problem is created in one province, but the impacts are felt in another) (Rosenberg et al. 1995). These assessments are highly complex, expensive, and require long-term databases from before and after the project, which are seldom available. Confounding these assessments is a warming climate. Winter flows will increase, spring freshet dates will advance, but peak flows will decline, as will summer flows due to enhanced evaporation (Woo et al. 2008; Price et al. 2013). It is predicted that such changes could reduce hydroelectric capacity by 15% by 2050 (NRTEE 2009).

3.5 Mining

3.5.1 Introduction

Canada is one of the leading mining nations in the world, producing more than 60 minerals and metals. In 2010, total mineral products had domestic exports valued at \$81.4B per year (21.8% of total exports) and mining and mineral-processing industries contributed another \$34.7B per year to the Canadian economy, representing 2.8% of the national GDP. Total direct employment in mining and mineral-processing industries was 308 000 or 2.1% of Canada’s total employment (NRCan 2011).

Eighty percent of the mining within Canada occurs in the boreal zone, where there are about 100 active mines, 6 smelters, and 9 coal mines (Fig. 2E, and Fig 6. in Brandt et al. 2013). In addition, there are over 10 000 mining and exploration sites across Canada in varying stages of decommissioning and remediation (Tremblay and Hogan 2006), including more than 1 300 abandoned mines in the boreal zone (Brandt et al. 2013). The types of materials mined and how they are mined differ across the boreal zone, including ferrous metals, precious metals, base metals, precious gems, and coal that are extracted from deep below the ground or at the surface (e.g., open pit or placer mining from rivers). Mining is not environmentally benign; impacts to water resources can occur throughout the mine cycle, from exploration, extraction, and smelting to waste management, including containment and remediation (Ptacek et al. 2004) and even in the creation of ice roads for winter access in remote mines (Cott et al. 2008). Most impacts of mining on water resources arise from discharge or seepage of mine effluents and acid mine drainage into receiving waters, and from atmospheric deposition of smelter-produced acidic and metal pollution (Gunn 1995; Urquizo et al. 2000). The resulting acidification and contamination of receiving waters have implications for the aquatic biodiversity in those systems (Kreutzweiser et al. 2013).

3.5.2 Water quantity

Various steps in the mine lifecycle involve water, potentially leading to disruptions in surface and groundwater quantity. The impacts are variable depending on type of mining (e.g., surface *vs.* subsurface), surface overburden (upland *vs.* wetland soils), underlying geology, mining methods, and the material being mined. During the exploration phase, water quantity is primarily affected by seismic lines, roads, excavations, field camps, and other exploration activities, all or any of which can disrupt surface water flow pathways (see section 3.1). Sampling and drilling activities may disrupt groundwater flow pathways if they come in contact with subsurface aquifers, and can release underground water sources to the surface if done improperly. These potential impacts of mining on water quantity in the boreal zone are described in at least two non-published reports (BCMOE 2008; NMC 2008); our literature search did not find recent peer-reviewed publications or reviews of potential mining impacts on water quantity in boreal basins. Younger et al. (2002) provides an extensive review of the impacts of mining on water resources, based largely on studies in non-boreal regions.

During mine operation, water quantity impacts vary with the mining method. Surface mining affects surface water flow pathways. Rivers and lakes may need to be diverted or drained to access the underlying materials. In the instances where extensive, shallow mineral deposits are located just below the ground surface, large-scale removal of overburden is required to access the underlying resource via open pit mining (Ptacek et al. 2004). When these surface mines are in wetland-rich areas, a large amount of wetland area will be drained and peat will be removed. Diamond mining at Victor Mine in Ontario's Hudson Bay lowlands is a current example of near-surface mining in wetland-rich areas that can impact the surface water and groundwater levels (CEAA 2004). Diamond mines are usually focused on kimberlite pipes that are often surrounded or overlain by water that must be diverted or displaced for mining operations (Foote and Krogman 2006).

The removal of surface mined substrate and overburden (i.e., soil or peat) decreases the hydraulic gradient. This causes a rise in the water table, and wells must be drilled to pump excess water. This "dewatering" increases vertical recharge of the surrounding land, causing groundwater levels to decline in the vicinity of the excavation, and potentially reducing surface water levels of local watercourses and wetlands to levels that could be environmentally damaging (Atkinson et al. 2010; CEAA 2011). The extent of water table decline depends on hydraulic properties of subsurface materials and pumping rates. For example, in wet areas, water losses in wetlands from mine dewatering can be significant and irreplaceable for the duration of the mine (Whittington and Price 2012).

Underground mining can also divert water sources and pathways. Underground openings at a gold mine in the Northwest Territories have caused changes to groundwater flow and to the circulation of surface flow to depth (Douglas et al. 2000). Placer mining removes deposits associated with stream beds, so this type of mining activity involves water damming and diversion that can influence natural drainage patterns and increase erosion and sedimentation rates (Pentz and Kostaschuk 1999). Placer mining accounts for about 5% of Canada's annual gold production, and Canadian placer gold mining is almost exclusively conducted in mountain regions of the Yukon Territory and British Columbia (McCracken et al. 2007).

Water quantity may also be affected by mining at the processing stage. During metal and mineral processing, water is extracted or diverted from local watercourses for processing, cooling, and diluting or for treating mined materials. During the remediation phases, replacement of overburden and creation of new features (e.g., diversion ditches) usually alters surface and subsurface drainage patterns (Ptacek et al. 2004; CEAA 2011).

3.5.3 Water quality

Most published literature on the impacts of mining in the boreal zone focused on water quality. The degree of impact of mining on water quality depends on the nature of the ore, its host rock, and the way that it is mined and processed (Wireman 2001). Mine related effluents, seepages, and emissions during mining and smelting cause the majority of water quality impacts, but impacts can also arise from leaching of waste rock and tailings well after mine closures (Leblanc et al. 2000; Urquizo et al. 2000; Ptacek et al. 2004). These waste materials and other exposed rock surfaces from mining often contain sulphide minerals that, when oxidized, produce acid drainage that elevates concentrations of metals in receiving waters (Blowes et al. 2003). Depending on the ore and its byproducts, these metal concentrations in water can include heavy metals as well as cyanide, arsenic, or radioactive compounds. In the absence of effective remedial actions, these waste impoundments have the potential to oxidize and produce acid mine drainage for centuries (Blowes and Jambor 1990; Moncur et al. 2005; Kalin et al. 2006).

Studies over the last two decades in or on the edge of Canada’s boreal zone showed that surface waters near historic and active mining and smelting operations often continue to contain elevated levels of various metals. Some of these elevated concentrations are from continuing discharges or leaching of effluents and tailings, and some reflect continuing mobilization of metals from saturated soils near smelters. Using an assessment approach based on region-specific objectives, de Rosemond et al. (2009) analyzed water quality from sampling sites downstream of mining effluent outputs and found that average water quality indices were significantly or measurably lower than at corresponding reference sites. Many (about 80%) of these mining sites (n = 71) were in or on the edge of the boreal zone, and the water quality indices were largely reduced by elevated concentrations of metals and nutrients. Some approached or exceeded values considered harmful to aquatic life.

Wang and Mulligan (2006) reviewed the environmental occurrence of arsenic in Canada, and reported that tailings from historic and recent gold mine operations in several regions continue to leach arsenic to surface waters resulting in concentrations more than 100 times acceptable levels, posing risk of harm to aquatic organisms. Historic silver mining in northern Ontario has also left large volumes of arsenic-bearing mine wastes that leach to nearby watercourses resulting in aqueous arsenic concentrations at least 10 times higher than Canadian drinking water standards, although the spatial extent of this exceedance appeared to be limited to within 1 km of the tailings outflow (Kwong et al. 2007).

Selenium concentrations in lake water at 5 to 10 times higher than reference levels were found up to 10 km downstream from uranium mining operations in northern Saskatchewan (Muscatello et al. 2008). In a separate study from the same region, Muscatello and Janz (2009) reported selenium concentrations in water of a lake at the outflow of an effluent management system of approximately 20 times above reference levels. Wiramanaden et al. (2010) found that although selenium concentrations in lakes downstream of uranium mining in their study area were fairly low some distance from the discharge point, they were above reference levels and accumulated in fish resulting in adverse effects.

Cyanide-containing tailings from an open pit gold mine near the boreal zone in eastern Canada that closed in 1992 continue to leach mercury and other metals (e.g., Cu, Zn, Pb) from a contaminated groundwater plume into a headwater stream, resulting in highly elevated mercury concentrations (Maprani et al. 2005; Al et al. 2006). An earlier study at the same site near the boreal zone also reported elevated aqueous gold concentrations in stream water originating from the weathering of tailings piles (Leybourne et al. 2000). Singh and Hendry (2013) concluded that there continues to be long-term risk of nickel and uranium leaching to surface and groundwater from waste-rock piles at an active uranium mine in northern Saskatchewan.

Moncur et al. (2006) found that 50 years after a zinc-copper mine in Manitoba closed, metal and sulphate concentrations in a lake below the outflow from a tailings impoundment were still elevated, with peak concentrations up to 8 500 mg L^{-1} of iron, 100 mg L^{-1} of aluminum, and 20 000 mg L^{-1} of sulphate, although concentrations varied spatially throughout the lake. A small lake in northwestern Ontario received tailings water with high concentrations of metals since the 1960s, but showed significant declines in metal loading, especially copper, zinc, and nickel, to the lake after improved water treatment and management practices by the mid 1990s. However, arsenic concentrations increased through that period to the end of the study, apparently the result of remobilization of arsenic stored in sediments (Martin and Pedersen 2002). Similarly, a lake near Flin Flon, Manitoba that received metal loadings for more than 50 years from metal mining and processing in the region showed increasing levels of zinc in recent years despite improved water treatment and reduced discharges of zinc (Bhavsar et al. 2004). They attributed the increased zinc concentrations to mobilization of historically deposited zinc in the lake drainage basin. Further evidence of the remobilization of stored metals in basins was shown by Outridge et al. (2011). They measured metal concentrations in cores from peat and lake sediments (reflecting trends in water chemistry) in the Flin Flon region and found that although mercury and other metal concentrations in peat declined over time in accordance with reduced smelter emissions, mercury and zinc increased in lake sediments. They attributed these increases to the mobilization of historically deposited metals in surrounding basins.

Nickel mining and smelting in the Sudbury region on the edge of the boreal zone in Ontario have left a legacy of metal-contaminated soils in regional basins that continue to export metals to receiving waters. Although it is clear from earlier studies (e.g., Keller and Pitblado 1986) that emission controls and other remedial actions have reduced metal loading to lakes within the region, recent studies continue to find elevated metal concentrations in surface waters. Rajotte and Couture (2002) and Eastwood and Couture (2002) reported that several lakes in the Sudbury region contained elevated levels of aluminum, cadmium, copper, nickel, and zinc, often exceeding provincial water quality objectives by five times or more and with negative implications for fish condition. Hruska and Dubé (2004) found that effluents being discharged to stream water contained several metals or their byproducts at concentrations 10 to 22 times higher than reference levels, with selenium 64 times higher than reference. Ponton and Hare (2009) reported that nickel concentrations in lakes near mining and smelting operations were up to 200 times higher than in reference lakes. Several metals were found at elevated concentrations in stream water receiving metal mining effluents, and these concentrations led to elevated metal body burden in fish (Weber et al. 2008). Further information on trends in metal contamination of lakes in the Sudbury region is found in Keller et al. (2007) and Keller (2009).

Coal mining also produces waste rock that can oxidize and become sources of acidity, metals, and salinity that leach to receiving waters (Zielinski et al. 2001). Open pit coal mines, by virtue of their associated overburden removal that can extend for tens of square kilometers (e.g., Strong 2000), can disrupt hydrologic flowpaths and affect downstream water quality. Although coal mining is active in the boreal zone, particularly in western regions, and those mining installations are under national regulatory assessment (e.g., EUB-CEAA 2000), our search of the published literature did not find any studies reporting the impacts of coal mining on water quantity or quality in the boreal zone. Studies from non-boreal regions show that coal mining can adversely affect surface water quality (e.g., Zielinski et al. 2001; MacCausland and McTammany 2007) and similar risks would be expected from coal mining activities in the boreal zone. The lack of published literature to assess these risks in the boreal zone remains a critical information gap for coal mining assessment.

3.5.4 Prognosis

Several aspects of mineral and metal mining activities have the potential to affect water quantity (i.e., water levels of natural water bodies) through water diversion, overburden removal and replacement, and intentional dewatering. However, it appears that compliance with current mining regulations and implementation of mitigation measures can reduce these impacts on water quantity. In at least one example, mitigation measures proposed for a major surface gold mine were considered effective at reducing the risks of adverse and long-lasting changes in water levels of local watercourses (CEAA 2011). Mitigation measures for impacts on water quantity include the establishment of water flow monitoring stations, water flow supplementation by pumping when necessary to maintain base flows within 15% of seasonal norms, active open pit flooding to increase aquifer recovery after mine closure, proper drainage channel installation to manage and minimize water diversion, and managing site infrastructure to minimize the overall footprint on water flow paths and wetlands (BCMOE 2008; CEAA 2011). The mining sector in general is not a significant consumer of water, and threats to water quantity from mining of metals and minerals across the boreal zone do not appear to be a constraint on future mining development (Ptacek et al. 2004; NRTEE 2010). Potential impacts to water quantity from current mining activities and the effectiveness of mitigation measures to minimize these impacts could not be assessed from the published literature. Peer-reviewed scientific studies on these issues around water quantity are lacking and represent an information gap.

Water quality is often impacted near mining and smelting operations in the boreal zone. Historic mining areas, tailings ponds, and older and abandoned mines and sites are more likely to contribute to elevated metal concentrations in receiving waters than newer installations (Urquizo et al. 2000; Ptacek et al. 2004). Stakeholder working groups that have been formed to assess and report on the effects of metal mining on aquatic ecosystems have concluded that recent metal mining effluent regulations and best practices have improved water quality at many sites, but that elevated concentrations of metals and other dissolved solids continue to be detected and are linked to adverse effects on aquatic organisms in many areas (AQUAMIN 1996; Ptacek et al. 2004; Environment Canada 2012b; Kreutzweiser et al. 2013). The related issues of atmospheric deposition of acidic pollutants from processing emissions and their effects on water quality are ongoing (see section 3.8). Emissions of acidifying pollutants have declined by about 50% nationally, and by much more in some areas (CCME 2011), but atmospheric deposition of acidic pollutants still exceeds critical loads, and acidifying emissions are still increasing in other areas (Environment Canada 2005*a*; FPTC 2010).

The impacts on water quality can be mitigated to some extent by careful mine operations, effluent and emission controls, and improved tailings treatment and management. Canada's National Orphaned/Abandoned Mines Initiative is supported by the Mining Association of Canada and other non-governmental and governmental bodies and is committed to the remediation of abandoned mines (www.abandoned-mines.org). However, with more than 10 000 exploration and mining sites across Canada that require varying degrees of remediation (Tremblay and Hogan 2006), including at least 1 300 abandoned mines in the boreal zone (Brandt et al. 2013; see also CESD 2002; CDL 2005; Cowan et al. 2010), the problem of persistent metal contamination to waterbodies from older mining sites will continue for some time. Several long-term studies have demonstrated that water quality conditions in the boreal zone can measurably improve with enhanced emission and effluent treatments and that these improved conditions can promote biological recovery (Keller and Yan 1991; Gunn et al. 1995; Keller et al. 1998, 2004, 2007). However, the recovery pathways are long and complex (Jeziorski et al. 2013; Luek et al. 2013; Szkokan-Emilson et al. 2013), often confounded by the remobilization of historically deposited or stored metal concentrations in soils and sediments as described above.

Recent studies have demonstrated that improved treatment of mine tailings and acid mine drainage can minimize the contamination of downstream water bodies. Sherriff et al. (2011) showed that despite high oxidation rates and metal concentrations in tailings ponds of a copper-zinc mine in Manitoba (closed in 2002), metal concentrations in a downstream lake receiving tailings outflow water were stable indicating that the tailings management activities at that site were effective to date. Kachhwal et al. (2011) also demonstrated that current water cover technologies and other tailings management efforts at a nickel-copper mining site in northwestern Ontario (closed in 1998) were effective at preventing downstream export of metals in the final effluent. Water cover technologies in rehabilitation efforts at uranium mining sites in northcentral Ontario (closed as recently as the mid 1990s) have successfully reduced acid generation such that only limited effluent treatment is required for pH control and radium removal (Davé 2011). A nickel-copper mining facility currently operating in Quebec has employed new technologies in waste management and water treatment, and reports that water discharge to downstream environments have metal and byproduct concentrations well below water quality limits (Kratochvil 2010). The use of natural and constructed wetlands for bioremediation of mining wastes is a reclamation technology that would appear to be well suited for aquatic systems in the boreal zone given the prevalence of natural wetlands (Sobolewski 1996; Zhang et al. 2010). Other biological remediation and ecological engineering technologies are also being explored and developed to prevent the formation or migration of mining waste leachates (Johnson and Hallberg 2005). Kalin et al. (2006) demonstrated that biological remediation efforts in a lake in northwestern Ontario affected by acid mine drainage were successful at preventing changes to water quality in the next lake downstream from the affected lake. Regardless of these demonstrated successes, biological remediation and other ecological engineering technologies have not been widely adopted by the Canadian mining industry (Kalin and Wheeler 2011), possibly because these technologies may be limited by cold temperatures in the boreal zone, or by the relatively low volumes of water that can be reasonably treated in this fashion.

Recent national mining impact assessments continue to report degraded water quality near mining sites (Ptacek et al. 2004; Environment Canada 2012b), indicating that mitigation measures for water quality are not always completely effective. Degraded water quality from mining activities can pose risks of harm to aquatic biodiversity, and national reporting regulations under the Environmental Effects Monitoring Program are intended to track these risks and impacts (Environment Canada 2012b; Kreutzweiser et al. 2013). To the extent that metal mining developments and their associated tailings ponds, waste rock, effluents, and emissions are likely to increase, the potential for contamination of regional water bodies could also increase and would be incremental to the legacy effects of many older sites. From the published studies, it was not always clear to what spatial extent the contamination of water bodies extended beyond mining or processing sites. Most studies reported impacts within a few kilometers of the mining sites and many showed declining metal concentrations or water quality issues with distance from the mining activities. Given the extensive distribution of metal mining across the boreal zone, the persistent nature of metal contamination, and the fact that mining and processing often occur in or near water-abundant areas, there is potential for significant cumulative impacts at regional scales. This potential for cumulative impacts has not been rigorously assessed in the published literature and remains an information gap. There are indications that the mining sector is responding to the challenges of environmental stewardship by implementing various beneficial management practices and mining reforms across Canada (MAC 2010; PDAC 2012). This includes new reporting requirements for improved tailings management, environmental performance, and biodiversity conservation management (MAC 2010).

3.6 Conventional oil and gas and oil sands development

3.6.1 Introduction

Fossil fuels account for the greatest share of Canada's production of primary energy, dominated by crude oil (41.4%), natural gas (36.5%), and coal (9.2%) (NRCan2011). The oil sands is an important component of our crude oil, representing 97% of the crude reserves and a little over half of crude production in 2010 (NRCan 2011), with the majority from surface extraction (King and Yetter 2011). Oil and gas developments are located primarily in the sedimentary regions of the western boreal zone (Fig. 9). It has been estimated that the oil and gas well sites, pipelines, and seismic lines (Fig 2F) have an industrial footprint of 46 million ha (Timoney and Lee 2001). Brandt et al. (2013) calculated from the Seismic Data Listing Service that there were about 441 000 km of pipelines and 1.7 million km of seismic lines. There are more than 220 000 active and inactive well sites drilled by the oil and gas industry in boreal Canada and this number is increasing at the rate of approximately 10 000 new wells per year (Wells et al. 2010; Brandt et al. 2013). Pasher et al. (2013) estimate that well sites and oil and gas infrastructure cover approximately 114 377 ha. Oil and gas developments can be characterized as being intensive (surface extraction of oil sands bitumen) or extensive (individual well sites). Similar to mining, water resources are impacted by oil and gas development at all stages of the development cycle, including exploration (seismic lines), drilling (roads and well pads, water consumption), and production (water removals and contamination).

Impacts on water resources differ depending on the mode of extraction. In older conventional oil fields, water is pumped down the well to stimulate production. In the oil sands, *in-situ* extraction uses steam to liquefy the viscous bitumen so it can be pumped to the surface (Gosselin et al. 2010; CAPP 2013). Surface extraction of oil sands occurs where bitumen reserves are within 70 m of the surface of the ground. After the removal of soils and overburden, bituminous sands are removed with conventional mining techniques (e.g., truck and shovel) (King and Yetter 2011; Mikula 2012). Bitumen is extracted by mixing with hot water and mechanical agitation. Typically, the bitumen is about 10% by weight of the raw ore and 12 volumes of water, sand, silt, and clay tailings are created for each volume of bitumen produced (Kasperski and Mikula 2011). The resulting tailings are pumped to tailings ponds where the sand settles quickly leaving a suspension of fine particulates and dissolved organics, minerals, and salts (Kasperski and Mikula 2011; CAPP 2013).

Parts of the boreal zone in Alberta and British Columbia have natural gas contained within shale that can only be accessed by a technique known as hydraulic fracturing (fracking) (National Energy Board 2009). Pressurized hydraulic fracturing fluid, which contains large volumes of water, mixed with sand and chemical additives, is pumped into the wellhead at high pressure. This creates cracks in the rock beds, allowing for increased extraction and recovery of gas (National Energy Board 2009; CAPP 2013; CSUG undated). Although currently shale gas extraction is limited to only a few places in the western boreal, this component of the oil and gas industry is likely to increase in the future. Although environmental impacts of shale gas extraction are known (Council of Canadian Academies 2014) , boreal-specific studies on the impacts on surface and groundwater resources are presently lacking.

3.6.2 Water quantity

Water is a critical component of oil and gas production in Canada. For example, about 75% of Alberta's oil production is water assisted (CAPP 2010). In 2010, 9% of all water allocated within the province of Alberta was licensed for use by the oil and gas industry (CAPP 2012). The source of water for oil and gas extraction may be from surface or groundwater sources (CAPP 2010, 2012; Government of Alberta 2013b). When surface water sources are used, the main concern for water quantity is the reduction of water supplies to downstream users and ecosystems (DFO 2010). Given that river flows are variable, removals have more impact on in-stream processes when water flow rates are low (DFO 2010). Groundwater can be sourced from shallow or deep sources or from fresh or saline sources (Government of Alberta 2012b, CEMA 2012). The use of groundwater for oil and gas extraction has the potential to deplete local aquifers that are needed for other important purposes (e.g., drinking water). These groundwater depletions may also alter pressure within the aquifer which influences local or regional groundwater flow patterns and recharge of surface waters and wetlands (Manitoba Environment 1993; Griffiths et al. 2006; Government of Alberta 2012b).

3.6.2.1 Oils sands example

All aspects of the oil sands developments, including surface mining, bitumen upgrading, and *in-situ* extraction, are dependent upon water (Gosselin et al. 2010; CEMA 2012). It is estimated that 523 million m^3 of water is used per year in the oil sands extraction process (Griffiths et al. 2006; Alberta Environment 2007). The production of 1 m^3 of synthetic crude oil (upgraded bitumen) from surface extraction requires about 12 m^3 of water, of which about 75% is recycled, meaning that 3 m^3 must be drawn from licensed sources (Kasperski and Mikula 2011; CAPP 2012; CEMA 2012). Water used to extract surface mined bitumen comes primarily (50-75%) from surface waters of the Athabasca River. Groundwater is also extracted during surface extraction, to decrease the groundwater table to access the surface deposits, and occasionally as an alternative water source for processing when river flows are low (King and Yetter 2011). Groundwater is used almost exclusively in *in-situ* extraction. About 0.5 m^3 of groundwater is required to produce 1 m^3 of bitumen (CAPP 2012; CEMA 2012). All bitumen extraction and production facilities are required to recycle water as much as possible; for example, 85% of water used in the surface extraction process is recycled (CAPP 2008) and 90-95% of water used in *in-situ* operations is recycled (Humphries 2008).

Current allocation of water from the Athabasca River, for all usages, is 3.5% of the total annual average river flow, with allocations for oil sands mining projects accounting for about 2.2% of total flow (AMEC 2007). Less than 10% of the water used for the oil sands industry is returned to the Athabasca River (Richardson 2007). Despite recycling, much of the water ends up in the engineered tailings ponds or evaporates from the pond’s surface (Griffiths et al. 2006; CEMA 2012). Several reports have found no trend or mildly increasing flow in the Athabasca River prior to oil sands development in the mid-1970s and declines in flow following development (Alberta Environment 2004; Schindler et al. 2007; Squires et al. 2009). In light of observed declines in flow rates of the Athabasca River, the percentage of flow removed increases even if water use remains constant. Flows within the Athabasca River are lowest during winter, due to headwaters being frozen (Kim et al. 2013); hence there is a high likelihood of water shortages during winter in the future (Mannix et al. 2010) or at other periods of low precipitation (Kim et al. 2013). The water management framework for the Lower Athabasca River (Ohlson et al. 2010) requires water removals be adjusted to meet in-stream requirements (DFO 2010).

Unlike river flows, groundwater flows are not easily observed. Groundwater dynamics within the oil sands regions are extremely complicated (CEMA 2012; Government of Alberta 2012b). The surficial geology in the Athabasca oil sands region consists of Holocene and Quaternary sediments in a complex distribution of tills, outwash sands, and buried valleys (King and Yetter 2011; Government of Alberta 2013a), creating a complex hydrogeologic system of aquifers (geologic formations that are permeable and contain or transmit groundwater) and aquitards (geologic formations with low permeability to groundwater) (Andriashek and Atkinson 2007). Many of the hydrogeology investigations date back to the 1970s and 1980s. However, there have been more efforts to better characterize the subsurface hydrogeology. For example, Andriashek and Atkinson (2007) acquired and interpreted more than 35 000 new borehole logs from the oil sands industry to construct a three-dimensional model of the subsurface including major buried aquifers contained within buried valleys and channels north of Fort McMurray. However, even with higher density of data, the narrow form and discontinuous nature of many of the subsurface features that may contain aquifers or be natural pathways of water and contaminants means that they remain undetected (Andriashek and Atkinson 2007).

Groundwater needs to be removed for surface oil sand extraction in order to lower the groundwater table to prevent seepage into the open-pit mine. This creates a draw-down zone that dries out the surrounding wetland and upland areas (King and Yetter 2011; Government of Alberta 2012b). Draw-downs of 15 to 25 m have been observed near active mining areas compared to 1 to 4 m in natural water level fluctuation (WorleyParsons 2010 in Government of Alberta 2012b). Groundwater removals in both surface and *in-situ* extraction can depressurize aquifers (King and Yetter 2011; CEMA 2012). Control of the hydraulic pressure within the basal aquifer must be maintained to prevent hydraulic connection between the basal aquifer and groundwater within overlying sediments (King and Yetter 2011). There are limits set on the volume of groundwater that can be extracted (Water Conservation and Allocation Guideline for Oilfield Injection; Alberta Environment 2006). Non-saline groundwater can be drawn-down in the production aquifer to 35% during the first year of operation and no more than 50% over the life of the project (Alberta Environment 2006). In 2009, the total licensed volume for groundwater removal was approximately 52×10^6 m^3 per year while the average annual volume of water actually withdrawn was 57% of that (30×10^6 m^3 per year) (Government of Alberta 2012b). Over half of those licenses were in shallow groundwater deposits (Government of Alberta 2012b).

Groundwater and surface water are interconnected. The depressurization of aquifers that accompanies groundwater removals affects local and regional groundwater flows patterns (King and Yetter 2011; CEMA 2012). Changes in both surface water and groundwater affect local and regional recharge and discharge to the wetlands and shallow lakes common in the area (Devito et al. 2005). Schmidt et al. (2010) showed that a combination of stable isotope and radon mass balance approaches provides information on water flow path partitioning that is useful for evaluating surface-groundwater connectivity (i.e., connectivity of lakes and rivers to underlying aquifers).

3.6.3 Water quality

Water quality concerns with oil and gas development are related to both surface and groundwater contamination. Contamination can occur both at the well or excavation site (e.g., oil and hydraulic facturing fluid spills), from tailings ponds (e.g., overflows or breaches, groundwater seepage), from pipelines during transport, or from groundwater contamination (e.g., saltwater intrusion, mixing of water with hydrocarbons, groundwater heating from steam extraction) (National Energy Board 2009; CAPP 2010, 2012; Environment Canada 2010; Foote 2012; Council of Canadian Academics 2014). Although some of these are specific to different oil and gas extraction methods, examples from surface and *in-situ* extraction of the oil sands are given below.

3.6.3.1 Oil sands example

Water quality can be impacted at various points in oil sands processing. Water quality is usually impaired from contaminated groundwater discharging into surface bodies (i.e., streams, rivers, lakes, wetlands) rather than direct effluent release (Gosselin et al. 2010; King and Yetter 2011). Tailings ponds are an important point source of contaminants

(Government of Alberta 2013a). After bitumen has been extracted from surface mining of oil sands, the residual material composed of solids and contaminated oil sands process water is discharged to tailings ponds (CAPP 2012). Water is removed from tailings (i.e., dewatered), producing a fine particulate suspension known as mature fine tailings. Generally, tailings contain approximately 70 to 80% weight basis water, 20 to 30% solids, and 1 to 3% bitumen (Allen 2008). The released water is reused in the oil sands extraction process. This remaining water contains elevated levels of soluble salts (due to continuous recycling of water), dissolved organics (including naphthenic acids), metals, trace elements, phenols, and low molecular weight hydrocarbons (e.g., polycyclic aromatic hydrocarbons) (Government of Alberta 2013a). All extraction wastes are contained in large tailings or settling ponds, constructed using overburden and tailings and retained by sand dykes constructed with drainage collections systems to intercept leaking water. The current volume of fluid tailings is estimated at 720×10^6 m^3 and covers 130 km^2 (Government of Alberta 2012a), an area likely to increase with further expansion of oil sands mining operations unless regulatory requirements reduce the 40 year historical inventory of tailings (Woynillowicz et al. 2005; Gosselin et al. 2010; Foote 2012). Directive 074 of Alberta Energy Regulator requires reduction of fluid tailings and conversion of tailings into trafficable deposits that are ready for reclamation five years after the deposits have ended (ERCB 2009). New technologies, such as Suncor's TRO™ process (http://www.suncor.com/en/responsible/3229.aspx), are steps towards meeting those targets by converting fluid fine tailings to solid materials that can be used in reclamation. Reduction in fluid fine tailings will also increase the water available for recycling, reducing demands for process water from licensed sources (Kasperski and Mikula 2011).

Water quality impacts to groundwater may occur due to the disruption of natural or engineered tailings pond barriers, or through vertical or horizontal connections in groundwater pathways (Gosselin et al. 2010; Government of Alberta 2012b). There are also naturally occurring geologic formations that impact surface water quality, including soluble hydrocarbons, salts, and trace elements that leach from bedrock, groundwater, and wetlands (Government of Alberta 2012b). The transport and persistence of various contaminants from tailings ponds and separating natural background levels of contaminants from process-affected contamination continues to be an important research focus. Leakage rates from tailings containment structures are gaps in knowledge.

A class of contaminants of particular concern because of their toxicity to aquatic organisms (Kreutzweiser et al. 2013) is naphthenic acid (NA), which is the primary class of hydrocarbons in the process water. Ambient concentrations of NA in northern Alberta rivers within the Athabasca Oil Sands region are generally below 1 mg/L, while concentrations of NA in tailings pond waters can reach over 100 mg/L (Headley and McMartin 2004). NAs are mobile and persistent in shallow groundwater of the oil sands mining area. Therefore, little attenuation, beyond dispersive dilution, can be anticipated along groundwater flow paths in the reclaimed landscape (Ferguson et al. 2009; Oiffer et al. 2009; Barker et al. 2010). Gervais (2004) found that low molecular weight NA could be biodegraded under aerobic but not anaerobic conditions. However, Janfada et al. (2006) found preferential sorption of the individual NAs in soil, which could be important from a toxicity standpoint because different NA species have varying degrees of toxicity. Ahad et al. (2013) highlighted the need for accurate characterization of the diversity of NA species in order to quantify potential seepage from tailings ponds. Savard et al. (2012) developed a new method of carbon isotopic analysis of carboxyl groups to distinguish mining-related contaminants from natural background organic acids. These data were combined with a two-dimensional conceptual model to simulate groundwater flow and mass transport between tailings ponds and the Athabasca River. Estimates suggest that mining-related acid extractable organics (e.g., NAs) may be reaching the river, but groundwater contamination is more of an issue at the local scale (Savard et al. 2012).

Metal contamination is another potential concern. Savard et al. (2012) developed lead and zinc isotopic methods to discriminate natural from mining-related metal inputs. They found extraction-related metals were attenuated along the groundwater flow path, with practically no load being delivered to the Athabasca River (Savard et al. 2012). Oiffer et al. (2009) also found minimal trace metal mobilization. Barker et al. (2010) found that process affected groundwater usually contains little trace metals, but mildly anaerobic process waters may leach toxic metals from the aquifer material and so could attain undesirable levels of selenium, arsenic, among other metals. However, studies of current plumes show little accumulation of such toxic constituents (Barker et al. 2010).

Gibson et al. (2011) assessed the potential for labeling process-affected water from oil sands operations using a suite of isotopic and geochemical tracers, including inorganic and organic compounds in water. Although selected isotopic and geochemical tracers were found to be definitive for labeling water sources in some locations, overall it was unreliable to attempt any universal labeling of water sources based solely on individual tracers or simple combinations of tracers. They concluded that understanding the regional hydrogeological system and interpretation of tracer variations in the context of a systems approach on a case-by-case basis offers the greatest potential for comprehensive understanding and labeling of water sources and pathways (Gibson et al. 2011).

In-situ oil sand extraction creates different water quality issues. *In-situ* wells are encased to prevent groundwater from moving up or down through vertical layers and to prevent contamination (saline water or bitumen) from moving into the groundwater (CEMA 2012). However, breaches in well encasements can degrade groundwater through disturbance and mixing of saline and freshwater aquifers. Water quality may also be impaired from improper waste disposal of salts from the water treatment and steam production process (CEMA 2012). Salt disposal is usually by deep injection, raising concern for subsurface contamination. Steam-assisted extraction leads to increases in groundwater temperatures; any increase in temperature is expected to result in negative effects because of the potential for enhanced solubility and/or mobility of chemical constituents that are stable at naturally-occurring temperatures (Gosselin et al. 2010; CEMA 2012). Dewatering in the subsurface can cause increased oxygen in the subsurface, which could cause oxidative release of some toxic elements, such as arsenic, into groundwater (Birkham et al. 2007; Gosselin et al. 2010; Holden et al. 2013).

