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Reclamation of wetland habitat in the Alberta oil sands: Generating assessment targets using boreal marsh vegetation communities

by

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Abstract

Thousands of hectares of wetlands are being destroyed by oil sands mining in Alberta, and the industry must undertake wetland reclamation to compensate for these losses. Wetland vegetation has developed at some previously mined sites, however reclamation is thus far exploratory, and limited in extent. To inform reclamation practices and assist compliance monitoring I examined vegetation communities in 25 natural boreal wetlands and 20 oil sands reclaimed wetlands, developed a Vegetation-based Index of Biological Integrity (vIBI) to quantify the ecological health of wetlands, and identified possible physical and chemical barriers to reclamation. The vIBI identified 6 reclaimed wetlands in fair to good health, within the range of natural wetlands, however reclaimed wetlands have different vegetation communities, do not produce the same level of aboveground biomass, and have lower levels of sediment nutrients than natural wetlands. To reclaim healthy wetlands, planning should focus on establishing appropriate species, and alleviate nutrient and sediment deficiencies.

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Glossary of terms and definitions

Bioassessment: evaluation of the health of an ecosystem through the direct measurement of biological assemblages, chemical, and physical conditions, wherein the biological attributes of a system reflect the underlying ecological health of their ecosystem.

Bitumen: a heavy, viscous form of crude oil.

Disturbance score: scores used to quantify the relative level of disturbance among all study sites on a synthetic abiotic physicochemical disturbance gradient. Scores were calculated based on physical or chemical variables measured locally at each site.

Ecological health: the presence of appropriate species, populations, and communities; the occurrence of ecological processes at appropriate rates and scales; the environmental conditions to support taxa and processes.

Equivalent land capability: the ability of the land to support various land uses after reclamation that are similar, although not necessarily identical, to those that existed before mining.

Indicator: superficial ecosystem attributes that act as surrogates for core ecosystem processes.

Invasive species: species regulated by the Alberta Weed Control Act (2001) as either restricted or noxious.

Metric: individual measure of an ecosystem component.

Multimetric bioassessment: the integration of multiple biological indicators, e.g., from the vegetation, macroinvertebrate, or fish communities, to provide a robust measure of ecological health.

Multivariate bioassessment: evaluation of the overall species composition of sites, grouping of sites on the basis of their vegetation communities, and comparison of test sites to a group of reference of sites.

Naphthenic acids: a family of saturated, polycyclic and acyclic carboxylic acids that occur in petroleum deposits that may become concentrated in process-affected water found on reclaimed landscapes. Naphthenic acids can be highly toxic to vegetation and aquatic organisms.

Oil sands process-affected (OSPA) treatment: wetlands that were subjected to both physical and chemical disturbance, by exposure to oil sands process water or substrate. These materials can be highly saline and can contain naphthenic acids, polycyclic aromatic hydrocarbons and heavy metals. This disturbance could have occurred once in the history of the wetland, or is ongoing, such as the case of some wetlands receiving seepage water from nearby tailings facilities.

Oil sands reference (OSREF) treatment: wetlands that were subjected only to physical disturbance on oil sands leases, such as gravel extraction or impoundment, or were formed on materials that were not considered process-affected.

Oil sands: a mixture of bitumen, sand, clay, saline water, naphthenic acids, and other hydrocarbons.

Reclamation: the process of returning an ecosystem that has been degraded, damaged, or destroyed, to a similar, though not necessarily the same ecological condition. Reclamation objectives are to stabilize terrain, improve aesthetics and public safety, and return degraded land to a useful purpose.

Reference (REF) treatment: shallow open water marshes, often located in protected areas, away from anthropogenic disturbance, that spanned a range in salinity from fresh to sub-saline.

Restoration: the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed, to its pre-existing condition.

1. General introduction and thesis overview

Vast expanses of pristine boreal forest and wetlands are being destroyed by oil sands mining in Alberta, and reclamation of the region into healthy self-sustaining ecosystems following mine closure is the greatest environmental challenge yet faced in the province. The terms *pristine*, *undisturbed*, and *ecologically healthy* are often used in ecology to refer to the pre-settlement biological or physical state of an ecosystem, however, these terms are more abstract than they are quantifiable. On the other hand, environmental protection, in the form of legal acts and policy, often mandates outcomes that are explicitly quantitative. Effective assessment is essential in informing, and enforcing policy, particularly in the context of compliance monitoring for large-scale industrial development.

The boreal plain, wetlands, and vegetation

The boreal plain, with its flat topography and large-scale but poorly understood groundwater patterns, is the dominant ecozone in northern Alberta. The pre-settlement boreal plain landscape was a matrix of forest, lakes, ponds and wetlands. Upland and forest vegetation in the region is characterized by *Picea glauca* (white spruce), *Abies balsamea* (balsam fir), *Populus tremuloides* (trembling aspen), *Populus balsamifera* (balsam poplar) and *Betula papyrifera* (white birch).

Wetlands are essential components of the boreal plain landscape; many species commonly associated with the boreal forest are dependent on wetlands for part, or all of their lifecycle, such as medicinal and traditional use plants, waterfowl, birds, fur-bearers and large mammals such as *Alces alsces* (moose). Marsh and shallow water wetlands, the least common of the boreal wetland types, are of disproportionately high value to wildlife, particularly migratory waterfowl (Zoltai et al. 1988). Wetlands in the region are primarily peat-producing fens and bogs, however, marshes and shallow open water wetlands also occur to a lesser extent, spread widely across the boreal plain. The predominant fen and bog wetland vegetation in the region is *Picea mariana* (black spruce), *Larix laricina* (tamarack), ericaceous shrubs, *Sphagnum* and other mosses, sedges (*Carex* spp.), rushes (e.g., *Scirpus* spp.), herbs and forbs. Marshes and shallow open water wetlands are characterized by willows (*Salix* spp.), *Typha latifolia* (broadleaf cattail), *Carex* spp., grasses (e.g., *Calamagrostis* spp., *Scholochloa fetucacea*), rushes, herbs, and forbs (Zoltai et al. 1988).

Wetland losses in the boreal region have been occurring since European settlement and industrialization, from such industries as forestry, agriculture, oil and gas extraction, and mining (Foote and Krogman 2006). Unfortunately, wetland functions and values are often not fully appreciated until after they have been lost, and the indirect nature of their economic importance is easily undervalued (Brander et al. 2006). Recently there has been progress to valuate wetlands in terms of biodiversity support, water quality improvement, flood abatement, carbon management, and nutrient cycling (Mitsch and Gosselink 2000, Zedler and Kercher 2005). When these services are taken into account, wetlands value higher on a per hectare basis than any terrestrial ecosystem (Costanza et al. 1997).

Vegetation is an indicator of overall wetland health due to its role as a link between abiotic environmental factors and higher levels of wetland biota such as macroinvertebrates and birds, its reflection of past and present hydrology, and its sensitivity to various other forms of physical and chemical disturbance (Kirkman et al. 2000, Fennessy et al. 2002). The definition of a wetland requires that the ecosystem must support hydrophytic vegetation, specially adapted to saturated soils and shallow water (Mitsch and Gosselink 2002). Being at the base of the food chain, wetland plants are critical for energy flow to higher trophic levels, and many higher taxa rely on vegetation structure for habitat. Wetland plants influence water quality by acting as nutrient sinks and sources. They have been shown to remove contaminants from water and sediment. Wetland vegetation is also essential to wetland processes that are of value to humans, such as flood and sediment control, shoreline stabilization, and carbon sequestration through peat accumulation (Cronk and Fennessy 2001).

Bioassessment and the Index of Biological Integrity

The enactment of the Clean Water Act (CWA) in the U.S. in the 1970s gave unprecedented protection to waters in the United States, with the goal of maintaining the biological, chemical, and physical integrity of US inland waters. With this protection came methods to monitor the wide range of aquatic ecosystems that fell under the CWA, including wetlands, in order to ensure the mandate was being met (Mack 2007). The development of techniques to assess the health of aquatic ecosystems has been ongoing since this period, and has led to modern bioassessment methods to measure ecological health,

Currently, bioassessment is primarily done in two ways: by multimetric and multivariate analyses. These methods have been compared in several studies (e.g., Reynoldson et al. 1997, Keleher and Rader 2008, Collier 2009), and the general consensus is that while each has its merits, the multimetric method is more easily applied, and thus more suited to a management context, where speed and affordability are important.

Multimetric bioassessment uses direct measurement of biological indicators to quantify the health of a water body. The biological indicators reflect the underlying chemical and physical conditions of an ecosystem (Karr and Chu 1999, Danielson 2002). The Index of Biological Integrity (IBI) is a well-known multimetric tool that has been used to measure the ecological health of intact natural wetlands, as well as the relative health of disturbed, restored, or reclaimed wetlands, to enforce wetland protection policies (Simon 2000). The method focuses on the integration of multiple biological indicators, e.g., from vegetation, macroinvertebrate, or fish communities, to provide a robust measure of ecological health. The single quantitative value produced by the multimetric IBI assessment of a wetland is easily compared to target values, and this is useful to managers enforcing or adhering to policy (Reynoldson et al. 1997).

