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REVIEW

# Wetlands in the Athabasca Oil Sands Region: the nexus between wetland hydrological function and resource extraction

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**Abstract:** Oil sands development within the Athabasca Oil Sands Region (AOSR) has accelerated in recent decades, causing alteration to natural ecosystems including wetlands that perform many vital ecosystem functions such as water and carbon storage. These wetlands comprise more than half of the landscape, and their distribution and local hydrology are the result of interactions among a subhumid climate, topography, and spatially heterogeneous surficial and bedrock geology. Since hydrology plays a fundamental role in wetland ecological functioning and determines wetland sensitivity to human disturbances, the characterization of anthropogenic impacts on wetland hydrology in the AOSR is necessary to assess wetland resilience and to improve current best management practices. As such, this paper reviews the impacts of oil sands development and related disturbances including infrastructure construction, gravel extraction, and land clearing on wetland function in the AOSR. Hydrologic disturbances in wetlands in the AOSR include changes to soil hydrophysical properties that control water table position, the interruption of recharge–discharge patterns, and alteration of micrometeorological conditions; these in turn govern wetland ecological structure and wetland ecosystem processes (e.g., evapotranspiration, nutrient cycling). Given that anthropogenic disturbance can affect natural wetland succession, long-term hydrological monitoring is crucial for predicting the response of these ecosystems to varying levels of human impact.

*Key words:* wetlands, hydrologic functioning, Athabasca Oil Sands Region, human disturbance, anthropogenic impact.

**Résumé :** L'exploitation des sables bitumineux de l'Athabasca (AOSR, *Athabasca Oil Sands Region*) s'est accélérée au cours des dernières décennies, provoquant une altération des écosystèmes naturels, dont les zones humides qui remplissent nombre de fonctions vitales dans les écosystèmes telles que le stockage d'eau et de carbone. Ces zones humides constituent plus de la moitié du paysage, et leur distribution et leur hydrologie locale sont le résultat des interactions entre le climat subhumide, la topographie et la géologie de surface et de substrat rocheux spatialement hétérogène. Puisque l'hydrologie joue un rôle fondamental dans le fonctionnement écologique des zones humides et qu'elle détermine la sensibilité des zones humides aux perturbations humaines, la caractérisation des impacts anthropiques sur l'hydrologie des zones humides de l'AOSR est nécessaire pour évaluer la résilience des zones humides et améliorer les meilleures pratiques de gestion actuelles. De ce fait, cet article fait la synthèse des impacts de l'exploitation des sables bitumineux et des perturbations associées, y compris la construction d'infrastructures, l'extraction de gravier et le défrichement sur le fonctionnement des zones humides de l'AOSR. Les perturbations hydrologiques dans les zones humides de l'AOSR comprennent des changements des propriétés hydrophysiques du sol qui contrôlent la position de la nappe phréatique, l'interruption des cycles de recharge/décharge et l'altération des conditions micrométéorologiques; celles-ci contrôlent en retour la structure écologique des zones humides et les processus de l'écosystème des zones humides (par exemple l'évapotranspiration, le cycle des nutriments). Puisque ces perturbations anthropiques peuvent affecter la succession naturelle de zones humides, le suivi hydrologique à long terme est essentiel pour prédire la réponse de ces écosystèmes aux différents niveaux d'impacts humains. [Traduit par la Rédaction]

*Mots-clés :* zones humides, fonctionnement hydrologique, région des sables bitumineux de l'Athabasca, perturbation humaine, impact anthropique.

## 1. Introduction

Wetlands represent a relatively small proportion (4%–9% of land area, 5.3–12.8 million km$^2$) of Earth's total land area (Zedler and Kercher 2005), yet they store roughly 20% (450 Gt C) of the total terrestrial carbon pool (Yu 2012). These ecosystems also serve important hydrological roles, including water storage and filtration and flood control (Ingram 1983), while providing unique assemblages of vegetation communities (Vitt et al. 1996) and natural habitat for wildlife (Mitsch and Gosselink 1993). Wetlands are vulnerable to a wide range of disturbances, including urbanization, peat extraction (for horticulture), livestock grazing, agriculture, forestry, mining, and resource exploration (Chimner et al. 2016).

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In the Athabasca Oil Sands Region (AOSR), wetlands dominate the landscape, covering more than 50 500 km² (~54% of total land cover) (AEP 2018) despite the subhumid climate with potential evapotranspiration (PET) often exceeding precipitation (Bothe and Abraham 1993; Devito et al. 2012). Wetlands vary in class, form, and type (Fig. 1a) and include mineral wetlands and peat-forming wetlands (peatlands), ranging from ombrogenous open bogs to treed extreme-rich fens (Vitt et al. 1996; Alberta Environment and Parks 2018), swamps (Locky et al. 2005), and saline fens (Grasby and Londry 2007; Trites and Bayley 2009*a*; Wells and Price 2015*a*). This variability of peatland type is influenced greatly by the deep and spatially heterogeneous topography and surficial geology (hereafter referred to as physiography) with varying sequences of soil textures and subsequent hydrophysical and geochemical properties that characterize the region (Fig. 1*b*) (Devito et al. 2005*a*; Andriashek 2003; Andriashek and Atkinson 2007).

The AOSR encompasses an area of 93 259 km² of the Western Boreal Plains (WBP) in northeastern Alberta (ABMI 2017); 5% of the AOSR (4750 km²) is deemed surface-mineable (Fig. 1*c*). As of 2017, an area of 895 km² has been impacted by open-pit oil sands mining (Government of Alberta 2017*b*). According to oil sands mine closure plans, the vast majority of peatlands destroyed by open-pit mining will not be reclaimed back to peatlands but rather reclaimed to upland forest (increase of 155 km²), end pit lakes (+36 km²), riparian shrubland (+23 km²), and marshes (+17 km²). This will result in the loss of over 295 km² of peatlands and the release of between 11.4 and 47.3 million metric tonnes of stored carbon and reduction of continued carbon sequestration by these peatlands of 5734–7241 metric tonnes C $y^{-1}$ (Rooney et al. 2012). However, recent efforts to reclaim peatlands on oil sands mines has shown preliminary establishment peatland plant communities and peat accumulation (Borkenhagen and Cooper 2019), although more monitoring is needed to determine long-term sustainability. The remainder of the AOSR has bituminous sands located deeper than 70 m that cannot be mined economically; in these areas, bitumen is extracted using in-situ techniques (e.g., steam-assisted gravity drainage (SAGD)) (Government of Alberta 2017*b*). SAGD is based on the injection of heated steam into the oil sands deposit through underground wells, thereafter the less viscous bitumen is pumped to the surface. In-situ bitumen extraction is less environmentally destructive as it has a significantly smaller human development footprint relative to open-pit mining, but it also requires construction of industrial facilities and infrastructure (e.g., well pads, tailing storages, access roads, etc.) (Jordaan et al. 2009). An inventory of wetland landscape transformation of in-situ oil sands lease areas (sensu Rooney et al. 2012) has not been performed.

Oil sands development is also associated with other human disturbances including gravel extraction, forestry, seismic exploration lines, transportation (permanent and semi-permanent access roads, highways, pipelines), and additional mining-related infrastructure and refining practices (Rooney et al. 2012; AEP 2016*a*). Urban development and road construction are relatively dispersed and have smaller spatial scales compared to mining; however, the total area of these disturbances exceeds 2100 km² (ABMI 2017). For example, recent urban expansion in Fort McMurray, based on historical satellite imagery (Google Earth), demonstrates more than 6 km² of wetlands have been removed since 2000. These disturbances can cause long-term changes to the hydrological function of peatlands and potentially further increase their vulnerability to other natural and human-induced drivers of change such as weather variability, climate change, wildfires, and invasive species that influence their ecosystem function and associated ecosystem services (Keshta et al. 2011; Rooney et al. 2012; Ireson et al. 2015; Waddington et al. 2015; Thompson et al. 2017; Ramsar Convention on Wetlands 2018).