Monitoring is essential to quantifying impacts of oil sands development and efficacy of mitigation measures. The Regional Aquatics Monitoring Program (RAMP) was established in 1997 to assess water quality in rivers and lakes of the oil sands region, including the Athabasca River Delta. RAMP monitors different elements and compounds within surface waters, including major ions, nutrients (ammonia, total nitrogen, total phosphorus, dissolved organic carbon), general organics (NAs and phenolics), total metals (e.g., aluminum, arsenic, cadmium, selenium, and zinc), polycyclic aromatic hydrocarbons (PAHs) and other constituents like phthalates, acrylamide, petroleum hydrocarbons, vanadium and dissolved oxygen.

A recent RAMP study (RAMP 2009) attributed the contaminants found in the Athabasca River and its tributaries to a natural origin. Detailed examination of the results revealed that there were deviations from the baseline at about 25% of the sites for one or more of the water quality parameters evaluated, but there were no consistent trends for which constituents were elevated. Other studies (e.g., Headley et al. 2005; McMaster et al. 2006*a*; Conly et al. 2007) also concluded that high concentrations of arsenic (Conly et al. 2007), metals (Headley et al. 2005), and hydrocarbons (McMaster et al. 2006*a*) were of natural origin. Several studies have been critical of the RAMP conclusions. Timoney and Lee (2009) and Kelly et al. (2009) showed higher PAHs downstream of the oil sands that were not attributed to natural oil sand. In a more recent paper, Kelly et al. (2010) showed that elevated priority pollutants (13 elements as identified by the United States Environmental Protection Agency: antimony, arsenic, beryllium, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, thallium, and zinc) were higher in the snowpack near oil sands developments than at more remote sites. They also showed that summer stream water priority pollutants were greater near developed and downstream areas than undeveloped and upstream areas.

Groundwater quality monitoring has been limited (Gosselin et al. 2010), with the Lower Athabasca Groundwater Management Framework (Government of Alberta 2012b) assessing groundwater quality data as fair to poor. There are limited spatial and temporal data for most aquifers. In particular, more groundwater quality monitoring data are needed for buried valley and buried channels to sufficiently frame baseline conditions (Government of Alberta 2012b).

3.6.4 Prognosis

Water is necessary for oil and gas extractions, particularly in the oil sands. Oil and gas companies are keenly aware of the need to conserve and recycle the water resources that are vital to the extraction process (CAPP 2012). The oil sands highlight the need for water management frameworks to be developed and/or implemented.

Management plans will likely need to be both sound and adaptive to meet future water needs of an expanding oil sands industry. In 2011, Canada’s oil sands production was 1.7 million barrels per day (Government of Alberta 2013b) and is forecasted to increase to be 4.3 million barrels per day by 2030 (Prebble et al. 2009). These increases in oil sands extraction rates, coupled with changes to water flows with a changing climate (Price et al. 2013) will place increasing pressure on the water resources of the Lower Athabasca River basin.

More surveys of water resources would support water resource decision making. Some of the fiercest criticisms of oil sands environmental management have been directed at the water monitoring program of RAMP, with recent reviews identifying many gaps in the RAMP monitoring design in the oil sands (Environment Canada 2010; Main 2011). Gaps included lack of sufficient temporal data to determine historical trends for key water indicators, limited monitoring coverage outside of the active mineable oil sands area, a lack of hydrologic and geologic data, and the absence of a regional groundwater model. As part of a response to these concerns, Environment Canada (2011) put forth a conceptual framework for an integrated oil sands environmental monitoring plan that would be holistic and comprehensive, scientifically rigorous, adaptive and robust, inclusive and collaborative, and transparent and accessible. In early 2012, Alberta's Minister of Environment and Sustainable Resource Development and Canada's Environment Minister announced the Joint Canada-Alberta Implementation Plan for Oil Sands Monitoring (http://environment.gov.ab.ca/info/library/8704.pdf). The plan, to be implemented by 2015, is aimed at enhancing the oil sands monitoring program for air, land, water and biodiversity. A key aspect of the plan is transparency, with data to be publicly accessible through the Oil Sands Information Portal (http://environment.alberta.ca/apps/osip/).

As highlighted in the Environment Canada review of RAMP, understanding groundwater resources both spatially and temporally is a significant knowledge gap, one echoed in the recently released Lower Athabasca Groundwater Management Framework (2013). Groundwater quantity and quality issues are only going to become more important in the future. As of October 2011, groundwater allocation had risen by approximately 40% from 2009 values to 72.8×10^6 m^3 per year (WorleyParsons 2012). Greater water demands coupled with requirements for maintaining in-stream flow requirements are likely to result in greater withdrawals of groundwater. Groundwater is a hidden resource and, because of uncertainties in its distributions, flows and recharge rates, may be vulnerable to over exploitation (Struzik 2013). Efforts such as the aquifer risk assessment methods presented in the Lower Athabasca Region Groundwater

Management Framework for North Athabasca Oil Sands Area (2013) are important first steps in identifying vulnerable groundwater aquifers.

Groundwater flows at much lower velocities (m per year) than surface water (m per second), and if contaminated, the time scale for recovery will be much longer (Gosselin et al. 2010). Thus, groundwater monitoring will be necessary for decades after operations cease. Much of the focus on groundwater research, and gains in understanding, has been on contaminant hydrogeology (Lyness and Fennell 2010). Knowledge of contaminant persistence and transfer is essential, but new collaborative efforts could be aimed at fingerprinting the natural and mining-related sources of organic and metallic contaminants in connection with the groundwater flow systems (Savard et al. 2012).

Surface and groundwater systems are inter-related. As reviewed above, recent studies have significantly expanded our understanding of shallow groundwater processes in this hydrogeologically complex terrain. However, much still needs to be done, particularly in understanding the pivotal role of wetlands, ponds and lakes as connectors between surface and groundwater systems (e.g., Ferone and Devito 2004; Smerdon et al. 2005; Devito et al. 2012). Further development of isotope and mass balance approaches, along with other remote sensing platforms that detect soil moisture through dielectric properties (e.g., Canadian Space Agency's Radarsat; Parks and Grunsky 2002) or gravity fields (e.g., NASA's Gravity Recovery and Climate Experiment (GRACE); Schmidt et al. 2008), may help to better understand the complex hydrogeologic processes and the patterns they create in this landscape.

Modeling surface and groundwater hydrology and contaminant persistence and transport will continue to be an important tool in understanding impacts of current and future scenarios. However, even the best models can be inaccurate and imprecise. Monitoring data are essential to improve our understanding of hydrologic processes and for calibrating and validating model outputs. Site-level modeling is often a patchwork of different models and development of regional groundwater models will be an essential step in understanding cumulative effects. Initial steps are being taken to develop standardized modeling practices in the Oil Sands (Jones and Mendoza 2013).

Cumulative effects assessments have been identified as essential in dealing with regional water issues (Government of Alberta 2012b). There are multiple disturbances within the Lower Athabasca watershed including surface and extensive oil sands developments, mines, and forest management disturbances that contribute to changes in water quantity and quality. Despite recognition of the importance of quantifying cumulative effects, a clear approach to achieve this has still to be developed.

A wealth of information has been collected on a variety of topics by consultants, academics, and government scientists related to oil sands development. The CEMA Oil Sands Environmental Management Bibliography (http://osemb.cemaonline.ca/rrdcSearch.aspx) and the Oil Sands Research and Information Network (OSRIN) library (http://www.osrin.ualberta.ca/) are examples of portals to access this information. Recent reports have done thorough reviews of the oil sands environmental and health impacts (e.g., Gosselin et al. 2010, commissioned by the Royal Society), but not without criticism (Timoney 2012) or rebuttal (Hrudey et al. 2012). These reviews and critiques highlight the immense scrutiny placed upon the oil sands industry both from within Canada, but also around the world, to demonstrate environmental stewardship and leadership in developing the oil sands. Progressive management frameworks, recognition of the importance of cumulative effects, risk assessments, and research databases are all steps in the right direction to addressing criticism, but benefits will only become realized once fully implemented.

3.7 Peat mining

3.7.1 Introduction

In some areas of the boreal zone, primarily in Quebec, Manitoba and Alberta, peat is commercially extracted for sale as a horticultural soil conditioner, personal hygiene, industrial absorbent, or as a biofuel source (Daigle and Gautreau-Daigle 2001; Poulin et al. 2004). In 2012, it was an industry that extracted 973 000 tonnes of peat in Canada with an economic value of approximately $186M (NRCan 2013*d*). Peat extraction affects a relatively small portion of the boreal landscape. Environment Canada (2014) estimated that 260 km^2 are, or were at some point in the past, drained for peat extraction, with 140 km^2 currently being actively managed and 110 km^2 no longer under production. Peat mining can pose risks to water resources at a local scale because the process almost always requires the draining and ditching of wetlands, hydrologic disruptions, and the removal of peat and associated vegetation.

There are two methods for peat harvesting with different environmental impacts (Gleeson et al. 2006). Dry harvesting involves draining the peatland first to allow solar drying over a period of time. Following draining, dry peat can be extracted by sod peat production (blocks of peat are cut and extracted to dry), milled peat production (cutting and shredding of the top surface, followed by turning over to promote drying), or vacuum peat production (pneumatic removal of dry upper milled peat). The mined peat is then further dewatered for production of briquettes or pellets. The alternative method is wet harvesting, which involves removing peat without any on-site solar drying and transporting it to a plant for dewatering and thermal drying. This method allows extraction to occur in areas where drainage is impossible, greatly increases the peat production season, and is relatively cost-effective. Wet mining, however, is a new extraction method for which there is little published information on the efficiency and impacts (Gleeson et al. 2006) and is not currently used on a large scale in Canada (Environment Canada 2014).

3.7.2 Water quantity. Impacts on water quantity from peat harvesting differ depending on the extraction method (Gleeson et al. 2006). Wet harvesting removes both the peat and water, thus water drains and evaporates away from the local site. Dry harvesting can result in losses of water to natural water bodies by diversion to drainage ditches prior to harvest; when harvesting is completed, the drainage ditches are blocked and water levels are slowly increased. For both methods, the large amount of peat material that is removed results in a permanent (i.e., over thousands of years) loss of water storage capacity (Laine et al. 1995).

Price et al. (2005) reviewed recent studies in peatland hydrology and found that peatland drainage often resulted in peat compression from seasonal drying and a decrease in the hydraulic conductivity of the remaining peat by over 75%, and that surface runoff from affected sites increased by about 25%. Over time, and in the absence of natural or planned drainage abatement, mined sites continue to get drier. The effects of drying can continue long after the site has been abandoned because of the altered water flow patterns, and because the developing shrub community can deposit annual leaf litter layers to the soil surfaces that restrict water flow to deeper soils (Van Seters and Price 2001, 2002; Price 2003; Price and Whitehead 2004). Colonization of mined sites by trees can cause further water losses at the site by evapotranspiration (up to 25% of precipitation) and by interception (up to 32% of precipitation) (Price et al. 2003). Price et al. (2005) suggest that these impairments of hydrologic function induced by peatland drainage may have broader implications for local or regional scales where disruptions to water flow paths within peat-dominated basins could interfere with natural hydrologic patterns and linkages between uplands and wetlands. In areas where peat mining is extensive, this could cause cumulative effects and altered hydrologic regimes at a broader scale (Bedford 1999; Buttle et al. 2005).

3.7.3 Water quality

Undisturbed peatlands affect downstream water quality by interception and accumulation of inorganic elements, thereby influencing their export to receiving waters (Weis and Weis 2004; Pelster et al. 2008). Additionally, peatlands are a primary source of dissolved organic matter that is created by anoxic and acidic conditions and is exported downstream (Schiff et al. 1998; Nyman 2011). Therefore, disruptions to these biogeochemical processes in peatlands through peat harvesting activities can influence the production and hydrologic transfer of such elements (Laine et al. 1995). The changes in hydrology induced by peat mining can increase levels of suspended sediments and cause changes in water quality (Daigle and Gautreau-Daigle 2001).

Of the few recent studies we found that reported effects of peat mining in or near the Canadian boreal zone on downstream water quality, most focused on suspended sediments. Peatland drainage ditches in New Brunswick (just outside the boreal zone in the hemiboreal subzone (*sensu* Brandt 2009)) delivered elevated concentrations of suspended sediments to downstream receiving waters, despite the use of sediment settling ponds in the peat mining area (Clément et al. 2009). In a separate study, St-Hilaire et al. (2006) also recorded elevated suspended sediment concentrations in downstream areas below sediment settling ponds of peat mining areas. Significantly higher concentrations of suspended sediments were found in streams draining other peat mining sites in comparison to non-mined peatlands and exceeded provincial guidelines for daily maximums to protect aquatic organisms about 70% of the time (Pavey et al. 2007). Prévost and Plamondon (1999) reported elevated concentrations of several elements in water draining from mined peatlands in Quebec than in nearby natural waters. Although these increases in sediment and element concentrations could have implications for biodiversity in receiving waters, there are few published reports on impacts of water quality changes from peat mining on aquatic organisms (Kreutzweiser et al. 2013).

3.7.4 Prognosis

Peat mining is not a serious threat to water quantity and quality in the Canadian boreal zone, given that only about 0.01% of boreal peatlands is being harvested for peat (Cleary et al. 2005). However, at a local scale, peat mining can measurably impact water quantity and quality in mined areas and in downstream receiving waters, with potential for adverse effects on some aquatic biota (Kreutzweiser et al. 2013). With increasing interest in peat as a biofuel source (Telford 2009), the area of peat mining in the boreal zone could expand, with potential impacts to water resouces and recovery of wetlands. Burning peat for heating has a long history in northern Europe, but it is a relatively new consideration in Canada. Some early trials for co-firing peat at a thermal generating station were, however, not very promising. Burning peat was not a clean source of energy, and other issues related to energy costs from drying and processing prior to burning are likely to limit its potential as biofuel feedstock (Gleeson et al. 2006).

Given that peat accumulates slowly over thousands of years (Ali et al. 2008), peat mining may not be a sustainable industry at local scales because of the profound alterations to the hydrologic and therefore ecological functions of peatlands, and because of the time required to replace similar volumes of peat. Price et al. (2003) point out that it is still uncertain whether the hydrologic, biogeochemical (carbon storage), and ecological functions of peatlands can be restored within a reasonable time frame, and suggest that efforts should be made to retain more water on site during peat extractions to minimize overall impacts. The peat-mining industry, under the auspices of the Canadian Sphagnum Peat Moss Association, acknowledges the inherent need for responsible peatland management to improve the sustainability of their industry (www.peatmoss.com). As part of their preservation and reclamation policy, they recommend that exploitation methods minimize the affected surface area, that some plots of natural peat be retained as a buffer and recolonization source, that vegetation removed should be retained and replanted on abandoned sites, and that surface re-profiling, mulches, and seepage reservoirs be applied to reduce hydrologic impacts and to assist vegetation re-establishment. Price et al. (2003) also recommended blocking ditches, constructing peat banks or terraces, and creating shallow depressions as means to retain water in rehabilitation efforts.

The International Peat Society's Strategy for Responsible Peatland Management (Clarke and Rieley 2010) outlines best management principles for sustainable peat management. The principles are based on conservation of biodiversity, mitigation of impacts on hydrology and water regulation, implications for carbon cycling and climate change, and application of the latest technologies in post-mining rehabilitation and reclamation. It also advocates for the transfer and sharing of knowledge, engagement with local interest groups, and good governance to ensure conservation of ecological goods and services from peatlands.

3.8 Atmospheric emission and deposition of pollutants

3.8.1. Introduction

Natural resource extraction activities within the boreal zone, as well as other industrial activities outside of the boreal zone, both from within Canada and around the world, create emissions that influence boreal water bodies. These emissions initiate from point sources such as industry, or diffuse sources such as forest fires. Emissions from industry (Fig. 10) are the gaseous and particulate products of combustions (e.g., nitric oxides (NO_x) and sulphuric oxides (SO_x)) from biomass and fossil fuel burning, and volatilized bi-products from extracting and processing resources (see Fig. 6 in Brandt et al. 2013). These emissions quickly spread from local sources and are transported throughout the airshed by the prevailing winds eventually falling as wet or dry deposition. Although the pollution may become diluted in the atmosphere, impacts have been observed to occur with chronic exposure (e.g., Sudbury, eastern North American seaboard).

3.8.2. Water quantity

Atmospheric emissions of pollutants do not have direct effects on water quantity. However, particulate emissions, aerosols, and greenhouse gases are known to affect atmospheric radiative balances (either absorb or reflect solar radiation depending on the particle) as well as serving as cloud condensation nuclei (Gieré and Querol 2010). These mechanisms have largely unknown (Cotton and Yuter 2009) but likely small impacts on local or regional precipitation patterns within the boreal zone.

3.8.3. Water quality

Surface water quality is influenced by wet and dry atmospheric deposition of pollutants either through direct input onto surface waters or indirectly through soil-mediated effects. When sulphate- and nitrate-based emissions combine with precipitation, the resulting acidic deposition can reduce pH in soil and surface waters. Acidification synergizes metal export and accumulation in receiving waters (LaZerte 1986) and confounds, often increasing, the bioavailability and toxicity of metals to aquatic organisms (Keller and Pitblado 1986; Schindler 1988). A vast literature exists on the impacts of acidification and associated metal toxicity on aquatic ecosystems, and demonstrates that when pH levels in receiving waters have dropped to or near 4, drastic and long-lasting effects on aquatic organisms can occur (e.g., Keller and Yan 1991).

Atmospheric nitrogen deposition also impacts the amount of nitrogen within forest soils having impacts on forest productivity, soil nutrient cycling, and nutrient (particularly C and N) exports (Hyvönen et al. 2007).

There is a long history of studying acid rain impacts in eastern North America. The eastern parts of the Canadian boreal zone and the hemiboreal subzone were traditionally the area of highest acidic deposition and acidification of surface waters (Environment Canada 2005*a*). From the 1960s through to the 1980s, emissions in eastern North America were high because of combustion of fossil fuels used in industry and in thermal hydroelectric generation within the industrial heartland of eastern United States and southern Canada. Since the implementation of the United States Clean Air Act (1990) and the Canada-United States Air Quality Agreement (IJC 1991), SO_x levels have dramatically been reduced, and NO_x has levelled off, or shown a slight decline. Controls on sulphur dioxide emissions in North America resulted in large reductions in sulphate concentrations and increases in pH in many eastern boreal lakes by the mid-1990s (Keller 2009). Numerous studies are tracking the chemical and biological recovery of water bodies from acidification (Keller et al. 1992; Gunn 1995; Carbone et al. 1998; Doka et al. 2003; Jeffries et al. 2003*a*, 2003*b*; Snucins and Gunn 2003; Clair et al. 2007; Keller et al. 2007; Gray and Arnott 2009). Overall, these show that the pH of surface waters in most areas near historic industrial operations is improving, but that metal concentrations often remain elevated above reference or target levels and that biological recovery usually lags behind chemical recovery.

Although improvements have been made at regional (northeastern United States and Canada) and local scales (e.g., Sudbury; Keller 2009), parts of the boreal zone in

Alberta and Saskatchewan have soils with relatively low buffering capacity, and therefore water bodies in that region are at risk of acidification from atmospheric pollutants (Hazewinkel et al. 2008; Aherne and Shaw 2010; Scott et al. 2010; Whitfield et al. 2010). Deposition of SO_x and NO_x in the oil sands region of Alberta has increased over the last 40 years, although levels have shown some decline over the period 2000 to 2005 with the introduction of sulphur capture technology (Alberta Environment 2008) and are low compared to areas of eastern North America (Whitfield et al. 2010). Scott et al. (2010) surveyed 259 headwater lakes within 300 km of Fort McMurray, AB. They found that 60% of the lakes were classified as sensitive and 8% as very sensitive to atmospheric deposition of acidic pollutants. They determined that although acidification appears not to be significantly advanced, many dilute oligotropic lakes with pH 6 to 6.5 are vulnerable to acidic pollutants.

The absence of evidence for acidification does not necessarily imply that emissions from the oil sands region are environmentally benign, but rather suggests that the biogeochemistry of these lakes differs fundamentally from well-studied acidified counterparts in eastern North America and northern Europe (Hazewinkel et al. 2008). Model simulations suggest limited risk of acidification primarily due to sulphur retention in the basins, however drought may induce episodic depression of acid neutralizing capacity (Whitfield et al. 2010). Thus, it has become apparent that the recovery of lakes from acidification is closely linked with the responses to, and interaction with, other large-scale environmental stressors like climate change and calcium declines (Keller 2009).

Across the boreal zone, trends in critical loads (i.e., an estimate of an exposure to one or more pollutants below which significant harmful effects on specified sensitive elements of the environment do not occur according to present knowledge) of acidity and their exceedances (i.e., acid deposition load minus critical load) show that many areas have high sensitivity to acidification and exceedance of critical loads (Carou et al. 2008) (Fig. 11). These areas are mostly characterized by non-carbonate bedrock and shallow, coarse-textured, upland soils that have a low buffering capacity. A primary goal of *The Canada-Wide Acid Rain Strategy for Post-2000* is to meet the environmental threshold of critical loads for acid deposition across Canada, by pursuing further acidifying emission reductions in Canada and the United States, preventing pollution and protecting ecosystems not yet impacted by acidifying emissions. These national maps of the sensitivity of ecosystems serve as a useful means to illustrate where and by how much forest ecosystems are at risk of sulphur and nitrogen deposition damage, and therefore where and by how much emission controls will be needed in order to protect sensitive ecosystems (Carou et al. 2008).

3.8.4. Prognosis

Investigations into atmospheric deposition of acidic pollutants from processing emissions and their impacts on water quality are ongoing. Emissions of acidifying pollutants have declined by about 50% nationally, and by much more in some areas (CCME 2011). However, atmospheric deposition still exceeds critical loads, and acidifying emissions are still increasing in other areas (FPTC 2010). To the extent that natural resource development activities and their related emissions are likely to increase in some areas of the boreal zone (e.g., oil sands region, Hudson Bay lowlands), the deposition of acidic pollutants on landscapes with concomitant impacts on water quality are likely to increase at local to regional scales. To further reduce emissions, consideration could be given to improve scrubbing technologies, increased processing efficiencies, and cleaner fuels.

4. Cumulative effects and non-linear, threshold and tipping point behaviors

Although much of the boreal zone remains unaffected by natural resource development activities, some regions, particularly in southern parts, have been affected substantially (Fig. 2). This anthropogenic footprint has resulted in fragmentation of the landscape (Schneider et al. 2003). Many continuing industrial activities have reduced their disturbances, but still produce measurable cumulative impacts to water quantity and quality over local to regional scales, and legacy impacts of former industrial installations in particular will persist for decades (Table 3; Niemelä 1999; Ptacek et al. 2004; Keller et al. 2007).

The release of pollutants and contaminants into surface waters can occur not only at the site of disturbance but can also be transported many kilometers downstream (Culp et al. 2003; Muscatello et al. 2008; Kelly et al. 2010). Individual effects of headwater disturbances can potentially accumulate and affect downstream ecosystems and their services (Townsend et al. 2008). Multiple effects of point and diffuse disturbances within and among industrial sectors that are hydrologically connected can also accumulate over the landscape (Keller 2009; Seitz et al. 2012), leading to greater uncertainty and unpredictability of ecosystem response (Dubé 2003; Gunn and Noble 2011). It is expected that both individual and cumulative effects resulting in non-linear and tipping point behaviour will become the new “norm” in boreal ecosystems in response to intensified and/or expanded natural resource development (Kinzig et al. 2006; Foote 2012).

Cumulative Effects Assessments (CEAs) are a requirement under various provincial Environmental Impact Assessment laws and regulations. CEAs examine the interactions between changes in basin structure and function that accumulate and the response of the river ecosystem under different futures scenarios of development within the basin (Seitz et al. 2011). A CEA demands that proponents examine the cumulative effects associated with their proposed development alongside relevant past, present, and future projects. The CEA is often done at the project scale, not at the regional scale (Squires et al. 2009). CEAs are methodologically complex as cumulative effects are frequently interactive or synergistic in nature (Dubé 2003) and there is not a single conceptual approach that is widely accepted by both scientists and managers (Squires et al. 2009). Overcoming challenges set forth by scaling issues, diverging views, different policies and legislations, and complex ecological pathways is in itself the main challenge for those who try to carry out an effective CEA (Gunn and Noble 2011; Seitz et al. 2011). Examples of effects-based CEAs in the boreal zone include the Moose River Basin study (Munkittrick et al. 2000), Northern Rivers Basin Study (Culp et al. 2000), Northern Rivers Ecosystem Initiative (Dubé et al. 2006), and Athabasca River Basin study (Squires et al. 2009).

Seitz et al. (2011) suggest that regional CEAs should take a multiple stakeholder approach, with government assuming leadership to establish management objectives as well as compliance and effectiveness monitoring at the regional basin scale. The Cumulative Effects Monitoring Association (CEMA), within the Regional Municipality of Wood Buffalo, AB, is an example of a multiple stakeholder group established to produce recommendations and management frameworks pertaining to the cumulative effects of oil sands development in northeastern Alberta, which are, once complete, forwarded to provincial and federal government regulators (http://cemaonline.ca/index.php/about-us). Technical and scientific work performed for CEMA is completed through the direction of working groups (Land, Reclamation, Air, Wetland, Traditional Knowledge) using smaller task groups to focus on specific issues (http://cemaonline.ca/index.php/working-groups). Creation of a regional CEA is, however, not part of its mandate.

Climate change complicates efforts to understand cumulative effects within ecosystems. For example, long-term increases in temperature and changes in precipitation, coupled with increased frequency of extreme events such as droughts and floods (as anticipated for Canada’s boreal zone by Price et al. 2013), may create ecosystem instabilities that promote non-linear responses (Keller et al. 2007; Vose et al. 2011).

Innovative research methods that use spatially explicit data and long-term monitoring networks would support efforts to quantify cumulative effects of natural resource development activities on basin processes at large spatial and temporal scales (Creed et al. 2011a; Vose et al. 2011). Development of predictive models may also help to make informed decisions on projected impacts of future natural resource extraction activities under different climate and possible mitigation scenarios (Chapin et al. 2004; Creed et al. 2011a), and require rigorous scientific data, and a tighter coupling between management and monitoring activities (i.e., adaptive management; see Gauthier et al. 2014). Furthermore, model outputs need to be better integrated into decision-making frameworks for prioritizing boreal forest management strategies that minimize impacts of natural resource development on boreal aquatic ecosystems and the services they provide. This is an active research focus of the Canadian Network of Aquatic Ecosystem Services in the Healthy Forests, Healthy Waters Theme (CNAES 2014).

5. Improving water stewardship

Natural resource industries have and will continue to require access to water, a need that will likely continue to expand (UNEP 2009; NRTEE 2011). This review has identified that water quantity and quality has been impacted by industrial development in some areas of the boreal zone. Although some natural resource developments have localized, short-term effects on water resources (e.g., forest harvesting at stand or sub-basin scales), others have localized, long-term effects (e.g., mining activities and their effluents), and still others have regional, long-term impacts (e.g., processing emissions, hydroelectric power generation, oil sands development). Certain areas of the boreal zone are at increased risk of cumulative effects of multiple natural resources development on water resources (e.g., Lower Athabasca Watershed). Effectively managing impacts of natural resource development across multiple spatial and temporal scales of Canada's boreal zone is challenging. Improving water stewardship performance in the natural resource sector could include the following considerations.

5.1 Awareness of water resource values

The places where humans benefit from ecosystem services is often different from the places that produce those services, thus feedbacks to ensure the continued provision of these services is often missing (Brauman et al. 2007; Keeler et al. 2012). This is particularly relevant for the water resources of the boreal zone, where water-related ecosystem services generated in source areas are often hundreds of kilometers away from the downstream human settlements that benefit from those services. The lack of a clear connection between the locations of the supply of services and the beneficiaries of those services may reduce the recognition of the value of water-related ecosystem services (Brauman et al. 2007).

Accurate valuation of water resources may increase the public awareness of their value and encourage more efficient water use (Hoover et al. 2007). Calculation of a water footprint, which is the volume of water appropriated to produce a product taking into account the water consumed and polluted in the different steps of the supply chain, is one approach to water valuation (Hoekstra and Mekonnen 2012; ISO 2012; Launiainen et al. 2013). Although there is some criticism of water footprint estimates, which vary dramatically depending on the methodology used, the concept has had notable success in raising awareness about water use by providing a previously unavailable and seemingly simple numerical indicator of water use (Chenoweth et al. 2013).

Economic valuation of water-related ecosystem services can be a powerful tool for policy evaluation because it provides a common metric with which to make comparisons between natural resources and natural capital (Heal 2000). However, values associated with ecosystem goods and services are challenging to estimate. Frequently, there are insufficient data to attempt to quantitatively incorporate all environmental value information into prices or to construct accurate full cost accounting (Adamowicz et al. 2007).

As one example within the boreal zone, Anielski and Wilson (2005) estimated Canada’s natural capital based on boreal ecosystem services at a total non-market value of $93.2B per year based on 2002 Canadian dollars, approximately 2.5 times greater than the net market value of boreal natural capital extraction (Anielski and Wilson 2005). They later revised the ratio upward to 13.8, primarily from a jump in nonmarket ecological service values due to revaluation of stored carbon in forest and wetlands (Anielski and Wilson 2009). The majority of the non-market values of the boreal zone was attributed to water-dependent services, in particular, the ecosystem services provided by wetlands (i.e., groundwater aquifer recharge, water purification, flood control, and biodiversity) (Anielski and Wilson 2005). The valuation of ecosystem services continues to be an active and important field of investigation for environmental economists (Liu et al. 2010; Molnar and Kubiszewski 2012). Considerable research effort is being focused on improving methodologies so that results are transparent and defensible (Liu et al. 2010; Salles 2011).

5.2 Governance of water resources

Water is one of the most challenging natural resources to govern and manage due to the division of power and shared responsibilities for management among all levels of government (de Loë 2008; NRTEE 2011; Canada West Foundation 2011). Water governance in Canada has been founded on a collection of statutes and policies, involving all levels of government (Saunders and Wenig 2007; Hipel et al. 2011). Water governance within the boreal zone is primarily under provincial or territorial responsibility that includes a mix of federal and provincial legislation, even though many boreal basins cross multiple jurisdictions and ecozones.

The current federal water policy dates back to 1987 (Environment Canada 1987) and the federal wetland conservation policy to 1991 (Government of Canada 1991). These policies were never fully implemented and have been criticized for not reflecting new pressures on water resources from expanding industries, water diversions, international water export, and climate change (Morris et al. 2007; Muldoon and McClenaghan 2007; Sanders and Wenig 2007; NRTEE 2011). National consistency on environmental issues concerning water could be achieved through the Canadian Council of Ministers of the Environment (CCME) (de Loë 2008; Hipel et al. 2011). The CCME serves as a principal forum for members to develop national strategies, norms, and guidelines for setting environmental standards (Hipel et al. 2011).

There have been repeated calls from a diverse range of groups and sectors for renewed federal action on water (Morris et al. 2007; Pollution Probe 2008; de Loë 2008, 2009; McLaughlin 2009; Hipel et al. 2011; NRTEE 2011). Constitutional and practical considerations require that leadership in water governance come from both federal and provincial governments (Morris et al. 2007; McLaughlin 2009). However, Hipel et al. (2011) suggest that national water governance is best led by the federal government given its experience in cooperatively managing water resources in shared basins with the United States *via* the International Joint Commission that was established under the Boundary Waters Treaty of 1909.

There are differing opinions as to the form in which federal action on water could take, whether it be a revised federal water policy, water framework, or pan-Canadian water strategy. Despite differences in the name attached to the action, most proposals share a similar vision and approach (Bakker 2007; Morris et al. 2007; de Loë 2008, 2009; NRTEE 2011). Their shared vision is to achieve a comprehensive and coordinated approach to water governance in Canada as a platform for addressing water-related challenges and opportunities that demand a national perspective. A guiding principle, as proposed by NRTEE (2011), is that water has economic, environmental and social values and should be managed in trust without harm to its sustainability or to that of the ecosystems in which it occurs or flows through. Proponents of a national water strategy suggest that a key element of the strategy could be a national water council that would include basin management authorities. These authorities would work with the provinces in undertaking the day-to-day water management functions. The national water strategy could focus on priority areas, such as enhancing national capacity for freshwater protection, responding to the impacts of climate change, ensuring sustainable natural resource development, securing safe drinking water for all Canadians, protecting aquatic ecosystems and aboriginal water rights, promoting water conservation, circumventing inter-jurisdictional and trade conflicts, and supporting world class water science (Morris et al. 2007).

Some proponents of national action on water resources (e.g., Bakker 2007; Morris et al. 2007) suggest that Canada's water strategy could follow the European Union (EU) Water Framework Directive (Commission of the European Communities 2007). Since 2000, the EU has applied a collaborative approach that facilitates action across political and cultural borders by investing in science, knowledge sharing, and a common operational approach to water management (Lagacé 2011). Under the guiding principles of harmonization and subsidiarity, the goal of the directive is to establish common standards and practices across the EU that safeguard water quantity and quality of for the future (Lagacé 2011).

Proponents of a national water strategy for Canada have suggested that it should include the following themes. First, water resources within basins would be treated as a whole within an integrated system, acknowledging that many interconnected water-related factors must be considered and that the trade-offs among competing stakeholder and environmental requirements be taken into account (Global Water Partnership and International Network of Basin Organizations 2009; Hipel et al. 2011). Second, water governance and related policies and management would be adaptive in order to handle the largely unpredictable behavior of the environment and society brought about by intrinsic complexity, uncertainty, and interconnectedness (Hipel et al. 2011) and by continually learning (through on-going assessments) from the success (or failure) of policies that are implemented. Third, water governance and management would be collaborative, integrating and coordinating federal, provincial, territorial, First Nation policies as well as a broad range of other stakeholders (Berkes and Folk 1998; Blatter and Ingram 2001; Berkes 2002; Finger et al. 2006; Bouleau 2008).

Despite widespread agreement in terms of the purpose and approach of a national water strategy, its development and implementation would not be straightforward (Bakker 2007). Experiences in jurisdictions such as Brazil, Australia, South Africa, and the EU that have developed or are developing water strategies illustrate these difficulties (de Loë 2008). Regardless of the challenges, de Loë (2008) points out that the benefits of a national water strategy would include: (1) an improved position for meeting growing international expectations and obligations; (2) a stronger capacity to respond to threats (e.g., climate change, extractions, contaminants, legacy effects) and opportunities (e.g., funding programs); (3) a clarification of responsibilities among jurisdictions; (4) a greater consistency in responding to concerns; and (5) more effective decision making when problems transcend jurisdictional boundaries.