The multivariate approach to bioassessment relies on the overall species composition of sites, grouping sites on the basis of their ecological communities, and comparing test sites to a group of reference of sites, the latter representing the ideal outcome of restoration or reclamation. The multivariate approach can also be useful in exploring trends in species presence and environmental factors (Reynoldson et al. 1997, Collier 2009).

Oil sands mining and reclamation

Forty years of oil sands mining have changed 520 km^2 of northern Alberta from intact boreal forest and peatlands to a landscape of open pits, vast ponds of saline tailings water, and stockpiles of glacial till, shale, sand, clay, and peat. The Athabasca oil sands lie 400 km northeast of Edmonton, Alberta, in the boreal plain, and represent the largest deposit found in the province. The deposit underlies 104 300 km^2 of land (ERCB 2009), or approximately one sixth of the total area of the province. Oil sands occur globally in subterranean deposits, and are a mixture of bitumen (heavy, viscous crude oil), sand, clay, and saline water, as well as toxic substances such as naphthenic acids and heavy metals. Once separated from other oil sands constituents, bitumen undergoes upgrading to crude oil.

Oil sands mining in northern Alberta occurs in two ways; open pit surface mining when deposits are within 65 m of the surface, and in-situ extraction when deposits lie deeper (ERCB 2009). Oil sands extraction has thus far been primarily a surface mining operation, with 3750 km^2 of the Athabasca oil sands deposit considered to be surface mineable. In order to access the oil sands by surface mining, layers of overburden are stripped away and stored for later use in reclamation operations. The bituminous sand is then mined and processed to separate the heavy crude from other constituents, leaving tailings materials that are up to 25% greater in volume than before mining (Harris 2007).

Oil sands companies are required by federal and provincial laws to reclaim the land they have disturbed to “equivalent land capability” (Harris 2007). This means the landscape must provide similar functions and values as pre-mining conditions, however the fundamental form of the landscape may be drastically changed. The original intact landscape in the vicinity of the Athabasca oil sands was greater than 50% wetland, 95% of which were fen or bog peatlands (Golder Associates 2002). Reclamation strategies currently project the post-reclamation landscape to be approximately 20-30% wetland (C. Qualizza, pers. comm.), and it is unlikely that fens and bogs will be possible to reclaim with the available materials, in the mandated timelines. Indeed, thus far reclaiming healthy, viable ecosystems from the stockpiled materials has proven to be challenging (Johnson and Miyanishi 2008). Reclamation of aquatic ecosystems such as wetlands has not yet been formally initiated, although some wetlands have formed on oil sands leases spontaneously, or have been constructed for research purposes.

It is important to differentiate between restoration and reclamation. The Society for Ecological Restoration (SER) defines restoration as "assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (SER 2004). When land is returned to an ecologically healthy state (such that it is stable, resilient, and requires little external support; Karr 1991), though not necessarily in the same form as the historical ecosystem, it is termed reclamation. Reclamation objectives, in general, are to stabilize terrain, improve aesthetics and public safety, and return degraded land to a useful purpose (SER 2004). As the trajectories of reclaimed sites towards ecological health are often unpredictable (Matthews et al. 2009), ongoing monitoring is essential to the provision of guidance to reclamation activities throughout the process.

Thesis objective and outline

The objectives of this study are to contribute to the state of knowledge on wetland reclamation monitoring and assessment of success in the post-mining oil sands landscape, and to provide recommendations for future reclamation practices by in-depth examination of vegetation and environmental factors as they vary among natural wetlands, and the reclaimed wetlands that currently exist on oil sands leases.

Chapter 2 develops a tool to assess wetland reclamation success in the oil sands. I chose to develop a vegetation-based IBI to measure the ecological health of reclaimed wetlands, and this tool provides feedback as to the health of individual wetlands in our study.

Chapter 3 contains a more in-depth examination of the among-site variation in vegetation and environmental factors that might cause certain reclaimed wetlands to reach higher levels of ecological health than others.

Chapter 4 provides an overview of the contributions this research makes to the field of ecology, and provides guidance for future work.

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2. Development of a vegetation-based Index of Biological Integrity to assess wetland reclamation success in the Alberta oil sands

2.1 Introduction

Karr (1991) defines an ecologically healthy system as one that is stable, resilient, and requires little external support. To determine if an ecosystem is ecologically healthy it must be compared to natural, undisturbed ecosystems on the basis of its biota, ecological processes, as well as chemical and physical structure.

Ecological indicators have long been used as measures of ecological health, and employ relatively superficial ecosystem attributes as surrogates for core ecosystem processes (Niemi and McDonald 2004). The concept that a single biological parameter can characterize the health of an ecosystem led to the ecological indicator (or bioindicator) approach. Bioassessment is the use of bioindicators to evaluate the health of an ecosystem through the direct measurement of biological assemblages, wherein the biological attributes of a system reflect the underlying chemical and physical conditions (Danielson 2002). Early bioassessments focused on the use of individual bioindicators to measure ecological health, however in doing so, they risked failing to quantify accurately ecological health across a broad range of disturbance, from pristine to severely degraded. Current methods in bioassessment focus on the integration of multiple bioindicators to provide more robust measures of ecological health.

The Index of Biological Integrity (IBI) is a measure of ecosystem health that relies on the biological community to diagnose underlying ecological condition of a site.

The IBI does so by incorporating multiple bioindicators (referred to as metrics), which are individual measures of ecosystem components, and the single numerical output of an IBI is comprised of the sum of many individual metrics. A wide array of metrics have greater power to characterize the biological integrity of an ecosystem than single indicators, and while redundant metrics can over-emphasize certain attributes, removing metrics that correlate strongly with each other can control for this (Stevens et al. 2006). To assess the health of ecosystems that experience some form of disturbance (e.g., from industry, urbanization, agriculture) comparisons are made among the biological communities of sites with varying degrees of degradation, and natural (reference) sites in the region. To calibrate the IBI, sites are ranked, or assigned values, along a gradient of disturbance from pristine to most degraded, developed from physical and chemical data at each site. Metrics are scored based on their response to the physical and chemical variation, and are then used in place of direct measurements of disturbance. The single score produced by a multimetric index is easily compared to a target value, which is especially appealing to reclamation managers (Reynoldson et al. 1997). Intensively sampling the biota, hydrology, and chemistry of aquatic ecosystems is often economically and logistically unfeasible, and the foundation of current monitoring approaches such as the IBI is that to be effective they must be inexpensive, simple, and include measurements that are sensitive to small changes in the ecological health of an ecosystem (Schindler 1987).

Large-scale open pit mining of oil sands is destroying large areas of boreal forest and wetlands in northern Alberta. Oil sands, a mixture of sand, clay, saline water, and bitumen, is extracted by surface mining when deposits are within 65 m of the surface, and is refined into crude oil by separating out sand, clays, and salts. The total area of surface-mineable oil sands in Alberta is 3750 km^2. Already 520 km^2 of boreal landscape has been destroyed to access this resource and more projects will begin operation within the next decade (ERCB 2009). The landscape overlying the oil sands deposits is greater than 50% wetland, 95% of which are fen or bog peatlands (Golder Associates 2002). Oil sands mining reclamation requirements demand that the post-mining landscape achieve "equivalent land capability" (Harris 2007), defined as the ability of the land to support various land uses after reclamation that are similar, although not necessarily identical, to those that existed before mining (Alberta Environment 1999).

Approximately 20-30% of reclaimed land is projected to be wetland (C. Qualizza, pers. comm.). Assuming the conservative estimate of 20% wetlands on the entire surface mineable oil sands area, this still means that 750 km^2 will need to be engineered into viable, self-sustaining wetlands. The post-mining landscape will initially be best suited to the development of marshes and shallow open water wetlands, and these have already begun to form on oil sands leases, both by design for research purposes, and spontaneously when surface and hydrologic conditions have allowed (Trites and Bayley 2009a). A complication to the reclamation process is that the excavated marine shale and other tailings present in the post-mining landscape contribute saline runoff to nearby aquatic ecosystems, and can lead to wetlands with elevated salinity. Reclaimed land must receive certification before industry is considered to have met their reclamation requirements (Harris 2007), however as of yet no wetlands have been certified as reclaimed. One reason for lack of certification may be the absence of a method to assess the ecological health of newly created oil sands wetlands. Marshes and shallow open water wetlands are infrequent in the boreal plain in comparison to fens and bogs, but may represent the best outcome for reclaimed wetlands (Purdy et al. 2005). Using natural fresh to sub-saline wetlands as benchmarks for fully functioning, healthy boreal marsh wetlands, the goal of this study was to create a tool for government and industry to assess the health of reclaimed wetlands in the oil sands. Once an assessment tool exists, managers will have tangible targets to reach in reclamation planning and execution.