To date, investigations into the degree of influence that oil sands development and the related disturbances have on the natural functioning of peatlands in the AOSR remain incomplete despite substantial efforts to summarize and synthesize hydrological functioning of northern peatlands (e.g., Devito et al 2005*a*, 2012; Pelster et al. 2008; Waddington et al. 2015), cumulative effects of multiple stressors related to human developments in the AOSR (e.g., Webster et al 2015; Dabros et al. 2018; Lima and Wrona 2019), and wetland restoration and construction (e.g., Graf 2009; Ketcheson et al. 2016). Generalization of results is confounded by the variability in ecohydrological function associated with different wetland classes and the duration of studies that are shorter than the decadal climate cycles typical of this region (Bothe and Abraham 1993; Marshall et al. 1999; Devito et al. 2012). This review summarizes the current state of knowledge with respect to peatland hydrological function in the AOSR and the impacts of oil sands development and related activities on peatland function. The overall goals of this paper are to: (*i*) outline the current state of knowledge with respect to peatland hydrologic functioning in the Boreal Plains ecozone, (*ii*) review the impacts of oil sands development and associated disturbances on peatland hydrology and identify research gaps, and (*iii*) use the subsequent framework to complement existing recommendations for monitoring the impacts of oil sands mining activities on wetlands in the AOSR. In this paper, "hydrologic functioning" refers to a complex of processes related to collecting, storing, contributing, and transmitting water that is fundamental to any landscape component (Black 1997).

## 2. Wetlands in the AOSR

### 2.1. Wetland distribution in the AOSR

The formation, distribution, and functioning of wetlands are a time-integrated result of interaction among climate, surface and bedrock geology, and topography over the past 10 000 years (Vitt et al. 1996; Halsey et al. 1998; Devito et al. 2005*a*). The combination of subhumid climate, heterogeneous surface sediments, highly variable topography, and complex hydrogeology cause a wide range of environmental conditions in AOSR wetlands that follow distinct hydrologic, chemical, and biotic gradients (Chee and Vitt 1989; Vitt et al. 1996; Smith et al. 2007; AESRD 2015). In the AOSR, the most abundant wetlands are fens (Vitt and Chee 1990; AESRD 2015) that occupy more than 22 540 km² (AEP 2018) (Fig. 2). Poor fens, that are more acidic and lower in dissolved ions, are common in headwaters (e.g., Stony Mountain, Birch Mountains), whereas moderate-rich fens tend to occur at topographical lows and areas with coarse-textured deposits (AESRD 2015). More than half of the fens are treed fens (poor and rich), whereas shrubby and graminoid fens (rich and poor) comprise about 25% (AESRD 2015; AEP 2018). Bogs are the second most abundant wetland class in the AOSR occupying more than 12 550 km² (AEP 2018); more than 80% of bogs in the AOSR are treed, whereas shrubby and open bogs are less common (AESRD 2015). Swamps occupy about 11 570 km² (AEP 2018) and are common at transitions between wetlands and adjacent uplands as well as in the bottom of river valleys (Stanek et al. 1977; Locky et al. 2005). More than half of the swamp areas in the AOSR are conifer-dominated, whereas shrubby, mixed-wood, and hardwood swamps are less abundant (AESRD 2015). Shallow water wetlands and marshes cover 3840 km² (AEP 2018) and tend to occur at riparian areas and along lakeshores as well as in areas with increased salinity (Bayley and Mewhort 2004; AESRD 2015). Marshes are generally characterized by complex surface water–groundwater interactions, large water table fluctuations, and temporally variable geochemical conditions (National Wetlands Working Group 1997).

**Fig. 1.** Map of the Athabasca Oil Sands Region showing: (*a*) wetland distribution (data from AEP (2018)); (*b*) surficial geology (modified from Fenton et al. (2013)); and (*c*) human disturbance (data from ABMI (2013)).

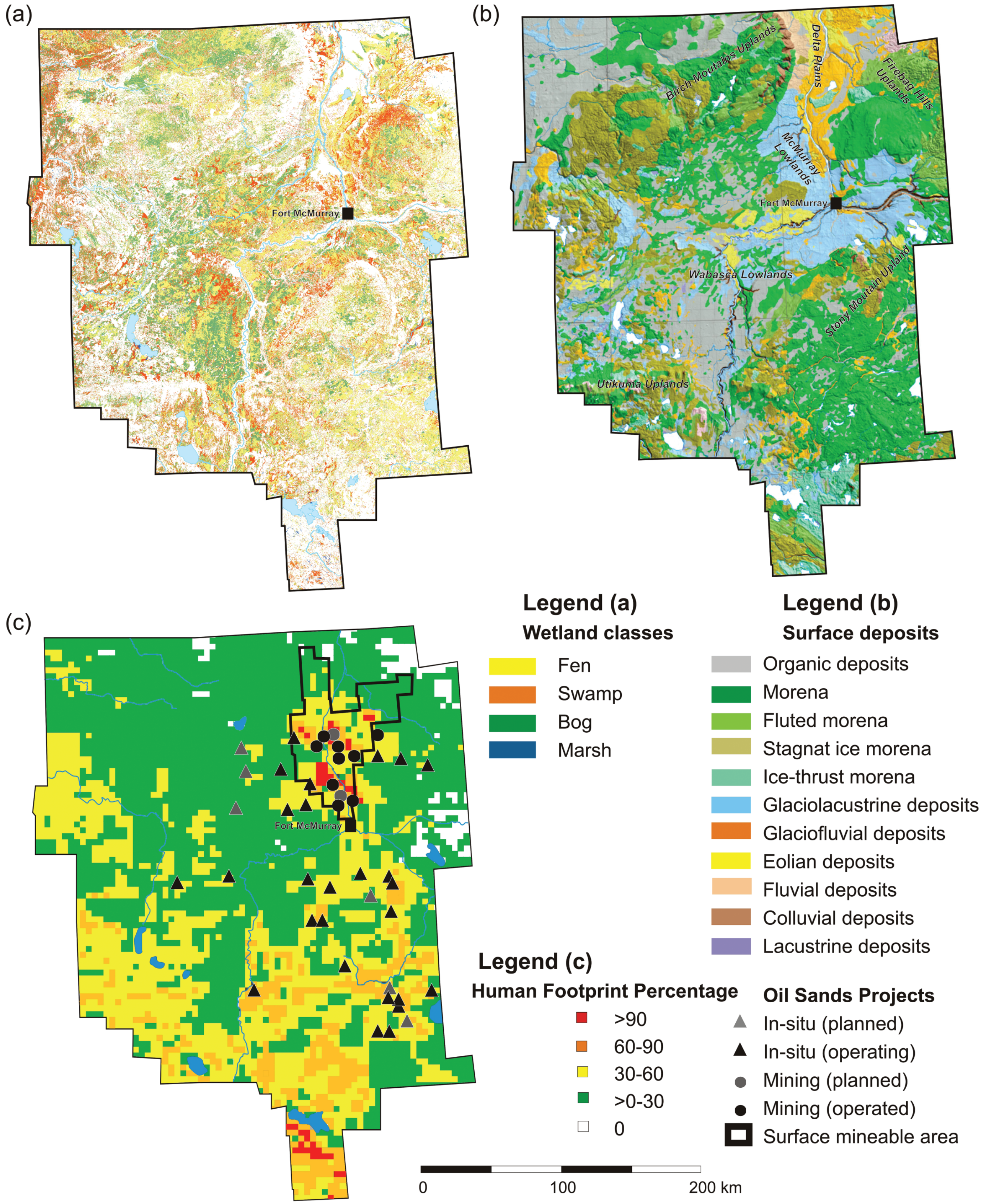
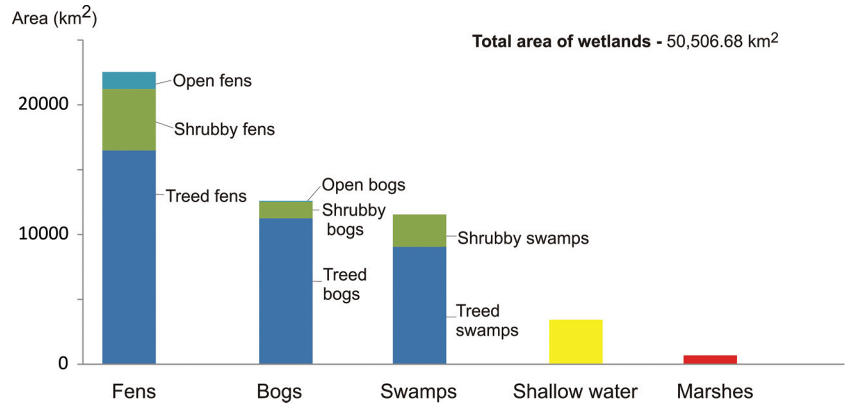