Although a formalized national water strategy could improve effective and consistent decision making for water stewardship (Bakker 2007), its absence is not necessarily a barrier to making progress in managing boreal water resources. A number of provinces and territories have recently developed water strategies including Quebec (2002), Alberta (2003), Manitoba (2003), British Columbia (2008) and Nova Scotia (2010) and NWT (2010) (Morris et al. 2007; de Loë 2008; NRTEE 2011). Most provinces or territories have some form of basin management organization, although the roles and responsibilities of these organizations vary among jurisdictions.

5.3 Innovative water management

Several reports have illustrated that water supply challenges can often be traced to misallocation rather than to water scarcity (Environment Canada 2004; Schoengold and Zilberman 2007; Muldoon and McClenaghan 2007; Hipel et al. 2011), although this may change in the future as a result of the anticipated consequences from climate change. Misallocation can be the result of poor or insufficient baseline data (NRTEE 2009). Accurate, complete, and current water-quantity data are critical in establishing water-management systems in which water is effectively allocated and efficiently used (McLaughlin 2009). Sustainable water management requires current and accurate information on water supply and demand, and reasonable estimates of how supply and demand may change in the future.

On the supply side, current data on water quantity, monitoring capacity, and reporting protocols are available through the National Hydrometric Program operated by the Water Survey of Canada, and often supplemented by provincial monitoring. However, the program has been criticized for lacking a long-term plan for expanding and maintaining monitoring stations, and lacking clear directions for adapting to water-quantity threats such as climate change (NRTEE 2011). These concerns are particularly relevant in the boreal zone where there is a low density of monitoring sites (Fig. 6), increasing industrial development, and high vulnerability to climate change (Schindler and Lee 2010).

On the demand side, there are gaps in availability and discrepancies in the quality of data collected across provinces (Hipel et al. 2011; NRTEE 2011). Demand estimates are often based on water allocation permits, but the actual quantity taken or returned to water bodies is typically unknown (NRTEE 2011). However, overall trends in water demand during the period 1981 to 2005 indicate that water extraction or use did not increase at the same rate as economic growth, reflecting improved water-use efficiencies and conservation among most natural resource industries (NRTEE 2011). These efficiencies may offset increasing water demand from expanding natural resource development in Canada, such that the overall demand for water by the natural resource sector at the national scale is expected to hold steady or increase marginally over the next couple of decades (NRTEE 2011). There may be exceptions at a regional scale. For example, the largest increases in industrial water use in Canada's boreal zone by 2030 that are expected to present water supply challenges are associated with oil sands development (both *in-situ* and surface mining) (Bruce et al. 2009; Hipel et al. 2011).

There is growing recognition across Canada's natural resource sectors that water stewardship be considered in business planning and that boreal ecosystem security be incorporated in water allocation decisions (McLaughlin 2009). Economic instruments could facilitate these considerations while addressing water stewardship performance outside of a government framework (Brauman et al. 2007). Economic instruments are emerging tools for water management that complement, rather than replace, traditional government control approaches (Shrubsole and Draper 2007). Economic instruments provide incentives for behavioural change, generate revenue for financing environmental initiatives, promote technological innovations, and reduce wasteful water usage at the lowest cost to society (Shrubsole and Draper 2007; NRTEE 2011). Economic instruments include water surcharges, tradable water permits, subsidies, and financial incentives (Shrubsole and Draper 2007).

Water pricing on a volumetric basis can help achieve water reduction objectives, with modest impacts to most sectors and the national economy (NRTEE 2011; McLaughlin 2009). Typically, industrial water-use systems run on a cost-recovery basis, with fees that are set very low and that do not reflect the true value of the water being used (Bruce et al. 2009). For example, the average cost of gross water use is about \$0.13 per m^3 across all natural resource sectors. To achieve a 40% reduction in water intake would require an increase in price ranging from \$0.50 to \$0.70 per m^3 (NRTEE 2011). Charging for the use of water at prices that more closely reflect the actual value of water used in natural resource development could promote responsible water use (McLaughlin 2009). For example, incentives through water pricing can lead to innovation in technologies for reducing water uptake by increasing recirculation or recycling (Adamowicz et al. 2007).

Water trading (the reallocation of water permits from license holders with a surplus of water to those with a need) is another economic instrument that exists in many parts of the world, but within Canada, it exists only in Alberta and only at a very limited scale (Adamowicz et al. 2007; Shrubsole and Draper 2007; NRTEE 2011). It can be a transformational strategy in the sense that trading water rights would be a fundamental shift in water management such that water managers and regulators would become market managers that could potentially drive up water prices. Such a transformational strategy warrants caution, time, and careful thought to implement (NRTEE 2011). Practical experience in water markets remains limited, with other countries having demonstrated both successes and failures in implementation (NRTEE 2011). A move toward a national water trading scheme would require an understanding of the level of acceptance for, and potential negative implications of, water trading, and appropriate institutional and legal safeguards would need to be included (Adamowicz et al. 2007).

Financial incentive programs are another form of economic instruments that can result in improved water stewardship performance. For example, the Pulp and Paper Green Transformation Program of Natural Resources Canada funded projects that promoted technologies for improved environmental performance in the pulp and paper sector through increased efficiencies (NRCan 2010). Incentive programs can be more effective than tax subsidies or other mechanisms that discount the product at the expense of the ecosystem (Anderson et al. 2010).

Voluntary initiatives can also promote water stewardship, and are taken on by industries in the absence of government intervention. Voluntary initiatives are guided by a shared commitment among participating organizations to achieve a desired outcome such as increasing requirements to respond to market and customer expectations, maintaining a social license to operate, enabling internal management and performance improvement, demonstrating environmental responsibility, and internally addressing knowledge gaps (NRTEE 2011). The NRTEE (2011) provided several examples of voluntary initiatives among natural resource sectors that included: 1) industry-driven performance initiatives with an emphasis on improved water-use efficiencies (e.g., Oil Sands Leadership Initiative, Global Social Compliance Programme, International Council on Mining and Metals); 2) standards and certification programs to improve environmental and social management practices, to promote brand recognition, and to demonstrate responsible resource development (e.g., Alliance for Water Stewardship, ISO 14046 Water Footprint, Forest Stewardship Council Certification for Ecosystem

Services); 3) international reporting initiatives to promote transparency and accountability among natural resource sectors (e.g., Global Reporting Initiative, Carbon Disclosure Project – Water Disclosure); and 4) accounting and management initiatives to identify risks and areas for performance improvement, and to demonstrate corporate social responsibility (e.g., WBCSD Global Water Tool, Global Environmental Management Initiative).

5.4 Integrated water management

Effective water management can be attained through adaptive and integrated ecosystem-based approaches that incorporate collaborative dialogue from multiple stakeholder groups (Berkes and Folk 1998; Blatter and Ingram 2001; Berkes 2002; Finger et al. 2006; G3 Consulting Ltd. 2009). Under this scheme, water is often managed at tertiary (local to regional) basin scales, but is considerate of implications at regional, national, and international scales. Integrated regional water planning brings together local management plans and integrates them over large areas that include multiple sectors. Undertaking integrated water management is challenging in that it requires knowledge of regional surface water and groundwater hydrology. Furthermore, natural resource sectors are often disproportionally dispersed over the region, and each sector may have disparate water stewardship standards and water allocation requirements such that the cumulative effects on water quantity and quality at a regional scale are unknown (NRTEE 2009).

Creed et al. (2011b) suggest that integrating water management plans at regional scales could be improved by following a common set of management-oriented hydrologic principles. They proposed six principles and complementary management actions that link natural resource development to water stewardship. These principles are: 1) determine hydrologic system boundaries and consider the entire hydrologic system where development or management actions take place; 2) conserve critical hydrologic features by minimizing disturbance to areas involved in the source, movement, and storage of water; 3) maintain connections between hydrologic features by minimizing disruptions to water, sediment, and nutrient flows; 4) respect the temporal variability in hydrologic processes, over short-term (i.e., day-to-day operations) and long-term time scales (i.e., 100-year planning horizons) (e.g., instream flow requirements); 5) respect the spatial heterogeneity in hydrologic processes among different scales within a basin (e.g., stand, hillslope, basin) and among different hydrologic regions (e.g., discharge dominated *vs.* evapotranspiration dominated); and 6) maintain redundancy and diversity of hydrologic form and function within basins. Maintaining undisturbed sub-basins or other contiguous areas of forest and peatlands within a basin can also help to protect water resources because they prevent or slow human-caused alteration of hydrology from industrial land-use activities and can prevent or slow the spread of pollutants and invasive species (Wells et al. 2010; Andrew et al. 2014).

Integrated water management at a regional scale requires knowledge of the fundamental hydrologic processes at work. As previously discussed (see section 2.1), key hydrologic processes differ across the boreal zone and these differences may influence the effectiveness of water management strategies. Therefore, integrated water management plans at regional scales across the boreal zone will need to adjust for these differences. However, this is increasingly challenging because knowledge gaps remain with regard to some of the underlying hydrologic processes, such as surface water and groundwater connections and their susceptibilities to natural resource development activities under differing regional conditions. In conjunction with these knowledge gaps, additional uncertainties are likely to arise from climate change impacts (Bruce et al. 2009; McLaughlin 2009). Further to these challenges for integrated water management, Canada's water science capacity to provide current and reliable data appears to have declined in recent decades (McLaughlin 2009) and resources for systematic water-related research and monitoring have not kept pace with expanding natural resource development (Bruce et al. 2009). In the absence of current data and regional-specific information, a precautionary approach to water resource management is required (Rosenberg International Forum 2013).

Even with adequate knowledge of hydrologic processes within the basin being managed, reliable data on water demand and the potential impacts of natural resource development across all sectors are required for integrated water management. Providing these data to managers for basins in the boreal zone, particularly in remote areas, is difficult because data sources are often incomplete and inconsistent (McLaughlin 2009). Others have pointed out that reliable measurements or estimates of basic water inventory metrics, such as wetland areas, lake volumes, river and glacial runoff, groundwater resources, and pollutant loadings are often scarce or absent at regional scales, thereby precluding the application of effective integrated water management strategies (Nowlan 2007; Bruce et al. 2009; Hipel et al. 2011).

In addition to regionally-specific information on hydrologic processes, water resource inventories, natural resource development activities, water demands, and risks to water quantity and quality, integrated water management approaches require data processing tools (Morris et al. 2007; Bruce et al. 2009). A critical gap is tools to synthesize information spatially and temporally, and to model cumulative effects, to make predictions about future scenarios and to forecast risks. Spatially distributed hydrologic models are available and can provide some assistance, but they are often incomplete and difficult to parameterize (Creed et al. 2011a). Although there have been advances in surface water models and improvements to groundwater models (Beckers et al. 2009), coupled surface water – groundwater models are still in their infancy and are important for understanding risks to water resources from natural resource development. Models that link natural resource development and water resources to economic endpoints are particularly uncommon (Bruce et al. 2009). Bruce et al. (2009) also point out that uncertainties in model predictions due to model assumptions and limited calibrations need to be clearly communicated to end-users.

There are several emerging examples of integrated water management in the boreal zone. Alberta Environment and Sustainable Resource Development has river management frameworks for many of their key boreal basins. For example, the Lower Athabasca Surface Water Quality Management Framework (Government of Alberta 2012c) and the Lower Athabasca Groundwater Management Framework (Government of Alberta 2012b) are comprehensive in their scope in assessing water resources and stressors to these water resources in the basin. Both of these frameworks are new, however, and it will take many years for them to be implemented and even more years to determine their effectiveness.

A further example of integrated water management planning is provided by a regional river planning initiative, the Mackenzie River Basin Board (MRBB), which has been in existence for over a decade. The Mackenzie River is the largest north-flowing river in North America and the longest river in Canada, draining nearly 20% of the country (Hipel et al. 2011). It is a unique global resource providing ecosystem services locally (biodiversity) and globally (climate regulation through carbon sequestration and freshwater flows to Arctic Ocean). Although it is among the least fragmented large-scale basins in North America, it is increasingly at risk from climate warming and from development pressures including oil and gas, mining, and hydroelectric power generation (Hipel et al. 2011; Rosenberg International Forum 2013). Initially, the basin's planning governance was complicated by jurisdictional discontinuity (de Loë 2010; Rosenberg International Forum 2013) and therefore a need for an effective transboundary water management plan was recognized (de Loë 2010). In response, the Mackenzie River Basin Transboundary Agreement was developed in 1997 among the governments of Canada, Saskatchewan, Alberta, British Columbia, Yukon and Northwest Territories. It was founded on guiding principles of equitable utilization, prior consultation, sustainable development, and maintenance of ecological integrity (www.mrbb.ca/information/31/index.html). Although the agreement has been in place since 1997, a recent report concluded that there has been little effective follow-through on the Master Agreement (Rosenberg International Forum 2013). However, the report did acknowledge that the MRBB was an appropriate model for integrated resource management, and could provide effective management if it was fully implemented (Rosenberg International Forum 2013). That report also suggested that the MRBB's effectiveness could be strengthened if it had increased authority for decision-making and an expanded scientific knowledge foundation for management actions with research priorities set through a science advisory panel (Rosenberg International Forum 2013). It is likely that this model could be adapted for additional basin management boards in the boreal zone. The experience of the MRBB highlights that integrated water management authorities will need to be properly resourced and empowered to turn their visions into effective action (Sanders and Wenig 2007; Wells et al. 2010; Rosenberg International Forum 2013).

6. Summary and Conclusion

The boreal zone hosts more than half of Canada's total renewable supply of freshwater and the majority of Canada's wetlands (bogs, fen, swamps, marshes and shallow open waters). Boreal water-related ecosystem services are important to the well-being of society. Natural resource industries within the boreal, which contribute significantly to Canada's GDP and employ hundreds of thousands of Canadians, are dependent on a renewable supply of water. Without water to draw, we can hew no wood, mine no coal, or extract no oil (McLaughlin 2009).

The boreal zone spans the continent and, consequently, there is substantial variation in the dominant hydrologic processes across this large biome. This variation is important to understanding how disturbances may (or may not) affect water resources. The western boreal zone is characterized by a sub-humid climate (precipitation $\cong$ evapotranspiration), deep, permeable soils and surficial geology, and relatively low relief. Thus, hydrologic processes are dominated by vertical flows and evapotranspiration, with little runoff generation. In contrast, the eastern boreal zone is characterized by a humid climate (precipitation > evapotranspiration), shallow confining layers or bedrock, and relatively high relief. Here the dominant hydrologic processes are lateral flows, with substantial runoff. These differences in hydrologic regimes must be taken into account to determine how a particular disturbance affects water quality or quantity.

The literature on the risks to water quantity and quality posed by natural resource development, including roads, forest management activities, pulp and paper operations, mining, oil and gas extraction, peat mining and electric power generation, were reviewed and prognoses developed. Natural resource sectors have and continue to be responsible for significant changes to boreal water quantity and quality through extractions, diversions, hydrologic disruptions, effluents, seepages, and emissions. In general, the evolving system of regulations, guidelines, and standards for natural resource development are improving the prognosis for water resources for some sectors in some areas, but in other areas legacy impacts persist, deposition of atmospheric pollutants to receiving waters continues, and expanding natural resource development poses new risks to both water quantity and quality. Additionally, the potential confounding influences of a changing climate may exacerbate natural resource development impacts on water resources.

The cumulative effects of natural resource development activities over space and time, coupled with the uncertainties related to climate change, have not been adequately examined in the published literature, and therefore critical information gaps remain (Gunn and Noble 2011). As natural resource development increases into previously unmanaged areas, the resilience of aquatic ecosystems in the boreal zone may be challenged, at least at local or regional scales. An integrated approach to natural resource management that includes economic and ecological valuation of water resources, improved water governance, and conservation-based basin management will lead to enhanced management of water resources (Schindler and Lee 2010). Continued efforts in coordination between regional, provincial, and federal governments and a unifying national water strategy could enhance the sustainability of Canadian boreal water resources for future generations.

7. Acknowledgements

The authors wish to acknowledge the contributions of Laura Sanderson, Benoit Hamel, and Stephanie Wilson (Canadian Forest Service) and David Aldred, Johnston Miller, and Adam Spargo (Western University) for their contributions to the preparation of this review. Jagtar Bhatti, James Brandt, Kate Edwards, Doug Maynard, and Brad Pinno reviewed an earlier draft of this manuscript. The support of the Canadian Forest Service, Natural Resources Canada to KLW, FDB and DPK is gratefully acknowledged. The support of NSERC's Canadian Network of Aquatic Ecosystem Services to IFC is also gratefully acknowledged. Finally, the anonymous reviewers are thanked for their contribution in improving the manuscript.

List of Tables

List of Figures

Figure 9: Oil and gas well site density per 100 km × 100 km grid (source: Provincial and territorial government databases).

Figure 10: Sulphur oxides (A) and nitrogen oxides (B) emissions for 10 × 10 km grids (source: NPRI, Air Pollutant Emissions; Environment Canada 2008).

Figure 11: Exceedance (eq ha^{-1} y^{-1}) of critical base cation: aluminum ratio = 10 (to protect against loss of soil base saturation) under modelled 2006 total nitrogen and sulphur deposition (source: Clair et al. 2011).

References

Adamowicz, V. 2007. Section 2: Water use and Alberta oil sands development–science and solutions: An analysis of options. *In* Running out of steam? Oil sands development and water use in the Athabasca River-Watershed. Science and market based solutions. *Edited by* D.J. Davidson and A.M. Hurley. University of Alberta, Edmonton, Alberta. pp. 40-58.

Ahad, J.M.E., Pakdel, H., Savard, M.M., Calderhead, A.I., Gammon, P.R., Rivera, A., Peru, K.M., and Headley, J.V. 2013. Characterization and quantification of mining-related "naphthenic acids" in groundwater near a major oil sands tailings pond. Environ. Sci. Technol. **47**(10): 5023−5030.

Aherne, J., and Shaw, P.D. 2010. Impacts of sulphur and nitrogen deposition in western Canada. J. Limnol. **69** (Suppl. 1): 1−3.

Al, T.A., Leybourne, M.I., Maprani, A.C., MacQuarrie, K.T., Dalziel, J.A., Fox, D., and Yeats, P.A. 2006. Effects of acid-sulfate weathering and cyanide-containing gold tailings on the transport and fate of mercury and other metals in Gossan Creek: Murray Brook mine, New Brunswick, Canada. Appl. Geochem. **21**(11): 1969−1985.

Alberta Chamber of Resources. 2004. Oil sands technology roadmap - unlocking the potential. Edmonton, Alberta Chamber of Resources.

Alberta Environment. 2004. Advisory committee on water use practice and policy: Final Report. Alberta Environment. Pub No.I/979. Edmonton, Alberta.

Alberta Environment. 2006. Alberta water conservation and allocation guideline for oilfield injection. Pub NO. I/969. Edmonton, Alberta.

Alberta Environment. 2007. Water management framework: instream flow needs and water management system for the Lower Athabasca River. Edmonton, Alberta: Alberta Environment. Fisheries and Oceans Canada. Available online at: http://environment.alberta.ca/documents/Athabasca_RWMF_Technical.pdf. [Accessed 4 December 2013]

Alberta Environment. 2008. Alberta air emissions, trends and projections. Prepared by the Air Policy Branch, Edmonton, Alberta. Available online at: http://environment.gov.ab.ca/info/library/7964.pdf. [Accessed 4 December 2013]

Ali, A.A., Ghaleb, B., Garneau, M., Asnong, H., and Loisel, J. 2008. Recent peat accumulation rates in minerotrophic peatlands of the Bay James region, eastern Canada, inferred by ^{210}Pb and ^{137}Cs radiometric techniques. Appl. Rad. Isotop. **66**(10): 1350−1358.

Alila, Y., and Beckers, J. 2001. Using numerical modelling to address hydrologic forest management issues in British Columbia. Hydrol. Processes, **15**(18): 3371−3387.

Allen, E.W. 2008. Process water treatment in Canada's oil sands industry: I. Target pollutants and treatment objectives. J. Environ. Eng. Sci. **7**(2): 123−138.

AMEC Earth & Environmental. 2007. Current and future water use in Alberta. Prepared for Alberta Environment. pp. 724.

Anderson J., Gomez, C.W., McCarney, G., Adamowicz, V., Chalifour, N., Weber, M., Elgie, S., and Howlett, M. 2010. Ecosystem services valuation, market-based instruments, and sustainable forest management: a primer. Sustainable Forest Managment Network State of Knowledge Report.

Andres, D., Van Der Vinne, G., Sterenberg, G. 1987. Hydrologic, hydrogeologic, thermal, sediment, and channel regimes of the Tri-Creeks Experimental Basin. Alberta Research Council.

Andrew, M.E., Wulder, M.A., and Cardille, J.A. 2014. Protected areas in boreal Canada: a baseline and considerations for the continued development of a representative and effective reserve network. Environ. Rev. *In press*.

Andriashek, L.D., and Atkinson, N. 2007. Buried channels and glacial-drift aquifers in the Fort McMurray Region, Northeast Alberta. Alberta Energy and Utilities Borard, EUB/AGS Earth Sciences Report 2007-01. Pp. 169.

Anielski, M., and Wilson, S. 2005. Counting Canada's natural capital: assessing the real value of Canada's boreal ecosystems. The Pembina Institute. Drayton Valley, Alberta.

Anielski, M., and Wilson, S. 2009. Counting Canada's natural capital: assessing the real value of Canada's boreal ecosystems. The Pembina Institute. Drayton Valley, Alberta.

AQUAMIN Working groups 7 & 8. 1996. Assessment of the aquatic effects of mining in Canada: final report. Evaluation and Interpretation Branch, Environment Canada, Ottawa, Ont.

Atkinson, L., Keeping, P., Wright, J., and Liu, H. 2010. The challenges of dewatering at the Victor Diamond Mine in Northern Ontario, Canada. Mine Water Environ. **29**(2): 99-107.

Aust, W.M., Reisinger, T.W., Burger, J.A., and Stokes, B.J. 1993. Soil physical and hydrological changes associated with logging a wet pine flat with wide-tired skidders. South. J. Appl. For. **17**(1): 22−25.

Aust, W.M., Schoenholtz, S.H., Zaebst, T.W., and Szabo, B.A. 1997. Recovery status of a tupelo-cypress wetland seven years after disturbance: silvicultural implications. For. Ecol. Manage. **90**(2-3): 161-169.

Bakker, K. 2007. Eau Canada: the future of Canada's water. UBC Press. Vancouver, British Columbia.

Baldocchi, D., Kelliher, F. M., Black, T. A., and Jarvis, P. 2000. Climate and vegetation controls on boreal zone energy exchange. Global Change Biology, **6**(S1), 69-83.

Band, L., Tague, C., Groffman, P., and Belt, K. 2001. Forest ecosystem processes at the watershed scale: hydrological and ecological controls of nitrogen export. Hydrol. Processes, **15**(10): 2013−2028.

Barker, J.F., Thompson, N.R., Oiffer, A.A.L., Mayer, K.U., and Tomkins, T.G. 2010. The groundwater vector in the reclaimed oil sand mining landscape: learnings from the operational phase. *In* GeoCanada 2010 – Working with the Earth. May 10-14, 2010, Calgary, Alberta. Canadian Society of Petroleum Geologists, Calgary, Alberta. 2 pp.

Barnett, T.P., Adam, J.C., and Lettenmaier, D.P. 2005. Potential impacts of a warming climate on water availability in snow-dominated regions. Nature, **438**(7066): 303−309.

Bates, B., Kundzewicz, Z.W., Wu, S., and Palutikof, J. 2008. Climate change and water. Technical paper VI. The Intergovernmental Panel on Climate Change, IPCC Secretariat, Geneva, 210 pp.

Baxter, R. 1977. Environmental effects of dams and impoundments. Annu. Rev. Ecol. Syst. **8**: 255-283.

Beckers, J.B., Smerdon, B., and Wilson, M. 2009. Review of hydrological models for forest management and climate change applications in British Columbia and Alberta. FORREX Series 25, Forum for Research and Extension in Natural Resources Society, Kamloops BC, 179, pp.

Bedford, B.L. 1999. Cumulative effects on wetland landscapes: links to wetland restoration in the United States and southern Canada. Wetlands, **19**(4): 775-788.

Bednarek, A.T. 2001. Undamming rivers: a review of the ecological impacts of dam removal. Environ. Manage. **27**(6): 803-814.

Beltaos, S., Prowse, T.D., Bonsal, B., MacKay, R., Romolo, L., Pietroniro, A., and Toth, B. 2006. Climatic effects on ice-jam flooding of the Peace-Athabasca Delta. Hydrol. Processes, **20**(19): 4031-4050.

Bergstrom, G.P., Nip, A.M.K. 1988. The Tri Creeks watershed study, (1965-1986) Hinton, Alberta. *In:* Proceedings of the Canadian Hydrology Symposium No.17.

Banff, Alberta, May 9-11, 1988. Canadian Research Basins: successes, failures and future. National Research Council of Canada, Associate Committee on Hydrology. pp.171-178.

Berkes, F. 2002. Cross-scale institutional linkages for commons management: perspectives from the bottom up. *In* The Drama of the Commons. *Edited by* E. Ostrom, T. Dietz, N. Dolšak, P.C. Stern, S. Stonich, and E.U. Weber. National Academy, Washington, D.C., USA. pp. 293-321.

Berkes, F., and C. Folk. 1998. Linking social and ecological systems for resilience and sustainability: management practices and social mechanisms for building resilience. Cambridge University Press, Cambridge, UK.

Berry, G.J., and J.K. Jeglum. 1991. Hydrology of drained and undrained black spruce peatlands: groundwater table profiles and fluctuations. COFRDA Report 3307.

Bertolo, A., and Magnan, P. 2007. Logging-induced variations in dissolved organic carbon affect yellow perch (Perca flavescens) recruitment in Canadian Shield lakes. Can. J. Fish. Aquat. Sci. **64**(2): 181-186.

Beven, K. 2000. On model uncertainty, risk and decision making. Hydrol. Processes, **14**(14): 2605-2606.

Bhavsar, S.P., Diamond, M.L., Evans, L.J., Gandhi, N., Nilsen, J., and Antunes, P. 2004. Development of a coupled metal speciation-fate model for surface aquatic systems. Environ. Toxicol. Chem. **23**(6): 1376-1385.

Biermann, C.J. 1993. Essentials of pulping and papermaking. Academic press, San Diego, CA.

Biermann, C.J. 1996. Handbook of pulping and papermaking. Academic press, San Diego, CA.

Birkham, T.K., Hendry, M.J., Wassenaar, L.I., and Mendoza, C.A. 2007. A transient model of vadose zone reaction rates using oxygen isotopes and carbon dioxide. Vadose Zone J. **6**(1): 67-76.

Blatter J., and Ingram, H.M. 2001. Reflections on water: new approaches to transboundary conflicts and cooperation. Massachusetts Institute of Technology Press, Cambridge, MA.

Blowes, D.W., and Jambor, J.L. 1990. The pore-water geochemistry and the mineralogy of the vadose zone of sulfide tailings, Waite Amulet, Quebec, Canada. Appl. Geochem. **5**(3): 327-346.

Blowes, D.W., Ptacek, C.J., Jambor, J.L., and Weisener, C.G. 2003. The geochemistry of acid mine drainage. *In* Treatise on geochemistry, volume 9. *Edited by* B.S.Lollar. Elsevier Science, Pergamon, Oxford. pp. 149-204.

Bodaly, R., Beaty, K.G., Hendzel, L.H., Majewski, A.R., Paterson, M.J., Rolfhus, K.R., Penn, A.F., Louis, V.L.S., Hall, B.D., and Matthews, C.J. 2004. Peer Reviewed: Experimenting with hydroelectric reservoirs. Environ. Sci. Technol. **38**(18): 346A-352A.

Bosch, J.M., and Hewlett, J. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. J. Hydrol. **55**(1): 3-23.

Bothwell, M.L. 1992. Eutrophication of rivers by nutrients in treated kraft pulp mill effluent. Water Poll. Res. J. Can. **27**(3): 447- 472.

Bouleau, G. 2008. The WFD dreams: between ecology and economics. Water Environ. J. **22**(4): 235-240.

Brandt, J.P. 2009. The extent of the North American boreal zone. Environ. Rev. **17**(1): 101-161.

Brandt, J.P., Flannigan, M.D., Maynard, D.G., Thompson, I.D., and Volney, W.J.A. 2013. An introduction to Canada's boreal zone: ecosystem processes, health, sustainability, and environmental issues. Environ. Rev. **21**(4): 207-226.

Branfireun, B., Krabbenhoft, D., Hintelmann, H., Hunt, R., Hurley, J., and Rudd, J. 2005. Speciation and transport of newly deposited mercury in a boreal forest wetland: a stable mercury isotope approach. Water Resour. Res. **41**(6).

Brauman, K.A., Daily, G.C., Duarte, T.K., and Mooney, H.A. 2007. The nature and value of ecosystem services: an overview highlighting hydrologic services. Annu. Rev. Environ. Resour. **32**: 67-98.

British Columbia Ministry of the Environment (BCMOE). 2008. Handbook for mineral and coal exploration in British Columbia. Ministry of energy, mines and petroleum resources. Victoria, British Columbia.

Brown, R.J.E., 1967. Permafrost in Canada. Geological Survey of Canada Map 1246a and Division of Building Research Map NRC-9769. Natural Resources Council of Canada, Ottawa,ON.

Brown, A.E., Zhang, L., McMahon, T.A., Western, A.W., and Vertessy, R.A. 2005. A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. J. Hydrol. **310**(1): 28-61.

Bruce, J.P., Cunningham, W., Freeze, A., Gillham, R., Gordon, S., Holysh, S., Hrudey, S., Logan, W., MacQuarrie, K., Muldoon, P., Nowlan, L., Pomeroy, J., Renzetti, S., Lollar, B.S., and Therrien, R. 2009. The sustainable management of groundwater in Canada. Report of the Expert Panel on Groundwater. Council of Canadian Academies. pp. 270.

Buttle, J.M. 2002. Rethinking the donut: the case for hydrologically relevant buffer zones. Hydrol. Process. **16**(15): 3093-3096.

Buttle, J.M., and Metcalfe, R.A. 2000. Boreal forest disturbance and streamflow response, northeastern Ontario. Can. J. Fish. Aquat. Sci. **57**(SUPPL. 2): 5-18.

Buttle, J.M., Creed, I.F., and Pomeroy, J.W. 2000. Advances in Canadian forest hydrology, 1995-1998. Hydrol. Processes, **14**(9): 1551-1578.

Buttle, J., Creed, I., and Moore, R. 2005. Advances in Canadian forest hydrology, 1999–2003. Hydrol. Processes, **19**(1): 169-200.

Buttle, J.M., Creed, I.F., and Moore, R.D. 2009. Advances in Canadian forest hydrology, 2003-2007. Can. Water Resour. J. **34**(2): 113-126.

Campbell, J.A. 1980. Oxygen flux measurements in organic soil. Can. J. Soil Sci. **60**(4): 641-650.

Canada West Foundation. 2011. Water pricing: Seizing a public policy dilemma by the horns. Canadian Policy Backgrounder 4. Available online at: http://cwf.ca/pdf-docs/publications/Water_Backgrounder_4_Sept_2011.pdf. [Accessed 2 December 2013]

Canadian Association of Petroleum Producers (CAPP). 2008; Environmental challenges and progress in Canada's oil sands. Calgary, Alberta

Canadian Association of Petroleum Producers (CAPP). 2010. Responsible water management in Canada's oil and gas industry. Available online at: http://www.capp.ca/getdoc.aspx?DocID=173950. [Accessed November 2011]

Canadian Association of Petroleum Producers (CAPP). 2012. Oils Sands Today: Water Use. Available online at: http://oilsandstoday.ca/topics/wateruse/pages/default.aspx. [Accessed 2 December 2012]

Canadian Association of Petroleum Producers (CAPP). 2013. The facts: On natural gas. Updated. Available online at: http://www.capp.ca/getdoc.aspx?DocId=220513&DT=NTV. [Accessed 2 December 2013]

Canadian Council of Ministers of the Environment (CCME). 2011. Progress report on the Canada-wide Acid Rain Strategy for post-2000.Winnipeg, MB. Report # PN-1458. pp. 40.

Canadian Environmental Assessment Agency (CEAA). 2004. Victor diamond project comprehensive study report. Available online at: www.ceaa-acaa.gc.ca/80C30413-docs/report_e.pdf. [Accessed 2 December 2013]

Canadian Environmental Assessment Agency (CEAA). 2011. Detour Lake gold mine comprehensive study report. Public Works and Government Services Canada, Ottawa, Ontario. Report no. EN106-105/2011E.

Canadian Hydropower Association. 2008. Hydropower in Canada past present and future. ISBN 978-0-9810346-0-7. Available online at: https://canadahydro.ca/reportsreference/cha-reports-and-publications. [Accessed 20 December 2013]

Canadian Society for Unconventional Gas (CSUG). undated. Understanding hydraulic fracturing. Available online at: http://www.csur.com/images/CSUG_publications/CSUG_HydraulicFrac_Brochure.pdf. [Accessed 2 December 2013]

Carbone, J., Keller, W., and Griffiths, R. 1998. Effects of changes in acidity on aquatic insects in rocky littoral habitats of lakes near Sudbury, Ontario. Restor. Ecol. **6**(4): 376-389.

Carignan, R., and Steedman, R.J. 2000. Impacts of major watershed perturbations on aquatic ecosystems. Can. J. Fish. Aquat. Sci. **57**(S.2): 1-4.

Carignan, R., D'Arcy, P., and Lamontagne, S. 2000. Comparative impacts of fire and forest harvesting on water quality in Boreal Shield lakes. Can. J. Fish. Aquat. Sci. **57**(S2): 105-117.

Carou, S., Dennis, I., Aherne, J., Ouimet, R., Arp, P.A., Watmough, S.A., DeMerchant, I., Shaw, M., Vet, B., Bouchet, V., and Moran, M. 2008. A national picture of acid deposition critical loads for forest soils in Canada. Prepared for the CCME Acid Rain Task Group. pp. 8.

Cal Data Ltd. (CDL). 2005. Capacity building for a national inventory of orphaned/abandoned mines in Canada. Final Report for the National Orphaned/Abandoned Mines Initiative Advisory Committee. Kelowna, British Columbia. Available online at: www.abandoned-mines.org [Accessed 20 December 2013].