IBIs have already been extensively adapted for use in measuring wetland ecosystem integrity. An early wetland-specific IBI from Massachusetts used comparisons between impacted and reference sites to diagnose the ecological health of wetlands that were in close proximity to land uses such as residential housing and agriculture (Carlisle et al. 1998). The United States Environmental Protection Agency produced a series of wetland bioassessment methodology papers focused on IBI development, and in one such document Teels and Adamus (2002) concentrate on the use of vegetation metrics as the backbone of an IBI. The vegetation community is an integral part of wetland ecological functions such as nutrient cycling and carbon storage (Mitsch and Gosselink 2000, Fennessy et al. 2002), and supports all the higher taxa. Vegetation community attributes, such as immobility (therefore exposure to local stressors), relatively high species richness and growth rate, ubiquity in wetlands, well documented taxonomy, life history and tolerances, and well developed sampling techniques are features that render vegetation useful as a source of ecological indicators (Teels and Adamus 2002). Vegetation-based IBIs (vIBIs) are powerful tools that facilitate rapid assessment of wetland health, with plant community metrics such as species richness of perennials, number of invasive species, and obligate wetland species being positively or negatively correlated with gradients of disturbance (Mack 2001, Simon et al. 2001, DeKeyser et al. 2003).

I chose to develop a vIBI to measure the ecological health of oil sands reclaimed wetlands. My detailed objectives were to 1) assemble a list of potential vegetation metrics from field data, 2) test metrics against a disturbance gradient for response to increasing abiotic stress, 3) combine sensitive metrics into a vIBI, and 4) use the vIBI to quantify the health of reclaimed wetlands on oil sands leases.

2.2 Methods

Study sites

I collected data in 2007 and 2008 within the North American boreal plains ecoregion (Fig. 2.1, Appendix Table A.1). The boreal plain is characterized by flat topography with surficial glacial deposits of loamy till and gravel-sand glaciofluvial 30-200 m deep overlying Mesozoic- and Cenozoic-age sedimentary bedrock (Johnson and Miyanishi 2008). The subhumid mid to high boreal climate annually averages -2 to +1 ºC, with 400 to 500 mm of wet precipitation and 150 to 200 cm of snow (Zoltai et al. 1988). The majority (70%) of the total annual precipitation occurs between May and September (Devito et al. 2000). The annual water deficit is 40-60 mm due to higher potential evapotranspiration than precipitation, and the groundwater patterns of this region are complex and poorly documented due to the low topography and deep glacial deposits (Zoltai et al. 1988, Price 2005). The pre-settlement landscape was composed of forest, wetlands, lakes and ponds. Forest vegetation in the region is characterized by *Picea glauca* (white spruce), *Abies balsamea* (balsam fir), *Populus tremuloides* (trembling aspen), *Populus balsamifera* (balsam poplar) and *Betula papyrifera* (white birch), with *Picea mariana* (black spruce) in areas with poor drainage. Wetlands in this region are primarily fens and bogs, however, marshes and shallow open water wetlands also occur, and range in salinity from fresh to saline. The fen and bog wetland vegetation in the region is predominantly *Picea mariana*, *Larix laricina* (tamarack), ericaceous shrubs, *Sphagnum* and other mosses, sedges (*Carex* spp.), rushes (e.g., *Scirpus* spp.), herbs and forbs. Marshes and shallow open water wetlands are characterized by willows (*Salix* spp.), *Typha latifolia* (broadleaf cattail), *Carex* spp., grasses (e.g., *Calamagrostis* spp., *Scholochloa fetucacea*), rushes, herbs and forbs (Zoltai et al. 1988).

I selected 20 reclaimed oil sands wetlands and 25 natural reference (REF) wetlands for a total of 45 study sites. Reclaimed wetlands were located on the oil sands leases of Syncrude Canada Limited and Suncor Energy Incorporated near Fort McMurray, Alberta, Canada (56.8531° N, 111.3180° W to 57.1150° N, 111.6833° W), and were predominately fresh to subsaline shallow open water wetlands with marsh fringes.

Reclaimed wetlands were classified by treatment a priori, on the basis of their construction and physicochemical history, taking into account whether a wetland was the subject of physical disturbance such as gravel extraction or impoundment (oil sands reference, or OSREF treatment), or both physical and chemical disturbance (oil sands process-affected, or OSPA treatment). Chemical disturbance occurs when the wetland is exposed to oil sands process water or substrate. These materials are often highly saline and can contain naphthenic acids, polycyclic aromatic hydrocarbons as well as other contaminants. Chemical disturbance could have occurred once in the history of the wetland, or is ongoing, such as the case of some wetlands receiving seepage water from nearby tailings facilities. Natural salinity of marine shale in the region may lead to increased salinity at OSREF sites, in the absence of tailings seepage water or materials. Reclaimed oil sands wetlands have been intentionally constructed for research, and have formed "opportunistically" where conditions have allowed. The 20 reclaimed wetlands included all oil sands wetlands on Suncor Energy Incorporated and Syncrude Canada Limited leases greater than seven years of age, with the exception of a single site that was approximately three years old.

The 25 natural reference (REF) wetlands were located at 6 loci across the boreal plain and were shallow open water marshes that spanned a range in salinity from fresh to subsaline (Fig. 2.1, Appendix Table A.1). The large spatial spread of the natural study sites was necessary as shallow marsh wetlands, in particular those that have elevated salinity, are rare in the boreal plain of northern Alberta (Fairbarns 1990, Trites and Bayley 2009a).

All study sites were considered Class V permanent open water wetlands (Stewart and Kantrud 1971). Three wetland vegetation zones occurred at my study wetlands: the open water, emergent, and wet meadow zone. Stewart and Kantrud (1971) refer to a shallow marsh zone, between the emergent and wet meadow zones, however my boreal wetlands had indistinguishable shallow marsh and wet meadow zones, therefore they were considered a single zone which I called wet meadow. Trites and Bayley (2009b) followed a similar protocol. The undisturbed wet meadow zone is comprised of sedge and grass species as well as obligate and facultative wetland forb species. Often this zone is flooded in the spring, however the water level drops to the sediment surface for the majority of the growing season. The emergent (or marsh) zone lies between the wet meadow and open water. This zone is comprised of robust wetland obligate emergent species, and is typically flooded for the duration of the growing season. The emergent and wet meadow zones were the focus of this study.

Wetland size (including open water, emergent, and wet meadow zones), ranged from 0.4 ha to 25 ha, with a mean of 4.3 (+/- 5.6) ha and median of 2.3 ha. I captured similar size ranges with the REF and OSREF wetlands (0.6 to 24 ha and 0.8 to 25 ha, respectively). OSPA wetlands were generally smaller in size (0.4 to 3.2ha).

Climate and interannual variability

Study wetlands were located within the same ecozone, the boreal plain, however there was a wide spatial distribution of my sites with a maximum distance of 950 km between sites. Temperature is an environmental variable that strongly influences standing crop of sedge meadows in northern and middle latitudes (Gorham 1974), so temperature and other climate factors were examined to determine if vegetation productivity patterns could be explained by the spatial extent of my study sites.

For each locus of sites the nearest meteorological station was identified, and climate data for the May-August growing season was obtained from the Environment Canada National Climate Data and Information Archive (Environment Canada 2009; Table 1). Growing degree-days (GDD) above 5ºC were calculated for the period from May 1 to August 31. Vegetation studies in the northern boreal region have previously used this as a standardized measure of heat units available for plant growth (e.g., Gorham 1974, Lumley et al. 2001, Hogg et al. 2002).

To examine if there was an effect of latitude or longitude on three climate variables (temperature, GDD, and cumulative precipitation) I performed linear regression in SPSS v.17 (SPSS Inc. 2008) to detect latitudinal and longitudinal climatic gradients. There was a weak, non-significant correlation between latitude and total May to August precipitation (R^2=0.41, *p*=0.12) and longitude and total May to August precipitation (R^2=0.44, *p*=0.10). These correlations were likely due to the above-average precipitation at the 2 Saskatchewan sites in the summer of 2007. The region received 84 mm, or one-third more precipitation than the 1971-2000 climate normal for May to August total precipitation, of 255 mm. There were no significant temperature or GDD gradients.

Interannual hydrologic variability may affect the vegetation communities of wetlands sufficiently to affect their vIBI scores on a year-to-year basis, and it has been proposed that different vIBIs may be necessary in years of drought or above average precipitation (Wilcox et al. 2002). To assess whether 2007 and 2008 vegetation communities were independent of each other I repeated sampling at 5 sites in both years, and performed a Mantel Test in PCOrd version 4 (McCune and Mefford 1999) with the randomization (Monte Carlo) method of interpretation to compare the wet meadow distance matrices, and the Bray-Curtis distance measure (Sokal and Rohlf 1995). The results of Mantel Test indicated non-independence, or similarity, of the 2007 and 2008 species composition data (n=5, r-statistic=0.374, *p*=0.19). Despite the non-significance, I excluded the 2007 repeated sites from data analyses, as I measured a greater number of variables in 2008.