**Fig. 2.** Areas of different wetland classes (based on Alberta Wetland Classification System) in the Athabasca Oil Sands Region (data from AEP (2018), Ducks Unlimited Canada (2019)).






### 2.2. Water balance of wetlands

As the AOSR is a part of the Boreal Plains ecozone, characterized by subhumid climate (Marshall et al. 1999; Petrone et al. 2007), wetlands are often functioning under a long-term moisture deficit (Bothe and Abraham 1993; Devito et al. 2012). However, actual evapotranspiration is often lower than PET due to landscape controls and autogenic feedbacks, which help conserve water during periods of low water availability (Petrone et al. 2007; Devito et al. 2017). Despite these controls, there are landscape configurations in the region, such as peatland–swamp dominated catchments, that can be major source areas of water (Devito et al. 2017). In addition, higher elevated areas (i.e., The Stoney Mountain Uplands) in the AOSR can receive above average precipitation for the region, and annual precipitation often exceeds PET (Wells et al. 2017; Government of Canada 2018). These headwater wetland-dominated basins can therefore be important sources of water for downstream ecosystems that may be experiencing a moisture deficit and acting as groundwater recharge areas, which drive formation of regional groundwater flow systems (Barson et al. 2001).

In the AOSR, the water budgets of wetlands (except bogs and wetlands underlaid by fine-grained deposits) is commonly reliant on groundwater inflow–outflow and storage changes because they are situated within deep sediments that can store and transmit large volumes of water (Devito et al. 2012). Groundwater discharge into wetlands is controlled by the scale of the groundwater system, as well as the hydraulic conductivity of the mineral substrates that are defined by the physiography of the region (Tóth 1999; Winter 1999; Winter et al. 2003; Elmes and Price 2019). This contrasts with wetlands in Eastern Boreal regions of Canada where groundwater inflow–outflow from surrounding uplands can be assumed to be small as these wetlands are underlined by low-permeability crystalline bedrock with limited water storage capacity and low hydraulic conductivity (Creed and Sass 2011). In the AOSR, the interaction between the subhumid climate and the near-surface sediments results in pronounced seasonal and decadal wet and dry cycles that have a strong effect on hydrological and geochemical functioning of wetlands (Devito et al. 2005*a*; Chasmer et al. 2018; Elmes and Price 2019). Further, with this subhumid climate, evapotranspiration (ET) is typically the largest negative water flux term in any wetland type in this region (Devito et al. 2005*a*; Petrone et al. 2007).

Many wetlands in the AOSR rely on autogenic water table feedbacks (i.e., water table-transmissivity feedbacks) that can reduce subsurface flow out of a peatland and limit subsequent water table drawdown during periods of low water availability. This feedback occurs when the water table recedes, where the upper highly permeable peat layers no longer contribute to the transmissivity of the fen peat column and subsequently reduce flow across the peatland (Waddington et al. 2014; Elmes and Price 2019). For example, peatlands in the region maintain shallow water tables by limiting ET through the low conductivity catotelm peat (Price et al. 2003).

### 2.3. Geologic and geomorphic controls on wetland hydrologic functioning

The AOSR region has heterogenous surficial geology due to multiple prior glaciations, most recently the Laurentide Ice Sheet, and subsequent deglaciation around 11 000 cal years B.P. (Dyke 2004) and by the formation of glacial lakes (e.g., Lake McConnell, Lake McMurray). Such heterogenous surficial geology strongly influences the variability in hydrological functioning of wetlands in the region. The associated glacial and postglacial processes were responsible for the deposition of moraines at the perimeter of the AOSR, glaciolacustrine sediments in the central areas, and fluvioglacial deposits throughout the region, as well as formation or modification of the major topographic features of the region (i.e., rolling and low-relief terrain with numerous shallow lakes and meandering streams) (Fenton et al. 2013). Drift thickness is variable in the region, ranging from <1 m in the central and northern parts to >200 m at topographic highs (Andriashek and Atkinson 2007). As a result, the surface topography of AOSR is characterized by presence of higher elevation uplands (Stony Mountain, Birch Mountains, Firebag Hills, Utikuma Uplands) situated along the perimeter of the central lowlands (e.g., Wabasca McMurray Lowlands) (Andriashek 2003; Andriashek and Atkinson 2007).

Areas influenced by glaciofluvial, fluvial, or eolian processes are composed of coarse-textured deposits rich in sand and gravel that have a relatively high capacity to transmit water (Freeze and Cherry 1979). At regional topographic lows where coarse-grained sediments are sufficiently thick, fens in the WBP can be connected to regional-scale groundwater flow systems and form expansive networks of individual fen systems with relatively stable water table positions (Winter et al. 2003; Smerdon et al. 2005; Devito et al. 2012). However, fens overlying relatively thin (10–15 m) coarse-grained sediment deposits in the AOSR have been shown instead to be connected to local flow systems (Elmes and Price 2019). This local hydrogeological setting renders fens more susceptible to flow reversals, as groundwater is vulnerable to head fluctuations in the presence and absence of precipitation-driven recharge from adjacent uplands (Elmes and Price 2019). In addition, local topographic highs in areas dominated by coarse-grained sediment have been shown to provide a source of lateral subsurface groundwater flow to adjacent fen areas (Smerdon et al. 2005; Elmes and Price 2019).

Till-rich moraines and flat areas that were occupied by postglacial lakes are composed of fine-textured sediments rich in silt and clay (Andriashek 2003) and thus have lower hydraulic conductivities resulting in relatively poor hydrologic connectivity (Devito et al. 2012). Discharge from uplands to adjacent wetlands can occur in these catchments; however, during periods of low water availability, wetlands can supply water to the uplands, either through groundwater flow reversals from wetland to upland (Ferone and Devito 2004; Devito et al. 2012; Brown et al. 2014; Wells et al. 2017) or through transpiration by aspen trees via hydraulic lift, where deep clonal roots have been shown to encroach into adjacent wetlands (Depante 2016). However, low hydraulic conductivity of the silt- and clay-rich sediment underlying the peatlands restricts groundwater recharge during flow reversals, limiting water table drawdown in the wetland (Ferone and Devito 2004; Wells et al. 2017). Runoff from uplands can occur; however, it has not been shown to represent a significant proportion of the annual water budget. The combination of high transpiration demands from aspen stands, combined with the relatively high storage capacity of the uplands, will result in low runoff ratios in most years (Devito et al. 2005*b*; Wells et al. 2017).