Commissioner of the Environment and Sustainable Development (CESD). 2002. Report to the Commissioner of the Environment and Sustainable Development to the House of Commons. Chapter 3: Abandoned mines in the north. Office of the Auditor General, Ottawa, Canada.

Chambers, P.A., Dale, A.R., Scrimgeour, G.J., and Bothwell, M.L. 2000. Nutrient enrichment of northern rivers in response to pulp mill and municipal discharges. J. Aquat. Ecosyst. Stress Recovery, **8**(1): 53-66.

Chambers, P.A., Guy, M., Roberts, E.S., Charlton, M.N., Kent, R., Gagnon, C., Grove, G., and Foster, N. 2001. Nutrients and their impact on the Canadian environment.

Agriculture and Agri-Food Canada, Environment Canada, Fisheries and Oceans Canada, Health Canada and Natural Resources Canada. pp. 241.

Chambers, P.A., Culp, J.M., Glozier, N.E., Cash, K.J., Wrona, F.J., and Noton, L. 2006. Northern rivers ecosystem initiative: nutrients and dissolved oxygen–issues and impacts. Environ. Monit. Assess. **113**(1-3): 117-141.

Chapin III, F.S., Callaghan, T.V., Bergeron, Y., Fukuda, M., Johnstone, J.F., Juday, G., and Zimov, S.A. 2004. Global Change and the Boreal Forest: Thresholds, Shifting States or Gradual Change. AMBIO. **33**(6):361-365.

Chapin III, F.S., Matson, P.A., and Vitousek, P.M. 2011. Principles of terrestrial ecosystem ecology. 2nd Edition. Springer.

Chenoweth, J., Hadjikakou, M., and Zoumides, C. 2013. Quantifying the human impact on water resources: a critical review of the water footprint concept. Hydrol. Earth Syst. Discuss. **10**(7): 9389-9433.

Clair, T.A., Dennis, I.F., Scruton, D.A., and Gilliss, M. 2007. Freshwater acidification research in Atlantic Canada: a review of results and predictions for the future. Environ. Rev. **15**: 153-167.

Clair, T.A., Dennis, I., Jeffries, D., Aherne, J. Wolniewicz, M. 2011. National Focus Centre Report for Canada. *In* Coordination Centre for Effects (CCE) Status Report 2011 modelling critical threshold and temporal changes of geochemistry and vegetation diversity. *Edited by* M. Posch, J. Slootweg, and J.P. Hettelingh. Report No. 630359003. Available online at: http://www.rivm.nl/media/documenten/cce/Publications/SR2011/CCE_Report-2011_Canada.pdf. [Accessed 4 December 2013]

Clarke, D., and Rieley, J. 2010. Strategy for Responsible Peatland Management. International Peat Society, Finland.

Cleary, J., Roulet, N.T., and Moore, T.R. 2005. Greenhouse gas emissions from Canadian peat extraction, 1990-2000: a life-cycle analysis. AMBIO: A Journal of the Human Environment, **34**(6): 456-461.

Clément, M., St-Hilaire, A., Caissie, D., Chiasson, A., Courtenay, S., and Hardie, P. 2009. An evaluation of mitigation measures to reduce impacts of peat harvesting on the aquatic habitat of the East Branch Portage River, New Brunswick, Canada. Can. Water. Resour. J. **34**(4): 441-452.

Canadian Network of Aquatic Ecosystem Services (CNAES). 2014. Research objectives. Available online at: http://cnaes.ca/research. [Accessed 31 October 2014]

Coffin, A.W. 2007. From roadkill to road ecology: a review of the ecological effects of roads. J. Transp. Geogr. **15**(5): 396-406.

Commission of the European Communities. 2007. Towards sustainable water management in the European Union - First stage in the implementation of the Water Framework Directive 2000/60/EC. Available online at: http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=COM:2007:0128:FIN:EN:PDF. [Accessed 2 December 2013]

Conly, F., Crosley, R., Headley, J., and Quagraine, E. 2007. Assessment of metals in bed and suspended sediments in tributaries of the Lower Athabasca River. J. Environ. Sci. Health, **42**(8): 1021-1028.

Cotton, W.R., and S.Yuter. 2009. Principles of Cloud and Precipiation Formation in Aerosol Pollution Impact on Precipitation. A Scientific Review. Springer, 386 pp.

Council of Canadian Academies, 2014. Environmental Impacts of Shale Gas Extraction in Canada. Ottawa (ON): The Expert Panel on Harnessing Science and Technology to Understand the Environmental Impacts of Shale Gas Extraction, Council of Canadian Academies.

Cowan, W.R., Mackasey, W.O., and Robertson, J.G.A. 2010. The policy framework in Canada for mine closure and management of long-term liabilities: a guidance document. Prepared for the National Orphaned/Abandoned Mines Initiative. Cowan Minerals Ltd., Sudbury, Canada. Available online at: www.abandoned-mines.org/pdfs/PolicyFrameworkCanforMinClosureand MgmtLiabilities.pdf. [Accessed 13 December 2013]

Creed, I.F., and Beall, F.D. 2009. Distributed topographic indicators for predicting nitrogen export from headwater catchments. Water Resour. Res. **45**(10): W10407.

Creed, I.F., and Sass, G.Z. 2011. Digital terrain analysis approaches for tracking hydrological and biogeochemical pathways and processes in forested landscapes. *In* Forest Hydrology and Biogeochemistry. *Edited by* D.F. Levia, Springer Netherlands. pp. 69-100.

Creed I.F., Band, L.E., Foster, N.W., Morrison, I.K., Nicolson, J.A., Semkin, R.S., and Jeffries, D.S. 1996. Regulation of nitrate-N release from temperate forests: A test of the N flushing hypothesis. Water Resour. Res. **32**(11): 3337-3354.

Creed, I.F., Sanford, S., Beall, F., Molot, L., and Dillon, P. 2003. Cryptic wetlands: integrating hidden wetlands in regression models of the export of dissolved organic carbon from forested landscapes. Hydrol. Processes, **17**(18): 3629-3648.

Creed, I.F., Sass, G., Wolniewicz, M., and Devito, K. 2008. Incorporating hydrologic dynamics into buffer strip design on the sub-humid Boreal Plain of Alberta. For. Ecol. Manage. **256**(11): 1984-1994.

Creed, I.F., Sass, G.Z., Beall, F.D., Buttle, J.M., Moore, R.D.,and Donnelly, M. 2011a. Scientific theory, data and techniques for conservation of water resources within a changing forested landscape. A State of Knowledge Report. Sustainable Forest

Management Network. 87 pages. ISBN: 978-1-55261-275-0. Avaialable online at: http://www.sfmn.ales.ualberta.ca/en/Publications/~/media/sfmn/Publications/StateofKnowledgeReports/Documents/SOK2011Water1CreedetalEn.pdf [Accessed 20 December 2013]

Creed, I.F., Sass, G.Z., Beall, F.D., Buttle, J.M., Moore, R.D.,and Donnelly, M. 2011b. Hydrological principles for conservation of water resources within a changing forested landscape. A State of Knowledge Report. Sustainable Forest Management Network. 84 pages. ISBN: 978-1-55261-261-3. Available online at: http://www.sfmn.ales.ualberta.ca/en/Publications/~/media/sfmn/Publications/StateofKnowledgeReports/Documents/SOK2011Water2CreedetalEn.pdf [Accessed 20 December 2013]

Culp, J.M., Cash, K.J., and Wrona, F.J. 2000. Cumulative effects assessment for the Northern River Basins Study. J. Aquat. Ecosyst. Stress and Recov. **8**: 87-94.

Culp, J.M., Cash, K.J., Glozier, N.E., and Brua, R.B. 2003. Effects of pulp mill effluent on benthic assemblages in mesocosms along the Saint John River, Canada. Environ. Toxicol. Chem. **22**(12): 2916-2925.

Cumulative Environnemental Management Association (CEMA). 2012. Backgrounder: The new CEMA. Available online at: http://cemaonline.ca/index.php/administration/cat_view/2-communications/20-cema-news/21-press-releases?start=30). [Accessed 2 December 2013]

Currie, D.V. 1976. Hydrogeology of the Tri-Creek Basin, Alberta. Alberta Research Council. pp.67.

Dai, A. 2011. Drought under global warming: a review. WIREs Clim. Change. **2**(1): 45-65.

Dai, A. 2013. Increasing drought under global warming in observations and models. Nat. Clim. Change. **3**(1): 52-58.

Daigle, J.Y., and Gautreau-Daigle, H. 2001. Canadian peat harvesting and the environment. 2nd ed. North American Wetlands Conservation Council Committee-Environment Canada, Ottawa.

Davé, N.K. 2011. Decommissioning of Elliot Lake uranium mines – site performance and long-term radium mobility. *In* The Uranium Mining Remediation Exchange Group (UMREG) Report, International Atomic Energy Agency, Vienna, Austria. ISBN 978-92-0-169310-5.

de Loë, R.C. 2008. Toward a Canadian national water strategy. Final report. Prepared for Canadian Water Resources Association. Guelph, ON, Rob de Loë Consulting Services.

de Loë, R.C. 2009. A Canadian vision and strategy for water in the 21st century. Policy Options July-August 2009. pp. 21-24.

de Loë, R.C. 2010. Transboundary water governance in the Mackenzie River Basin, Canada. Biennial Rosenberg international forum on water.

de Rosemond, S., Duro, D.C., and Dubé, M. 2009. Comparative analysis of regional water quality in Canada using the Water Quality Index. Environ. Monit. Assess. **156**(1-4): 223-240.

Déry, S.J., and Wood, E. 2005. Decreasing river discharge in northern Canada. Geophys. Res. Lett. **32**(10): L10401.

Desrosiers, M., Planas, D., and Mucci, A. 2006. Short-term responses to watershed logging on biomass mercury and methylmercury accumulation by periphyton in boreal lakes. Can. J. Fish. Aquat. Sci. **63**(8): 1734-1745.

Devito, K., Creed, I.F., Rothwell, R., and Prepas, E. 2000. Landscape controls on phosphorus loading to boreal lakes: implications for the potential impacts of forest harvesting. Can. J. Fish. Aquat. Sci. **57**(10): 1977-1984.

Devito, K., Creed, I.F., Gan, T., Mendoza, C., Petrone, R., Silins, U., and Smerdon, B. 2005. A framework for broad-scale classification of hydrologic response units on the Boreal Plain: Is topography the last thing to consider? Hydrol. Processes, **19**(8): 1705-1714.

Devito, K., Mendoza, C., and Qualizza, C. 2012. Conceptualizing water movement in the Boreal Plains. Implications for watershed reconstruction. Synthesis report prepared for the Canadian Oil Sands Network for research and development.

Department of Fisheries and Oceans (DFO). 2010. Science evaluation of instream flow needs (IFN) for the Lower Athabasca River. Fisheries and Oceans Canada, National Capital Region, Ottawa, ON.

Dillon, P., and Molot, L. 1997. Effect of landscape form on export of dissolved organic carbon, iron, and phosphorus from forested stream catchments. Water Resour. Res. **33**(11): 2591-2600.

Doka, S.E., Mcnicol, D.K., Mallory, M.L., Wong, I., Minns, C.K., and Yan, N.D. 2003. Assessing potential for recovery of biotic richness and indicator species due to changes in acidic deposition and lake pH in five areas of southeastern Canada. Environ. Monit. Assess. **88**(1-3): 53-101.

Donahue, W., Allen, E., and Schindler, D. 2006. Impacts of coal-fired power plants on trace metals and polycyclic aromatic hydrocarbons (PAHs) in lake sediments in central Alberta, Canada. J. Paleolimnol. **35**(1): 111-128.

Douglas, M., Clark, I.D., Raven, K., and Bottomley, D. 2000. Groundwater mixing dynamics at a Canadian Shield mine. J. Hydrol. **235**(1): 88-103.

Douglas, T. 2007. “Green” Hydro Power: Understanding Impacts, Approvals, and Sustainability of Run-of-River Independent Power Projects in British Columbia. Watershed Watch Salmon Society. Available online at: http://www.watershed-watch.org/publications/files/Run-of-River-long.pdf. [Accessed 2 December 2013]

Dubé, M.G. 2003. Cumulative effect assessment in Canada: a regional framework for aquatic ecosystems. Environ. Impact Assess. Rev. **23**(6): 723-745.

Dubé, S., Plamondon, A.P., and Rothwell, R.L. 1995. Watering up after clear-cutting on forested wetlands of the St. Lawrence lowland. Water Resour. Res. **31**(7): 1741-1750.

Dunne, T., and Leopold, L.B. 1978. Water in environmental planning. W.H. Freeman and Company, New York. pp. 818.

Dutton, A.L., Loague, K., and Wemple, B. 2005. Simulated effect of a forest road on near-surface hydrologic response and slope stability.

Dynesius, M. and Nilsson, C. 1994. Fragmentation and flow regulation of river systems in the northern third of the world. Science, **266**(5186): 753-753.

Eastwood, S., and Couture, P. 2002. Seasonal variations in condition and liver metal concentrations of yellow perch (*Perca flavescens*) from a metal-contaminated environment. Aquatic Toxicol. **58**: 43-56.

Egré, D., and Milewski, J.C. 2002. The diversity of hydropower projects. Energy Policy, **30**:1225-1230.

Elliott, J., Toth, B., Granger, R., and Pomeroy, J. 1998. Soil moisture storage in mature and replanted sub-humid boreal forest stands. Can. J. Soil Sci. **78**(1): 17-27.

Energy and Utilities Board, Canadian Environmental Assessment Agency (EUB-CEAA). 2000. Report of the EUB-CEAA Joint Review Panel Cheviot Coal Project Mountain Park Area, Alberta Energy and Utilities Board, Calgary/Canadian Environmental Assessment Agency, Hull. 161 pp. Applications No. 960313 and 960314 Cardinal River Coals Ltd. Available online at: http://www.ceaa.gc.ca/D9BC55EE-docs/report-eng.pdf. [Accessed 2 December 2013]

Environment Canada. 1987. Federal Water Policy. Available online at: http://www.ec.gc.ca/eau-water/default.asp?lang=En&n=E05A7F81-1#federalWP. [Accessed 25 November 2014]

Environment Canada. 2004. Threats to Water Availability in Canada. National Water Research Institute, Burlington, Ontario NWRI Scientific Assessment Report Series No. 3 and ACSD Science Assessment Series No. 1: pp. 128.

Environment Canada. 2005*a*. Canadian acid deposition science assessment and summary of key results. Meteorological Service of Canada, Ottawa, Ont. 440 pp.

Environment Canada. 2005*b*. Improving the effectiveness and efficiency of pulp and paper environmental effects monitoring: a smart regulation opportunity. Available online at:http://www.ec.gc.ca/eem/pdf_publications/english/EEM_Smart_Regulation.pdf. [Accessed 2 December 2013]

Environment Canada. 2008. Air Pollution Emissions. National polutant release inventory (NPRI). 2008. Available online at : http://www.ec.gc.ca/inrp-npri/default.asp?lang=en&n=0EC58C98-. [Accessed 4 December 2013]

Environment Canada - Oil Sands Advisory Panel. 2010. A foundation for the future: Building and environmental monitoring system for the oil sands.

Environment Canada. 2011. An integrated Oil Sands Environment Monitoring Plan. Available online at: http://www.ec.gc.ca/pollution/EACB8951-1ED0-4CBB-A6C9-84EE3467B211/Integrated%20Oil%20Sands_low_e.pdf. [Accessed 2 December 2013]

Environment Canada. 2012a. Fifth National Assessment of Environmental Effects Monitoring Data from Pulp and Paper Mills Subject to the Pulp and Paper Effluent Regulations. En14-84/2013E-PDF. Public Works and Government Services of Canada, Ottawa, Ontario. 27 pp.

Environment Canada. 2012b. Second national assessment of environmental effects monitoring data from metal mines subjected to the Metal Mining Effluent Regulation. Report No. En14-64/2012E. Public Works and Government Services of Canada, Ottawa, Ontario. 31 pp.

Environment Canada. 2014. Greenhouse Gas Sources and Sinks in Canada. National Inventory Report 1990-2012. The Canadian Government's Submission to the UN Framework Convention on Climate Change.

Energy Resources Conservation Board (ERCB). 2009. Directive 074. Tailings performance criteria and requirements for oil sands mining schemes. Available online at: http://www.aer.ca/documents/directives/Directive074.pdf. [Accessed 4 December 2013]

Environment and Sustainable Resource Development (ESRD). 2013. 2010 Reclamation Criteria for Wellsites andAssociated Facilities for Forested Lands (Updated July 2013). Edmonton, Alberta. 81pp. ISBN: 978-0-7785-8984-6. Available online at:

http://www.environment.gov.ab.ca/info/library/8364.pdf. [Accessed 28 December 2013]

Evans, J.E., Prepas, E.E., Devito, K.J., and Kotak, B.G. 2000. Phosphorus dynamics in shallow subsurface waters in an uncut and cut subcatchment of a lake on the boreal plain. Can. J. Fish. Aquat. Sci. **57**(S.2): 60-72.

Faria, D.A., Pomeroy, J.W., and Essery, R.L.H. 2000. Effect of covariance between ablation and snow water equivalent on depletion of snow-covered area in a forest. Hydrol. Processes, **14**(15): 2683-2695.

Ferguson, G.P., Rudolph, D.L., and Barker, J.F. 2009. Hydrodynamics of a large oil sand tailings impoundment and related environmental implications. Can. Geotech. J. **46**(12): 1446-1460.

Ferone, J., and Devito, K. 2004. Shallow groundwater–surface water interactions in pond–peatland complexes along a Boreal Plains topographic gradient. J. Hydrol. **292**(1): 75-95.

Finger, M., Tamiotti, L., and Allouche, J.. 2006. The Multi-Governance of Water: Four Case Studies. State University of New York Press.

Food and Agriculture Organization (FAO). 2013. AQUASTAT. Available online at: http://www.fao.org/nr/water/aquastat/main/index.stm [Accessed 2 December 2013]

Foote, L. 2012. Threshold Considerations and Wetland Reclamation in Alberta's Mineable Oil Sands. Ecol. Soc. **17**(1): 35.

Foote, L., and Krogman, N. 2006. Wetlands in Canada's western boreal forest: Agents of Change. For. Chron. **82**(6): 825-833.

Forman, R.T.T., and Alexander, L.E. 1998. Roads and their major ecological effects. Annu. Rev. Ecol. Syst. **29**: 207-231.

Fortin, P., and G. Collu. 2008. Hydro development in Canada: An update. Available online at: http://www.hydroworld.com/articles/hr/print/volume-27/issue-7/feature-articles/hydro-development-in-canada-an-update.html. [Accessed 2 December 2013]

Forest Products Association of Canada (FPAC). 2011. The next generation of Canada's Forest Products Industry. Available online at: http://www.fpac.ca/publications/fpac-sustainability-2011-en-final.pdf. [Accessed 2 December 2013]

Federal, Provincial, and Territorial Governments of Canada (FPTC). 2010. Canadian biodiversity: ecosystem status and trends 2010. Federal, Provincial, and Territorial Governments of Canada, Canadian Council of Resource Ministers, Ottawa, Ont. Available online at: www.biodivcanada.ca/ecosystems. [Accessed 2 December 2013]

France, R., Culbert, H., and Peters, R. 1996. Decreased carbon and nutrient input to boreal lakes from particulate organic matter following riparian clear-cutting. Environ. Manage. **20**(4): 579-583.

France, R., Steedman, R., Lehmann, R., and Peters, R. 2000. Landscape modification of DOC concentration in Boreal Lakes: Implications for UV-B sensitivity. Water Air Soil Pollut. **122**(1-2): 153-162.

Fulton, R.J. 1995. Geological Survey of Canada, "A" Serie Map 1880A. 1 sheet, doi: 10.4095/205040. Available online at: http:// geoscan.nrcan.gc.ca/starweb/geoscan/servlet.starweb. [Accessed 20 December 2013]G3 Consulting Ltd. 2009. Ecosystem Needs of Water Resources. Prepared for the National Round Table on the Environment and Economy (NRTEE), Ottawa. pp. 141.

Garcia, E., and Carignan, R. 2000. Mercury concentrations in northern pike (Esox lucius) from boreal lakes with logged, burned, or undisturbed catchments. Can. J. Fish. Aquat. Sci. **57**(S2): 129-135.

Garcia, E., Carignan, R., and Lean, D.R. 2007. Seasonal and inter-annual variations in methyl mercury concentrations in zooplankton from boreal lakes impacted by deforestation or natural forest fires. Environ. Monit. Assess. **131**(1-3): 1-11.

Gervais, F.J.M. 2004. Fate and transport of naphthenic acids in glacial aquifers. M.Sc. Thesis, Department of Earth Sciences, University of Waterloo, Waterloo, Ontario. pp. 530.

Ghassemi, F., and White, I. 2007. Inter-basin water transfer: case studies from Australia, United States, Canada, China and India. Cambridge University Press.

Gibson, J.J., Birks, S.J., Moncur, M., Yi, Y., Tattrie, K., Jasechko, S., Richardson, K., and Eby, P. 2011. Isotopic and geochemical tracers for fingerprinting process-affected waters in the oil sands industry: A pilot study. Oil Sands Research and Information Network, University of Alberta, School of Energy and the Environment, Edmonton, Alberta. OSRIN Report No. TR-12.

Gieré, R., and Querol, X. 2010. Solid particulate matter in the atmosphere. Elements, **6**(4): 215-222.

Gillies, C. 2011. Water management techniques for resource roads in wetlands: A state of practice review. FPInnovations Contract Report for Ducks Unlimited Canada CR-652.

Gleeson, J., Zeller, A., and McLaughlin, J.W. 2006. Peat as a fuel source in Ontario: A preliminary literature review. Forest Research Info Paper, Ontario Forest Research Institute, Sault Ste. Marie, Ontario.

Global Water Partnership (GWP), and International Network of Basin Organizations (INBO). 2009. A handbook for integrated water resources management in basins. Available online at: http://www.gwp.org/Global/ToolBox/References/A%20Handbook%20for%20Integrated%20Water%20Resources%20Management%20in%20Basins%20(INBO,%20GWP,%202009)%20ENGLISH.pdf. [Accessed 5 December 2013]

Glooschenko, W., Roulet, N., Barrie, L., Schiff, H., and McAdie, H. 1994. The northern wetlands study (NOWES): An overview. J. Geophys. Res.: Atmos. **99**(D1): 1423-1428.

Golding, D.L., and Swanson, R.H. 1986. Snow distribution patterns in clearings and adjacent forest. Water Resour. Res. **22**(13): 1931-1940.

Gorham, E. 1991. Northern peatlands: role in the carbon cycle and probable responses to climatic warming. Ecol. Appl. **1**(2): 182-195.

Gosselin, P., Hrudey, S.E., Naeth, M.A., Ploudre, A., Therrien, R., Van Der Kraak, G., and Xu, Z. 2010. Environmental and health impacts of Canada's oil sands industry. Royal Society of Canada, Ottawa, ON. pp. 438.

Government of Alberta. 2012a. Oil sands environmental management quick facts. Available online at: http://www.oilsands.alberta.ca/FactSheets/Environmental_Management_Quick_Facts_Oct_2012.pdf. [Accessed September 2013]

Government of Alberta. 2012b. Lower Athabasca Region Groundwater Management Framework. Edmonton, Alberta. ISBN: 978-1-4601-0534-4.

Government of Alberta. 2012c. Lower Athabasca Region Surface Water Quality Management Framework. Edmonton, Alberta. ISBN: 978-1-4601-0530-6.

Government of Alberta. 2013a. Lower Athabasca Region Groundwater Management Framework supporting document for the North Athabasca Oil Sands (NAOS) Area. Edmonton, Alberta. ISBN: 978-1-4601-1118-5.

Government of Alberta 2013b. About Oil Sands – Facts and Statistics. Available online at: http://www.energy.alberta.ca/OilSands/791.asp. [Accessed 2 December 2013]

Government of Canada. 1991. The federal policy on wetland conservation. Canadian Wildlife Service, Environment Canada. Cat. No. CW66-116/1991E. ISBN 0-662-18940-X

Gray, D.K., and Arnott, S.E. 2009. Recovery of acid damaged zooplankton communities: measurement, extent, and limiting factors. Env. Rev. **17**(NA): 81-99.

Griffiths, M., Woynillowicz, D., and Taylor, A. 2006. Troubled waters, troubling trends: Technology and policy options to reduce water use in oil and oil sands development in Alberta. Pembina Institute.

Guillemette, F., Plamondon, A.P., Prévost, M., and Lévesque, D. 2005. Rainfall generated stormflow response to clearcutting a boreal forest: Peak flow comparison with 50 world-wide basin studies. J. Hydrol. **302**(1-4): 137-153.

Gummer, W.D., Cash, K.J., Wrona, F.J., and Prowse, T.D. 2000. The northern river basins study: Context and design. J. Aquat. Ecosyst. Stress Recovery, **8**(1): 7-16.

Gunn, J.M. 1995. Restoration and recovery of an industrial region. Springer-Verlag, New York.

Gunn, J.M., and Noble, B.F. 2011. Conceptual and methodological challenges to integrating SEA and cumulative effects assessment. Environ. Impact Assess. Rev. **31**(2): 154-160.

Gunn, J.M., Keller, W., Negusanti, J., Potvin, R., Beckett, P., and Winterhalder, K. 1995. Ecosystem recovery after emission reductions: Sudbury, Canada. Water, Air, Soil Pollut. **85**(3): 1783-1788.

Hall, B.D., St. Louis, V.L., Rolfhus, K.R., Bodaly, R.A., Beaty, K.G., Paterson, M.J., and Cherewyk, K.A.P. 2005. Impacts of reservoir creation on the biogeochemical cycling of methyl mercury and total mercury in boreal upland forests. Ecosystems, **8**(3): 248-266.

Hart, D.D., Johnson, T.E., Bushaw-Newton, K.L., Horwitz, R.J., Bednarek, A.T., Charles, D.F., Kreeger, D.A., and Velinsky, D.J. 2002. Dam removal: challenges and opportunities for ecological research and river restoration: we develop a risk assessment framework for understanding how potential responses to dam removal vary with dam and watershed characteristics, which can lead to more effective use of this restoration method. Bioscience, **52**(8): 669-682.

Hausmann, S., and Pienitz, R. 2009. Seasonal water chemistry and diatom changes in six boreal lakes of the Laurentian Mountains (Québec, Canada): impacts of climate and timber harvesting. Hydrobiologia, **635**(1): 1-14.

Hazewinkel, R.R.O., Wolfe, A.P., Pla, S., Curtis, C., and Hadley, K. 2008. Have atmospheric emissions from the Athabasca Oil Sands impacted lakes in northeastern Alberta, Canada?. Can. J. Fish. Aquat. Sci. **65**(8): 1554-1567.

Hazlett, P.W., Gordon, A.M., Sibley, P.K., and Buttle, J.M. 2005. Stand stocks and soil carbon and nitrogen storage for riparian and upland forests of boreal lakes in northeastern Ontario. For. Ecol. Manage. **219**: 56-68.

Hazlett, P.W., Gordon, A.M., Voroney, P., and Sibley, P.K. 2006. Nitrogen mineralization and immobilization from soils in upland boreal forest cutovers. Soil Biol. Biochem. **64**: 43-57.

Hazlett, P.W., Gordon, A.M., Voroney, R.P., and Sibley, P.K. 2007. Impact of harvesting and logging slash on nitrogen and carbon dynamics in soils from upland spruce forests in northeastern Ontario. Soil Biol. Biochem. **39**(1): 43-57.

Headley, J.V., and McMartin, D.W. 2004. A review of the occurrence and fate of naphthenic acids in aquatic environments. J. Environ. Sci. Health, Part A: Toxic/Hazard. Subst. Environ. Eng. **39**(8): 1989-2010.

Headley, J., Crosley, B., Conly, F., and Quagraine, E. 2005. The characterization and distribution of inorganic chemicals in tributary waters of the lower Athabasca river, Oilsands region, Canada. J. Environ. Sci. Health, Part A: Toxic/Hazard. Subst. Environ. Eng. **40**(1): 1-27.

Heal, G. 2000. Nature and the marketplace: capturing the value of ecosystem services. Island Press.

HEC Montréal. 2011. Job Creation and Economic Development Opportunities in the Canadian Hydropower Market.

Hecky, R., Bodaly, R., Ramsey, D., Ramlal, P., and Strange, N. 1987. Evolution of limnological conditions, microbial methylation of mercury and mercury concentrations in fish in reservoirs of Northern Manitoba. Technical Appendices to the Summary Report of the Canada-Manitoba Agreement on the Study and Monitoring of Mercury in the Churchill River Diversion **3**: 1-7.

Hewitt, L.M., Parrott, J.L., and McMaster, M.E. 2006. A decade of research on the environmental impacts of pulp and paper mill effluents in Canada: Sources and characteristics of bioactive substances. J. Toxicol. Environ. Health, Part B. **9**(4):341-356.

Heyes, A., Moore, T.R., Rudd, J.W.M., and Dugoua, J.J. 2000. Methyl mercury in pristine and impounded boreal peatlands, Experimental Lakes Area, Ontario. Can. J. Fish. Aquat. Sci. **57**(11): 2211-2222.

Hillman, G.R., Feng, J.C., and Feng, C.C. 1997. Effects of watershed characteristics and disturbances on the water quality of two boreal streams. Canadian Forest Service and the Alberta Environmental Protection, Land and Forest Service Project No. 1116-150. pp. 1-75.

Hipel, K.W., Miall, A.D., and Smith, D.W. 2011. Water resources in Canada: A strategic viewpoint. Report to the Inter American National Academies of Science (IANAS). pp. 21-53.

Hoekstra, A.Y., and Mekonnen, M.M. 2012. The water footprint of humanity. Proc. Natl. Acad. Sci. **109**(9): 3232-3237.

Hogg, E.H. 1994. Climate and the southern limit of the western Canadian boreal forest. Can. J. For. Res. **24**(9): 1835-1845.

Holden, A., Haque, S., Mayer, K., and Ulrich, A. 2013. Biogeochemical processes controlling the mobility of major ions and trace metals in aquitard sediments beneath an oil sands tailings pond: Laboratory studies and reactive transport modeling. J. Contam. Hydrol. **151**: 55-67.

Holmes, W.E., and Zak, D.R.. 1999. Soil microbial control of nitrogen loss following clear-cut harvest in northern hardwood ecosystems. Ecol. App. **9**(1): 202-215.

Hoover, G., Howatson, A., Churchill, J., and Roberts, J. 2007. Navigating the Shoals: Assessing Water Governance and Management in Canada. The Conference Board of Canada, Ottawa, Ontario.

Hornberger, G., Bencala, K., and McKnight, D. 1994. Hydrological controls on dissolved organic carbon during snowmelt in the Snake River near Montezuma, Colorado. Biogeochemistry, **25**(3): 147-165.

Hrudey, S. E., Naeth, M. A., Therrien, R., Van Der Kraak, G., Gosselin, P., Plourde, A., and Xu, Z. 2012. Response to Timoney critique of Royal Society of Canada expert panel on oil sands. Environ. Sci. Technol. **46**(8): 4257-4258.

Hruska, K.A., and Dubé, M.G. 2004. Using artificial streams to assess the effects of metal-mining effluent on the life cycle of the freshwater midge (*Chironomus tentans*) in situ. Environ. Toxicol. Chem. **23**(11): 2709-2718.

Humphries, M. 2008. North American oil sands: history of development, prospects for the future. Library of Congress, Washington DC, Congressional Research Service.

Hunter, M.L. 1993. Natural fire regimes as spatial models for managing boreal forests. Biol. Conserv. **65**(2): 115-120.

Hutchinson, D.G., and Moore, R. 2000. Throughflow variability on a forested hillslope underlain by compacted glacial till. Hydrol. Processes, **14**(10): 1751-1766.

Hyvönen, R., Ågren, G.I., Linder, S., Persson, T., Cotrufo, M.F., Ekblad, A., Freeman, M., Grelle, A., Janssens, I.A., Jarvis, P.G., Kellomäki, S., Lindroth, A., Loustau, D., Lundmark, T., Norby, R.J., Oren, R., Pilegaard, K., Ryan, M.G., Sigurdsson, B.D., Strömgren, M., van Oijen, M., and Wallin, G. 2007. The likely impact of elevated [CO2], nitrogen deposition, increased temperature and management on carbon sequestration in temperate and boreal forest ecosystems: a literature review. New Phytol. **173**(3): 463-480.

Ice, G.G., Schilling, E., and Vowell, J. 2010. Trends for forestry best management practices implementation. J. For. **108**(6): 267-273.

International Hydropower Association. 2007. World’s largest hydroelectric plants (over 4,000 MW capacity. Available online at: http://www.infoplease.com/ipa/A0001336.html . [Acessed 20 December 2013]

International Joint Comission (IJC). 1991. Canada-U.S. Air Quality Agreement Available online at: http://www.ijc.org/rel/agree/air.html. [Accessed 3 December 2013]

ISO. 2012. Environmental management - Water footprint - Principles, requirements and guidelines. ICS 13.020.10; 13.020.60. pp. 34.

Janfada, A., Headley, J. V., Peru, K. M., and Barbour, S. L. 2006. A laboratory evaluation of the sorption of oil sands naphthenic acids on organic rich soils. J. Environ. Sci. Health, Part A: Toxic/Hazard. Subst. Environ. Eng. **41**(6): 985-997.

Jeffries, D.S., Clair, T.A., Couture, S., Dillon, P.J., Dupont, J., Keller, W., McNicol, D.K., Turner, M.A., Vet, R., and Weeber, R. 2003*a*. Assessing the recovery of lakes in southeastern Canada from the effects of acidic deposition. AMBIO: A Journal of the Human Environment, **32**(3): 176-182.

Jeffries, D.S., Brydges, T.G., Dillon, P.J., and Keller, W. 2003*b*. Monitoring the results of Canada/USA acid rain control programs: some lake responses. Environ. Monit. Assess. **88**(1-3): 3-19.

Jeziorski, A., Keller, B., Patterson, A.M., Greenaway, C.M., and Smol, J.P. 2013. Aquatic ecosystem responses to rapid recovery from extreme acidification and metal contamination in lakes near Wawa, Ontario. Ecosystems **16**: 209-223.

Johnson, D.B., and Hallberg, K.B. 2005. Acid mine drainage remediation options: A review. Sci. Tot. Environ. **338**(1): 3-14.