Sampling technique

Vegetation data

Macrophyte data was collected in August each year, which is considered the peak biomass period of the growing season. Three transects perpendicular to the edge of open water were established at each site, roughly dividing the wetland into thirds. Transects were used as points of reference for the measurement of zone widths, water depths at zone interfaces, and vegetation quadrat placement. Perpendicular to each transect in the centre of each zone I placed 2 community composition quadrats (1 m^2) approximately 5 m apart, which gave a total of 6 community composition plots per zone per wetland. When large differences in community structure among sites were expected, as I did among oil sands and reference wetlands, 5-10 vegetation plots were recommended based on a multivariate power analysis of wetland vegetation plots (James-Pirri et al. 2007). The assessment of 6 plots per zone was chosen as a necessary trade-off between extensive sampling at each wetland and the ability to visit a greater number of geographically widespread wetlands during the short period of peak biomass. In the wet meadow zone 2 biomass clipping quadrats (0.25 m^2) were positioned between the 1 m^2 community composition plots. In the emergent zone only one 0.25 m^2 plot was clipped per transect.

All macrophyte species were identified in the 1 m^2 community composition plots, and percent cover was estimated by species. In addition, a time-restricted species diversity walk-around (Locky et al. 2005) increased the probability of encountering less common species at each site. When it was not possible to identify macrophytes in the field, voucher specimens were collected and later identified. When species could not be determined on a small number of specimens, genus was used. Nomenclature followed the Flora of Alberta (Moss 1983).

The amount of bare ground, with no vegetation growth or litter, appeared to be an important difference between sites. Hence, in 2008, a walk-around was conducted in the wet meadow zone to note at 5 m intervals whether the ground was <25% vegetated (bare) or >25% vegetated, to characterize the amount of bare ground at study sites beyond the quadrat level.

To measure vegetation production in the wet meadow and emergent zones live macrophyte aboveground standing crop was clipped to within 1 cm of the substrate surface in the 0.25 m^2 biomass plots. Biomass samples were dried to constant mass and weighed for total aboveground biomass. Visual obstruction of vegetation in the wet meadow was estimated using a Robel pole, a non-destructive measure that has been used as a surrogate for biomass in grasslands and recently wetlands (Robel 1970, Whitbeck and Grace 2006). Robel measurements were compared with aboveground biomass plot clipping using linear regression in SPSS v.17 (SPSS Inc. 2008) to determine the concordance of the two measures of aboveground biomass production. Robel height was significantly correlated with clipped aboveground biomass (R^2=0.73, p<0.0001, Fig. 2.2).

Environmental data

Water and sediment were sampled in August of each year. Water conductivity and pH were measured in situ using a handheld YSI MPS 556. To obtain a composite water column sample, water was collected from the centre of the wetland using an integrated water sampler. Water was analyzed for total nitrogen (TN), total dissolved nitrogen (TDN), total phosphorus (TP), total dissolved phosphorus (TDP), soluble reactive phosphorus (SRP), dissolved organic carbon (DOC), total dissolved solids (TDS), total suspended solids (TSS), alkalinity, anions and cations, following methods described in Bayley and Prather (2003). Naphthenic acids (NAs) in water were measured by the Fourier-transform infrared spectroscopy method (Jivraj et al. 1996).

Sediment cores were taken at 3 locations in the centre of each zone using a suction-corer to a depth of 10 cm. Sediment samples were composited by zone to account for patchiness of the sediment around the perimeter of the wetland. Sediment was homogenized, oven-dried at 60°C for 48 hours to determine water content, and analyzed for total carbon (TC) and TN by combustion in an Exeter Analytical CE40 Elemental Analyzer. Total P was determined using the peroxide/sulfuric acid digestion method (Parkinson and Allen 1975) and spectrophotometric analysis. Oil content of sediment was determined using refluxing toluene in a soxhlet extraction apparatus, described in Rooney and Bayley (2010a).

HOBO water level loggers were installed at each wetland from May to August to record water level amplitude. Secchi depth and mean water depth were taken in the open water zone at each site. Open-water area was determined from aerial photographs and satellite imagery.

vIBI approach

Development of the vIBI was done in multiple stages (Appendix Figure A.1). The first stage identified and calculated metrics from vegetation data collected in the field. Following this, metrics were tested for response to an abiotic gradient of disturbance, with highly correlated metrics being selected for inclusion in the vIBI. This ensured that the vIBI would reflect the underlying physical and chemical conditions of each site. Metrics were then scored based on the range of their actual values, which allowed metrics that had varying scales, and positive or negative correlations with disturbance to be combined. Next, the total vIBI scores were calculated by combining the individual metric scores to create the multimetric index. Testing the total vIBI scores against the abiotic disturbance gradient then validated the vIBI, by ensuring that the metrics, in combination, were effective in diagnosing ecological health. Finally, the full range of vIBI scores was divided into categories of ecological health, from poor to good, using undisturbed natural wetlands as the benchmark for ecologically healthy wetlands.

Metric identification

An extensive list of metrics was developed from vegetation data collected in the field. Metrics with high sensitivity to disturbance have been shown to be transferable among jurisdictions (Karr and Chu 1999, pp. 57), thus lists of vegetation metrics from previous work developing IBIs across North America (e.g., Simon et al. 2001, Fennessy et al. 2002, DeKeyser et al. 2003, Miller et al. 2006, Brazner et al. 2007) were used as guides when considering metrics for this study. Metrics based on the Floristic Quality Assessment Index (FQAI; Wilhelm and Ladd 1988), a plant-based measure of habitat quality, are often used in plant-based IBIs in jurisdictions where species have been assigned coefficient-of-conservatism (C) values. The methods and rationale for assignment of C-values is described in Andreas and Lichvar (1995), and was recently done for wetland plant species in the aspen parkland and boreal regions of Alberta (Forrest 2010). As such, I included mean site C-value, the FQAI, and the adjusted FQAI (Miller and Wardrop 2006) as candidate metrics.

The full list of calculated metrics include measures of vegetation community structure (e.g, vegetation zone width, FQAI), vegetation functional groups (e.g., perennial species richness, % obligate wetland species), and species-specific metrics (e.g., proportion cover *Carex aquatilis*). Both untransformed and arcsin square root transformed data were calculated for metrics in the wet meadow, emergent and combined wet meadow/emergent zones. This is a standard and relatively straightforward data transformation for proportional data (Sokal and Rohlf 1995).

Metric testing and selection

Candidate metrics for an IBI are tested for their response (positive or negative) to disturbance using dose-response curves (Karr and Chu 1999, pp. 48, Mack et al. 2000, Miller et al. 2006), which assesses the correlation of metrics to a gradient of wetland condition from healthy to disturbed. Typically sites have been ranked along this gradient using combinations of best professional judgment (e.g., Simon et al. 2001, Helgen and Gernes 2002, DeKeyser et al. 2003, Hering et al. 2006), landscape-scale factors such as land-use intensity (e.g., Carlisle et al. 1998, Miller et al. 2006), and rapid assessment methods such as those developed for use in Ohio, USA (Mack et al. 2000, Mack 2007). DeKeyser et al. (2003) identified the use of best professional judgement to rank individual wetland condition as a weakness in their methodology.

For this study, sites were assigned scores on an abiotic, physicochemical disturbance gradient that spread them across a range of disturbance, and largely avoided the use of best professional judgement. Using the same sites as I did, Rooney and Bayley (2010b) calculated site disturbance scores to quantify the relative level of disturbance among all study sites on a synthetic physicochemical disturbance gradient.

Metrics were tested for correlation with the disturbance gradient by linear regression using SPSS version 17.0 (SPSS Inc. 2008). Metrics that correlated well with disturbance (R^2>0.25) were selected for inclusion in the multimetric index. Certain metrics were measured in 2008 only, which excluded them from being used in the final index calculation. If they were highly correlated with the disturbance gradient they were listed as metrics showing strong potential for future vIBI development. As there is the possibility for redundancy among metrics (e.g., among vegetation height and aboveground biomass), especially when testing a large number of similar attributes, redundant metrics were eliminated to prevent over-emphasis of differences in the final vIBI scores.

Metric scoring

Before calculating the final vIBI scores metric values must normalized. A scoring system is used in order to weight all metrics equally, before they are combined. I chose to score metrics based on 3 value ranges that represent the vegetation metric response to low, medium, and high disturbance, as this is the most common method for scoring metrics in the literature (e.g., Reynoldson et al. 1997, DeKeyser 2000, Simon et al. 2001, Keleher and Rader 2008). There are, however, multiple ways to define the 3 value ranges, which include multivariate techniques such as cluster analysis (e.g., DeKeyser et al. 2003) and trisection of the range of metric scores (e.g., Helgen and Gernes 2001, Simon et al. 2001, Keleher and Rader 2008).