Transitions between areas composed of fine- and coarse-textured material are often characterized by a presence of veneer-type deposits with layered sequences of coarse- and fine-grained sediments. Such areas have complex hydrological interactions, and wetlands are more abundant at the margins and zones that cross the local water table due to the sediment cover thinning or surface lowering (Devito et al. 2012). In areas where coarse-texture sediments are underlined by fine-texture ones, the water table position is more likely to reflect the distribution of impermeable layers rather than the topography (Devito et al. 2005*a*; Riddell 2008). The layering of fine- over coarse-grained sediments can lead to the existence of wetlands that are perched and isolated from the regional water table (perched wetlands; Riddell 2008) or connected to the regional water table despite being elevated, with recharge limited by the low hydraulic conductivity fine-grained material overlying the aquifer (Elmes and Price 2019).

The AOSR is situated within a region characterized by infrequent (0%–10% of land area) and isolated patches of permafrost (Brown et al. 1997, Vitt et al., 1994), and as such, local hydrology can be influenced by the presence of a frozen soil layer. For example, permafrost impedes flow by reducing the transmissivity of the peat column, limiting the cycling of subsurface water, and therefore isolating peatlands (primarily bogs) on the landscape (O'Donnell et al. 2011). More elevated areas in the AOSR are characterized by lower winter temperatures and therefore have a greater extent of permafrost (Lindsay and Odynsky 1965; Ozoray et al. 1980; Beilman et al. 2001). Extensive peat plateaus have been noted in the Birch Mountains (Fig. 1*b*) that are underlain by more continuous permafrost (Vitt et al. 1996). However, these areas have been subject to degradation, as evidenced by collapse scars that form internal lawns (Vitt et al. 1996), causing changes to the proportion of land-cover types as well as subsurface and groundwater flow patterns (Connon et al. 2014). Permafrost degradation is often associated with enhanced flow in the upper peat column, as frozen peat with previously restricted flow thaws and becomes hydrologically conductive, resulting in enhanced connectivity of peatland types and drainage networks, ultimately changing the hydrology of a basin (Wright et al. 2009; Quinton et al. 2011; Gibson et al. 2019).

Bedrock geology can also influence wetland hydrologic functioning because physical properties of bedrock (e.g., thickness, orientation, fissuring, strength of material) and their composition have a strong influence on surficial topography and formation of parent material that contribute to wetland distribution and function (Mitsch and Gosselink 1993; Devito et al. 2012). In addition, bedrock geology exerts a direct control on the groundwater chemistry and soil physical properties, as well as direction of regional groundwater flow (Bachu 1995). In the AOSR, the regional hydrodynamic regime follows a south to north direction primarily through Devonian carbonates overlain by Cretaceous strata (e.g., siltstones, shales, sandstones, sands) (Bachu 1995; Andriashek 2003). Yet, a number of higher elevation uplands (Stony Mountain, Birch Mountains) that are underlain by cretaceous shales and sandstones act as regional recharge areas, creating confined regional aquifers that eventually discharge into the Athabasca River (Andriashek 2003; Grasby and Chen 2005). Because the region is located on the northeastern edge of the Alberta Basin (Wright et al. 1994), and the thickness of sedimentary cover decreases in a northwestern direction from about 1500 m to near 0 m, several discharge zones of deep aquifer associated with erosional edges of Devonian and Cretaceous strata occur (Bachu 1995). By introducing base ions, deep groundwater input governs the geochemical properties of wetlands; however, the potential for a wetland to receive groundwater from deep regional groundwater will be dependent on its positioning relative to the terminus of deep regional aquifers in the area (Tóth 1999). For example, south of Fort McMurray, the existence of a high salinity groundwater discharge zone from Grand Rapids formation resulted in formation of saline fen peatlands (Purdy et al. 2005; Wells and Price 2015*b*).

## 3. Alteration of wetland hydrologic function

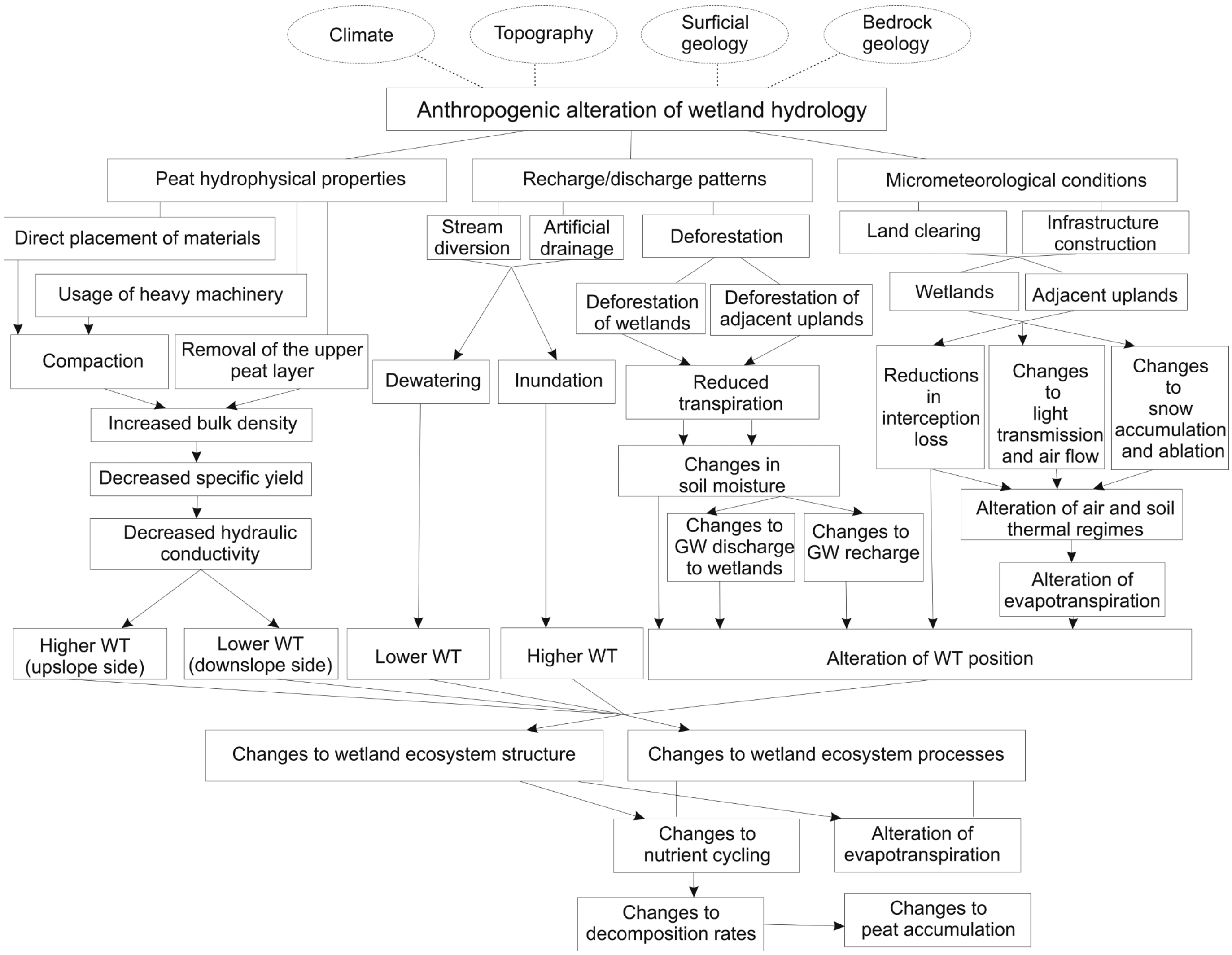
The spatial extent and degree of anthropogenic influence on wetlands will differ depending on the wetland type, extent of area impacted, as well as the type of disturbance. However, in general, ground disturbance occurs at local level through changes to soil hydrophysical properties and related wetland connectivity and at watershed level through changes to wetland water table and groundwater recharge–discharge patterns. In addition, wetland hydrologic functioning can be affected by transformation of water and energy balances associate with changes in micrometeorological conditions that govern evapotranspiration (Petrone et al. 2007), one of the key components of a wetland water balance (Devito et al. 2012). The key disturbances and associated changes are summarized in Fig. 3, and will be discussed in the following subsections. Because the AOSR is strongly dominated by peat-forming wetlands (Vitt et al. 1996), the focus of this section will be on alteration of peatland hydrology.