Johnson, S.L., and Jones, J.A. 2000. Stream temperature responses to forest harvest and debris flows in western Cascades, Oregon. Can. J. Fish. Aquat. Sci. **57**(S2): 30-39.

Jones, J., and Grant, G.E. 1996. Peak flow responses to clear-cutting and roads in small and large basins, western Cascades, Oregon. Water Resour. Res. **32**(4): 959-974.

Jones, J.P., and Mendoza, C. 2013. Groundwater Modelling Guidlines for the Alberta Oil Sands: Overview and Discussion. Alberta Innovates Technology Futures. Presentation to Water Tech 2013 Conference Banff, Alberta, April 10-12.

Jones, J.A., Swanson, F.J., Wemple, B.C., and Snyder, K.U. 2000. Effects of roads on hydrology, geomorphology, and disturbance patches in stream networks. Cons. Biol. **14**(1): 76-85.

Jutras, S., Begin, J., and Plamondon, A.P. 2002. Impact du drainage forestier après coupe sur la croissance de l'épinette noire en forêt boréale. Can. J. For. Res. **32**(9): 1585-1596.

Kachhwal, L.K., Yanful, E.K., and Lanteigne, L. 2011. Water cover technology for reactive tailings management: a case study of field measurement and model predictions. Wat. Air Soil Pollut. **214**(1-4): 357-382.

Kalin, M., and Wheeler, W.N. 2011. Ecological perspectives in restoring mine waste management areas. Procedia Environ. Sci. **9**: 90-95.

Kalin, M., Wheeler, W.N., and Olaveson, M.M. 2006. Response of phytoplankton to ecological engineering remediation of a Canadian Shield Lake affected by acid mine drainage. Ecol. Eng. **28**: 296-310.

Kasperski, K.L. and Mikula, R.J. 2011.Waste streams of mined oil sands: Characteristics and remediation. Elements **7**(6): 387-392.

Keeler, B. L., Polasky, S., Brauman, K. A., Johnson, K. A., Finlay, J. C., O'Neill, A., Kovacs, K., and B. Dalzell. 2012. Linking water quality and well-being for improved assessment and valuation of ecosystem services. Proc. Natl. Acad. Sci. **109**(45): 18619-18624.

Keller, W. 2009. Limnology in northeastern Ontario: from acidification to multiple stressors. Can. J. Fish. Aquat. Sci. **66**(7): 1189–1198.

Keller, W., and Pitblado, J.R. 1986. Water quality changes in Sudbury area lakes: a comparison of synoptic surveys in 1974–1976 and 1981–1983. Water, Air, Soil Pollut. **29**(3): 285–296.

Keller, W., and Yan, N.D. 1991. Recovery of crustacean zooplankton species richness in Sudbury lakes following water quality improvements. Can. J. Fish. Aquat. Sci. **48**(9): 1635-1644.

Keller, W., Gunn, J.M., and Yan, N.D. 1992. Evidence of biological recovery in acid-stressed lakes near Sudbury, Canada. Environ. Pollut. **78**(1): 79-85.

Keller, W., Heneberry, J.H., and Gunn, J.M. 1998. Effects of emission reductions from the Sudbury smelters on the recovery of acid- and metal-damaged lakes. J. Aquat. Ecosyst. Stress Recovery, **6**(3): 189–198.

Keller, W.B., Heneberry, J., Gunn, J.M., Snucins, E., Morgan, G., and Leduc, J. 2004. Recovery of acid and metal-damaged lakes near Sudbury, Ontario: trends and status. Cooperative Freshwater Ecology Unit, Laurentian University, Sudbury, Ont.

Keller, W., Yan, N.D., Gunn, J.M., and Heneberry, J. 2007. Recovery of acidified lakes: lessons from Sudbury, Ontario, Canada. Water, Air, Soil Pollut: Focus, **7**(1): 317–322.

Kelly, C.A., Rudd, J.W.M., Bodaly, R.A. Roulet, N.P., St. Louis, V.L., Heyes, A., Moore, T.R., Schiff, S., Aravena, R., Scott, K.J., Dyck, B., Harris, R., Warner, B., and Edwards, G.1997. Increases in fluxes of greenhouse gases and methyl mercury following flooding of an experimental reservoir. Environ. Sci. Technol. **31**: 1334-1344.

Kelly, E.N., Short, J.W., Schindler, D.W., Hodson, P.V., Ma, M., Kwan, A.K., and Fortin, B.L. 2009. Oil sands development contributes polycyclic aromatic compounds to the Athabasca River and its tributaries. Proc. Natl. Acad. Sci. U. S. A. **106**(52): 22346-22351.

Kelly, E.N., Schindler, D.W., Hodson, P.V., Short, J.W., Radmanovich, R., and Nielsen, C.C. 2010. Oil sands development contributes elements toxic at low concentrations to the Athabasca River and its tributaries. Proc. Natl. Acad. Sci. U. S. A. **107**(37): 16178-16183.

Kidd, K.A., Paterson, M.J., Hesslein, R.H., Muir, D.C., and Hecky, R.E. 1999. Effects of northern pike (*Esox lucius*) additions on pollutant accumulation and food web structure, as determined by δ13 C and δ15 N, in a eutrophic and an oligotrophic lake. Can. J. Fish. Aquat. Sci. **56**(11): 2193-2202.

Kim, Y. J., Hipel, K. W., and Bowman, C. W. 2013. Water security problems in Canada's oil sands. Can. Water Resour. J. **38**(1): 61-72.

King, K.S., and J. Yetter. 2011. Groundwater and Alberta's Oil Sands Mines. Groundwater, **49**(3):316-318.

Kinzig, A.P., Ryan, P., Etienne, M., Allison, H., Elmqvist, T., and Walker, B.H. 2006. Resilience and regime shifts: assessing cascading effects. Ecol. Soc. **11**(1).

Knapp, C.W., Graham, D.W., Steedman, R.J., and DeNoyelles Jr., F. 2003. Deep chlorophyll maxima in small boreal forest lakes after experimental catchment and shoreline logging. Boreal Environ. Res. **8**(1): 9-18.

Kondolf, G.M. 1997. PROFILE: Hungry water: Effects of dams and gravel mining on river channels. Environ. Manage. **21**(4): 533-551.

Kozlowski, T.T. 1984. Flooding and plant growth. Academic Press, Inc. (London).

Kratochvil, D. 2010. Huge mine produces clean water too. Can. Min. J. **131**(2): 20-21.

Kreutzweiser, D.P., Hazlett, P.W., and Gunn, J.M. 2008. Logging impacts on the biogeochemistry of boreal forest soils and nutrient export to aquatic systems: A review. Env. Rev. **16**(NA): 157-179.

Kreutzweiser, D., Capell, S., Good, K., and Holmes, S. 2009*a*. Sediment deposition in streams adjacent to upland clearcuts and partially harvested riparian buffers in boreal forest catchments. For. Ecol. Manage. **258**(7): 1578-1585.

Kreutzweiser, D.P., Capell, S.S., and Holmes, S.B. 2009*b*. Stream temperature responses to partial-harvest logging in riparian buffers of boreal mixedwood forest watersheds. Can. J. For. Res. **39**(3): 497-506.

Kreutzweiser, D.P., Sibley, P.K., Richardson, J.S., and Gordon, A.M. 2012. Introduction and a theoretical basis for using disturbance by forest management activities to sustain aquatic ecosystems. Freshwater Sci. **31**(1): 224-231.

Kreutzweiser, D., Beall, F., Webster, K., and Creed, I. 2013. Impacts and prognosis of natural resource development on aquatic biodiversity in Canada's boreal zone. Environ. Rev. **21**(4): 227-259.

Krezek, C.C., Buttle, J.M., Beall, F.D., Moore, R.D., Creed, I.F., Sibley, P.K., Silins, U., Devito, K.J., and Mendoza, C.A. 2008. HydroEcological Landscapes and Processes Project: A National-scale Forest Hydrology Initiative. Streamline Watershed Management Bulletin **12**(1): 33-38.

Kumar, A., Schei, T. Ahenkorah, A., Caceres Rodriguez,R. Devernay, J.-M. Freitas, M.. Hall, D. Killingtveit, Å. Liu, Z. 2011: Hydropower. In IPCC Special Report on Renewable Energy Sources and Climate Change Mitigation [O. Edenhofer, R. Pichs-Madruga, Y. Sokona, K. Seyboth, P. Matschoss, S. Kadner, T. Zwickel, P. Eickemeier, G. Hansen, S. Schlömer, C. von Stechow (eds)], Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Kurz, W.A., Shaw, C.H., Boisvenue, C., Stinson, G., Metsaranta, J., Leckie, D., Dyk, A., Smyth, C., and Neilson, E.T. 2013. Carbon in Canada's boreal forest — A synthesis. Environ. Rev., **21**(4): 260-292.

Kwong, Y.T.J., Beauchemin, S., Hossain, M.F., and Gould, W.D. 2007. Transformation and mobilization of arsenic in the historic Cobalt mining camp, Ontario, Canada. J. Geochem. Explor. **92**(2): 133-150.

La Marche, J.L., and Lettenmaier, D.P. 2001. Effects of forest roads on flood flows in the Deschutes River, Washington. Earth Surf. Processes Landforms. **26**(2): 115-134.

Lagacé, E. 2011. Shared water, one framework : What Canada can learn from EU water governance. Forum for Leadership on Water. pp. 7.

Laine, J., Vasander, H., and Sallantaus, T. 1995. Ecological effects of peatland drainage for forestry. Environ. Rev. **3**(3-4): 286-303.

Lamontagne, S., Carignan, R., D'Arcy, P., Prairie, Y.T., and Pare, D. 2000. Element export in runoff from eastern Canadian Boreal Shield drainage basins following forest harvesting and wildfires. Can. J. Fish. Aquat. Sci. **57**(S2): 118-128.

Lasserre, F. 2007. Drawers of water: water diversions in Canada and beyond. Eau Canada: The Future of Canada's Water. pp. 143-162.

Laudon, H., Hedtjärn, J., Schelker, J., Bishop, K., Sørensen, R., and Ågren, A. 2009. Response of dissolved organic carbon following forest harvesting in a boreal forest. AMBIO: A Journal of the Human Environment, **38**(7): 381-386.

Launiainen, S., Futter, M.N., Ellison, D., Clarke, N., Finér, L., Högbom, L., Laurén, A., and Ring, E. 2013. Is the Water Footprint an Appropriate Tool for Forestry and Forest Products: The Fennoscandian Case. AMBIO: 1-13.

Lavoie, M., Paré, D., Fenton, N., Groot, A., and Taylor, K. 2005. Paludification and management of forested peatlands in Canada: A literature review. Env. Rev. **13**(2): 21-50.

LaZerte, B. 1986. Metals and acidification: An overview. Water Air Soil Pollut. **31**(3-4): 569-576.

Leblanc, M., Morales, J.A., Borrego, J., and Elbaz-Poulichet, F. 2000. 4,500-year-old mining pollution in southwestern Spain: long-term implications for modern mining pollution. Econ. Geol. **95**(3): 655-662.

Lewis, M.W. 2012. The controversial global boom in dam-building and hydroelectric power. Geocurrents.info. Available online at: http://www.geocurrents.info/environmental-geography/the-controversial-global-boom-in-dam-building-and-hydroelectric-power. [Accessed 20 December 2013]

Lewis, J., Mori, S.R., Keppeler, E.T., and Ziemer, R.R. 2001. Impacts of logging on storm peak flows, flow volumes and suspended sediment loads in Caspar Creek, California. *In* Land use and watersheds: human influence on hydrology and geomorphology in urban and forest areas. *Edited by* M.S. Wigmosta and S.J. Burges. American Geophysical Union, Washington, DC. pp. 85-125.

Leybourne, M.I., Goodfellow, W.D., Boyle, D.R., and Hall, G.E.M. 2000. Form and distribution of gold mobilized into surface waters and sediments from a gossan tailings pile, Murray Brook massive sulphide deposit, New Brunswick, Canada. Appl. Geochem. **15**(5): 629-646.

Lieffers, V. 1988. Sphagnum and cellulose decomposition in drained and natural areas of an Alberta peatland. Can. J. Soil Sci. **68**(4): 755-761.

Lieffers, V., and Rothwell, R. 1986. Effects of depth of water table and substrate temperature on root and top growth of *Picea mariana* and *Larix laricina* seedlings. Can. J. For. Res. **16**(6): 1201-1206.

Lieffers, V., and Rothwell, R. 1987. Rooting of peatland black spruce and tamarack in relation to depth of water table. Can. J. Bot. **65**(5): 817-821.

Liu, S., Costanza, R., Farber, S., and Troy, A. 2010. Valuing ecosystem services: Theory, practice, and the need for a transdisicplinary synthesis. Annals of the New York Academy of Sciences **1185**(2010): 54-78.

Locky, D.A., and Bayley, S.E. 2007. Effects of logging in the southern boreal peatlands of Manitoba, Canada. Can. J. For. Res. **37**(3): 649-661.

Long, J.N. 2009. Emulating natural disturbance regimes as a basis for forest management: A North American view. For. Ecol. Manage. **257**(9): 1868-1873.

Luek, A., Morgan, G.E., Wissel, B., Gunn, J.M., and Ramcharan, C.W. 2013. Impaired littoral energy pathways cause a shift to pelagic resources by fish in recovering lake food webs. Ecol. Freshwat. Fish **22**: 348-360.

Lugo, A.E., and Gucinski, H. 2000. Function, effects, and management of forest roads. For. Ecol. Manage. **133**(3): 249-262.

Lyness, L.S., and Fennell, J.W. 2010. Groundwater in harmony with oil sands development: A blueprint. *In* GeoCanada 2010 – Working with the Earth, May 10-14, 2010, Calgary, AB. Canadian Society of Petroleum Geologists, Calgary, AB.

The Mining Association of Canada (MAC). 2010. Towards sustainable mining progress report 2010. Available online at: http://extras.newswire.ca/smr/PDF/MAC_TSM_Progress_Report.pdf. [Accessed 3 December 2013]

MacCausland, A., and McTammany, M.E. 2007. The impact of episodic coal mine drainage pollution on benthic macroinvertebrates in streams in the Anthracite region of Pennsylvania. Environ. Pollut. **149**" 216-226.

MacDonald, J.S., Beaudry, P.G., MacIsaac, E.A., and Herunter, H.E. 2003*a*. The effects of forest harvesting and best management practices on streamflow and suspended sediment concentrations during snowmelt in headwater streams in sub-boreal forests of British Columbia, Canada. Can. J. For. Res. **33**(8): 1397-1407.

MacDonald, J.S., MacIsaac, E.A. and Herunter, H.E. 2003*b*. The effect of variable-retention riparian buffer zones on water temperatures in small headwater streams in sub-boreal forest ecosystems of British Columbia. Can. J. For. Res. **33**(8): 1371-1382.

Madden, N., Lewis, A., and Davis, M. 2013. Thermal effluent from the power sector: an analysis of once-through cooling system impacts on surface water temperature. Environ. Res. Lett. **8**(3): 035006.

Magilligan, F.J., and Nislow, K.H. 2005. Changes in hydrologic regime by dams. Geomorphology, **71**(1): 61-78.

Mailman, M., and Bodaly, R. 2005. Total mercury, methyl mercury, and carbon in fresh and burned plants and soil in Northwestern Ontario. Environ. Pollut. **138**(1): 161-166.

Mailman, M., and Bodaly, R. 2006. The burning question: Does burning before flooding lower methyl mercury production and bioaccumulation? Sci. Total Environ. **368**(1): 407-417.

Mailman, M., Stepnuk, L., Cicek, N., and Bodaly, R. 2006. Strategies to lower methyl mercury concentrations in hydroelectric reservoirs and lakes: A review. Sci. Total Environ. **368**(1): 224-235.

Main, C. 2011. Regional aquatics monitoring program (RAMP) scientific review. Alberta Innovates-Technology Futures, Calgary, AB.

Mallik, A.U., and Teichert, S. 2009. Effects of forest management on water resources in Canada: A research review. NCASI Tech. Bull. **969**: 1-84.

Manitoba Environment. 1993. State of Environment Report.

Mannix, A. E., Dridi C., and Adamowicz, W.L. 2010. Water availability in the oil sands under projections of increasing demands and a changing climate: An assessment of the Lower Athabasca Water Management Framework (Phase 1). Can. Water Res. J. **35**(1): 29-52.

Maprani, A.C., Al, T.A., MacQuarrie, K.T., Dalziel, J.A., Shaw, S.A., and Yeats, P.A. 2005. Determination of mercury evasion in a contaminated headwater stream. Environ. Sci. Technol. **39**(6): 1679-1687.

Marcotte, P., Roy, V., Plamondon, A.P., and Auger, I. 2008. Ten-year water table recovery after clearcutting and draining boreal forested wetlands of eastern Canada. Hydrol. Processes, **22**(20): 4163-4172.

Martin, A.J., and Pedersen, T.F. 2002. Seasonal and interannual mobility of arsenic in a lake impacted by metal mining. Environ. Sci. Technol. **36**: 1516-1523.

Martz, L.W. 1988. Sub-Watershed contributions to the suspended sediment yield of Spring Creek Watershed, Alberta. Can. Water Resour. J. **13**:74-84.

Matilainen, T., Verta, M., Niemi, M., and Uusi-Rauva, A. 1991. Specific rates of net methylmercury production in lake sediments. Water, Air, Soil Pollut. **56**(1): 595-605.

Maynard, D.G., Paré, D., Thiffault, E., Lafleur, B., Hogg, K.E., and Kishchuk, B. How do natural disturbances and human activities affect soil and tree nutrition and growth in the Canadian boreal forest? Environ. Rev. *In Press*.

McCracken, A.D., Macey, E., Monro Gray, J.M., and Nowlan, G.S. 2007. Placer gold. Popular Geoscience, Natural Resources Canada, Ottawa, Ontario.

McDonnell, J.J., and Woods, R. 2004. On the need for catchment classification. J. Hydrol. (Amsterdam, Neth.) **299**(1): 2-3.

McGuire, K., McDonnell, J., Weiler, M., Kendall, C., McGlynn, B., Welker, J., and Seibert, J. 2005. The role of topography on catchment-scale water residence time. Water Resour. Res. **41**(5).

McLaughlin, D. 2009. Water and the future of Canada’s natural resource sectors. Policy Options. pp.70.

McLaughlin, J.W., and Webster, K.L. 2013. Effects of a changing climate on peatlands in permafrost zones: a literature review and application to Ontario's far north. Climate Change Research Report, Ontario Forest Research Institute, (CCRR-34).

McMaster, M.E., Evans, M., Alaee, M., Muir, D., and Hewitt, L. 2006*a*. Northern rivers ecosystem initiative: Distribution and effects of contaminants. Environ. Monit. Assess. **113**(1-3): 143-165.

McMaster, M.E., Hewitt, L.M., and Parrott, J.L. 2006*b*. A decade of research on the environmental impacts of pulp and paper mill effluents in Canada: Field studies and mechanistic research. J. Toxicol. Environ. Health, Part B: **9**(4): 319-339.

Mengistu, S.G., Creed, I.F., Webster, K.L., Enanga, E., and Beall, F.D. 2013. Searching for similarity in topographic controls on carbon, nitrogen and phosphorus export from forested headwater catchments. Hydrol. Processes, DOI: 10.1002/hyp.9862.

Metcalfe, R., and Buttle, J. 1999. Semi-distributed water balance dynamics in a small boreal forest basin. J. Hydrol. **226**(1): 66-87.

Metcalfe, R., and Buttle, J. 2001. Soil partitioning and surface store controls on spring runoff from a boreal forest peatland basin in north-central Manitoba, Canada. Hydrol. Processes, **15**: 2305-2324.

Mikula, R. 2012. Advances in oil sands tailings handling: building the base for reclamation. *In* restoration and reclamation of boreal ecosystems: attaining sustainable development. *Edited by* D. Vitt and J. Bhatti. Cambridge University Press. pp. 412.

Millenium Ecosystem Assessment. 2005. Ecosystems and Human Well-Being: Synthesis. Island Press, Washington, DC. pp. 155.

Mitsch, W. J., and Gosselink, J.G. 2007. Wetlands. John Wiley and Sons, Inc. Hoboken, NJ.

Molnar, J.L., and Kubiszewski, I. 2012. Managing natural wealth: Research and implementation of ecosystem services in the United States and Canada. Ecosystem Services **2**(2012): 45-55.

Moncur, M.C., Ptacek, C.J., Blowes, D.W., and Jambor, J.L. 2005. Release, transport and attenuation of metals from an old tailings impoundment. Appl. Geochem. **20**(3): 639-659.

Moncur, M.C., Ptacek, C.J., Blowes, D.W., and Jambor, J.L. 2006. Spatial variation in water composition at a northern Canadian lake impacted by mine drainage. Appl. Geochem. **21**(10): 1799-1817.

Moore, R. D., and J.S. Richardson. 2012. Natural disturbance and forest management in riparian zones: comparison of effects at reach, catchment, and landscape scales. Freshwater Sci. **31**(1): 239-247.

Moore, R.D., and Wondzell, S.M. 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: A review. J. Am. Water Resour. Assoc. **41**(4): 763-784.

Morris, D.M. 1997. The role of long-term site productivity in maintaining healthy ecosystems: a prerequisite of ecosystem management. For. Chron. **73**(6): 731-740.

Morris, T.J., Boyd, D.R., Brandes, O.M., Bruce, J.P., Hudon, M., Lucas, B., Maas, T., Nowland, L., Pentland, R., and Phare, M. 2007. Changing the flow: A blueprint for federal action on freshwater. The Gordon Water Group of Concerned Scientists and Citizens. pp. 55.

Muldoon, P., and McClenaghan, T. 2007. A tangled web: Reworking Canada's water laws. Eau Canada: The Future of Canada's Water. pp. 245-261.

Muscatello, J.R., Belknap, A.M., and Janz, D.M. 2008. Accumulation of selenium in aquatic systems downstream of a uranium mining operation in northern Saskatchewan, Canada. Environ. Pollut. **156**(2): 387-393.

Muscatello, J.R., and Janz, D.M. 2009. Selenium accumulation in aquatic biota downstream of a uranium mining and milling operation. Sci. Total Environ. **407**(4): 1318-1325.

National Energy Board. 2009. Understanding Canadian Shale Gas. Energy Brief. Available online at: http://www.neb-one.gc.ca/clf-nsi/rnrgynfmtn/nrgyrprt/ntrlgs/prmrndrstndngshlgs2009/prmrndrstndngshlgs2009nrgbrf-eng.pdf. [Accessed 3 December 2013]

National Research Council. 2008. Hydrologic effects of a changing forest landscape. National Academies Press, Washington, D.C. pp. 168.

National Wetlands Working Group. 1997. The Canadian wetland classification system. Wetlands Research Centre, Waterloo, ON. pp. 76.

Nanos, N. 2009. Canadians overwhelmingly choose water as our most important natural resource. Policy Options. Special Issue. Available online at: www.nanosresearch.com/library/polls/POLNAT-S09-T382E.pdf. [Accessed 13 December 2013]

National Council for Air and Stream Improvement (NCASI). 2010. Water Profile of the Canadian Forest Products Industry Technical Bulletin No. 0975.

Naylor, B. J., Mackereth, R.W., Kreutzweiser, D.P., and Sibley, P.K. 2012. Merging END concepts with protection of fish habitat and water quality in new direction for riparian forests in Ontario: a case study of science guiding policy and practice. Freshwater Sci. **31**(1): 248-257.

Newbury, R.W., McCullough, G.K., and Hecky, R.E. 1984. The Southern Indian Lake impoundment and Churchill River diversion. Can. J. Fish. Aquat. Sci. **41**(4): 548-557.

Niemelä, J. 1999. Management in relation to disturbance in the boreal forest. For. Ecol. Manage. **115**(2): 127-134.

Nitschke, C.R. 2005. Does forest harvesting emulate fire disturbance? A comparison of effects on selected attributes in coniferous-dominated headwater systems. For. Ecol. Manage. **214**(1-3): 305-319.

Northwatch and MiningWatch Canada (NMC). 2008. The boreal below: mining issues and activities in Canada's boreal forest. Available online at: http://www.miningwatch.ca/sites/www.miningwatch.ca/files/Boreal_Below_2008_web.pdf. [Accessed 12 December 2013]

Nowlan, L. 2007. Out of sight, out of mind? Taking Canada's groundwater for granted. Eau Canada: The Future of Canada's Water. University of British Columbia Press, Vancouver.

Natural Resources Canada (NRCan). 2002. Peatlands of Canada database. Geogratis. Available online at: http://geogratis.gc.ca/api/en/nrcan-rncan/ess-sst/d5a3860c-2920-50b2-87ca-7e748778d4f0.html. [Accessed 4 December 2013]

Natural Resources Canada (NRCan). 2003*a*. Atlas of Canada1, 000,000 National frameworks data, hydrology-dams. Available online at: http://geogratis.gc.ca/api/en/nrcan-rncan/ess-sst/0c78d7fe-100b-5937-b74e-7590a03a6244.html. [Accessed 12 December 2013]

Natural Resources Canada (NRCan). 2003*b*. Atlas of Canada 1, 000,000 National frameworks data, hydrology-drainage areas. Available online at:

http://geogratis.gc.ca/api/en/nrcan-rncan/ess-sst/f4f51a17-ba98-57ae-9ad8-ff6878aea2d0.html. [Accessed 18 December 2013]

Natural Resources Canada (NRCan). 2007. Groundwater Mapping Program. Hydrogeological Regions. Available online at: http://atlas.nrcan.gc.ca/site/english/maps/water.html. [Accessed 18 December 2013]

Natural Resources Canada (NRCan). 2009. Small hydropower. Available online at: http://canmetenergy.nrcan.gc.ca/renewables/small-hydropower/617. [Accessed 3 December 2013]

Natural Resources Canada (NRCan). 2011. Important facts on Canada's Natural Resources (as of October 2011). Available online at: http://www.nrcan.gc.ca/statistics-facts/forests/897. [Accessed 3 December 2013]

Natural Resources Canada (NRCan). 2013*a*. Statistical data. Available online at: http://cfs.nrcan.gc.ca/statsprofile/economicimpact/ca. [Accessed 3 December 2013]

Natural Resources Canada (NRCan). 2013*b*. Transformative technologies. Available online at: cfs.nrcan.gc.ca/pages/166. [Accessed 3 December 2013]

Natural Resources Canada (NRCan). 2013*c*. Area of wetlands by ecozone. Available online at: http://cfs.nrcan.gc.ca/pages/353. [Accessed 3 December 2013]

Natural Resources Canada (NRCan). 2013*d*. Annual Statistics. Preliminary estimate of the mineral production of Canada, by province, 2012. Available online at: http://sead.nrcan.gc.ca/prod-prod/2012-eng.aspx. [Accessed 13 December 2013]

National Round Table on the Environment and the Economy (NRTEE). 2009. Charting a Path: Water and Canada's Natural Resource Sectors. Discussion Paper.

National Round Table on the Environment and the Economy (NRTEE). 2010. Changing Currents: Water Sustainability and the Future of Canada's Natural Resource Sectors. National Round Table, Ottawa, ON.

National Round Table on the Environment and the Economy (NRTEE). 2011. Charting a Course: Sustainable Water Use by Canada's Natural Resource Sectors. National Round Table.

Nicholls, K.H., Steedman, R.J., and Carney, E.C. 2003. Changes in phytoplankton communities following logging in the drainage basins of three boreal forest lakes in northwestern Ontario (Canada), 1991-2000. Can. J. Fish. Aquat. Sci. **60**:43-54.

Nip, A.M.K. 1991. Changes in hydrology and water quality following forest harvesting and their impact on salmonoid fishes in the Tri-Creeks Basin. Research Branch

Technical Report No. 4 ENR Technical Report T/199, 66. Alberta Forestry, Lands and Wildlife, Forest Service.

Nyman, J.A. 2011. Ecological functions of wetlands. *In* Wetlands. *Edited by* B.A. LePage. Springer, Dordrecht, Netherlands. pp. 115–128.

Observ'ER. 2011. Available online at: http://www.energies-renouvelables.org/observ-er/html/inventaire/pdf/14e-inventaire-Chap02.pdf. [Accessed 3 December 2013]

Ohlson, D., Long, G., and Hatfield, T. 2010. Phase 2 Framework Committee Report. pp. 238.

Oiffer, A.A.L., Barker, J.F., Gervais, F.M., Mayer, K.U., Ptacek, C.J., and Rudolph, D.L. 2009. A detailed field-based evaluation of naphthenic acid mobility in groundwater. J. Contam. Hydrol. **108**(3): 89-106.

Oke, T.R. 1978. Boundary layer climates. 2nd Ed. Methuen and Co., Ltd., London.

Ontario Ministry of Natural Resources (OMNR). Undated. Harvestining in the boreal forest. Available online at: http://www.mnr.gov.on.ca/stdprodconsume/groups/lr/@mnr/@forests/documents/document/241212.pdf. [Accessed 20 December 2013].

Ontario SFI Implementation Committee (OSIC). 2011. A guide to best management practices in northern Ontario: For private landowners and logging contractors. (SFI-01139). Available online at: http://www.sfiontario.org/BMP_Guide_for_web_v2_Reader_Friendly.pdf. [Accessed 3 December 2013]

Outridge, P.M., Rausch, N., Percival, J.B., Shotyk, W., and McNeely, R. 2011. Comparison of mercury and zinc profiles in peat and lake sediment archives with historical changes in emissions from the Flin Flon metal smelter, Manitoba, Canada. Sci. Tot. Environ. **409**(3): 548-563.

Paavilainen, E., and Päivänen, J. 1995. Peatland forestry: ecology and principles. Springer-Verlag, Berlin; New York. pp. 248.

Paish, O. 2002. Small hydro power: technology and current status. Renewable and sustainable energy reviews. **6**(6), 537-556.

Päivänen, J. 1980. The effect of silvicultural treatments on the ground water table in Norway spruce and Scots pine stands on peat. pp. 433-438.

Paré, D., Bernier, P., Thiffault, E., and Titus, B.D. 2011. The potential of forest biomass as an energy supply for Canada. For. Chron. **87**(1): 71-76.

Parks, K.P., and Grunsky, E. 2002. RADARSAT imager: A new way to help map discharge and recharge, Athabasca oil sands area. *In* Canadian Society of

Petroleum Geologists Annual Meeting – 2002. Canadian Society of Petroleum Geologists, Calgary, Alberta.

Partington, M., and Gillies, C. 2010. Resource roads and wetlands: Opportunities to maintain hydrologic function. FPInnovations Internal Report IR-2010-11-01.

Pasher, J., Seed, E., and Duffe, J. 2013. Development of boreal ecosystem anthropogenic disturbance layers for Canada based on 2008 to 2010 Landsat imagery. Can. J. Remote. Sens. **39**(1): 1-17.

Pavey, B., Saint-Hilaire, A., Courtenay, S., Ouarda, T., and Bobée, B. 2007. Exploratory study of suspended sediment concentrations downstream of harvested peat bogs. Environ. Monit. Assess. **135**(1-3): 369-382.

Prospectors and Developers Association of Canada (PDAC). 2012. E3plus: A framework for responsible exploration, principle and guidance notes. Available online at: www.pdac.ca/programs/e3-plus. [Accessed 3 December 2013]

Pelster, D., Burke, J.M., Couling, K., Luke, S.H., Smith, D.W., and Prepas, E.E. 2008. Water and nutrient inputs, outputs, and storage in Canadian boreal forest wetlands: a review. J. Environ. Eng. Sci. **7**(S1): 35-50.

Pentz, S., and Kostaschuk, R. 1999. Effect of placer mining on suspended sediment in reaches of sensitive fish habitat. Environ. Geol. **37**(1-2): 78-89.

Peters, D., and Buttle, J. 2010. The effects of flow regulation and climatic variability on obstructed drainage and reverse flow contribution in a Northern river–lake–Delta complex, Mackenzie basin headwaters. River Res. Appl. **26**(9): 1065-1089.

Peters, D.L., and Prowse, T.D. 2001. Regulation effects on the lower Peace River, Canada. Hydrol. Processes, **15**(16): 3181-3194.

Peters, D.L., Prowse, T.D., Pietroniro, A., and Leconte, R. 2006. Flood hydrology of the Peace-Athabasca Delta, northern Canada. Hydrol. Processes, **20**(19): 4073-4096.

Pinel-Alloul, B., Planas, D., Carignan, R., and Magnan, P. 2002. Review of ecological impacts of forest fires and harvesting on lakes of the boreal ecozone in Québec. Revue des Sciences de l'Eau **15**(1): 371-395.

Plamondon, A.P., and Ouellet, D.C. 1980. Partial clearcutting and streamflow regime of ruisseau des Eaux- Volees experimental basin. The influence of man on the hydrological regime with special reference to representative and experimental basins. Proc.Helsinki symposium, June 1980, (International Association of Hydrological Sciences, Washington DC; IAHS-AISH Publication, 130). pp. 129-136.

Poff, N.L., and Hart, D.D. 2002. How dams vary and why it matters for the emerging science of dam removal. Bioscience, **52**(8): 659-668.

Poff, N.L., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., Sparks, R.E., and Stromberg, J.C. 1997. The natural flow regime: A paradigm for river conservation and restoration. Bioscience, **47**(11): 769-784.

Pokhrel, D., and Viraraghavan, T. 2004. Treatment of pulp and paper mill wastewater-a review. Sci. Total Environ. **333**(1): 37-58.

Pollution Probe. 2008. A new approach to water management in Canada. pp.16.

Pomeroy, J.W., and Granger, R.J. 1997. Sustainability of the western Canadian boreal forest under changing hydrological conditions. I. Snow accumulation and ablation. IAHS Publication no. 240. pp. 237-242.

Pomeroy, J.W., and Granger, R.J. (eds). 1999. Wolf Creek Research Basin: Hydrology, Ecology, Environment. National Water Research Institute, Saskatoon, SK. 160p.

Pomeroy, J.W., Parviainen, J., Hedstrom, N., and Gray, D.M. 1998. Coupled modelling of forest snow interception and sublimation. Hydrol. Processes, **12**(15): 2317-2337.

Ponton, D.E., and Hare, L. 2009. Assessment of nickel contamination in lakes using the phantom midge Chaoborus as a biomonitor. Environ. Sci. Technol. **43**(17): 6529-6534.

Porvari, P., Verta, M., Munthe, J., and Haapanen, M. 2003. Forestry practices increase mercury and methyl mercury output from boreal forest catchments. Environ. Sci. Technol. **37**(11): 2389-2393.

Pothier, D., Prévost, M., and Auger, I. 2003. Using the shelterwood method to mitigate water table rise after forest harvesting. For. Ecol. Manage. **179**(1): 573-583.