I chose to examine 3 techniques for assigning value ranges to score metrics. I used a straightforward trisection approach to the range of observed metric values, which I termed the 'metric range' (MR) method. I scored each range 1, 3, or 5; 1 being a value expected when a site is highly disturbed, 5 being a value expected at a site that is relatively undisturbed. In the second technique, I calculated value ranges by trisection of the disturbance gradient, which I termed the 'disturbance gradient range' (DGR) method. The median metric value within each of the 3 disturbance gradient bins was calculated, and the midpoint between each median then defined the value ranges for individual metrics to assign scores of 1, 3, and 5. Lastly, I used multivariate cluster analysis to create value ranges, which was termed the 'metric clustering' (MC) method. This was done by clustering sites based on 7 non-redundant metrics selected for potential inclusion in the vIBI (Arcsin√x transformed proportion *Sonchus* spp., *Equisetum* spp., and *Carex atherodes* cover, proportion total vegetation cover, Adjusted FQAI, and log transformed zone width and aboveground biomass). Hierarchical clustering was done in PCOrd version 4 (McCune and Mefford 1999) with the Bray-Curtis distance measure and flexible beta (-0.25) linkage type. There was low chaining (4.5%), and the dendrogram was pruned at 70% information remaining to yield 3 clusters. The 3 clusters were defined as low, medium and highly disturbed by ranking them by the mean disturbance score of each cluster. The median metric value for each cluster was calculated, and the value ranges for 1, 3, and 5 were then determined using the same method as for the DGR. The low numbers associated with many of the metric values resulted in the median value of the lowest third of the range often being 0. In these cases a score of either 1 or 5, depending on whether the metric response to stress was positive or negative, was given to values of 0, while any value between zero and the upper cutoff for the medium disturbance category was given a score of 3.

vIBI calculation and verification

Once each metric is scored 1, 3 or 5, the sum of the metric scores at a particular site becomes the vegetation community score, or the vIBI score. Two vIBI tools were created from the selected metrics, a basic vIBI (BvIBI) and an advanced vIBI (AvIBI). The BvIBI was composed of metrics that could easily be measured in the field by a technician without botanical expertise such as species identification skills. The AvIBI was composed of the BvIBI metrics as well as additional metrics that would require a more intensive sampling methodology, longer periods of time, and species identification skills by the field technician. The vIBI scores for sites were then assessed for correlation with the disturbance gradient using linear regression to determine whether these tools, as sums of metric scores, correlate well with disturbance. In order to compare the order of site ranking by the AvIBI and BvIBI, I used the Wilcoxon Signed Ranks test.

vIBI health classification

The final step in developing the vIBI is to identify appropriate wetland health categories for ranges of vIBI scores, and set the level of performance required for a wetland to be considered reclaimed. I chose to do this by classifying wetlands within 2 standard deviations below the mean REF site vIBI score as fair health. Two standard deviations roughly approximates the 95% confidence intervals for inclusion in the range of the REF site distribution, and approximates the range of variation in health of the reference wetlands. Wetlands that score above the mean REF site value are considered to demonstrate good health, and wetlands that score lower than 2 standard deviations below the mean REF site score were considered to be poor health. Wetlands were considered reclaimed when in fair or good health.

2.3 Results

General vegetation results

The total number of species (or genus when species was not determined) encountered at the 45 study wetlands, including quadrat and walk-around assessments, was 166 (Appendix Table A.2). Of these, 105 species are considered by the USDA to have either obligate or facultative wetland indicator status in the northwest region of North America (USDA, NRCS 2010). Within the species composition plots I found a total of 134 species in the wet meadow and 53 species in the emergent zone. Aboveground biomass in each vegetation zone varied widely among sites, from 53 g m^{-2} to 988 g m^{-2} in the wet meadow and from 51 g m^{-2} to 1128 g m^{-2} in the emergent zone.

Metric identification, testing, and selection

To select metrics for inclusion in the vIBI, greater than 600 individual metrics calculated from data collected in 2007 and 2008 were tested for correlation with the disturbance gradient. Of the tested metrics 26 had R^2>0.25 (Table 2.2). These included metrics from all 3 metric categories: vegetation community structure, vegetation functional groups, and species-specific metrics.

The majority of metrics (90%) with R^2 values >0.25 were from the wet meadow zone. To simplify field data collection for future use of this tool, when there was a redundancy issue among metrics from the wet meadow and combined wet meadow/emergent zones I eliminated the combined zone metric. All combined zone metrics were eliminated in this way, and the index from this point on focused entirely on wet meadow vegetation.

Robel height and proportion bare ground were measured at fewer than 45 sites, and were not included in the final vIBI as they could not have been incorporated without decreasing my wetland sample size, however I calculated their value ranges for potential inclusion in a future version of the vIBI. Robel height and aboveground biomass are redundant; both are estimate vegetation standing crop. I included only clipped aboveground biomass as a standing crop metric in the final vIBIs.

The two metrics, proportion bare ground and total percent cover were not considered redundant due to the different vegetation community attributes they were measuring, and the different methods by which they were measured. Total percent cover was measured on a fine scale within 1 m^2 quadrats, whereas proportion bare ground was a more qualitative structural metric, characterizing the amount of open bare ground within the wet meadow zone by walking the perimeter.

After selecting metrics and removing redundancy, 11 metrics remained (Table 2.3). The redundant Robel height and proportion bare ground, which were not used in the construction of the vIBI, were still carried through to the next steps in case they may be used in later versions of the vIBI. Four of the final 11 metrics had positive responses to increasing disturbance, and 7 had negative responses to increasing disturbance.

Metric scoring, vIBI calculation, and verification

Metrics were scored based on value ranges that were calculated using 3 different techniques (Table 2.4), I devised 2 versions of the vIBI (Table 2.5) and calculated final vIBI scores using each of the 3 metric scoring methods (Table 2.6). The AvIBI contained the greatest number of metrics, and was developed by selecting the 6 metrics that had the greatest correlations with the disturbance gradient and were non-redundant. Additional metrics added to the AvIBI did not improve the correlation with the disturbance gradient, thus the number of metrics was kept to 6. The second index was the BvIBI, and the intent of this index was to provide an assessment tool that could be easily and rapidly executed without extensive botanical knowledge (particularly species identification skills). The BvIBI is composed of 3 metrics, and while one of these is total aboveground biomass, which requires laborious clipping, drying, and weighing of vegetation, this metric is intended as a surrogate for Robel height which could not be included due to lack of data in 2007. Proportion of bare ground could be added to this index as a fourth metric.

The AvIBI R^2 correlation with the disturbance gradient was similar for the 3 metric scoring methods, and all 3 methods were significantly and highly correlated with the disturbance gradient (Table 2.6). The BvIBI scores had lower correlations with the disturbance gradient than the AvIBI, but were still significant. The disturbance gradient range (DGR) method of scoring produced the highest R^2 values, followed by the metric clustering (MC) method, and then the metric range (MR) method. The R^2 correlations of the AvIBI and BvIBI with the disturbance gradient were higher than those observed for any individual metric used to build each vIBI. The ranked ordering of sites by the AvIBI and BvIBI according to health was not significantly different (Wilcoxon Signed Ranks test; $Z=-0.282$, $p=0.778$).

vIBI health classification

The DGR, MR, and MC methods required similar ranges of AvIBI scores to classify sites as fair or good health, whereas for the BvIBI, the DGR method required higher scores to rate sites as fair or good health (Table 2.7). I chose to focus the remaining analyses on the DGR scoring method, as it had higher R^2 correlations with the disturbance gradient than the other two methods, and was more conservative in placing sites in the fair or good health classes for the BvIBI scores. The three bins of wetland health distributed the sites among health classes fairly evenly (Fig. 2.3, Appendix Table A.1), with the AvIBI scoring 13 of the 45 sites in good health, 17 sites in fair health, and 15 sites in poor health. The BvIBI scored 18 sites in good health, 11 sites in fair health, and 16 sites in poor health. Greater than 60% of all sites were considered fair or good health, regardless of which vIBI was used, however this was not surprising, as 56% of study sites were of the natural REF treatment, and considered relatively free of anthropogenic disturbance. Despite the massive mining and landscape disturbance, the AvIBI scored 5 OSREF sites and 1 OSPA site as fair or good health, and the BvIBI scored 4 OSREF and 1 OSPA site as fair or good health. The AvIBI and BvIBI classified one REF site as being in poor health. This site was sampled in a year of above average rainfall, and was affected by shoreline disturbance from off-road vehicles around part of the perimeter.

2.4 Discussion

My work indicates that the vIBI has the potential to be a valuable tool for reclamation managers and government regulators to assess the health of reclaimed wetlands in the oil sands. Assessment of reclaimed oil sands wetlands using the vIBI demonstrates that it is possible for reclaimed wetlands to reach levels of ecological health within the range of natural, undisturbed wetlands, although most reclaimed sites do not perform to this standard.

Metric identification, testing, and selection

Only a small fraction of all metrics I tested had acceptable correlations (R^2>0.25) with the disturbance gradient, and were included in the final multimetric vIBIs. Dale and Beyeler (2001) state that ecological health requires the presence of appropriate species, populations, and communities, and the occurrence of ecological processes at appropriate rates and scales. The AvIBI includes within its 6 metrics measures of productivity, vegetation community structure, and presence of sensitive or invasive species, and effectively touches on all aspects of ecological health identified by Dale and Beyeler (2001).