### 3.1. Alteration of peat hydrophysical properties

The natural, undisturbed hydrophysical properties of a peat profile impose an important control on water table position and hydrological fluxes over periods of varying hydrometeorological conditions (Price et al. 2003). These properties (e.g., porosity, specific yield, bulk density, and hydraulic conductivity) are highly variable at both regional and field scales, and typically vary with peat depth, with trends controlled primarily by the degree of decomposition, and botanical origin (Boelter 1969; Levesque and Dinel 1977). The near-surface peat is composed of living and poorly decomposed plant material and has a structure characterized by relatively large pores, which results in a relatively high porosity (Carey et al. 2007; Rezanezhad et al. 2010), specific yield (Chason and Siegel 1986; Price 1992) and hydraulic conductivity (Fraser et al. 2001; Quinton et al. 2008; Branham and Strack 2014; Baird et al. 2016), and lower bulk density (Carey et al. 2007) and water retention capacity (Boelter 1969). These properties promote water infiltration and high permeability, while restricting capillary flow thus water loss from evaporation (Price 1996; McCarter and Price 2017). The degree of decomposition generally increases with depth below surface (Ingram 1978), consequently reducing pore-size (Boelter 1969), and therefore specific yield (Masch and Denny 1966) and hydraulic conductivity (Mualem 1976; Price and Maloney 1994; Hoag and Price 1995; Fraser et al. 2001, Beckwith et al. 2003; Whittington and Price 2006) while bulk density (Boelter 1968; Carey et al. 2007) and water retention capacity (Price et al. 2008) typically increase. These depth-dependent properties influence soil moisture profiles resulting in capillary rise and the productivity of mosses and vascular plants (McNeil and Waddington 2003).

Although not unique to the region, the depth-specific peat properties discussed above can provide important water conservation

**Fig. 3.** Conceptual model of human alteration of wetland hydrology in the Athabasca Oil Sands Region. Although natural settings (i.e., sub-humid climate, topography, heterogeneous surficial and bedrock geology; shown as ovals) are not always directly involved in anthropogenic alteration of wetland hydrology, their interactions (shown in dashed lines) define the sensitivity of wetland hydrology to human alteration at the local level. Ground disturbance occurs at the local level through alteration of peat hydrophysical properties and related wetland connectivity and at the watershed level through changes to wetland discharge and recharge. In addition, wetland hydrologic functioning can be affected by the transformation of water and energy balances associated with changes in micrometeorological conditions that govern evapotranspiration. The model represents simplified relationships between anthropogenic alteration, wetland components and processes. Detailed analysis of hydrological feedbacks to natural and anthropogenic disturbances in peatlands can be found in Waddington et al. (2015). Connections with arrows suggest cause–effect relationships.




mechanisms for peatlands in the subhumid AOSR. For example, decreases in water table position during periods of low rainfall intersects the water table to depths characterized by increasingly lower hydraulic conductivities ($10^{-5}$–$10^{-7}$ m $s^{-1}$; Ferone and Devito 2004; Wells et al. 2017; Elmes and Price 2019). This subsequently produces a water table feedback mechanism (Waddington et al. 2015), which reduces lateral drainage across a peatland, subsequently storing greater volumes of water. Conversely, during periods of high water availability, increases in water table position intersects the water table to shallow depths characterized by higher hydraulic conductivities ($10^{-3}$–$10^{-4}$ m $s^{-1}$; Ferone and Devito 2004; Elmes and Price 2019; Wells et al. 2017), allowing for greater transmissivity and thus connectivity to the surrounding landscape (McCarter and Price 2017; Elmes and Price 2019).

The peat profile is vulnerable to ground disturbance through compaction caused by direct placement of materials or infrastructure (e.g., pipelines; Enbridge Pipelines Inc. 2012), and (or) the heavy machinery used during construction (Johnson et al. 1991; McNabb et al. 2001; Whitson et al. 2003; Cambi et al. 2015). Compaction alters the hydrophysical properties of the upper peat column by reducing specific yield (causing flashier water table responses), reducing hydraulic conductivity (thus subsurface flow), and increasing bulk density (thus the water retention properties), influencing moss productivity and subsequent peat accumulation (Waddington et al. 2014). Seismic line (Severson-Baker 2003; Petrone et al. 2008), road (Trombulak and Frissell 2000; Strack et al. 2017), and pipeline (Soon et al. 2000; Olsen and Doherty 2012) construction have all been shown to increase the bulk density of peat and other near-surface soils throughout the boreal landscape. Moreover, belowground pipeline construction requires the removal of the upper highly transmissive peat layer (AEP 2017). For example, within the immediate location of a pipeline, a minimum of 40 cm of peat is removed during installation; although peat is replaced over the excavated area after installation, the natural peat structure is not maintained and the density and pore structure is altered (Enbridge Pipelines Inc. 2012). Both peat compaction and upper peat layer removal result in a decreased specific yield and hydraulic conductivity at the ground surface, as well as an increased bulk density and water retention capacity of the upper peat layer that amplify water table fluctuations (Trombulak and Frissell 2000; Plach et al. 2017; Strack et al. 2017; Fig. 3). Despite the widespread use of belowground pipelines in the oil sands industry, there are few hydrological studies that address the potential changes they cause to subsurface flow patterns within peatlands in the AOSR. Studies conducted on conifer swamps in north-central Minnesota have shown that belowground oil and gas pipelines impede lateral subsurface flow, through peat compaction that reduces hydraulic conductivity, impeding water flow through the upper peat layer (Boelter and Close 1974). Olsen and Doherty (2012) measured soil properties across pipeline corridors in several wetlands in southeast Wisconsin, reporting soil bulk density was 63% higher and moisture content 19% lower than values measured outside of the impacted perimeter. The authors also noted large differences in vegetation species composition, with pipeline corridors having a lower species diversity and a higher proportion of species that were not native to the wetland (Olsen and Doherty 2012).

Similar changes to hydrophysical properties follow the direct placement of filling materials (e.g., sand, clay, gravel) over the peat. For example, resource road development and additional associated features (e.g., cutbanks, ditches) constructed through fens can impede natural subsurface flow conditions, leading to waterlogging on the upslope side of the peatland and subsequent drying on the downslope side (Stoeckeler 1967; Bocking et al. 2017; Plach et al. 2017); however, the degree of subsurface flow alteration can vary depending on the orientation (e.g., perpendicular, parallel) of roads with respect to subsurface flow (Saraswati et al. 2019) and substrate type (i.e., fine- vs. coarse-grained deposits) (Willier 2017). Appropriate culvert placement can reduce interruption of water flow, but insufficient maintenance, especially in response to beaver activity, can severely restrict drainage function and result in vegetation community shifts (Bocking et al. 2017). Newer roads may include engineered features (e.g., corduroy conduits (Ducks Unlimited Canada 2014)) and geogrid and geocell synthetics (FPInnovations 2016), which can help in promoting natural flow patterns underneath the road surface. A decrease in water table position on the downslope side can cause peat subsidence because of increase effective stress associated with decreased pore-water pressure (Silins and Rothwell 1998; Kennedy and Price 2005; Petrone et al. 2008). Such compression results in the deformation of pore spaces (Rezanezhad et al. 2016; Baird and Gaffney 1995), increasing bulk density and reducing hydraulic conductivity (Price 2003; Whittington and Price 2006) and specific yield (Price and Schlotzhauer 1999). Decreasing the hydraulic conductivity of peat can reduce ET (Petrone et al. 2007), affecting the movement of water, nutrients, and gases through peatland ecosystem, and it can increase oxidation (Strack et al. 2006; Petrone et al. 2008), reducing peat transmissivity and the and thus transport of nutrients (Petrone et al. 2008) and ebullition of methane gas (Moore and Dalva 1993; Roulet et al. 1992).