Potvin, F., Courtois, R., and Bélanger, L. 1999. Short-term response of wildlife to clear-cutting in Quebec boreal forest: multiscale effects and management implications. Can. J. For. Res. **29**(7): 1120-1127.

Poulin, M., and S. Pellerin. 2001. La conservation, in Payette S., Rochefort L. (eds.), Écologie des tourbières du Québec-Labrador, Les presses de l'Université Laval, Québec, Canada. pp. 505-518.

Poulin, M., Rochefort, L., Pellerin, S., and Thibault, J. 2004. Threats and protection for peatlands in eastern Canada. Geocarrefour, **79**: 331-331-344.

Prebble, P., Coxworth, A., Simieritsch, T., Dyer, S., Huot, M., and Walsh, H. 2009. Carbon copy: Preventing oil sands fever in Saskatchewan. Oil Sands Fever Series. 14–20. Drayton Valley, Alberta: Pembina Institute, Saskatchewan Environmental Society, Canadian Parks and Wilderness Society.

Prepas, E.E., Burke, J.M., Chanasyk, D. S., Smith, D. W., Putz, G., Gabos, S., Chen, W., Millions, D., and Serediak, M. 2003. Impact of wildfire on discharge and phosphorus export from the Sakwatamau watershed in the Swan Hills, Alberta, during the first two years. J. Environ. Eng. Sci. **2**: S63-S72.

Prepas, E.E., Pinel-Alloul, B., Planas, D., Méthot, G., Paquet, S., and Reedyk, S. 2001*a*. Forest harvest impacts on water quality and aquatic biota on the Boreal Plain: Introduction to the TROLS lake program. Can. J. Fish. Aquat. Sci. **58**(2): 421-436.

Prepas, E.E., Planas, D., Gibson, J.J., Vitt, D.H., Prowse, T.D., Dinsmore, W.P., Halsey, L.A., McEachern, P.M., Paquet, S., Scrimgeour, G.J., Tonn, W.M., Paszkowski, C.A., and Wolfstein, K. 2001*b*. Landscape variables influencing nutrients and phytoplankton communities in Boreal Plain lakes of northern Alberta: A comparison of wetland- and upland-dominated catchments. Can. J. Fish. Aquat. Sci. **58**(7): 1286-1299.

Prepas, E.E., Burke, J.M., Whitson, I.R., Putz, G., and Smith, D.W. 2006. Associations between watershed characteristics, runoff, and stream water quality: Hypothesis development for watershed disturbance experiments and modelling in the Forest Watershed and Riparian Disturbance (FORWARD) project. J. Environ. Eng. Sci. **5**(S. 1).

Prévost, M., and Plamondon, A. 1999. Streamflow, water quality and substrate conditions of a drained black spruce peatland, Québec, Canada. Int. Peat J. **9**: 21-37.

Prévost, M., Barry, R., Stein J., and Plamondon, A.P. 1990. Snowmelt runoff modeling in a balsam fir forest with a variable source simulator (VSAS2). Water Resour. Res. **26**(5): 1067-1077.

Prévost, M., Belleau, P., and Plamondon, A. 1997. Substrate conditions in a treed peatland: responses to drainage. Ecoscience, **4**(4): 543-554.

Prévost, M., Plamondon, A.P., and Belleau, P. 1999. Effects of drainage of a forested peatland on water quality and quantity. J. Hydrol. **214**(1): 130-143.

Price, A.G., Dunham, K., Carleton, T., and Band, L. 1997. Variability of water fluxes through the black spruce (Picea mariana) canopy and feather moss (Pleurozium schreberi) carpet in the boreal forest of Northern Manitoba. J. Hydrol. **196**(1–4): 310-323.

Price, D.T., Alfaro, R.I., Brown, K.J., Flannigan, M.D., Fleming, R.A., Hogg, E.H., Girardin, M.P., Lakusta, T., Johnston, M., McKenney, D.W., Pedlar, J.H., Stratton, T., Sturrock, R.N., Thompson, I.D., Trofymow, J.A., and Venier, L.A. 2013. Anticipating the consequences of climate change for Canada's boreal forest ecosystems. Environ. Rev. **21**(4): 322-365.

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

Price, J.S., and Waddington, J.M. 2000. Advances in Canadian wetland hydrology and biogeochemistry. Hydrol. Processes, **14**(9): 1579-1589.

Price, J.S., and Whitehead, G.S. 2004. The influence of past and present hydrological conditions on Sphagnum recolonization and succession in a block-cut bog, Québec. Hydrol. Processes, **18**(2): 315-328.

Price, J., Heathwaite, A., and Baird, A. 2003. Hydrological processes in abandoned and restored peatlands: an overview of management approaches. Wetlands Ecol. Manage. **11**(1-2): 65-83.

Price, J.S., Branfireun, B.A., Waddington, M.J., and Devito, K.J. 2005. Advances in Canadian wetland hydrology, 1999–2003. Hydrol. Processes, **19**(1): 201-214.

Price, J. S., McLaren, R.G., and Rudolph, D.L. 2010. Landscape restoration after oil sands mining: conceptual design and hydrological modelling for fen reconstruction. Int. J. Min., Reclam. Environ. **24**(2): 109-123.

Prowse, T. D., and Conly, F. M. 2000. Multiple-hydrologic stressors of a northern delta ecosystem. Journal of Aquatic Ecosystem Stress and Recovery. **8**(1): 17-26.

Ptacek, C., Price, W., Smith, J.L., Logsdon, M., and McCandless, R. 2004. Land-use practices and changes – mining and petroleum production. In Threats to water availability in Canada. NWRI Scientific Assessment Report series No. 3 and ACSD Science Assessment Series No. 1, National Water Research Institute, Burlington, Ontario. pp. 67-75.

Rajotte, J.W., and Couture, P. 2002. Effects of environmental metal contamination on the condition, swimming performance, and tissue metabolic capacities of wild yellow perch (Perca flavescens). Can. J. Fish. Aquat. Sci. **59**(8): 1296-1304.

Regional Aquatics Monitoring Program (RAMP). 2009. Technical Design and Rationale. Available online at: http://www.ramp-alberta.org/UserFiles/File/RAMP_Design_&_Rationale.pdf. [Accessed 28 December 2013]

Reid, L.M., and Dunne, T. 1984. Sediment production from forest road surfaces. Water Resour. Res. **20**(11): 1753-1761.

Renöfält, B., Jansson, R., and Nilsson, C. 2010. Effects of hydropower generation and opportunities for environmental flow management in Swedish riverine ecosystems. Freshwat. Biol. **55**(1): 49-67.

Richardson, L. 2007. The oil sands: Toward sustainable development. In Report of the Standing Committtee on Natural Resources, 39^{th} Parliament, 1^{st} session. 34-47. Ottawa, ON: House of Commons.

Richardson, A.D., Hollinger, D.Y., Dail, D.B., Lee, J.T., Munger, J.W., and O'Keefe, J. 2009. Influence of spring phenology on seasonal and annual carbon balance in two contrasting New England forests. Tree Physiol. **29**(3): 321-331.

Richardson, J.S., Naiman, R.J., and Bisson, P.A. 2012. How did fixed-width buffers become standard practice for protecting freshwaters and their riparian areas from forest harvest practices? Freshwater Sci. **31**(1): 232-238.

Rosenberg, D.M., Bodaly, R.A., and Usher, P.J. 1995. Environmental and social impacts of large scale hydroelectric development: Who is listening? Global Environ. Change, **5**(2): 127-148.

Rosenberg, D.M., Berkes, F., Bodaly, R.A., Hecky, R.E., Kelly, C.A., and Rudd, J.W.M. 1997. Large-scale impacts of hydroelectric development. Env. Rev. **5**(1):27-54.

Rosenberg International Forum. 2013. The Mackenzie River Basin. Report of the Rosenberg International Forum's Workshop on Transboundary relations in the Mackenzie River Basin. pp. 43.

Rothwell, R. L., Woodard, P.M., and Rivard, P.G. 1993. The effect of peatland drainage and planting position on the growth of white spruce seedlings. North. J. Appl. For. **10**(4): 154-160.

Roy, V., Jeglum, J.K., and Plamondon, A.P. 1997. Water table fluctuations following clearcutting and thinning on Wally Creek wetlands. Lewis Publishers, CRC Press.

Roy, V., Ruel, J., and Plamondon, A.P. 2000. Establishment, growth and survival of natural regeneration after clearcutting and drainage on forested wetlands. For. Ecol. Manage. **129**(1): 253-267.

Rubec, C.D.A. 1991. Peat resources use in Canada: A national conservation issue. *In* Proceedings, International Peat Symposium, Duluth, Minnesota. *Edited by* D. Grubich.

Salles, J-M. 2011. Valueing biodiversity and ecosystem services: Why put economic values on Nature. C.R. Biologies **334**(2011): 469-482.

Sass, G. Z., Creed, I.F., Bayley, S.E., and Devito, K.J. 2007. Understanding variation in trophic status of lakes on the Boreal Plain: A 20 year retrospective using Landsat TM imagery. Remote Sens. Environ. **109**(2): 127-141.

Sass, G.Z., Creed, I.F., Bayley, S.E., and Devito, K.J. 2008*a*. Interannual variability in trophic status of shallow lakes on the Boreal Plain: Is there a climate signal? Water Resour. Res. **44**(8). doi:10.1029/2007WR006310.

Sass, G.Z., Creed, I.F., and Devito, K.J. 2008*b*. Spatial heterogeneity in trophic status of shallow lakes on the Boreal Plain: Influence of hydrologic setting. Water Resour. Res. **44**(8).

Saunders, J.O., and Wenig, M.M. 2007. Whose water? Canadian water management and the challenges of jurisdictional fragmentation. Eau Canada: The future of Canada's water. pp. 119-141.

Savard, M.M., Ahad, J.M.E., Gammon, P., Calderhead, A.I., Rivera, A., Martel, R., Klebek, M., Headley, J.V., Lefebvre, R.,Welsh, B., Smirnoff, A., Pakdel, H., Benoit, N., Liao, S., Jautzy, J., Gagnon, C., Vaive, J., Girard, I., and Peru, K. 2012. A local test study distinguishes natural from anthropogenic groundwater contaminants near an Athabasca oil sands mining operation. Geological Survey of Canada, Open File 7195**.** pp. 140. doi: 10.4095/292074

Schelker, J., Eklöf, K., Bishop, K., and Laudon, H. 2012. Effects of forestry operations on dissolved organic carbon concentrations and export in boreal first-order streams. J. Geophys. Res.: Biogeosci. **117**(G1). doi: 10.1029/2011JG001827.

Schiff, S., Aravena, R., Mewhinney, E., Elgood, R., Warner, B., Dillon, P., and Trumbore, S. 1998. Precambrian Shield wetlands: hydrologic control of the sources and export of dissolved organic matter. Climat. Change, **40**(2): 167-188.

Schindler, D.W. 1988. Effects of acid rain on freshwater ecosystems. Science, **239**(4836): 149-157.

Schindler, D.W., and Lee, P.G. 2010. Comprehensive conservation planning to protect biodiversity and ecosystem services in Canadian boreal regions under a warming climate and increasing exploitation. Biol. Conserv. **143**(7): 1571-1586.

Schindler, D., Donahue, W., and Thompson, J.P. 2007. Future water flows and human withdrawals in the Athabasca River. *In* Running out of Steam? Oil Sands Development and Water Use in the Athabasca River Watershed: Science and Market based Solutions. pp. 1-39.

Schmidt, A., Gibson, J.J., Santos, I.R., Schubert, M., Tattrie, K., and Weiss, H. 2010. The contribution of groundwater discharge to the overall water budget of two typical Boreal lakes in Alberta/Canada estimated from a radon mass balance. Hydrol. Earth Syst. Sci. **14**(1): 79-89.

Schmidt, R., Flechtner, F., Meyer, U., Neumayer, K.H., Dahle, C., König, R., and Kusche, J. 2008. Hydrological signals observed by the GRACE satellites. Surv. Geophys. **29**(4-5): 319-334.

Schneider, R.R., Stelfox, J.B., Boutin, S., and Wasel, S. 2003. Managing the cumulative impacts of land-uses in the western Canadian sedimentary basin: a modeling approach. Conserv. Ecol. **7**(1): 8.

Schoengold, K. and Zilberman, D. 2007. The economics of water, irrigation, and development. Handbook of agricultural economics. Vol. **3**. pp. 2933-2977.

Scott, K.A., Wissel, B., Gibson, J.J., and Birks, S.J. 2010. Chemical characteristics and acid sensitivity of boreal headwater lakes in northwest Saskatchewan. J. Limnol. **69** (Suppl.1): 33-44.

Seitz, N.E., Westbrook, C.J., and Noble, B.F. 2011. Bringing science into river systems cumulative effects assessment practice. Environ. Impact Assess. Rev. **31**(3): 172-179.

Seitz, N.E., Westbrook, C.J., Dube, M.G., and Squires, A.J. 2012. Assessing large spatial scale landscape change effects on water quality and quantity response in the lower Athabasca River basin. Int. Environ. Assess. Manag. **9**(3): 392-404.

Sellers, P., Hall, F., Margolis, H., Kelly, B., Baldocchi, D., Hartog, G., Cihlar, J., Ryan, M.G., Goodison, B., Crill, P., Ranson, K.J., Letterenmaier, D., and Wickland, D.E. 1995. The boreal ecostsyem-atmosphere study (BOREAS): an overview and early results from the 1994 year. Bull. Am. Meteorol. Soc. **76**(9): 1549-1577.

Sellers, P., Kelly, C.A., Rudd, J.W.M., and MacHutchon, A.R. 1996. Photodegradation of methylmercury in lakes. Nature, **380**: 694–697.

Shabbar, A., Skinner, W. 2004. Summer Drought Patterns in Canada and the Relationship to Global Sea Surface Temperatures. J. Climate. **17**(14): 2866-2880.

Sheehy, G. 1993. Conserving wetlands in managed forests. Issues Paper No. 1993-2**.** North American Wetlands Conservation Council, Ottawa, ON.

Sherriff, B.L., Etcheverry, D.J., Sidenko, N.V., and van Guick, J. 2011. Spatial and temporal evolution of Cu-Zn mine tailings during dewatering. Appl. Geochem. **26**(11): 1832-1842.

Shrubsole, D., and Draper, D. 2007. On guard for thee? Water (ab) uses and management in Canada. Eau Canada: The Future of Canada's Water. pp. 37-54.

Sibley, P.K., Kreutzweiser, D.P., Naylor, B.J., Richardson, J.S., and Gordon, A.M. 2012. Emulation of natural disturbance (END) for riparian forest management: synthesis and recommendations. Freshwat. Sci. **31**(1): 258-264.

Silins, U., and Rothwell, R.L. 1998. Forest peatland drainage and subsidence affect soil water retention and transport properties in an Alberta peatland. Soil Sci. Soc. Am. J. **62**(4): 1048-1056.

Silins, U., and R.L. Rothwell, R. L. 1999. Spatial patterns of aerobic limit depth and oxygen diffusion rate at two peatlands drained for forestry in Alberta. Canadian J. For. Res. **29**(1): 53-61.

Singh, S.P., and Hendry, M.J. 2013. Solid-phase distribution and leaching behaviour of nickel and uranium in a uranium waste-rock pile. Wat. Air Soil Pollut. **224**(1): 1360. DOI 10.1007/s11270-012-1360-9.

Smerdon, B., Devito, K., and Mendoza, C. 2005. Interaction of groundwater and shallow lakes on outwash sediments in the sub-humid Boreal Plains of Canada. J. Hydrol. **314**(1): 246-262.

Smerdon, B.D., Redding, T., and Beckers, J. 2009. An overview of the effects of forest management on groundwater hydrology. J. Ecosyst. Manage. **10**(1): 22-44.

Smith, J.B., and Tirpak, D.A. 1989. The potential effects of global climate change on the United States: Report to Congress. US Environmental Protection Agency, Office of Policy, Planning, and Evaluation, Office of Research and Development.

Smith, V. H., Tilman, G.D., and Nekola, J.C. 1999. Eutrophication: impacts of excess nutrient inputs on freshwater, marine, and terrestrial ecosystems. Environmental Pollution 100. pp. 179-196.

Smith, D.W., Prepas, E.E., Burke, J.M., Chanasyk, D.S., and Putz, G. 2003. The Forest Watershed and Riparian Disturbance study: a multi-discipline initiative to evaluate and manage watershed disturbance on the Boreal Plain of Canada. J. Environ. Eng. Sci. **2**(Suppl.1):S87-S101.

Snucins, E., and Gunn, J.M. 2003. Use of rehabilitation experiments to understand the recovery dynamics of acid-stressed fish populations. AMBIO. **32**(3): 240-243.

Sobolewski, A. 1996. Metal species indicate the potential of constructed wetlands for long-term treatment of metal mine drainage. Ecol. Eng. **6**(4): 259–271.

Soil Classification Working Group.1998. The Canadian system of soil classification. NRC Research Press.

Sørensen, R., Meili, M., Lambertsson, L., von Brömssen, C., and Bishop, K. 2009. The effects of forest harvest operations on mercury and methylmercury in two boreal streams: relatively small changes in the first two years prior to site preparation. AMBIO: A Journal of the Human Environment, **38**(7): 364-372.

Sprague, J.B. 2007. Great wet north? Canada's myth of water abundance.

Squires, A.J., Westbrook, C.J., and Dubé, M.G. 2009. An approach for assessing cumulative effects in a model river, the Athabasca River basin. Int. Environ. Assess. Manage. **6**(1): 119-134.

St-Hilaire, A., Courtenay, S.C., Diaz-Delgado, C., Pavey, B., Ouarda, T.B., Boghen, A., and Bobée, B. 2006. Suspended sediment concentrations downstream of a harvested peat bog: Analysis and preliminary modelling of exceedances using logistic regression. Can. Water Res. J. **31**(3): 139-156.

St. Louis, V.L., Rudd, J.W., Kelly, C.A., Bodaly, R., Paterson, M.J., Beaty, K.G., Hesslein, R.H., Heyes, A., and Majewski, A.R. 2004. The rise and fall of mercury methylation in an experimental reservoir. Environ. Sci. Technol. **38**(5): 1348-1358.

Stanley, E. H., and Doyle, M.W.. 2003. Trading off: The ecological effects of dam removal. Front. Ecol. Environ. **1**(1): 15-22.

Starr, M., Ukonmaanaho, L., Sibley, P.K., and Gordon, A.M. 2006. Deposition in boreal forests in relation to type, size, number, and placement of collectors. Environ. Monit. Assessm. **125**:123-136.

Statistics Canada. 2010. Human Activity and the Environment - Freshwater supply and demand in Canada, 2010 16-201-XIE.

Stednick, J.D. 1996. Monitoring the effects of timber harvest on annual water yield. J. Hydrol. **176**(1): 79-95.

Steedman, R.J. 2000. Effects of experimental clearcut logging on water quality in three small boreal forest lake trout (*Salvelinus namaycush*) lakes. Can. J. Fish. Aquat. Sci. **57**(S2): 92-96.

Steedman, R.J., and France, R.L. 2000. Origin and transport of aeolian sediment from new clearcuts into Boreal Lakes, northwestern Ontario, Canada. Water Air Soil Pollut. **122**(1-2): 139-152.

Steedman, R.J., and Kushneriuk, R.S. 2000. Effects of experimental clearcut logging on thermal stratification, dissolved oxygen, and lake trout (Salvelinus namaycush) habitat volume in three small boreal forest lakes. Can. J. Fish. Aquat. Sci. **57**(SUPPL. 2): 82-91.

Steedman, R.J., Kushneriuk, R.S., and France, R.L. 2001. Littoral water temperature response to experimental shoreline logging around small boreal forest lakes. Can. J. Fish. Aquat. Sci. **58**(8): 1638-1647.

Stevens, N. 1979. Hydrologic simulation of Tri-Creeks Watershed. Tri-Creeks Watershed Study report No. 4- 23. Alberta Energy and Natural Resources, Forest Land Use Branch, Watershed Management.

Strong, W.L. 2000. Vegetation development on reclaimed lands in the Coal Valley Mine of western Alberta, Canada. Can. J. Bot. **78**(1): 110-118.

Struzik, E. 2013. Underground Intelligence: The need to map, monitor, and manage Canada's groundwater resources in an era of drought and climate change. For the Program on Water Issues, Munk School of Global Affairs at the University of Toronto.

Surette, C., Lucotte, M., and Harris, R. 2003. How does fishing intensity affect the efficiency of a mercury bioaccumulation model in predicting fish Hg concentration? J.Phys. IV France**, 107**: 1445-1445.

Swanson, R.H, and Hillman, G.R. 1977. Predicted increased water yield after clear-cutting verified in west-central Alberta. Fish. Environ. Can., Can. For. Serv., North. For. Res. Cent. Inf. Rep. NOR-A-198.

Swanson, R.H., and Rothwell, R.L. 2001. Hydrologic recovery of aspen clearcuts in northwestern Alberta. *In:* Sustaining aspen in western landscapes: symposium procreedings. Shepperd, W.D., Binkly, D., Bartos, D.L., Stohlgren, T.J., and Eskew, L.G. (compliers). U.s Department of Agriculture Forest Service, Rocky Mountain Research Station, Fort Collins, Colo. RMRS-P-**18**: 121-136.

Swanson, R.H., Golding, D.L., Rothwell, R.L., and Bernier, P.Y. 1986. Hydrologic effects of clear-cutting at Marmot Creek and Streeter watersheds, Alberta. Can. For. Serv., North. For. Cent. , Edmonton, Alberta. Inf. Rep. NOR-X-278.

Szkokan-Emilson, E.J., Kielstra, B., Watmough, S., and Gunn, J. 2013. Drought-induced release of metals from peatlands in watersheds recovering from historial metal and sulphur deposition. Biogeochemistry **116**: 131-145.

Tague, C., and Band, L. 2001. Simulating the impact of road construction and forest harvesting on hydrologic response. Earth Surf. Process. Landforms, **26**(2): 135-151.

Tarnocai, C. 1980. Canadian wetland registry. *In* Procedings of a workshop on Canadian wetlands: meeting of the Natl. Wetl. Work. Group, Saskatoon, Saskat., 11-13 June 1979/comp. *Edited by* C.D.A. Rubec, and F.C. Pollett. Compte-rendu d'un atelier sur les terres humides du Canada. Ottawa: Environment Canada, Lands Directorate **12**: 9-21.

Tarnocai, C., and Stolbovoy, V. 2006. Northern peatlands: their characteristics, development and sensitivity to climate change. Dev. Earth Surface Processes, **9**: 17-51.

Telford, P.G. 2009. Peat fuel–a sustainable bioenergy resource. *In* Proceedings of the IASTED International Conference. Vol. **650**. pp. 815.

Thiffault, E., Paré, D., Brais, S., and Titus, B.D. 2010. Intensive biomass removals and site productivity in Canada: A review of relevant issues. For. Chron. **86**(1): 36-42.

Tilton, D.L. 1978. Comparative growth and foliar element concentrations of Larix laricina over a range of wetland types in Minnesota. J. Ecol. **66**: 499-512.

Timoney, K. 2006. Landscape cover change in the Peace-Athabasca Delta, 1927–2001. Wetlands, **26**(3): 765-778.

Timoney, K. 2012. Environmental and health impacts of Canada's bitumen industry: In search of answers. Env. Sci. Technol. **46**: 2496-2497.

Timoney, K., and Lee, P. 2001. Environmental management in resource-rich Alberta, Canada: First world jurisdiction, third world analogue? J. Environ. Manage. **63**(4): 387-405.

Timoney, K.P. and Lee, P. 2009. Does the Alberta tar sands industry pollute? The scientific evidence. Open Conserv. Biol. J. **3**: 65-81.

Townsend, C.R., Uhlmann, S.S., and Matthaei, C.D. 2008. Individual and combined responses of stream ecosystems to multiple stressors. J. Appl. Ecol. **45**(6): 1810-1819.

Tremblay, G.A., and Hogan, C.M. 2006. Initiatives at Natural Resources Canada to deal with orphan and abandoned mines. *In* Proceedings of the 30th annual British Columbia mine reclamation symposium. British Columbia Technical and Research Committee on Reclamation, 19-22 June 2006, Smithers, B.C.

Tremblay, A., Lambert, M. and Gagnon, L. 2004. Do hydroelectric reservoirs emit greenhouse gases? Environ. Manage. **33**(SUPPL. 1): S509-S517.

Tremblay, Y., Rousseau, A.N., Plamondon, A.P., Lévesque, D., and Jutras, S. 2008. Rainfall peak flow response to clearcutting 50% of three small watersheds in a boreal forest, Montmorency Forest, Québec. J. Hydrol. **352**(1): 67-76.

Tremblay, Y., Rousseau, A.N., Plamondon, A.P., Lévesque, D., and Prévost, M. 2009. Changes in stream water quality due to logging of the boreal forest in the Montmorency Forest, Québec. Hydrol. Processes, **23**(5): 764-776.

Trombulak, S.C., and Frissell, C.A. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conserv. Biol. **14**(1): 18-30.

Turchenek, L., Abboud, S.A., Dowey, U., and Alliance, C.A.S. 1998. Critical loads for organic (peat) soils in Alberta. Alberta Research Council.

Turner, M.A., and Rudd, J.W. 1983. The English-Wabigoon River system: III. Selenium in lake enclosures: its geochemistry, bioaccumulation, and ability to reduce mercury bioaccumulation. Can. J. Fish. Aquat. Sci. **40**(12): 2228-2240.

United Nations Environment Programme (UNEP). 2009. Water security and ecosystem services: The critical connection. A contribution to the United Nations World Water Assessment Program. Available online at: http://www.unep.org/themes/freshwater/pdf/the_critical_connection.pdf. [Accessed 3 December 2013]

United States Environmental Protection Agency (EPA). 1990. US Clean Air Act. Available online at: http://www.epa.gov/air/caa/index.html. [Accessed 12 December 2013]

Urquizo, N., Bastedo, J., Brydges, T., and Shear, H. (eds). 2000. Ecological assessment of the boreal shield ecozone. Environment Canada, Ottawa, Ont. Available online at: www.ec.gc.ca/Publications/1F4C0C47-4E18-4988-8514-A842EED6F774%5CEcologicalAssessmentOfTheBorealShieldEcozone.pdf. [Accessed 3 December 2013]

Van Der Vinne, G. 1992. Hydrologic impacts of forest harvesting in the Tri-Creek Watershed. *In:* Quality of aquatic habitats in Alberta: Proceedings of the third annual applied aquatic studies workshop, Edmonton, Alberta, February 17-18, 1987, editors, Moore, J.W., and Thomas A. pp. 34-54.

Van Geen, A., Adkins, J.F., Boyle, E.A., Nelson, C.H., and Palanques, A. 1997. A 120-yr record of widespread contamination from mining of the Iberian pyrite belt. Geology, **25**(4) : 291-294.

Van Seters, T.E., and Price, J.S. 2001. The impact of peat harvesting and natural regeneration on the water balance of an abandoned cutover bog, Quebec. Hydrol. Processes, **15**(2): 233-248.

Van Seters, T.E., and Price, J.S. 2002. Towards a conceptual model of hydrological change on an abandoned cutover bog, Quebec. Hydrol. Processes, **16**(10): 1965-1981.

Venier, L.A., Thompson, I., Fleming, R., Malcolm, J., Aubin, I., Trofymow, J.A., Langor, D., Sturrock, R., Patry, C., Outerbridge, R.O., Holmes, S.B., Haeussler, S., De Grandpré, L., Chen, H.Y.H., Bayne, E., Arsenault, A., and Brandt, J.P. 2014. Effects of natural resource development on the terrestrial biodiversity of Canadian boreal forests. Environ. Rev. *In press*.

Verta, M. 1990. Mercury in Finnish forest lakes and reservoirs: anthropogenic contribution to the load and accumulation in fish. Publications of the Water and Environment Research Institute. National Board of Waters and the Environment, Finland, No.6.

Vose, J.M., Sun, G., Ford, C.R., Bredemeier, M., Otsuki, K., Wei, X., Zhang, Z., and Zhang, L. 2011. Forest ecohydrological research in the 21st century: What are the critical needs? Ecohydrology, **4**(2): 146-158.

Wagener, T., Sivapalan, M., Troch, P., and Woods, R. 2007. Catchment classification and hydrologic similarity. Geogr. Compass, **1**(4): 901-931.

Walden, C.C. 1976. The toxicity of pulp and paper mill effluents and corresponding measurement procedures. Water Res. **10**(8): 639-664.

Wang, S., and Mulligan, C.N. 2006. Occurrence of arsenic contamination in Canada: Sources, behavior and distribution. Sci. Total Environ. **366**(2): 701-721.

Waring, R.H., and Schlesinger, W.H. 1985. Forest ecosystems. Concepts and management. Academic Press.

Watertight Solutions Ltd. 2005. Variability of precipitation and streamflow, Grande Cache-Grande Prairie and discussion of guidelines for water yield and peak flow increases. Prepared by Watertight Solutions for Weyerhaeuser Canada, Grande Prairie. FRIP Report.

Weber, L.P., Dubé, M.G., Rickwood, C.J., Driedger, K., Portt, C., Brereton, C., and Janz, D.M. 2008. Effects of multiple effluents on resident fish from Junction Creek, Sudbury, Ontario. Ecotoxicol. Environ. Saf. **70**(3): 433-445.

Webster, K.L., and McLaughlin, J.W. 2013. Application of a Bayesian belief network for assessing the vulnerability of permafrost to thaw and implications for greenhouse gas production and climate feedbacks. Environmental Science and Policy doi:10.1016/j.envsci.2013.10.00.

Weis, J.S., and Weis, P. 2004. Metal uptake, transport and release by wetland plants: implications for phytoremediation and restoration. Environ. Internat. **30**(5): 685-700.

Wells, J., Roberts, D., Lee, P., Cheng, R., and Darveau, M. 2010. A forest of blue - Canada's boreal forest: the world's waterkeeper. pp. 74.

Wemple, B. C., and Jones, J.A. 2003. Runoff production on forest roads in a steep, mountain catchment. Water Resour. Res. **39**(8): 1220. doi:10.1029/2002WR001744.

Wemple, B.C., Jones, J.A., and Grant, G.E. 1996. Channel network extension by logging roads in two basins, Western Cascades, Oregon. J. Am. Water Resour. Assoc. **32**(6): 1195-1207.

Wemple, B.C., Swanson, F.J., and Jones, J. 2001. Forest roads and geomorphic process interactions, Cascade Range, Oregon. Earth Surf.: Process. Landforms, **26**: 191-204.

White, B., Ogilvie, J., Campbell, D.M., Hiltz, D., Gauthier, B., Chisholm, H.K.H., Wen, H.K., Murphy, P.N., and Arp, P.A. 2012. Using the Cartographic Depth-to-Water Index to Locate Small Streams and Associated Wet Areas across Landscapes. Can. Water Resour. J. **37**(4): 333-347. Available online at: http://srd.alberta.ca/MapsPhotosPublications/Maps/ResourceDataProductCatalogue/Hydrological.aspx. [Accessed 4 December 2013]

Whitfield, C.J., Aherne, J., Cosby, B.J., and Watmough, S.A. 2010. Modelling boreal lake catchment response to anthropogenic acid deposition. J. Limnol. **69**(SUPPL. 1): 135-146.

Whitman, A. 2005. The outlook for forest products markets in Europe and implications for the profitability of plantation forestry in the United Kingdom. Food and Agriculture Organization of the United Nations. Available online at: http://www.fao.org/forestry/5538-0d7bb3918ab2b96944a4664ab039db25f.pdf. [Accessed 4 December 2013]

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

Wiener, J.G., Fitzgerald, W.F., Watras, C.J., and Rada, R.G. 1990. Partitioning and bioavailability of mercury in an experimentally acidified Wisconsin lake. Environ. Toxicol. Chem. **9**(7): 909-918.

Winkler, G., Leclerc, V., Sirois, P., Archambault, P., and Bérubé, P. 2009. Short-term impact of forest harvesting on water quality and zooplankton communities in oligotrophic headwater lakes of the eastern Canadian Boreal Shield. Boreal Environ. Res. **14**(2): 323-337.

Winkler, R.D. 1999. A preliminary comparison of clearcut and forest snow accumulation and melt in south-central British Columbia. *In* Confronting uncertainty, managing change in water resources and the environment, Conference Proceeding. Canadian Water Resources Association, 27-29 October,1999. pp. 294-300.

Winkler, R.D., Spittlehouse, D.L., and Golding, D.L. 2005. Measured differences in snow accumulation and melt among clearcut, juvenile, and mature forests in southern British Columbia. Hydrol. Processes, **19**(1): 51-62.

Winter, T.C. 2001. The concept of hydrologic landscapes. J. Am. Water Resour. Assoc. **37**(2): 335-349.

Wiramanaden, C.I.E., Forster, E.K., and Liber, K. 2010. Selenium distribution in a lake system receiving effluent from a metal mining and milling operation in Northern Saskatchewan, Canada. Environ. Toxicol. Chem. **29**(3): 606-616.

Wireman, M. 2001. Potential water-quality impact of hard-rock mining. Groundwater Monit. Rem. **21**(3): 40-51.

Wolfe, B.B., Karst-Riddoch, T.L., Vardy, S.R., Falcone, M.D., Hall, R.I., and Edwards, T.W. 2005. Impacts of climate and river flooding on the hydro-ecology of a floodplain basin, Peace-Athabasca Delta, Canada since AD 1700. Quatern. Res. **64**(2): 147-162.

Woo, M., and Thorne, R. 2009. Effects of reservoirs on streamflow in the boreal region. In “Threats to Global Water Security”. J.A.A. Jones et al. eds. Springer Netherlands, pp. 339-348.

Woo, M. K., and Young, K. L. 2003. Hydrogeomorphology of patchy wetlands in the High Arctic, polar desert environment. Wetlands. **23**(2), 291-309.

Woo, M., Thorne, R., Szeto, K., and Yang, D. 2008. Streamflow hydrology in the boreal region under the influences of climate and human interference. Philos. Trans. R. Soc., B: Biol. Sci. **363**(1501): 2249-2258.

Woodward, G. 2009. Biodiversity, ecosystem functioning and food webs in fresh waters: assembling the jigsaw puzzle. Freshwat. Biol. **54**(10): 2171-2187.

WorleyParsons. 2010. Lower Athabasca Regional Plan: Groundwater model calibration and 4 Mbpd Scenario. Prepared for Alberta Environment.