My measure of vegetation production as harvested and dried aboveground growth over the growing season has been used by few other studies as a metric. This is likely due to the laborious and destructive nature of collecting and processing plant tissue. The strong negative correlation of aboveground biomass and the disturbance gradient indicate that some measure of vegetation productivity, an important ecosystem process, should not be excluded when developing a vIBI to assess reclaimed wetland health. The clipped aboveground biomass metric could be replaced by Robel height, which had even higher correlation with the disturbance gradient (Table 2.2). This is unlikely to weaken the overall vIBI, and could reduce field sampling and laboratory preparation time, as well as offer a non-destructive method of quantifying this ecological process.

Wet meadow vegetation zone width, a vegetation structure metric, is influenced by the slope of the open water to upland transition (Zampella and Laidig 2003, Forrest 2010). While natural wetlands occasionally have narrow wet meadow zones due to their landscape setting, sites at the disturbed end of the spectrum have narrower wet meadow zones overall, as indicated by the negative correlation of this metric with increasing disturbance. If the goal of reclamation is to establish wetlands that are functionally equivalent to those found in natural settings then the overall vIBI score should increase if wetlands are being constructed in a way that promotes the establishment of wide wet meadow zones on gradually sloping wetland to upland transitions.

A second vegetation structural metric, though not included in final vIBI scores because I lacked data in 2007, is proportion of bare ground. This coarse assessment of vegetation cover at the zone-level, rather than quadrat-level could be indicative of a number of natural or anthropogenic disturbances. Flooding and water level drawdown, due to such events as above or below average precipitation or beaver activity can increase the area of bare ground in the wet meadow zone. While this is a natural process it is important to capture the natural range of disturbance in developing the vIBI to avoid bias against reclaimed wetlands in the ecological assessment process. In the oil sands reclamation landscape bare ground is likely more indicative of such stresses as contamination, unsuitability of substrate, or dispersal limitation of vegetation propagules. At a number of reclaimed sites the exposed sediment had noticeable contamination by thick oil, or was fine grained and fully saturated composite tailings material, an unstable surface for vegetation growth (Cooper 2004). As oil sands wetlands will likely be constructed from raw materials, with the expectation that vegetation will colonize and stabilize raw substrates (Johnson and Miyanishi 2008), intuitively the proportion of bare ground should decrease as reclaimed sites age, and approach functional equivalency to natural reference wetlands. To control for the lack of vegetation in very young sites, nineteen of the 20 reclaimed wetlands in this study were greater than 7 years of age. In fact the average age of all oil sands reclaimed wetlands (OSREF and OSPA) was 16 years. Thus, the higher amount of bare ground encountered at oil sands reclaimed sites may be more indicative of unsuitability of substrate than low colonization and establishment due to young age.

Proportion of total vegetation cover at the quadrat-level was not considered redundant with proportion of bare ground. Should any redundancy exist, it would not affect the vIBIs I developed, as the proportion of bare ground was excluded from the calculations. Proportion of total vegetation cover provides a fine-scale examination of vegetation cover within the wet meadow zone, and provides insight into revegetation and colonization success by assessing stem density within the vegetated zone.

The adjusted FQAI metric is a measure of habitat quality derived from the plant community richness, ecological conservatism of species (where ecological conservatism is a measure of plant species tolerance to disturbance), and presence of non-native species (Miller and Wardrop 2006). The calculation of the adjusted FQAI requires all species that are encountered to have pre-defined coefficients of conservatism, and is truly independent of the other metrics. A drawback of this metric is the potential that new species not yet assigned coefficients of conservatism may be encountered. In such cases expert botanical opinion would be required to assign values to these species.

It is not surprising that metrics based on relative species abundances were better correlated with the disturbance gradient than those based on absolute abundance, as natural reference sites can have fewer species than those that experience intermediate disturbance (Simon et al. 2001). This trend may be particularly true in the boreal plain, as I found near-monocultures of graminoids at many of the natural sites, with much lower numbers of many other species.

Relative diversity of dicot species is a measure of herbaceous vegetation, and this metric decreased with increasing levels of disturbance. Higher relative diversity could be indicative of either higher numbers of dicot species, or lower monocot species richness, which are primarily graminoids. The natural sites, at the healthy end of the spectrum, had fairly low graminoid species richness, and this may be causing the relationship.

Relative cover of invasive species predictably increased with disturbance. Of the species encountered in this study, only 1 species and 1 genus were considered invasive (*Cirsium arvense* and *Sonchus* spp.). Species regulated in Alberta as either restricted or noxious (Alberta Weed Control Act 2001) were considered to be invasive to the riparian zone by the local Cows and Fish program (Alberta Riparian Habitat Management Society 2007, Appendix Table A.3), and I used this list to remain consistent with other ecological monitoring programs in Alberta.

Although certain species-specific metrics had acceptably high correlations with disturbance (transformed proportion cover *Sonchus* spp., *Equisetum* spp., *Carex atherodes*), I eliminated them from the final vIBI tools. The broad geographic range across which the reference sites were distributed, and the potential for individual species to vary in abundance across this range, means metrics that examine vegetation functional groups rather than particular species may be more robust.

Metric scoring

The scoring of metrics is a crucial step in integrating ecosystem attributes that have differing units and scales into a single vegetation index. There are multiple ways to score metric values as low, medium, and high disturbance, and my choice of scoring method was made after examining 3 possibilities. Splitting of the full disturbance gradient into 3 bins (DGR method) to score metrics provided the highest R^2 correlation between the vIBIs and the disturbance gradient, and this method was the most conservative in classifying sites in fair or better health. The outcomes of oil sands wetland reclamation are not fully known (Harris 2007, Johnson and Miyanishi 2008), and reclamation trajectories can often be unpredictable within the first decade following reclamation (Matthews et al. 2009). In light of this, I chose the metric scoring technique that was most conservative in diagnosing ecological health.

vIBI verification

Both the AvIBI and BvIBI were highly and significantly correlated with abiotic disturbance (AvIBI R^2=0.68, p<0.001; BvIBI R^2=0.57, p<0.001). This level of correlation was considered good compared to other studies validating IBIs by assessing their correlation with gradients of disturbance (Mack et al. 2000, Miller et al. 2006, Keleher and Rader 2008, Stevens and Council 2008).

By developing two tools, one of which can be calculated using data collected in the field by a technician with minimal training in species identification, and one that requires more advanced botanical expertise, two tiers of assessment are possible. Reclamation success can be monitored on a more frequent basis by less-skilled technicians using the BvIBI to track site scores and determine if wetland vegetation communities are approaching certifiable levels of ecological health. When wetlands score within the fair to good range of health, then field technicians for government or other regulatory agencies can collect data for the calculation of the AvIBI score. If wetlands are failing to approach acceptable levels of health, individual metric scores can be used to identify deficiencies in development or design, and modify sites to guide recovery. The concordance of the two vIBIs, as confirmed by the Wilcoxon Rank Sums test, verifies that these tools can be used in tandem and will likely chart similar courses towards ecological health during long term monitoring, which is being seen as increasingly important in ensuring reclamation success (Campbell et al. 2002, Gutrich et al. 2009).

vIBI health classification

Requiring reclaimed wetlands to reach a level of function similar to the lower range of natural wetlands sets a more realistic goal for reclamation, rather than requiring sites reach a level of function near that of the healthiest natural sites. It is well documented that restored and reclaimed wetlands struggle to reach an equivalent level of health to pre-existing natural wetlands, at least within the initial 5-10 years following reclamation (Campbell et al. 2002, Seabloom and van der Valk 2003, Spieles 2005, Gutrich et al. 2009). This is due in part to the persistence of non-native or terrestrial species (Campbell et al. 2002, Spieles 2005, Aronson and Galatowitsch 2008) and slow soil development, a function of the macrophyte community biomass production and decomposition (Cole et al. 2001, Balcombe et al. 2005, Ballantine and Schneider 2009). Furthermore, the natural range of variability in local wetland condition can be high, both spatially and temporally. Yearly climatic and precipitation variability affects the biological community and thus the perceived ecological health of natural undisturbed wetlands (Wilcox et al. 2002, Euliss et al. 2004, van der Valk 2005) and can cause changes in local environmental conditions such as salinity (Arndt and Richardson 1993). Wet-dry climatic cycles affect overall wetland area and connectivity in boreal Alberta (Sass and Creed 2008), and in the prairie pothole region to the south (Kahara et al. 2009). Above average rainfall affected the 2 easternmost study sites, visibly reducing the area of wet meadow due to inundation. This natural, but destructive event underscores the necessity of accounting for such variation in the natural reference set of wetlands when setting them as benchmarks of fully functioning, healthy wetlands.

The AvIBI and BvIBI identified the same 3 oil sands sites as the healthiest reclaimed wetlands sampled in this study, however there was no overlap of the next 3 healthiest sites identified by each vIBI. This discrepancy could be attributed to the lack of vegetation functional group metrics included in the BvIBI, which assessed only vegetation presence and productivity. The AvIBI, being the more comprehensive tool, with a greater number of metrics that assess not only presence of a vegetation community but also the composition of the community, should be considered the more reliable method of assessing ecological health. The use of too few metrics has been cautioned against, as they may not encompass the full complexity of the system being assessed (Dale and Beyeler 2001).