### 3.2. Alteration of local and regional recharge–discharge patterns

The hydrologic function of fens in the AOSR are susceptible to change from the alteration of local and regional groundwater recharge and discharge patterns. In the oil sands industry, such changes are of particular concern from land clearance and (or) excavation for open-pit mining and construction of industrial infrastructure and living facilities (Gorrell 1976), gravel extraction (Smerdon et al. 2005), timber harvesting (Devito et al. 2005*b*; Redding and Devito 2008), and stream diversion and reservoir creation (Golder Associates Ltd. 2007). These disturbances can alter the water table and groundwater connectivity between peatlands and groundwater flow systems. For example, mine dewatering lowers the regional water table causing a drawdown cone around the mine (Whittington and Price 2012, 2013; AEP 2014) that, over the long term, causes changes to the hydrophysical properties of the upper peat column (Fig. 3). By dewatering and subsequent wetland water table decline, peat can lose its natural pore structure and can compress, which in turn can increase the bulk density and decreases its storage capacity (Clymo 1983; Boelter 1969). It is likely that the deformation of the natural peat structure will decrease the transmissivity of the peat, which will ultimately disrupt the natural hydrological connectivity between a peatland and its adjacent upland (Kompanizare et al. 2018). Such changes can enhance groundwater recharge from nearby fens, leading to enhanced water table drawdown and subsequent desiccation (Whittington and Price 2012, 2013; King and Yetter 2011; AEP 2014; Webster et al. 2015). Water table drawdown can be as high as 15–25 m near mining sites (Government of Alberta 2012), but can be as low as 1 m (AEP 2014), while still influencing the hydrologic function of nearby fens. Similar effects have been related to stream diversion and reservoir creation. For example, Beaver River Diversion System and reservoir creations divert water that formerly flowed north through Syncrude's Mildred Lake Lease 17. The diversion altered the catchment boundaries of Beaver Creek and Poplar Creek watersheds, influencing streamflow to the Athabasca River (Golder Associates Ltd. 2007). Stream diversion may therefore alter the water balance of peatlands within these watersheds, which has the potential to alter peatland water table position and thus landscape connectivity.

Oil sands operations require significant quantities of sand and gravel for infrastructure (primarily roads); therefore, many gravel-rich deposits have been depleted in the region (Fisher and Smith 1993). Gravel extraction alters the local topography and relief and can cause reductions in evapotranspiration and increases in groundwater recharge (Lemmelâ 1990) resulting in changes to patterns in groundwater flow (Kompanizare et al. 2018). For example, in a pond–peatland complex in the WBP, gravel extraction was shown to enhance the recharge of meteoric water to a regional groundwater flow system connected to the complex, altering the chemistry of groundwater discharging into the pond and peatland (Smerdon et al. 2012).

Timber harvesting associated with the oil and gas industry in the AOSR involves the removal of large tracts of mature forested boreal uplands (Mackendrick et al. 2001; Natural Regions Committee 2006). Research conducted in the WBP has illustrated that timber harvesting typically leads to little or no increases in runoff, primarily due to the subhumid climate and low-relief nature of the region, which favours vertical flow and soil moisture storage (Redding and Devito 2008; Devito et al. 2005*b*). Thus, hydrological changes due to timber harvesting in the WBP are mostly related to alterations of the water balance through reductions in transpiration (Satterlund and Adams 1992), resulting in enhanced infiltration, soil moisture storage, and presumably groundwater recharge (Redding and Devito 2008).

Despite widespread gravel extraction and timber harvesting in the region, few studies have addressed the potential long-term changes to groundwater recharge in watersheds in the AOSR (Smerdon et al. 2009; Jasechko et al. 2012; Hwang et al. 2018), especially with respect to fens connected only to local groundwater flow systems. Due to the heterogeneous forest cover (Beckingham and Archibald 1996) underlying surficial geology (Devito et al. 2012), and therefore variable transmissive properties and storage potential (Devito et al. 2005*b*) of forested uplands in the WBP, changes associated with deforestation are expected to be largely site- and region-specific (Devito et al. 2000, 2005*a*). For watersheds characterized primarily by coarse-grained surficial deposits, groundwater recharge has been shown to increase following harvest (Redding and Devito 2011). However, for areas characterized by fine-textured surficial deposits, enhanced recharge may be limited by the relatively higher soil moisture storage as well as enhanced evaporation due to capillary forces imposed by fine-grained sediment (Smerdon et al. 2009) as well as the greater potential for concrete ground frost to develop prior to snowmelt that enhances runoff (Redding and Devito 2011). Moreover, in forested areas dominated by aspen species, rapid regeneration following harvesting will likely mask the effects within the first few years (Redding and Devito 2008). In areas dominated by spruce and (or) pine, these effects may persist for longer as canopies may require more time to redevelop (Lieffers et al. 1996).

Predicting the response of land clearing on recharge–discharge patterns in the AOSR is largely site specific, which is made even more complex by the variability in the scale and strength of groundwater connectivity of peatlands throughout the region. Peatlands in the region that are connected to local groundwater flow systems will likely respond to changes from disturbance to adjacent forested uplands faster than peatlands connected to regional groundwater sources associated with longer travel times (Winter et al. 2003; Devito et al. 2005*b*). For example, Elmes and Price (2019) detected multiple vertical flow reversals in a moderate-rich fen in the AOSR overlying a local groundwater discharge area, which corresponded to short-term (diurnal and seasonal) fluctuations in precipitation-driven recharge to adjacent uplands. Conversely, Wells and Price (2015*b*) suggested that trends in groundwater discharge to a saline fen were influenced by much longer time scales (centuries to millennia), as groundwater was sourced by a saline spring discharge zone within the Grand Rapids formation, a known regional aquifer.

### 3.3. Wetland hydrology and alteration of micrometeorological conditions

Ground disturbance can also affect wetland hydrology indirectly through changes in the micrometeorological conditions that affect the water and energy balances (Fig. 3). Seismic lines built through peatlands, although less invasive than roads, can create differences in snow cover and the timing in which disturbed and undisturbed locations become ice-free (Fig. 3). This has been shown to significantly alter local subsurface flow patterns in the early parts of the growing season (Petrone et al. 2008). For permafrost peatlands in the Northwest Territories, characterized by discontinuous permafrost, seismic lines have been shown to enhance permafrost thaw, establishing perennial ice-free layers (Braverman and Quinton 2016). This thawing alters the ground thermal regime and energy balance (Williams and Quinton 2013), and can lead to peat subsidence and changes in subsurface flow patterns between bog and fen and the surrounding landscape (Braverman and Quinton 2016).

The placement of fill and (or) stripping of vegetation can change light transmission, air and soil temperature (MacFarlane 2003), and wind speed (Chen et al. 1992), all of which can influence the ratio of precipitation to evapotranspiration (Fig. 3). For example, seismic lines and seasonal roads in the WBP can reduce snow interception and subsequently increase snow depth, delaying melt and influencing the spatial variability of snow melt (Haag and Bliss 1974; Lafleur et al. 1997; Petrone et al. 2008; Strack et al. 2017). In addition, higher snow depth in seismic lines alters thermal regimes, typically by insulating the peat from subsequent freezing winter temperatures, ultimately reducing ice thickness (Zhang 2005). Reductions in ice thickness can cause an earlier spring thaw, leaf-out, and onset of transpiration (Repo et al. 2014; Van Huizen et al. 2019). Given the sparse information that does exist across Canada, there is still a significant knowledge gap addressing the range of variability in the relationships between land disturbance, micrometeorological conditions and wetland hydrological functioning in the AOSR.