WorleyParsons. 2012. Groundwater flow model for the Athabasca Oil Sands, North of Fort McMurray; Phase 1 Conceptual and Numerical Model Development. Report prepared for Alberta Environment and Water.

Woynillowicz, D., Severson-Baker, C, and M. Raynolds. 2005. Oil Sands Fever-The Environmental Implications of Canada's Oil Sands Rush. The Pembina Institute.

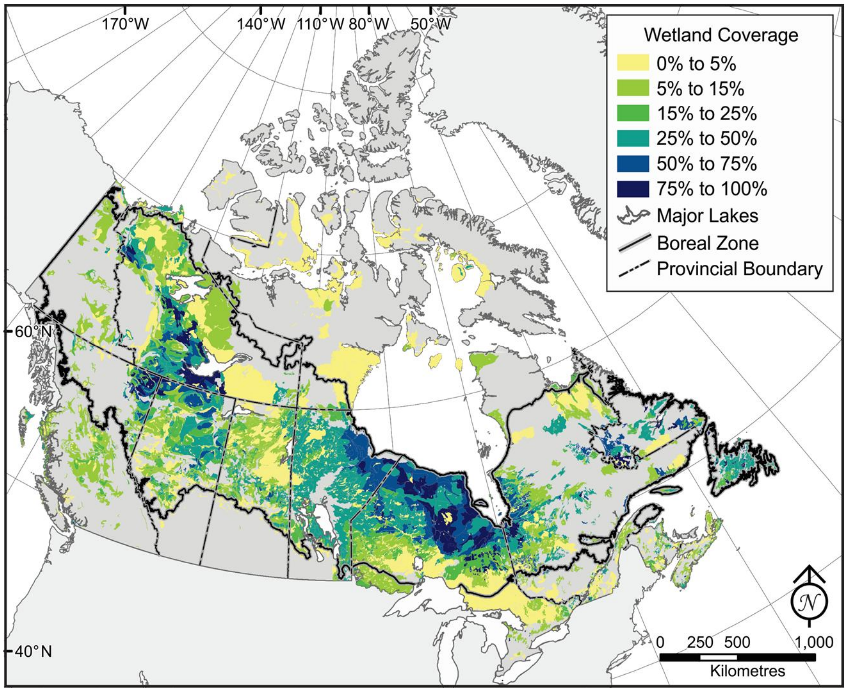
Yu, B., and Zwiers, F. W. 2007. The impact of combined ENSO and PDO on the PNA climate: a 1,000-year climate modeling study. Clim. Dynam. **29**(7-8): 837-851.

Younger, P. L., Banwart, S. A., and Hedin, R. S. (Eds.). 2002. Mine water: Hydrology, pollution, remediation (Vol. 5). Springer.

Zhang, B.Y., Zheng, J.S., and Sharp, R.G. 2010. Phytoremediation in engineered wetlands: Mechanisms and applications. Procedia Environ. Sci. **2**: 1315–1325.

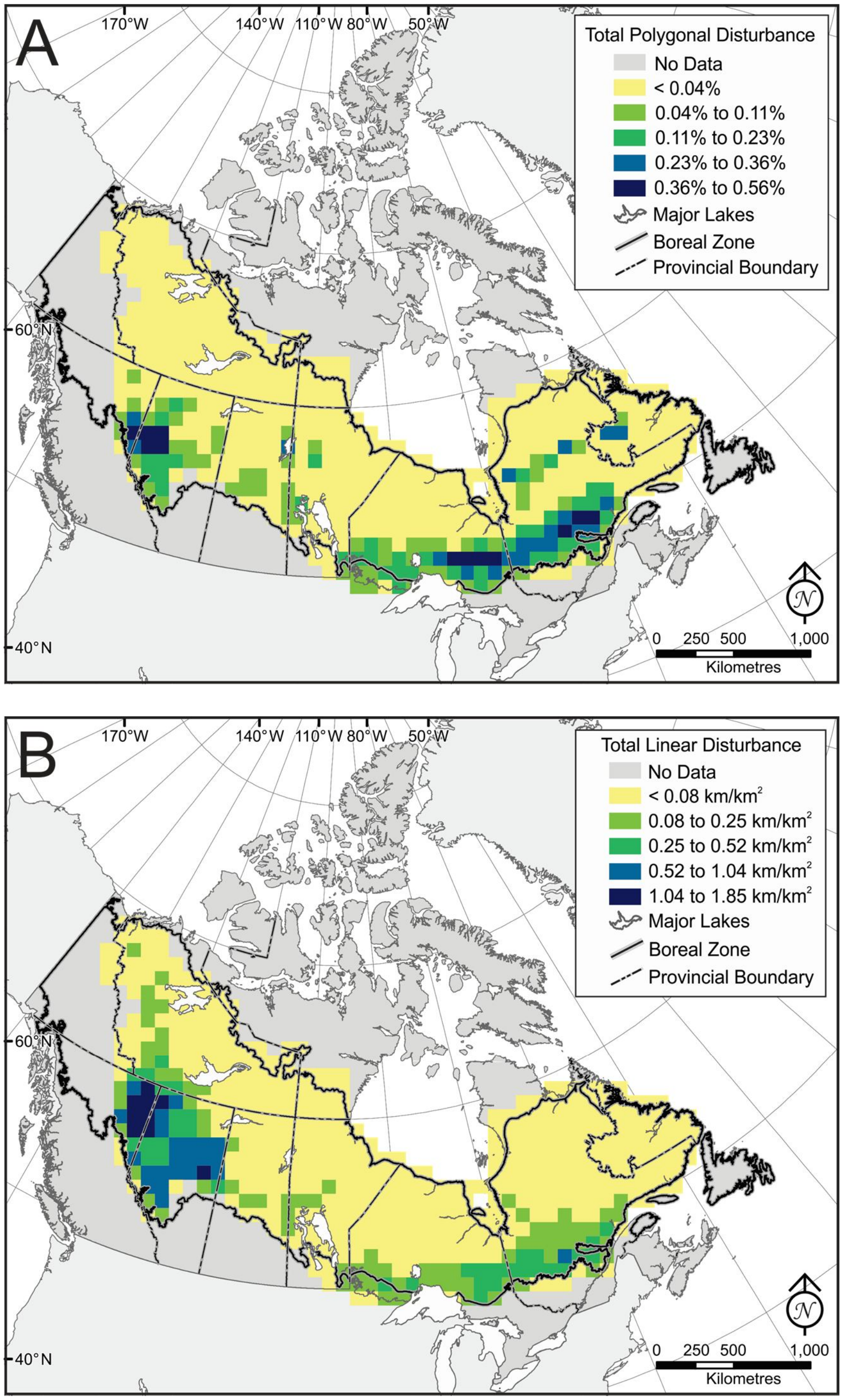
Zielinski, R.A., Otton, J.K., and Johnson, C.A. 2001. Sources of salinity near a coal mine spoil pile, north-central Colorado. J. Environ. Qual. **30**(4), 1237-1248.

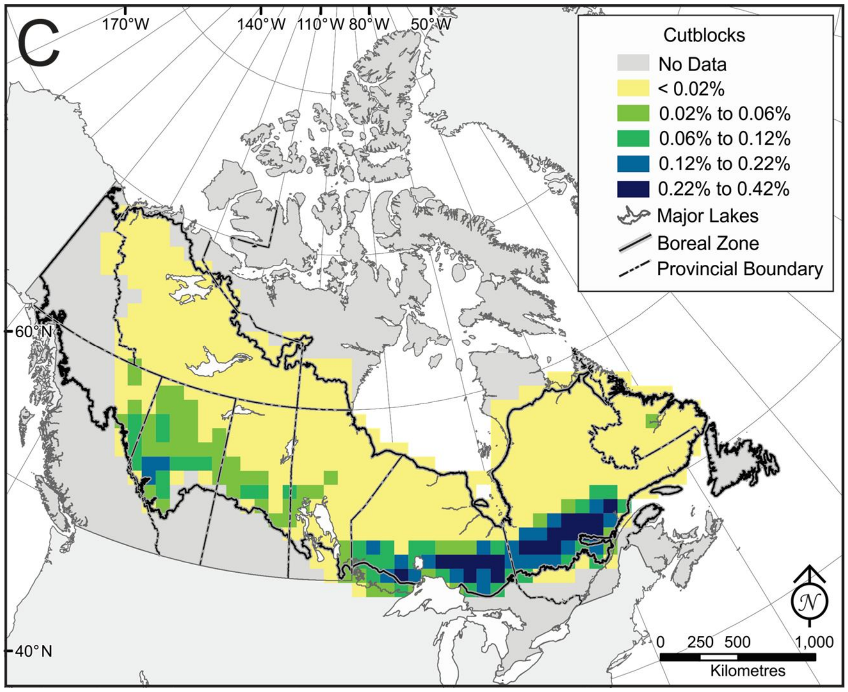
Zoltai, S.C., Taylor, S., Jeglum, J.K., Mills, G.F., and Johnson, J.D. 1988. Wetlands of Boreal Canada. *In* Wetlands of Canada. *Edited by* National Wetlands Working Group, Canada Committee on Ecological Classification. Montreal, Quebec, Polyscience Publications Inc. pp. 97-154.

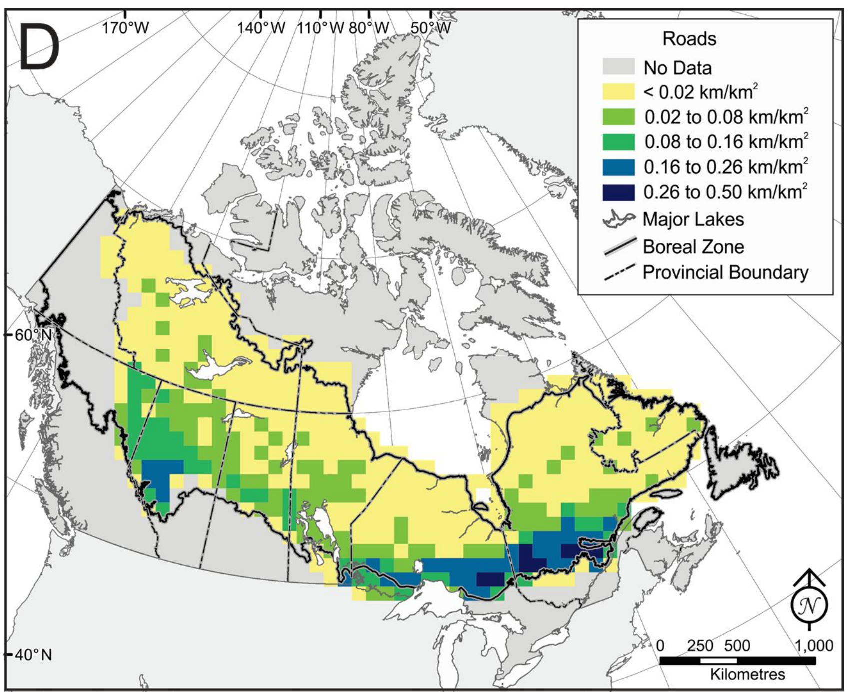


170°W
140°W
110°W
80°W
50°W
60°N
40°N
Wetland Coverage
0% to 5%
5% to 15%
15% to 25%
25% to 50%
50% to 75%
75% to 100%
Major Lakes
Boreal Zone
Provincial Boundary
0
250
500
1,000
Kilometres

Figure 1:


A
170°W
140°W
110°W
80°W
50°W
60°N
40°N
Total Polygonal Disturbance
No Data
< 0.04%
0.04% to 0.11%
0.11% to 0.23%
0.23% to 0.36%
0.36% to 0.56%
Major Lakes
Boreal Zone
Provincial Boundary
0 250 500 1,000
Kilometres
B
170°W
140°W
110°W
80°W
50°W
60°N
40°N
Total Linear Disturbance
No Data
< 0.08 km/km²
0.08 to 0.25 km/km²
0.25 to 0.52 km/km²
0.52 to 1.04 km/km²
1.04 to 1.85 km/km²
Major Lakes
Boreal Zone
Provincial Boundary
0 250 500 1,000
Kilometres


C
170°W
140°W
110°W
80°W
50°W
60°N
40°N
Cutblocks
No Data
< 0.02%
0.02% to 0.06%
0.06% to 0.12%
0.12% to 0.22%
0.22% to 0.42%
Major Lakes
Boreal Zone
Provincial Boundary
0
250
500
1,000
Kilometres


D
170°W
140°W
110°W
80°W
50°W
60°N
40°N
Roads
No Data
< 0.02 km/km²
0.02 to 0.08 km/km²
0.08 to 0.16 km/km²
0.16 to 0.26 km/km²
0.26 to 0.50 km/km²
Major Lakes
Boreal Zone
Provincial Boundary
0
250
500
1,000
Kilometres

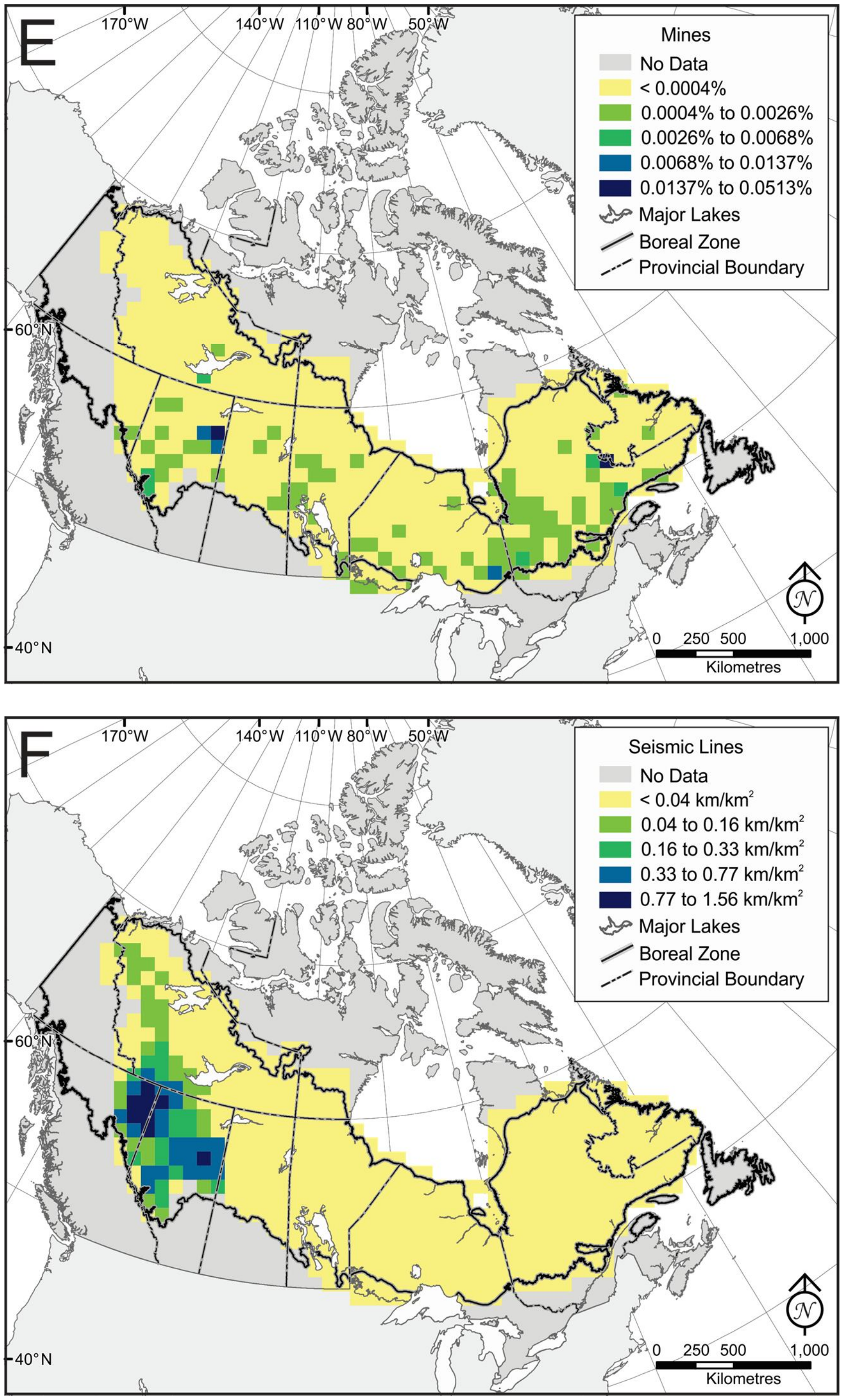
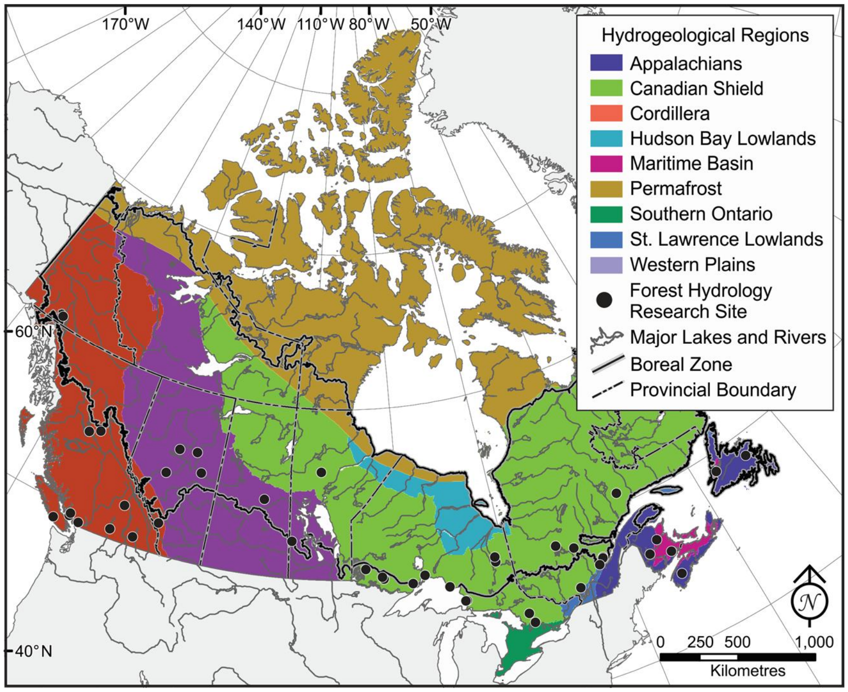
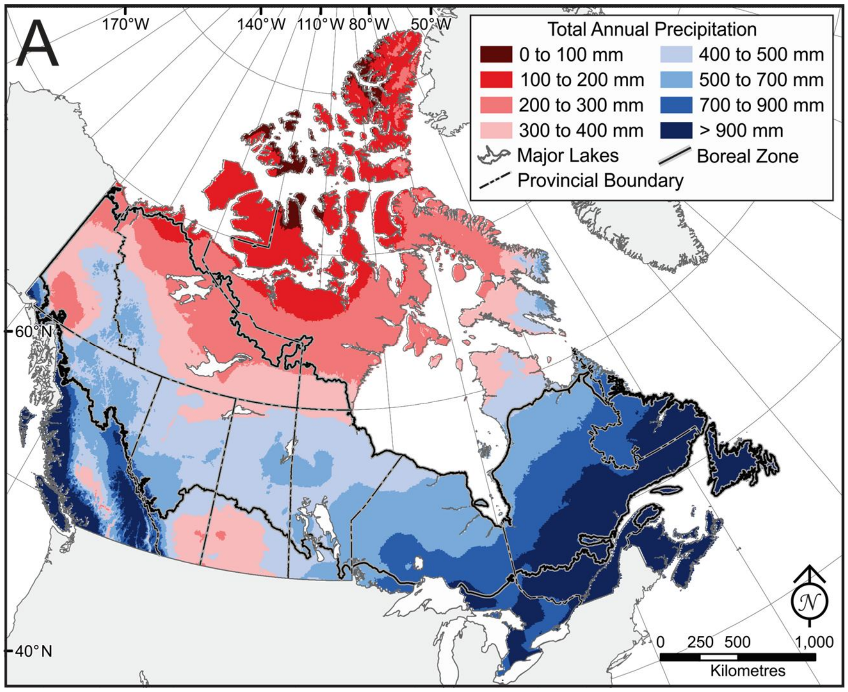

E
170°W
140°W
110°W
80°W
50°W
60°N
40°N
Mines
No Data
< 0.0004%
0.0004% to 0.0026%
0.0026% to 0.0068%
0.0068% to 0.0137%
0.0137% to 0.0513%
Major Lakes
Boreal Zone
Provincial Boundary
0 250 500 1,000
Kilometres
F
170°W
140°W
110°W
80°W
50°W
60°N
40°N
Seismic Lines
No Data
< 0.04 km/km²
0.04 to 0.16 km/km²
0.16 to 0.33 km/km²
0.33 to 0.77 km/km²
0.77 to 1.56 km/km²
Major Lakes
Boreal Zone
Provincial Boundary
0 250 500 1,000
Kilometres

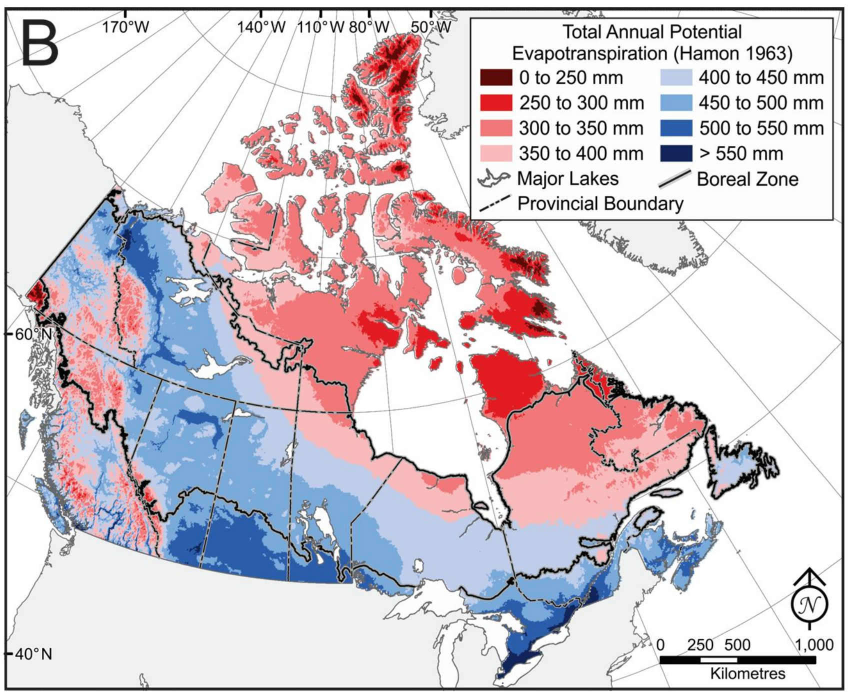
Figure 2:



170°W
140°W
110°W
80°W
50°W
60°N
40°N
Hydrogeological Regions
Appalachians
Canadian Shield
Cordillera
Hudson Bay Lowlands
Maritime Basin
Permafrost
Southern Ontario
St. Lawrence Lowlands
Western Plains
Forest Hydrology Research Site
Major Lakes and Rivers
Boreal Zone
Provincial Boundary
0 250 500 1,000
Kilometres
N

Figure 3:


A
170°W
140°W
110°W
80°W
50°W
60°N
40°N
Total Annual Precipitation
0 to 100 mm
100 to 200 mm
200 to 300 mm
300 to 400 mm
400 to 500 mm
500 to 700 mm
700 to 900 mm
> 900 mm
Major Lakes
Boreal Zone
Provincial Boundary
N
0 250 500 1,000
Kilometres


B
170°W
140°W
110°W
80°W
50°W
60°N
40°N
Total Annual Potential Evapotranspiration (Hamon 1963)
0 to 250 mm
250 to 300 mm
300 to 350 mm
350 to 400 mm
400 to 450 mm
450 to 500 mm
500 to 550 mm
> 550 mm
Major Lakes
Boreal Zone
Provincial Boundary
N
0 250 500 1,000
Kilometres

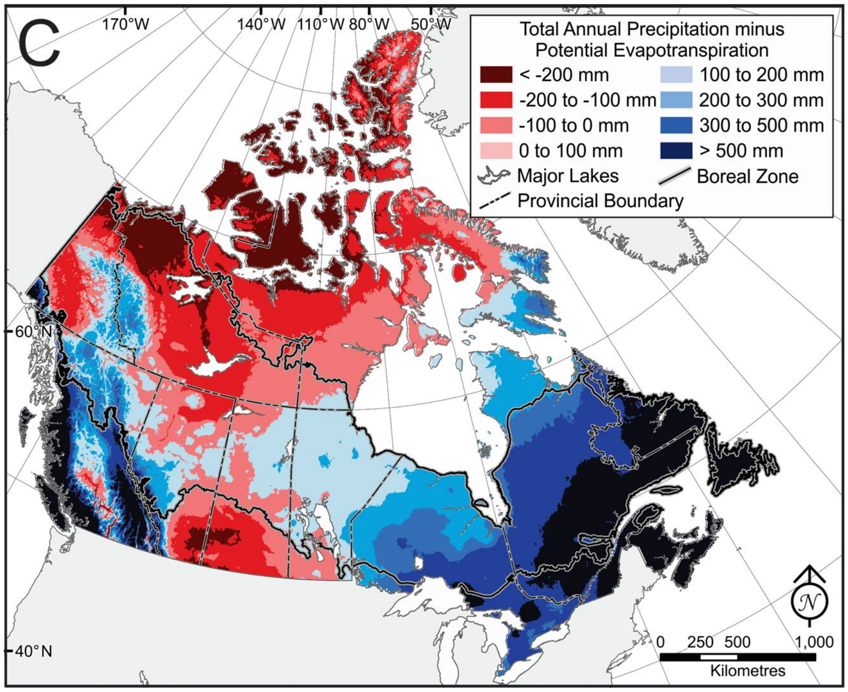

C
170°W
140°W
110°W
80°W
50°W
60°N
40°N
Total Annual Precipitation minus
Potential Evapotranspiration
< -200 mm
-200 to -100 mm
-100 to 0 mm
0 to 100 mm
100 to 200 mm
200 to 300 mm
300 to 500 mm
> 500 mm
Major Lakes
Boreal Zone
Provincial Boundary
N
0 250 500 1,000
Kilometres

Figure 4:

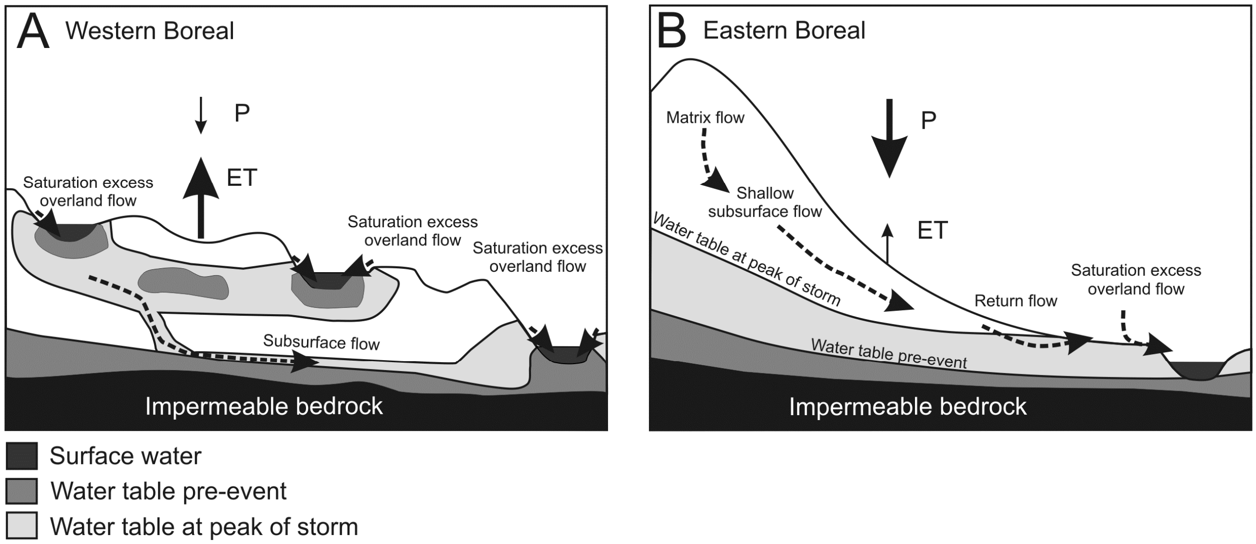
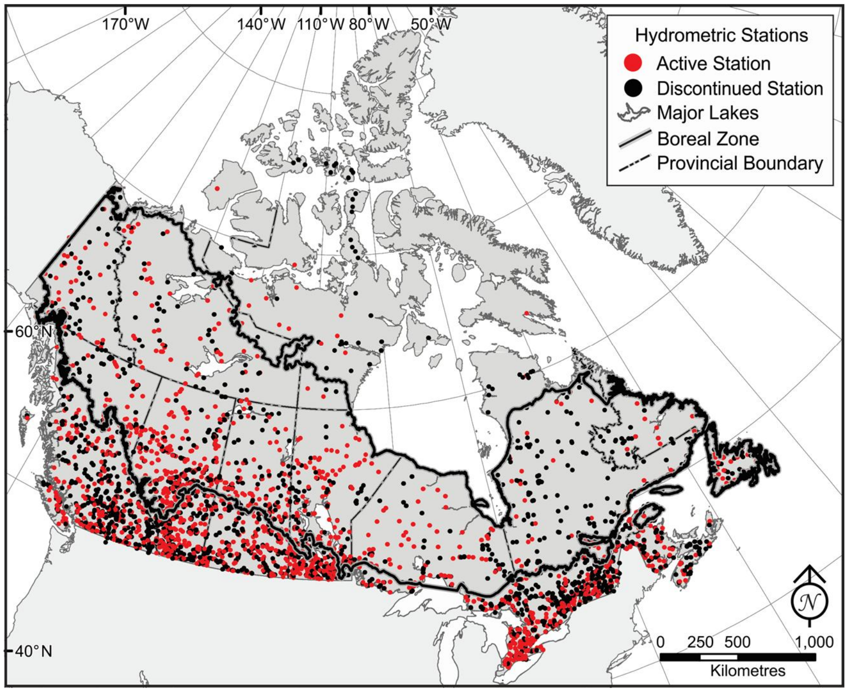
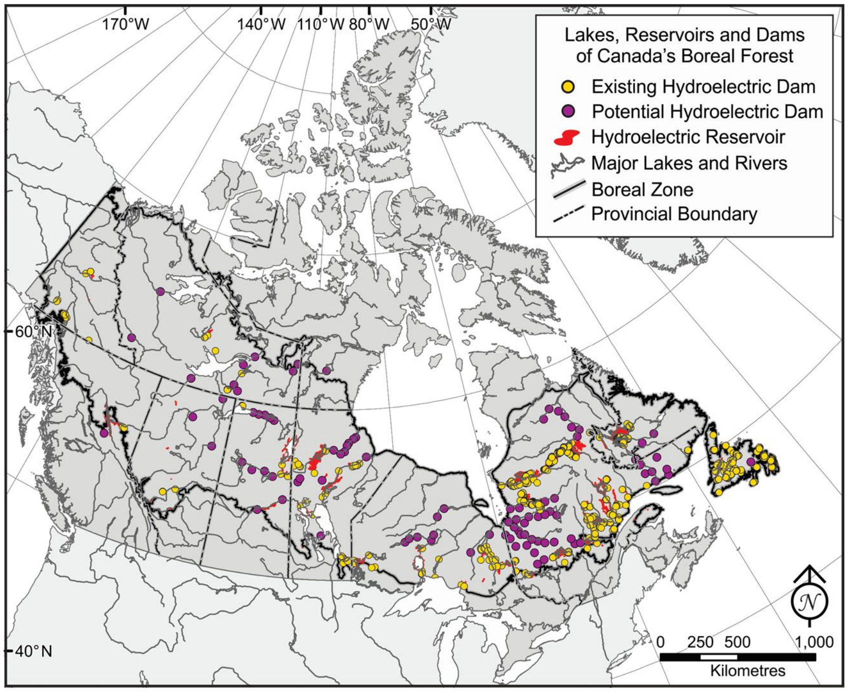

A
Western Boreal
P
ET
Saturation excess
overland flow
Saturation excess
overland flow
Saturation excess
overland flow
Subsurface flow
Impermeable bedrock
B
Eastern Boreal
Matrix flow
P
Shallow
subsurface flow
ET
Water table at peak of storm
Return flow
Saturation excess
overland flow
Water table pre-event
Impermeable bedrock
Surface water
Water table pre-event
Water table at peak of storm

Figure 5:



170°W
140°W
110°W
80°W
50°W
60°N
40°N
Hydrometric Stations
Active Station
Discontinued Station
Major Lakes
Boreal Zone
Provincial Boundary
N
0
250
500
1,000
Kilometres

Figure 6:



170°W
140°W
110°W
80°W
50°W
60°N
40°N
Lakes, Reservoirs and Dams
of Canada's Boreal Forest
Existing Hydroelectric Dam
Potential Hydroelectric Dam
Hydroelectric Reservoir
Major Lakes and Rivers
Boreal Zone
Provincial Boundary
N
0
250
500
1,000
Kilometres

Figure 7:

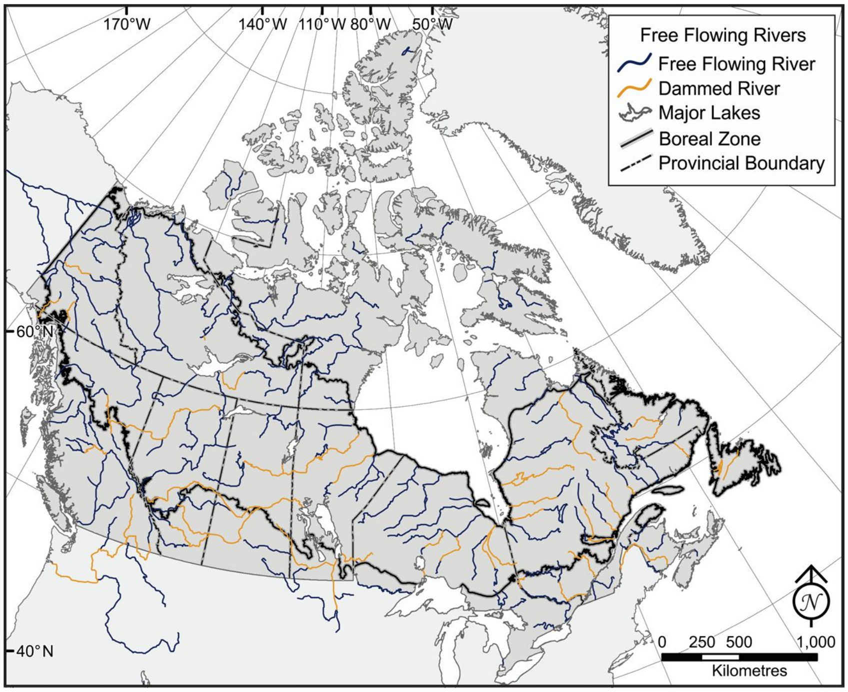
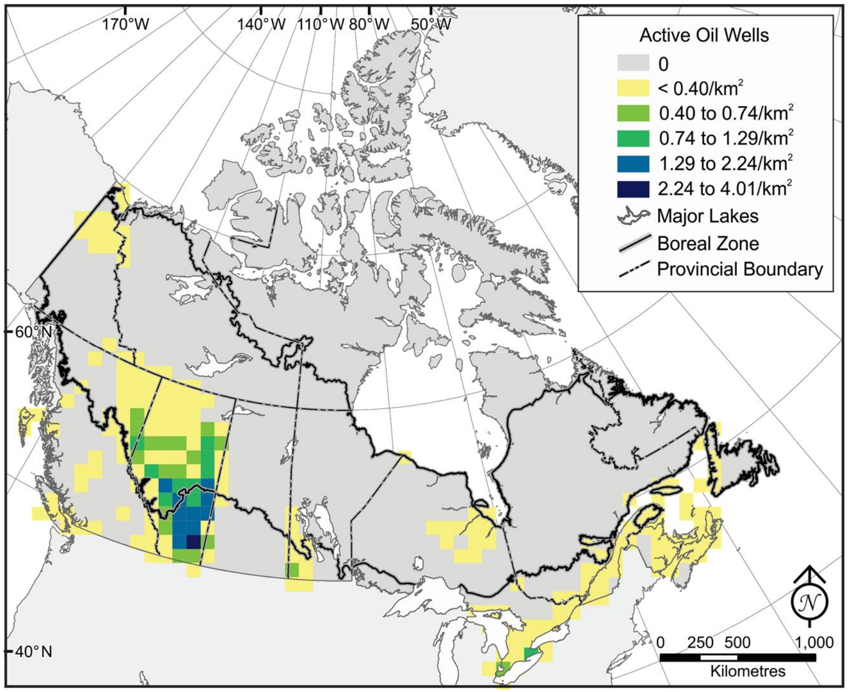

170°W
140°W
110°W
80°W
50°W
60°N
40°N
Free Flowing Rivers
Free Flowing River
Dammed River
Major Lakes
Boreal Zone
Provincial Boundary
0
250
500
1,000
Kilometres

Figure 8:



170° W
140° W
110° W
80° W
50° W
60° N
40° N
Active Oil Wells
0
< 0.40/km²
0.40 to 0.74/km²
0.74 to 1.29/km²
1.29 to 2.24/km²
2.24 to 4.01/km²
Major Lakes
Boreal Zone
Provincial Boundary
N
0 250 500 1,000
Kilometres

Figure 9:

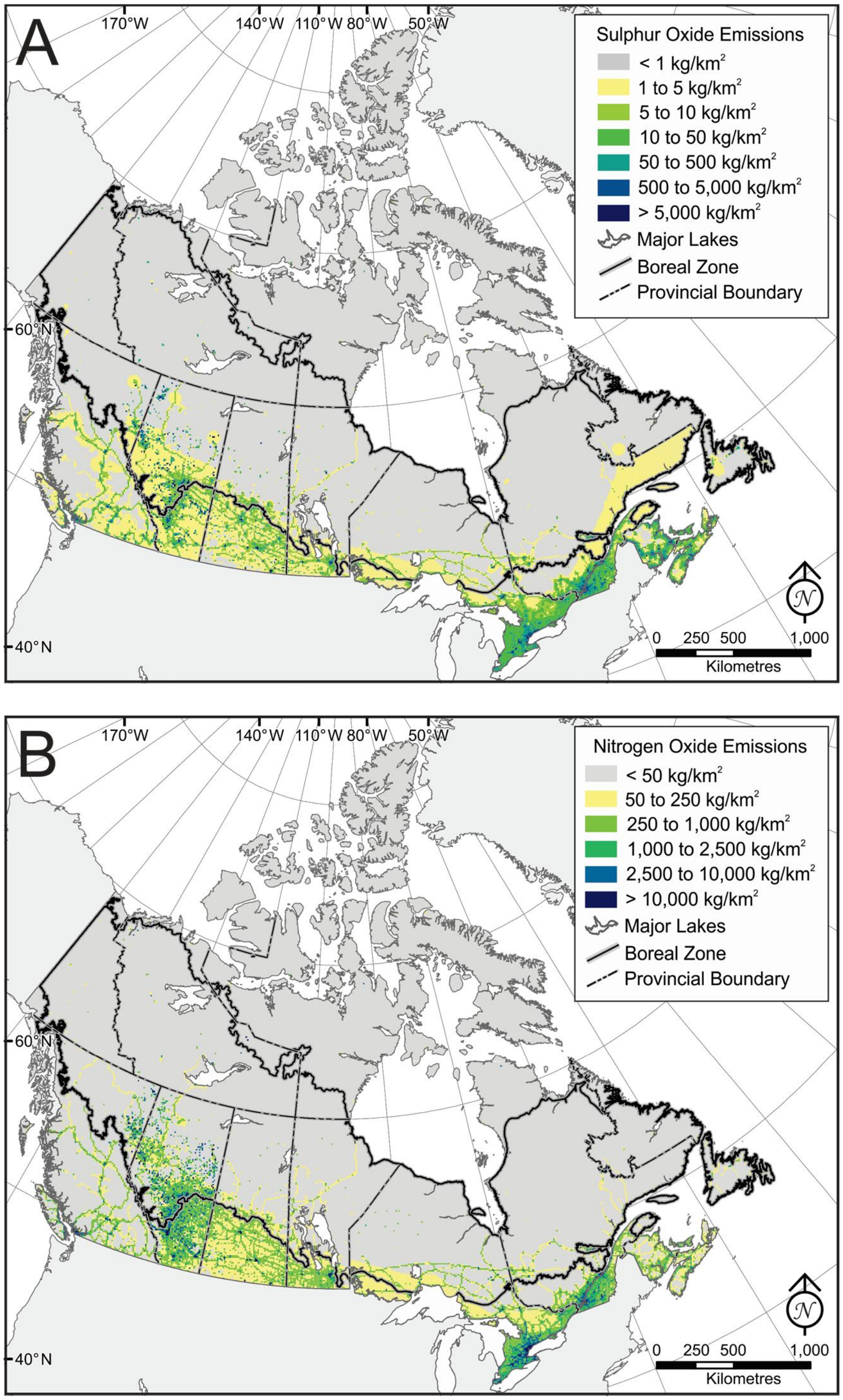

A
170°W
140°W
110°W
80°W
50°W
60°N
40°N
Sulphur Oxide Emissions
< 1 kg/km^2
1 to 5 kg/km^2
5 to 10 kg/km^2
10 to 50 kg/km^2
50 to 500 kg/km^2
500 to 5,000 kg/km^2
> 5,000 kg/km^2
Major Lakes
Boreal Zone
Provincial Boundary
0 250 500 1,000
Kilometres
B
Nitrogen Oxide Emissions
< 50 kg/km^2
50 to 250 kg/km^2
250 to 1,000 kg/km^2
1,000 to 2,500 kg/km^2
2,500 to 10,000 kg/km^2
> 10,000 kg/km^2
Major Lakes
Boreal Zone
Provincial Boundary
0 250 500 1,000
Kilometres

Figure 10:

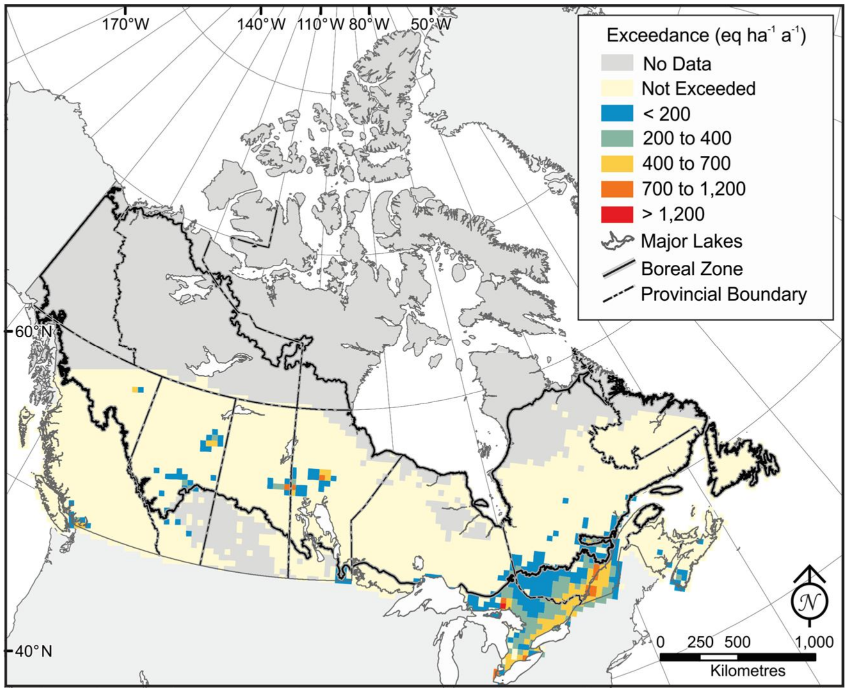

170°W
140°W
110°W
80°W
50°W
60°N
40°N
Exceedance (eq ha⁻¹ a⁻¹)
No Data
Not Exceeded
< 200
200 to 400
400 to 700
700 to 1,200
> 1,200
Major Lakes
Boreal Zone
Provincial Boundary
N
0 250 500 1,000
Kilometres

Figure 11:

Table 1: Boreal aquatic ecosystem services.

Service	Function		
Provisioning services	Wetland	Surface waters	Groundwater
Fibre and fuel/energy	Treed peatlands for forestry. Peat mining for peat moss products and bioenergy.	Water used in hydroelectric power generation and for cooling in thermal power generation.	NA[1.]
Food	Used as food for people and domestic animals/wood, fur and medicine	Used as food for people and domestic animals, fur and medicine	NA[1.]
Fresh water	Public and industrial water supply is obtained from reservoirs draining peatlands. Peatlands lost as freshwater source when drained for agriculture or forestry. Water quality compromised by waste disposal or landfill. Flooding of wetlands during reservoir creation lead to methyl mercury production.	Surface water used for public water supply, irrigation and resource extraction and processing. Transport of nutrients, organic matter and contaminants. Water quality compromised by waste disposal.	Groundwater used for public water supply, irrigation and resource extraction and processing. Surface water recharge. Transport of nutrients, organic matter and contaminants. Water quality compromised by waste disposal.
Regulating services			
Climate regulation	Regulation of greenhouse gases, regulation of climatic processes	Temperature moderation.	Temperature moderation indirectly through connections to surface water.
Water regulation	Water storage, groundwater recharge and discharge	Water storage, groundwater recharge and discharge	Water storage, groundwater recharge and discharge
Water purification and waste treatment	Retention, recovery and removal of excess nutrients and pollutants	Retention, recovery and removal of excess nutrients and pollutants	Retention, recovery and removal of excess nutrients and pollutants
Erosion protection	Peat blanket protecting the underlying soils from erosion	Local scouring of shoreline and re-deposition.	Maintenance of base flows.
Cultural Services			
Recreational and aesthetic	Opportunities for recreation and tourism; appreciation of nature	Opportunities for recreation and tourism; appreciation of nature	NA[1.]
Spiritual and inspirational	Personal feelings and well-being; religious significance	Personal feelings and well-being; religious significance	Personal feelings and well-being; religious significance
Educational	Opportunities for education, training and research	Opportunities for education, training and research	Opportunities for education, training and research
Supporting services			
Biodiversity	Habitats for species	Habitats for species	Habitats for species
Soil formation	Accumulation of organic matter	Accumulation of sediments.	NA[1.]
Nutrient cycling	Storage, recycling, processing and acquisition of nutrients	Transport and instream processing of nutrients	Transport and modification of nutrients.

[1.] Ecosystem service is not applicable.

Table 2: Water use in Canada's natural resource sectors. Sources: NRTEE (2010), Statistics Canada (2010) and NRCan (2011).

	Forestry and Pulp and Paper	Electrical power generation		Mining	Oil & Gas
		Hydro Power	Thermal Power		
Contribution to Gross Domestic Product (%)	1.9	2.3		2.8	3.4
National Gross Water Use (%)	5	Unknown	64	4	9.6[1.]
National Consumptive Water Use (%)	2	Unknown	12	3	6.3[1.]
Regions Most Implicated	All provinces	BC, AB, MB, ON, QC, Atlantic	Territories, AB, ON, Atlantic	BC, AB, SK, Territories	AB, BC, SK

[1.] Petroleum and coal product manufacturing

Table 3: Summary of impacts of natural resource development on water resources in the boreal.

Development type	Local area impacted	Cumulative boreal area Impacted	Primary effects on water quantity	Primary effects on water quality	Length of effect (years)	Key references
Roads	Narrow, linear disturbance	Large (when you incorporate density) 93×10^6 ha (Anielski and Wilson 2005) 600 000 km (Pasher et al. 2013)	Change to volume, timing and routing. Raise upslope water table, lower downslope water table	Sediment, contaminants and nutrients	100+	Trombulak and Frissel 2000; Whemple and Jones 2003; Partington and Gilles 2010
Forest Management	Large, irregular patches	Large 14.4×10^6 ha (Pasher et al. 2013)	Increase ("watering up" from vegetation removal) or decrease (increase solar heating or draining as occurs in treed wetlands)	Change in nutrient export.	5-10 yr	Buttle and Metcalfe 2000; Buttle et al. 2000, 2005, 2009; Mallik and Teichert 2009; Smith et al. 2007; See also Table 5
Pulp and Paper Industry	Small site disturbance but discharge of effluents has greatest impact over larger scale	Small to Medium (when incorporate downstream effects) 17 mills × 100 ha approximate footprint = 1 700 ha but downstream impacts 20+ km	Small decrease (most flow returned)	Contaminants and nutrients Temperature increase	>10 yr	Chambers 2001; NCASI 2010
Electricity Generation	Medium (thermal) to large (hydroelectric)	Large (including downstream effects 130 000 km of rivers affected by large dams (McAllister 2000) 5.2×10^6 ha reservoir area (Lee et al. 2012)	Increase upstream (flooding) and decrease downstream (lower and irregular flows)	MeHg and nutrient export (hydroelectric) Temperature alterations (thermal)	100+	Baxter 1977; Urquizo et al. 2000; Dynesius and Nilsson 1994; Urquizo et al. 2000
Minerals and Metal Mining	Medium to large depending on mine size and mode of extraction	Large 123 active + 1300 abandoned mines × 1000 ha approximate footprint = 1.4×10^6 ha	Decrease in stream flow due to removals. Increase where tailings ponds created Groundwater table declines	Contaminants and nutrients	1000+	Ptacek et al. 2004; Keller et al. 2007

			due to dewatering			
Oil and gas and oil sands development	Small (individual wells) to large (oils sand intensive surface extraction)	Large (when you incorporate density) 46 $\times 10^6$ ha (Timoney and Lee 2001) with 56 000 ha of actively mined oil sand (Alberta Environment http://environment.alberta.ca/02863.html)	Decrease in stream flow due to removals. For surface extraction of oil sand end pit lakes created, but groundwater table declines due to dewatering	Contaminants and nutrients	1000+	Woynillowicz et al. 2005; Gosselin et al. 2010; Kasperski and Mikula 2011; CEMA 2012; Government of Alberta 2012b, 2013a
Peat mining	Small local impact although draining may affect larger area	Small 17 000 ha (Daigle and Gautreau-Daigle 2001)	Decrease (during drainage) and increase (when drains plugged)	Nutrients	1000+	Daigle and Gautreau-Daigle 2001; Price et al. 2005; Gleason et al. 2006

Table 4. Description of boreal hydrology research sites in boreal including information on duration, treatments, and key references.

Site	Location	Area	Project Duration	Pre-harvest years	Year of Harvest	Post-harvest	Forest Operations	Research Infrastructure	Hydrologic and water quality metrics	Primary Reference (study design)
Wolf Creek, YT	60°32'N 135°07W	14.5 – 195 km^2	1992-present	NA	NA	NA	undisturbed system	-3 met. stations -piezometers -soil moisture meas. -3 stream gauges (nested) -stream chemistry -25 snow courses (monthly) -water quality (weekly) -contaminant sampling: water, snow & fish	-daily avg. SWE	Pomeroy & Granger 1999
Spring Creek, AB	54°56'N 117°43'W	1,860 – 1,971 ha	1966 - 2000	1966 - 1993	1994	1977 - 2002	control and clearcut	-Discharge -Air Temp -Precipitation -Other Met Variables - soil moisture and temp	- evapotranspiration - sediment yield	Swanson and Rowell 2001 Martz 1988 Watertight Solutions Ltd. 2005
Tri Creek, AB	53°16'N 117°14'W	1,480 – 2,820 ha	1969 - 1985	3	1974 - 1983	1983 - 1985	yes, unknown treatments	-Discharge -Air Temp -Precipitation -Water Chemistry -Water Temp -Sediment -Other Met Variables	NA	Currie 1976 Stevens 1979 Bergstrom and Nip 1988 Nip 1991 Van Der Vinne 1992 Andres et al. 1987
Hydrology, Ecology and Disturbance (**HEAD**) (aka USRA **Utikuma** Study Research Area), AB	56˚06'N 116˚32'W	0.05 – 5 km^2	2001-2004	NA	NA	NA	undisturbed system	@ *24 lakes* -lake level -standard limnological variables: TP, TN, chlorophyll a, DOC, conductivity (3 x/yr, May, Jul, Aug) @ *3 intensively studied lakes + catchments* -met station: T, P, wind -hundreds of piezometers (some w. continuous head meas.) (manual meas. 1-2x per month during ice free season, otherwise every two months) -some deep groundwater wells -lake w seepage meters -lake levels, inflow and outflow meas.	-no flow data -groundwater fluctuation at one intensively measured site -saturated and inundated areas measured from ERS satellite	Smerdon et al. 2005
Terrestrial and Riparian Organisms, Lakes, and Streams (**TROLS**), AB	54°46'N 111°59'W	189-5669 ha	1994-1998	1994-1995	1996-1997	1998	-3 controls with little to no logging -8 logged basins ranging from 9-35% clear cut	-12 lakes with limnological & aquatic ecology -3 WSC stream gauges -met. data from nearby EC sites -water quality measurements (2x/month) -chemical analysis -aquatic ecology -fish studies	-median total Phosphorus concentration [TP] used to illustrate influence of surface and subsurface hydrologic connections and impacts of logging on lakes	Devito et al. 2000 Prepas et al. 2001a

Site	Location	Area	Project Duration	Pre-harvest years	Year of Harvest	Post-harvest	Forest Operations	Research Infrastructure	Hydrologic and water quality metrics	Primary Reference (study design)
Forest Watershed and Riparian Disturbance (**FORWARD**), Swan Hills, AB	54°12'N 115°46'W	131 – 248 km^2	1998-2004	1983-1997 (WSC)	1998	1999-2000	-89% burned 1998, 10% logged	-3 WSC gauges (day) -4 met. stations (P, T, rel H, wind speed & direction., incoming solar, plus 8 additional P gauges) -stream chemistry – ISCO's (1-2 week grab samples) -1 EC met. station (Whitecourt)	-baseflow, summer stormflow -TP export	Smith et al. 2003 Prepas et al. 2003
Duck Mountain, MB	51˚01'W 100˚39'N	103-4230 km^2	-earliest WSC site is 1954; range is 10 – 40 years -3 active gauges, 11 discontinued				-Louisiana Pacific is license holder	-14 WSC gauges drain Duck Mountain, seasonal (day) -9 MSC met. stations in the region	-water yield -seasonal runoff -peak flows -daily Q	unpublished draft report by Watertight Solutions Ltd. 2005
BOREAS, Thompson MN, & Prince Albert SK	55.91°N 98.49°W 53.86˚N 104.62˚W	**NSA** 8000 km^2 **SSA** 11170 km^2	1993-1997				undisturbed system	- discharge - air temp - precipitation - other met variables - numerous stream gauges -numerous met. measurements -remote sensing -soil moisture -snow measures -flux towers	-water balance -ksat -SWE -daily hydrograph separations	Sellers et al. 1995 Metcalfe & Buttle 1999, 2001
Coldwater Lakes, ON	49˚10'N 92˚10'W	70-194 ha	1990-1999	1990-1995	1996 & 1998	1999	**L26** – 45% clear cut no shoreline disturbance **L39** – 77% clear cut with shoreline disturbance **L42** – 74% clear cut with shoreline disturbance -2 phase logging 1996 & 1998 - feller buncher & chain saw -aerially seeded in 1998	-stream gauges at inflow and outflow of 3 headwater lakes -1 land met. Station (1993), additional floating met stations on lakes -water quality & chemistry measurements (bi-weekly) -aquatic ecology (montly May to Sep) -lakes not hydrologically independent, single lake approach	-limnological study of water quality	Steedman 2000 Nicholls et al. 2003
Pukaskwa River (aka **White River)**, ON	48°03'N 85°48'W	537 km^2 (total area)	2002-present WSC: 1940 - present	2002-2005	2006	2007	-logging within the watershed starting 2000 -logging within experimental catchments in 2006	-1 WSC & 3 OMNR seasonal stream gauges -1 OMNR & 1 GLFC seasonal met. stations	NA	-no publications to date

Site	Location	Area	Project Duration	Pre-harvest years	Year of Harvest	Post-harvest	Forest Operations	Research Infrastructure	Hydrologic and water quality metrics	Primary Reference (study design)
Current River, ON	48°33'N 89°14'W	403 km^2 (total area)	2002-present WSC: 1968 - present	2002-2003	2004	2005-present	-extensive logging began 1995 in main watershed -logging in experimental watershed 2004	-1 WSC (1968) & 2 OMNR (2002) stream gauges -1 met. station at Lakehead University	NA	-no publications to date
Little Abitibi, ON	49.07˚N 81.03˚W	1408 km^2 (total area)	2003-present WSC: 1955 - present			2003 - present	-extensive logging using various methods throughout	-20 Trent stream gauges (2003) -3 met. stations -10 standard rain gauges	NA	-no publications to date
Esker lakes Research Area, ON	49.38˚N 81.01˚W	200 km^2 (total area)	2001-present	2001-2004	2005 (winter)	2005 - present	-extensive logging using clear cut (dominant) and variable retention methods	- 14 lakes with piezometers 10 lakes each with 3 zero-tension lysimeters and 6 tension (porous cup) lysimeters - Three rain gauges in study area - Water level indicators on 21 lakes	- Deep and shallow soil groundwater movement and chemistry - O^{18} measurements on groundwater - Lake water chemistry (2001-present) - Seepage meter measurements (2006) - Mini piezometer measurements of lake sediments - Snow melt and runoff data for 3 intensive lakes (2004-present) - Throughfall (terrestrial and nearshore)	1) Hazlett et al 2005 2) Hazlett et al 2006 3) Starr et al 2006
Lac Laflamme, **Montmorency** Forest, QC	47°19'N 71°07'W	68 ha	1981-1997	NA	NA	NA	undisturbed system	-18 -27 snow sampling stations -9 stream gauging stations (hourly) -1 met. station: air T & P -snow & soil moisture collected 2-7 days Mar-May -lake levels -modelling: VSAS2	-daily Q and snow water content -peak flow	Prévost et al. 1990

Site	Location	Area	Project Duration	Pre-harvest years	Year of Harvest	Post-harvest	Forest Operations	Research Infrastructure	Hydrologic and water quality metrics	Primary Reference (study design)
Ruisseau des Eaux-Vollées Experimental Watershed (**REVEW)**, Montmorency Forest, QC	47°16'N 71°09'W	122 - 917 ha	1967-1998	*1)* 1967-1974 *2)* 1985-1992	*1)* 1974-1975 *2)* 1993	*1)* 1976-1978 *2)* 1994-1998	*1)* 1974-1976 single patch cutting 31% of **6** *2)* 1993 85% basal area removal thru clearcut **7A** -harvester & forwarder -5-20 m buffers	-3 stream gauges (nested): **Main** (**5D**), **6**, **7A**; V-notch weirs (hr) -paired-watershed & single basin approaches *1)* **6** was treatment & **7** was control *2)* Second experiment was reverse	-peak flows -quick flow volume -baseflow volume -total storm flow volume -lag time -concentration time -rise time -falling time -base time	Guillemette et al. 2005 Plamondon & Ouellet 1980
Cote Nord & Haute – Mauricie, QC	50˚43'N 67˚30'W 48˚31'N 73˚25'W	0.83 – 4.09 km^2	1974-1977	1974	1975 1977	NA	**Cote Nord** (1975) -**7A**, **7B**, **7C, 5A** & **5C** controls -**5G** 62% clearcut with 10 m buffer -**5H** 26% clearcut no buffer -chain saws & rubber tire skidders -**5G** & **5H** are nested in **5C** **Haute Mauricie** (1977) -**Gilbert** 100% clearcut no buffer -**Huguette** 40% clearcut 30 m buffer -**Wajusk** control -chain saw & skidders and feller buncher	-suspended sediments -water T -water quality and chemistry -paired basin	-concentration of inorganic suspended sediments - CA, K, DO, FE -pH, colour and conductivity	Plamondon et al. 1982
Copper Lake, NF	48˚50'N 57˚50'W	114 – 124 ha **Copper Lake** – 13.5 km^2	1993-1997	1993	1994-1996	1996-1997	1994- road construction through both wsheds **T1-1** – 18% clearcut no buffer **T1-2**- 26% clearcut w. 20m buffer **T1-3** & **T1-5**, controls -2 phases winter 1994-1995 & summer 1996	-stream surveys -lake bathymetry -stream gauge at outflow from **Copper Lake** (WSC) -water chemistry (monthly) -fish studies -stream T (hourly), 10 recorders -suspended sediment sampling (~2x/year) -benthic invert surveys (annual)	-average monthly discharge -seasonal T	Curry et al. 2002 Clarke et al. 1997

Site	Location	Area	Project Duration	Pre-harvest years	Year of Harvest	Post-harvest	Forest Operations	Research Infrastructure	Hydrologic and water quality metrics	Primary Reference (study design)
Triton Brook, NL & LB	48°36'N 54°35'W	23,297 ha	July 2001 – present	2001	2002	2002-present	-mechanical harvesting, fire, infestation over last 100 years -no experimental harvest -reforestation efforts ongoing (scarification & replanting)	-water quality at 3 sites, manual, (bi-weekly Jun to Oct) -WSC stream gauge (daily) -modelling; single basin	-daily Q	NA

Table 5: Impacts of forest management practices on water quality.

Parameter	Disturbance	Location	Length of study	Impact	Author
Temperature of water	Wind; forestry	BC, Stuart-Takla	6 years	+ ; + +	Macdonald et al, 2003
	Wind; forestry	Ontario, Coldwater lakes	8 years, 5 years pre-harvest, 3 years post-harvest	+ Thermocline depth	Steedman and Kushneriuk, 2000
	Forestry	Ontario, White River	4 years, 2 pre-harvest 2 post-harvest	+ + without buffers	Kreutzweiser et al, 2009b
	Forestry	Ontario, Coldwater lakes	4 years, 1 year pre-harvest, 3 years post-harvest	0	Steedman et al, 2001
	Forestry	Quebec, Montmorency forest	5 year study, 2 years pre-harvest, 3 years post-harvest	0	Tremblay et al, 2009
pH	Fire; forestry	Quebec, Gouin Reservoir	3 years immediately post-fire; post-harvest	0; 0	Carignan et al, 2000a
	Fire; forestry	Quebec	3 years immediately post-fire; post-harvest	0; 0	Garcia et al, 2007
	Beaver dam; forestry	Alberta; Rocky Creek	3 years	+; 0	Hillman et al, 1997
	Forestry	Quebec, Laurentian Mountains	5 months, post-harvest	0	Hausmann and Pienitz, 2009
	Forestry	Alberta, TROLS study area	4 years, during harvest	0	Prepas et al, 2001a
	Forestry	Ontario, ELA	3 years, post-harvest	0	Steedman, 2000
	Forestry	Quebec, Montmorency Forest	5 years, 2 years pre-harvest, 3 years post-harvest	-	Tremblay et al, 2009
Nutrient (N,P)	Fire; forestry	Quebec, Gouin Reservoir	3 years, post-fire; post-harvest	+ P, + N; + P, + N	Carignan et al, 2000
	Fire; forestry	Quebec, Gouin Reservoir	3 years, post-fire; post-harvest	+ P, + N; + P, + N	Garcia et al, 2007
	Fire; forestry	Quebec, Gouin Reservoir	3 years, post-fire, post-harvest	+ P, + N; 0 P, 0 N	Garcia and Carignan, 2000
	Fire; forestry	Quebec, Haute-Mauricie	3 years, post-fire; post-harvest	+ P, + N; + P, + N	Lamontagne et al, 2000
	Fire; forestry	Alberta, Caribou Mountains	1 year, 2 years post-fire	+ P, + N; + P, + N	McEachern et al, 2000
	Forestry	Quebec, Gouin Reservoir	3 years, some during harvest	0 P 0 N	Bertolo and Magnan, 2007
	Forestry	Alberta, Moose Lake	1 year, post-harvest	0 P	Evans et al, 2000
	Forestry	Ontario, Experimental Lakes	1 year	- - P	France et al, 1996

Parameter	Disturbance	Location	Length of study	Impact	Author
	Forestry	Quebec	5 months, post-harvest	+ P + N	Hausmann and Pienitz, 2009
	Forestry; beaver dams; roads	Alberta	3 years, during harvest	0 N; + N; 0 N	Hillman et al, 1997
	Forestry	Ontario, White River	5 weeks, post-harvest	0 P 0 N	Kreutzweiser et al, 2008a
	Forestry	Alberta	4 years, 2 years pre-harvest, 2 years post-harvest	+ P 0 N	Prepas et al, 2001a
	Forestry	Ontario, Coldwater Lakes	8 years, 5 years pre-harvest, 3 years post-harvest	0 P + N	Steedman, 2000
	Forestry	Quebec, Montmorency Forest	5 years, 3 years pre-harvest, 2 years post-harvest	0 P + + N	Tremblay et al, 2009
	Forestry	Quebec	2 years, 1 year pre-harvest, 1 year post-harvest	+ P 0 N	Winkler et al, 2009
DOC	Fire; forestry	Quebec, Gouin Reservoir	3 years post-fire; post-harvest	0 ; +	Carignan et al, 2000
	Fire; forestry	Ontario, Quebec	13 year study, 4-13 years post-burn or harvest	+ ; +	France et al, 2000
	Fire; forestry	Quebec	3 years, post-fire and post-harvest	0 ; +	Garcia et al, 2007
	Fire; forestry	Quebec, Haute-Mauricie	3 years, post-fire and post-harvest	0 ; +	Lamontagne et al, 2000
	Forestry	Quebec	3 years, during harvest	+	Bertolo and Magnan, 2007
	Forestry	Ontario, Experimental Lakes	1 year	- -	France et al, 1996
	Forestry	Quebec, Laurentian Mountains	5 months, post-harvest	+ +	Hausmann and Pienitz, 2009
	Forestry; beaver dams; road	Alberta	3 years, 1 year pre-harvest, 2 years post-harvest	0 ; - ; 0	Hillman et al, 1997
	Forestry	Ontario, Cold-water lakes	4 years, post-harvest	0 (any increase seen could not be attributed to cut)	Knapp et al, 2003
	Forestry	Ontario, Turkey Lakes	5 weeks, post-harvest	0	Kreutzweiser et al, 2008
	Forestry	Ontario, Cold-water lakes	4 years, 2 years pre-harvest, 2 years post-harvest	+	Steedman, 2000
	Forestry	Quebec	2 years, 1 year pre-harvest, 1 year post-harvest	+	Winkler et al, 2009

Parameter	Disturbance	Location	Length of study	Impact	Author
	Forestry	Quebec	2 years, 1 year pre-harvest, 1 year post-harvest	0	Desrosiers et al, 2006a
Dissolved oxygen	Forestry	Ontario, Coldwater lakes	8 years, 5 years pre-harvest, 3 years post-harvest	0	Steedman and Kushneriuk, 2000
	Forestry	Quebec	2 years, 1 year pre-harvest, 1 year post-harvest	0	Winkler et al, 2009
Base cations	Fire; forestry	Quebec, Gouin Reservoir	3 years post-fire; post-harvest	+ ; 0	Carignan et al, 2000
	Fire; forestry	Quebec	3 years post-fire; 3 years post-harvest	+ ; 0	Garcia et al, 2007
	Fire; forestry	Quebec, Gouin Reservoir	3 years post-fire; post-harvest	+ ; +	Lamontange et al, 2000
	Fire; forestry	Quebec, Gouin Reservoir	3 years post-fire; post-harvest	0 ; 0	Garcia and Carignan, 2000
	Beaver dam; forestry; roads	Alberta	3 years	+ ; 0; 0	Hillman et al, 1997
	Forestry	Ontario	5 weeks post-harvest	0	Kreutzweiser et al, 2008a
	Forestry	Ontario, Coldwater Lakes	8 years, 5 years pre-harvest, 3 years post-harvest	-	Steedman, 2000
	Forestry	Quebec, Montmorency Forest	5 years, 3 years pre-harvest, 2 years post-harvest	+	Tremblay et al, 2009
Acid anions	Fire; forestry	Quebec, Gouin Reservoir	3 years post-fire; post-harvest	+ + ; 0	Carignan et al, 2000
	Fire; forestry	Quebec, Gouin Reservoir	3 years post-fire; post-harvest	+ ; 0	Lamontagne et al, 2000
	Fire; forestry	Canada, meta-analysis	Varied	+ ; -	Nitschke, 2005
	Forestry	Quebec, Laurentian Mountains	5 months post-harvest	0	Hausmann and Pienitz, 2009
	Forestry	Alberta, TROLS area	4 years, harvest at year 2	0	Prepas et al, 2001a
	Forestry	Ontario, ELA	8 years, 5 years pre-harvest, 3 years post-harvest	0	Steedman and Kushneriuk, 2000
	Forestry; fire	Quebec, Gouin Reservior	3 years, post-harvest; 3 years, post-fire	0 ; +	Garcia and Carignan, 2000