Of 6 oil sands wetlands identified by the AvIBI as being in fair to good ecological health, 3 were 15 years of age or younger at the time of sampling. While it has been suggested that mitigation projects should be given 15-20 years post-reclamation before they are judged for their success (Mitsch and Wilson 1996), 6 oil sands sites found to be in poor ecological health were greater than 15 years old, indicating that age alone cannot guarantee the development of a healthy vegetation community. I therefore recommend that reclamation monitoring begin earlier than the suggested 15-20 year age, as it could become clear early in the process whether sites are developing healthy vegetation communities. Those sites that are responding poorly may benefit from early intervention and further reclamation activity (Gutrich et al. 2009).

General discussion and conclusions

The final phase of the vIBI implementation will be to test the index on an independent set of reclaimed wetlands for its reliability in predicting disturbance scores. Reclamation of wetlands on oil sands leases is still in its infancy (Harris 2007), and it may take a number of years before such sites are available, and the vIBI can be tested.

The use of multimetrics for ecosystem assessment has occasionally been criticized for its perceived disadvantages: they may discard information, some metrics are redundant, and some can compound error (Reynoldson et al. 1997). My procedure in developing the oil sands vIBI took steps to avoid these pitfalls by ensuring metrics captured both structural and compositional elements of the vegetation community, as well as eliminating potentially redundant metrics.

Another criticism is that information may be lost when many variables are collapsed into a single index could mask real differences between groups of sites (McCoy and Mushinsky 2002), or that an overall good score could mask low scores of individual metrics (Paul 2001). Karr and Chu (1999) argue that the IBI condenses, integrates, and summarizes information, but does not lose it. As well, there is a tradeoff between detailed vegetation community assessment, and rapid assessment. The purpose of this vIBI is to facilitate the relatively rapid assessment of reclaimed wetland health, with respect to appropriate natural analogues of ecologically healthy wetlands. Wetlands that score as fair may warrant a more detailed examination of individual metric scores to determine if a particular metric, or indicator category is performing below minimum expectations.

One difficulty in using wetlands currently found on oil sands leases to develop an assessment tool is the possibility that reclamation practices and materials being used to construct wetlands will change in the coming years. Mining, extraction, and reclamation methods are constantly evolving as new technologies are developed. Thus far only a fraction of disturbed land has been reclaimed, and an even smaller fraction of this reclaimed land is wetland. The oil sands sites that currently score as ecologically healthy are possibly not representative of the eventual reclaimed wet-landscape, which will utilize enormous stockpiles of process-affected materials and tailings water in their construction. Upwards of 75 000 ha of wetlands will need to be reclaimed, and the development of this vIBI is an important first step in establishing an approach for monitoring wetland reclamation success in the Alberta oil sands.

2.5 Literature cited

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Table 2.1. Environment Canada climate data for the May 1 – August 31 growing season for all site loci visited in 2007 and 2008.

Locus	Year	Station latitude	Station longitude	Station elevation (m)	Avg. daily max temp. (ºC)	Avg. daily min temp. (ºC)	Mean temp. (ºC)	Total precip. (mm)	Degree days (>5ºC)
La La Biche	2007	54° 46.2' N	111° 58.2' W	559	20.7	8.2	14.5	195.2	2470
Utikuma	2007	55° 54.0' N	115° 45.0' W	678	19.4	4.8	12.0	260.8	2187
Oil sands	2007	57° 2.4' N	111° 33.6' W	310	21.3	9.9	15.6	163.5	2631
Saskatchewan	2007	53° 13.2' N	105° 40.2' W	428	21.5	8.6	15.1	339.4	2603
Child Lake	2007	58° 22.8' N	116° 2.4' W	289	21.9	8.6	15.3	186.2	2656
Oil Sands	2008	57° 2.4' N	111° 33.6' W	310	21.7	10.5	16.1	175.1	2728
Child Lake	2008	58° 22.8' N	116° 2.4' W	289	22.9	8.9	15.9	145.7	2777
Elk Island	2008	53° 41.0' N	112° 52.1' W	716	22.0	7.8	15.0	192.9	2611
Hay River	2008	58° 37.3' N	117° 9.9' W	338	21.1	7.3	14.2	160.8	2483
Mean (+/- SD)	-	-	-	474 (178)	21.3 (1.0)	8.0 (1.7)	14.7 (1.4)	210 (68)	2551 (197)

Table 2.2. Metric correlations with the disturbance gradient. All metrics from the wet meadow (WM) and combined wet meadow/emergent (WMEM) zones with R^2 correlations >0.25 are listed.

Metric category	Vegetation zone	Metric	R^2 (*p*-value)	n
Community structure	WM	Robel height	0.61 (<0.001)	16
	WM	Proportion bare ground	0.50 (<0.001)	26
	WM	Adjusted FQAI	0.42 (<0.001)	45
	WM	Total aboveground biomass	0.42 (<0.001)	45
	WM	Proportion total vegetation cover	0.36 (<0.001)	45
	WMEM	Adjusted FQAI	0.35 (<0.001)	45
	WM	Vegetation zone width	0.31 (<0.001)	45
	WM	Mean C-value	0.28 (<0.001)	45
Functional group	WMEM	Relative diversity dicot spp.	0.43 (<0.001)	45
	WM	Relative diversity dicot spp.	0.41 (<0.001)	45
	WM	Aboveground biomass obligate wetland spp.	0.39 (<0.001)	45
	WM	Proportion total perennial spp, cover	0.38 (<0.001)	45
	WM	Proportion monocot spp. cover	0.36 (<0.001)	45
	WMEM	Relative diversity monocot spp.	0.33 (<0.001)	45
	WM	Relative diversity monocot spp.	0.30 (<0.001)	45
	WM	Relative cover invasive spp.	0.29 (<0.001)	45
	WM	Aboveground biomass *Carex* spp.	0.28 (<0.001)	45
	WM	Proportion obligate wetland spp. cover	0.27 (<0.001)	45
	WM	Proportion non-native spp.	0.27 (<0.001)	45
	WM	Proportion disturbance-tolerant spp. cover	0.25 (<0.001)	45
Species-specific	WM	Proportion *Sonchus* spp. cover (Arcsin$\sqrt{x}$)	0.38 (<0.001)	45
	WM	Relative cover *Sonchus* spp.	0.33 (<0.001)	45
	WM	Proportion *Scirpus validus* cover (Arcsin$\sqrt{x}$)	0.31 (<0.001)	45
	WM	Proportion *Equisetum* spp. cover (Arcsin$\sqrt{x}$)	0.30 (<0.001)	45
	WM	Proportion *Sonchus* spp. cover	0.29 (<0.001)	45
	WM	Proportion *Carex atherodes* cover (Arcsin$\sqrt{x}$)	0.27 (<0.001)	45

Table 2.3. Wet meadow vegetation metric direction of response to disturbance (RTD), mean value, standard error, range, and correlation by linear regression with the disturbance gradient for 45 natural boreal and oil sands reclaimed wetlands.

Metric	RTD	Mean (±SE)	Range	R^2 (*p*-value)	n
Robel height (cm) *	(-)	47 (7.6)	5-92	0.61 (<0.001)	16
Proportion bare ground *	(+)	0.13 (0.047)	0-0.88	0.50 (<0.001)	26
Adjusted FQAI	(-)	38 (0.5)	29-44	0.42 (<0.001)	45
Total aboveground biomass ($g\ m^{-2}$)	(-)	454 (35.4)	53-988	0.42 (<0.001)	45
Vegetation zone width (cm)	(-)	1920 (255)	140-7000	0.31 (<0.001)	45
Proportion total vegetation cover	(-)	0.64 (0.033)	0.09-1.00	0.36 (<0.001)	45
Relative diversity dicot spp.	(-)	0.56 (0.019)	0.18-0.72	0.41 (<0.001)	45
Relative cover invasive spp.	(+)	0.03 (0.007)	0-0.25	0.29 (<0.001)	45
Proportion *Sonchus* spp. cover (Arcsin√x)	(+)	0.04 (0.007)	0-0.18	0.38 (<0.001)	45
Proportion *Equisetum* spp. cover (Arcsin√x)	(+)	0.02 (0.004)	0-0.12	0.30 (<0.001)	45
Proportion *Carex atherodes* cover (Arcsin√x)	(-)	0.21 (0.032)	0-0.67	0.27 (<0.001)	45

* metrics included for calculation of value ranges only

Table 2.4. Value ranges for scoring metrics before inclusion in final vIBI tools. The 3 value range calculation methods are shown: disturbance gradient range (DGR), metric clustering (MC), and metric range (MR).