## 4. Wetland mitigation

Since wetlands have been recognized as economically, ecologically, and environmentally valuable components to the boreal landscape (Brander et al. 2013; Belyea and Malmer 2004; Vitt et al. 2000), mitigation of wetland losses has been required since 2015 (Government of Alberta 2015). Wetland mitigation includes avoidance of wetland disturbance, minimization of negative effects to wetlands, and wetland replacement (Government of Alberta 2015, 2016). Best management practices to minimize the impact of human disturbance on wetland ecohydrological functioning include, but are not limited to: (*i*) minimization of disturbance size (COSIA 2013; Strack et al. 2017; Lovitt et al. 2018; Abib et al. 2019), (*ii*) avoiding mixing of peat and mineral soil for reclamation upon replacement (Ryder et al. 2004; Sakhalin Energy 2005; COSIA 2013; Nwaishi et al. 2015), (*iii*) retaining natural flow by installation of cross drainage manufactured products (e.g., corduroy, geogrid, geocell) (COSIA 2013; Osko et al. 2018), and (*iv*) placement of necessary linear disturbances parallel to the direction of water flow (Wood et al. 2016; Plach et al. 2017; Saraswati et al. 2019).

Wetland restoration has been increasingly performed in the AOSR to recover degraded wetlands after seismic explorations and construction of well pads and roads (Alberta Environment and Water 2012; COSIA 2013; Lamers et al. 2015; AEP 2016*b*). Restoration includes "the re-establishment of hydrology, vegetation and wetland processes" (Government of Alberta 2013) and can be achieved by blocking drainage (Cooper et al. 1998; Price et al. 2003; Zedler and Kercher 2005), changing the basin morphology and microtopography (Price et al. 2002; Vitt et al. 2011; Pouliot et al. 2011; Lieffers et al. 2017; Bourgeois et al. 2018), and modification of surface cover and vegetation (Price et al. 2003; Cobbaert et al. 2004; Rochefort et al. 2003; COSIA 2013). Recent studies in the WBP have focused mainly on changes in geochemistry (including carbon and methane fluxes) (Strack et al. 2014, 2016; Wood et al. 2016; Murray et al. 2017; Engering 2018) and the re-establishment of vegetation in peatlands that undergo restoration (Vitt et al. 2011; Gauthier 2014; Caners and Lieffers 2014; Shunina 2015; Lieffers et al. 2017; Gauthier et al. 2018). Although the recovery of hydrologic function of wetlands is an important step toward peatland restoration, there is a lack of in-depth hydrological studies looking at spatial and temporal changes in vertical water fluxes and local and regional drainage patterns at and around restored well pads, roads, cutlines etc. Several studies (e.g., Caners and Lieffers 2014; Lieffers et al. 2017) have shown that redevelopment of microtopography is important for restoration of peatland vegetation; however, better understanding of effects of landscape positions and surficial geology on restoration of hydrologic functioning is required. In addition, developing a better understanding of wetland hydrologic functioning can be useful for more effective peatland reforestation that is crucial for many habitat quality goals (e.g., for woodland caribou) (Dyer et al. 2001; Jordaan et al. 2009).

Restorative replacement can also be achieved through wetland construction and returning disturbed area to wetlands using appropriate reclamation techniques (Government of Alberta 2017*a*). Oil companies are instructed to reclaim approximately one-third of the boreal landscape impacted by oil sands mining and processing to wetlands according to provincial guidelines (OSWWG 2000; Alberta Environment 2008; CEMA 2014). The first wetland reclamation attempts in the AOSR have led to the initiation of several small, shallow, open-water wetlands and marshes as a result of planned construction (e.g., Suncor Wapisiw Marsh); spontaneous colonization of disturbed poorly drained areas by natural wetland vegetation; and following evaluation (e.g., Bill's Lake, Syncrude S4 Beaver Pond, Suncor CT's natural wetland) (CEMA 2014; Daly et al. 2012). These opportunistic wetlands formed over a range of slopes, aspects, and topographic positions across contrasting fine- and coarse-textured landforms (Little-Devito et al. 2019). Such variability in wetland establishment existing across landscape types revealed an importance of internal feedback mechanisms for wetland formation and maintenance. The first assessments of these opportunistic wetlands demonstrated that they had a lower vegetation diversity, ecological health status, soil organic content, peat accumulation potential, and nutrient concentrations compared with natural wetlands in the AOSR (Thormann et al. 1999; Trites and Bayley 2009*b*; Rooney and Bayley 2011; Raab and Bayley 2013; Roy et al. 2016).

The subsequent reclamation efforts have been focused on construction of fens (Price et al. 2010; Daly et al. 2012; Pollard et al. 2012; Wytrykush et al. 2012; Vitt and House 2015; Borkenhagen and Cooper 2016; Vitt et al. 2016); however, the development of these constructed fens is greatly challenged by salinization due to salt input from reclamation materials used in their construction (Ketcheson et al. 2016; Nicholls et al. 2016; Biagi et al. 2019; Kessel et al. 2018; Simhayov et al. 2017, 2018) and their hydrologic functioning (Elshorbagy et al. 2005; Negley and Eshleman 2006; Ketcheson and Price 2016*a*, 2016*b*). Constructed fens must create a system capable of maintaining a hydrological regime suitable for the development of the appropriate peat-forming vegetation (Ketcheson et al. 2016); however, the hydrologic functioning of these fens and reclaimed landscapes (Spennato et al. 2018) will differ from that of undisturbed areas (Elshorbagy et al. 2005; Negley and Eshleman 2006; Ketcheson and Price 2016*a*, 2016*b*). In many mine closure plans, the postmining landscape will be dominated by a combination of hummocks (low hills or mounds) and shallow channels composed of coarse-textured tailing sands (BGC Engineering Inc. 2010; Devito et al. 2012; CEMA 2014) to encourage the formation of localized recharge–discharge zones (Price et al. 2010; Ferone and Devito 2004). Thus, the hydrologic functioning of constructed fens in such settings can be expected to represent wetland functioning within coarse-textured fluvio-glacial landscapes. However, significant changes in hydrophysical properties and vegetation of reclaimed slopes can occur over time (Guebert and Gardner 2001; Kelln et al. 2007; Carey 2008; Meiers et al. 2011), which may cause a shift in their hydrological functioning (e.g., a shift from water transportation to water storage) (Ketcheson and Price 2016*b*), which in turn will affect their hydrology. Further uncertainty in constructed wetland functioning relates to the effects of variable topography and materials on water distribution within adjacent reclaimed areas (Leatherdale et al. 2012; Huang et al. 2015). Sutton and Price (2019) showed a cover soil had relatively little spatial variability in its hydraulic properties, but spatial variability in placement depth had a big impact on recharge rates, with thicker soils retaining more water to support vegetation.

The future trajectory of a constructed wetland also depends on the complex feedback mechanisms related to the water chemistry (e.g., salt concentration, heavy metal concentration; Oswald and Carey 2016; Simhayov et al. 2017, 2018; Kessel et al. 2018; Biagi et al. 2019) that will influence vegetation and evapotranspiration (Rezanezhad et al. 2012) and thus the water balance of the wetlands (Ketcheson et al. 2017; Nicholls et al. 2016). Precipitation inputs have been shown to moderate salt concentrations at the surface in constructed wetlands (Kessel et al. 2018; Biagi et al. 2019), but it is expected that future constructed wetland designs will be connected to deeper seepage zones on tailings sand with elevated salinity (BGC Engineering Inc. 2010; Ketcheson et al. 2016), and relations between salinity and precipitation within these wetlands can be different. This relationship is not straightforward, because while rainfall recharge can dilute near-surface porewater (Kessel et al. 2018), it can simultaneously enhance the inflow of saline groundwater from regional aquifers, as was observed in a local natural saline fen where increased precipitation intensified input of saline groundwater (Wells and Price 2015*b*; Volik et al. 2017). The complexity of this is illustrated in a constructed fen, where an enhanced flux of salt occurred during dry periods with less dilution of source water (Kessel et al. 2018).

Returning ecosystem function of postmined areas in the AOSR requires integrated landscape-scale (e.g., regional or watershed) reconstruction and consolidation of isolated and fragmented wetland systems (Johnson and Miyanishi 2008; Choi et al. 2008). It has been suggested that incorporation of landscape connectivity into the postmined landscape design plans can be useful not only for maintaining the water balance of reclaimed areas, but also for salinity control (Ketcheson et al. 2016; Volik et al. 2017). Moreover, parallel to the consideration of constructed wetlands as water sinks in low-lying areas within the postmining landscape, is a growing recognition of the wetlands as water sources (Ketcheson et al. 2016), because wetlands can store water during wet periods and release it to the neighbouring areas during dry periods (Ferone and Devito 2004; Petrone et al. 2008; Riddell 2008; Barr et al. 2012; Thompson et al. 2015). However, functioning of constructed wetlands as components of the postmining landscape and their possible effect on adjacent ecosystems are uncertain and require in-depth studies including numerical modeling (e.g., Elshorbagy et al. 2005). The reclaimed landscape can benefit from wetland functions such as water storage, nutrient transformation, and sediment retention, and constructed wetlands can also be incorporated into a constructed wetland treatment system (McQueen et al. 2017; Hendrikse et al. 2018) and contribute to reduction of water contamination.

## 5. Wetland hydrological monitoring

Several regional environmental monitoring efforts (e.g., Alberta Oil Sands Environmental Research Program, Regional Aquatics Monitoring Program, Cumulative Effects Management Association, Cumulative Environmental Management Association, Alberta Biodiversity Monitoring Institute, Wood Buffalo Environmental Association, Regional Groundwater Monitoring Network, Joint Oil Sands Monitoring) have been made since 1975, and their history and evolution were summarized by Cronmiller and Noble (2018). Considering the sensitivity of wetlands to anthropogenic alteration and to mitigate negative impacts on them, a wetland monitoring program under the Oil Sands Monitoring (OSM) Program has been initiated (Alberta Environment and Parks (AEP), and Environment and Climate Change Canada (ECCC) 2018). The program is in the development stage with implementation anticipated in 2020–2021. There are currently 22 wetland monitoring sites in the AOSR focused on monitoring stressors associated with oil sands development including measures of hydrologic conditions and predicted wetland ecosystem responses. Previous and ongoing wetland monitoring in the AOSR includes the Alberta Biodiversity Monitoring Program, which monitors wetlands for changes in wetland vegetation and other biota, with no associated measures of wetland hydrologic conditions (Ficken et al. 2019). The only other wetland monitoring in the region is local-scale wetland compliance monitoring on oil sands leases, which are inconsistent in their measures of hydrologic conditions and other stressors.

Hydrological monitoring of wetlands is an important part of wetland monitoring in the AOSR (Ciborowski et al. 2012; Eaton and Charette 2016; Roy et al. 2016) as trends in water levels and water movement directly and indirectly control all biogeochemical and ecological processes that occur in wetlands (Updegraff et al. 1995; Szumigalski and Bayley 1996; Laiho et al. 1999; Sundström et al. 2000; Salonen 1994; Bridgham et al. 1998). Water table depth coupled with soil moisture has an effect on nutrient transformation in peatlands in the AOSR (Bridgham et al. 1996; Wood et al. 2016) and thus on decomposition rates and plant productivity (Vitt et al. 2009; Peichl et al. 2014; Goetz and Price 2015) and community composition (Bridgham et al. 1996). While decomposition and plant productivity define peat properties and rates of peat accumulation (Waddington et al. 2015), composition of plant communities is considered a key driver of variability in evaporative losses (Williams and Flanagan 1996; Brown et al. 2010; Bubier et al. 1998; Petrone et al. 2011). Enhanced evapotranspiration can promote water table drawdown in disturbed wetlands (Price 1996; Van Seters and Price 2001) and can therefore influence the long-term succession of vegetation assemblages (Szumigalski and Bayley 1996). Taking into account such interconnections between wetland components and processes, intense long-term hydrologic monitoring is necessary for better understanding of consequences of anthropogenic alteration of wetland hydrology and related ecohydrological feedbacks including changes in water quality, water balance and regime, organic matter decomposition and accumulation, and vegetation succession (Waddington et al. 2015).

In addition to existing wetland monitoring recommendations (e.g., Ciborowski et al. 2012; Eaton and Charette 2016; Roy et al. 2016), the following points should be considered:

1. Since heterogeneous surface sediments and highly variable topography greatly influence the degree of peatland–upland connectivity and groundwater influence at the watershed scale (Ferone and Devito 2004; Smerdon et al. 2005; Scarlett and Price 2013; Wells and Price 2015*a*, 2015*b*; Lukenbach et al. 2015; Hokanson et al. 2016; Wells et al. 2017; Elmes and Price 2019), it is crucial to develop a monitoring network with respect to physiography of the area. Such an approach is necessary for creating a basis for improved predictions of the fate of different wetland types associated with coarse- to fine-grained glacial–fluvial and glacial–lacustrine surficial deposits (Devito et al. 2005*a*).
2. Throughout the AOSR, evapotranspiration is a key component of the wetland water balance (Bothe and Abraham 1993; Petrone et al. 2007; Devito et al. 2012; Phillips et al. 2016); therefore, evapotranspiration has to be a component of hydrologic monitoring. The implementation of eddy covariance towers providing continuous measurements of water, carbon, and energy fluxes (Baldocchi 2003) can be considered as an effective method of evapotranspiration monitoring in the AOSR (Waddington et al. 2009; Brown et al. 2010; Nicholls et al. 2016).
3. Existence of decadal wet and dry cycles is one of the most important features of the AOSR climate (Bothe and Abraham 1993; Marshall et al. 1999; Devito et al. 2012). Since the degree of groundwater connectivity between upland and wetland is variable over these 10–15-year climate cycles (Devito et al. 2005*a*; Chasmer et al. 2018), long-term monitoring of groundwater patterns is essential to be able to characterize their true range of hydrologic responses to natural and anthropogenic changes.

## Conclusion

Over the last 50 years, the AOSR landscape has been altered significantly due to extensive development of the oil sands industry and the associated infrastructure. Wetlands are a dominant landscape feature in the AOSR and are particularly vulnerable to disturbance because they rely on specific hydrological conditions that are susceptible to change from human disturbances. These disturbances can result in ecohydrological feedbacks that can alter subsurface flow patterns, influence water table and soil moisture trends, evapotranspiration rates, vegetation community composition, peat decomposition rates, and the hydrophysical properties of the peat.

Despite the ubiquity of disturbances in the AOSR, knowledge of their cumulative impacts on peatland hydrologic functioning remains incomplete. Better understanding of the impacts of construction of open mines, well pads, industrial and living facilities and the effects of linear disturbances (specifically pipeline construction) on recharge–discharge patterns in wetlands in the AOSR is required. The exact impact that disturbances will have on wetlands will be dependent on the hydrogeologic setting. Consequently, better understanding of variability at the scale of groundwater connection in the region will be an important first step before properly characterizing the impacts of disturbances on the recharge–discharge function of fens in the AOSR.

Boreal wetlands are dynamic systems that evolve over time; however, anthropogenic disturbance can alter this trajectory causing less straightforward stages in wetland development. The anticipated trajectory from such development is uncertain, and a sound understanding of the cause–effect relations between different types and intensities of human activities and shifts in wetland functioning is necessary. Thus, long-term wetland monitoring is a crucial component for answering these questions, most importantly in predicting the response of wetlands to varying levels of anthropogenic disturbance.

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