Metric	Score	Value range method		
		DGR	MR	MC
Robel height (cm)*	1	<27	<29	<21
	3	27-56	29-53	21-48
	5	>56	>53	>48
Proportion bare ground *	1	>0.30	>0.60	>0.40
	3	>0-0.30	0.3-0.60	>0-0.40
	5	0	<0.30	0
Adjusted FQAI	1	<36	<34	<35
	3	36-39	34-39	35-40
	5	>39	>39	>40
Total aboveground biomass ($g\ m^{-2}$)	1	<289	<364	<338
	3	289-518	364-676	338-513
	5	>518	>676	>513
Vegetation zone width (cm)	1	<800	<2430	<720
	3	800-2190	2430-4710	720-2480
	5	>2190	>4710	>2480
Proportion total vegetation cover	1	<0.51	<0.42	<0.46
	3	0.51-0.72	0.42-0.74	0.46-0.75
	5	>0.72	>0.74	>0.75
Relative diversity dicot spp.	1	<0.51	<0.36	<0.54
	3	0.51-0.63	0.36-0.54	0.54-0.61
	5	>0.63	>0.54	>0.61
Relative cover invasive spp.	1	>0.03	>0.17	>0.03
	3	>0-0.03	0.08-0.17	>0-0.03
	5	0	<0.08	0
Proportion *Sonchus* spp. cover (Arcsin√x)	1	>0.060	>0.120	>0.050
	3	0.010-0.060	0.060-0.120	0.010-0.050
	5	<0.010	<0.060	<0.010
Proportion *Equisetum* spp. cover (Arcsin√x)	1	>0.020	>0.079	>0.013
	3	>0-0.020	0.039-0.079	>0-0.013
	5	0	<0.039	0
Proportion *Carex atherodes* cover (Arcsin√x)	1	<0.078	<0.222	<0.112
	3	0.078-0.251	0.222-0.444	0.112-0.167
	5	>0.251	>0.444	>0.167

* metrics included for calculation of value ranges only

Table 2.5. Metrics selected for inclusion in final AvIBI and BvIBI tools following removal of redundant metrics.

Index	Component metrics
AvIBI	Adjusted FQAI
	Total aboveground biomass
	Vegetation zone width
	Proportion total vegetation cover
	Relative diversity dicot spp.
	Relative cover invasive spp.
BvIBI	Total aboveground biomass
	Vegetation zone width
	Proportion total vegetation cover

Table 2.6. vIBI validation by linear regression with the disturbance gradient, mean vIBI scores, and potential range of scores.

Index	Value range method	R^2 (*p*-value)	Mean score	Score range
	DGR	0.68 (<0.001)	19	6-30
AvIBI	MR	0.64 (<0.001)	20	6-30
	MC	0.65 (<0.001)	19	6-30
	DGR	0.57 (<0.001)	10	3-15
BvIBI	MR	0.50 (<0.001)	8	3-15
	MC	0.57 (<0.001)	9	3-15

Table 2.7. Wetland health categories for vIBI scores indicated by reference site confidence intervals.

Index	Health	Value range method DGR	MR	MC
	Good	24-30	24-30	24-30
AvIBI	Fair	18-22	18-22	16-22
	Poor	6-16	6-16	6-14
	Good	13-15	11-15	13-15
BvIBI	Fair	9-11	5-9	7-11
	Poor	3-7	3	3-5

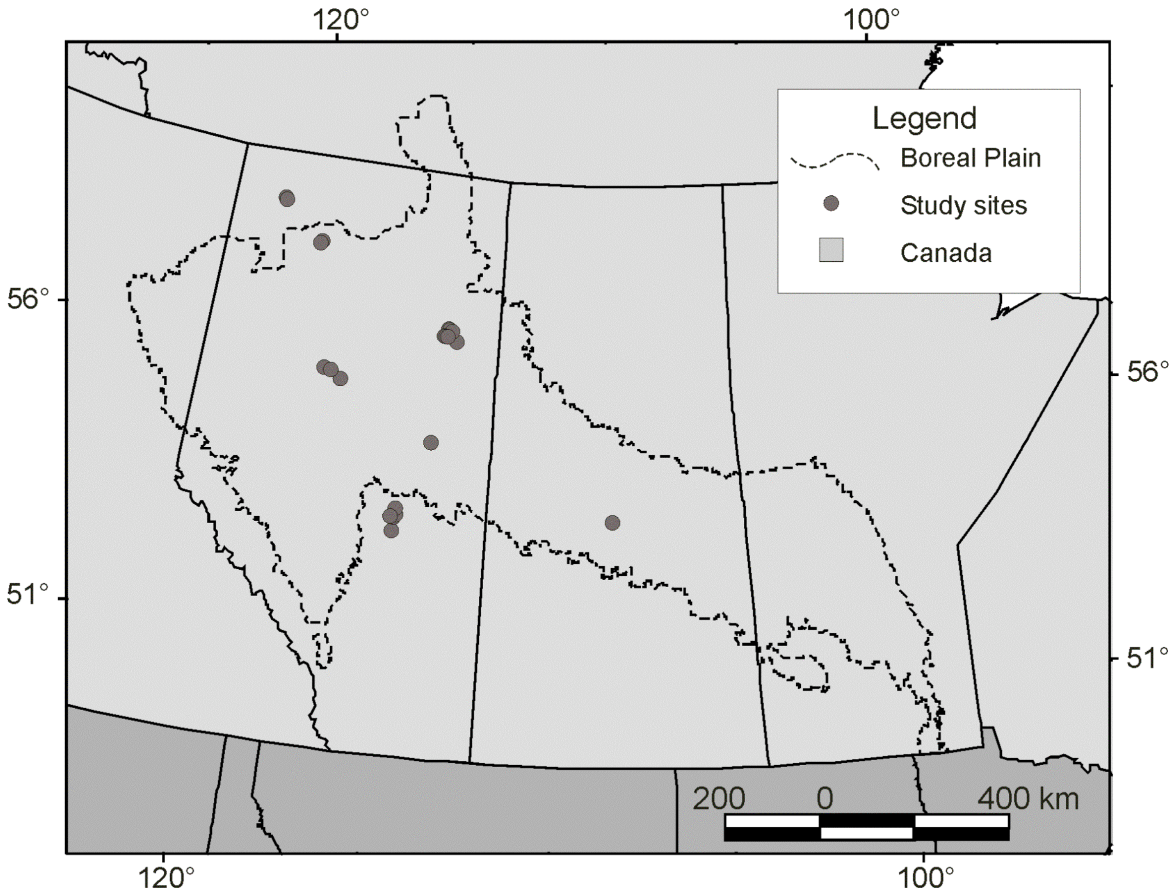

Figure 2.1. Study site locations within the boreal plain ecozone of northern Alberta and Saskatchewan. Sites located to the south of the zone in central Alberta are found within a small pocket of boreal forest, discontinuous with the rest of the zone. Sites to the north of the boreal plain in northern Alberta are within a zone of transition to the taiga plains.

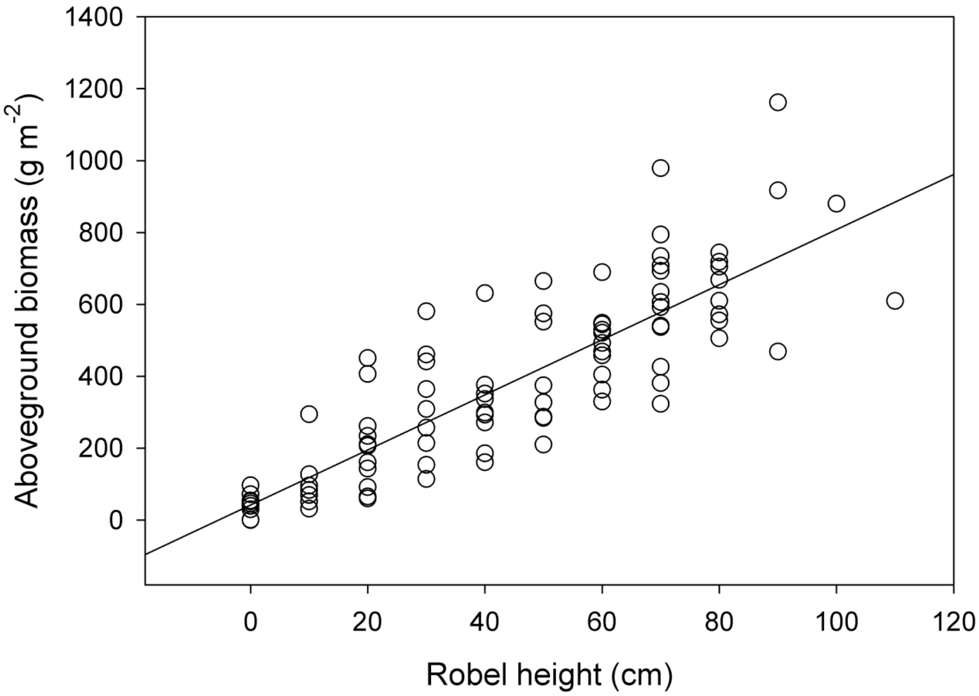

Figure 2.2. Robel height correlation with wet meadow aboveground biomass. n=96, R^2=0.73, p<0.0001.

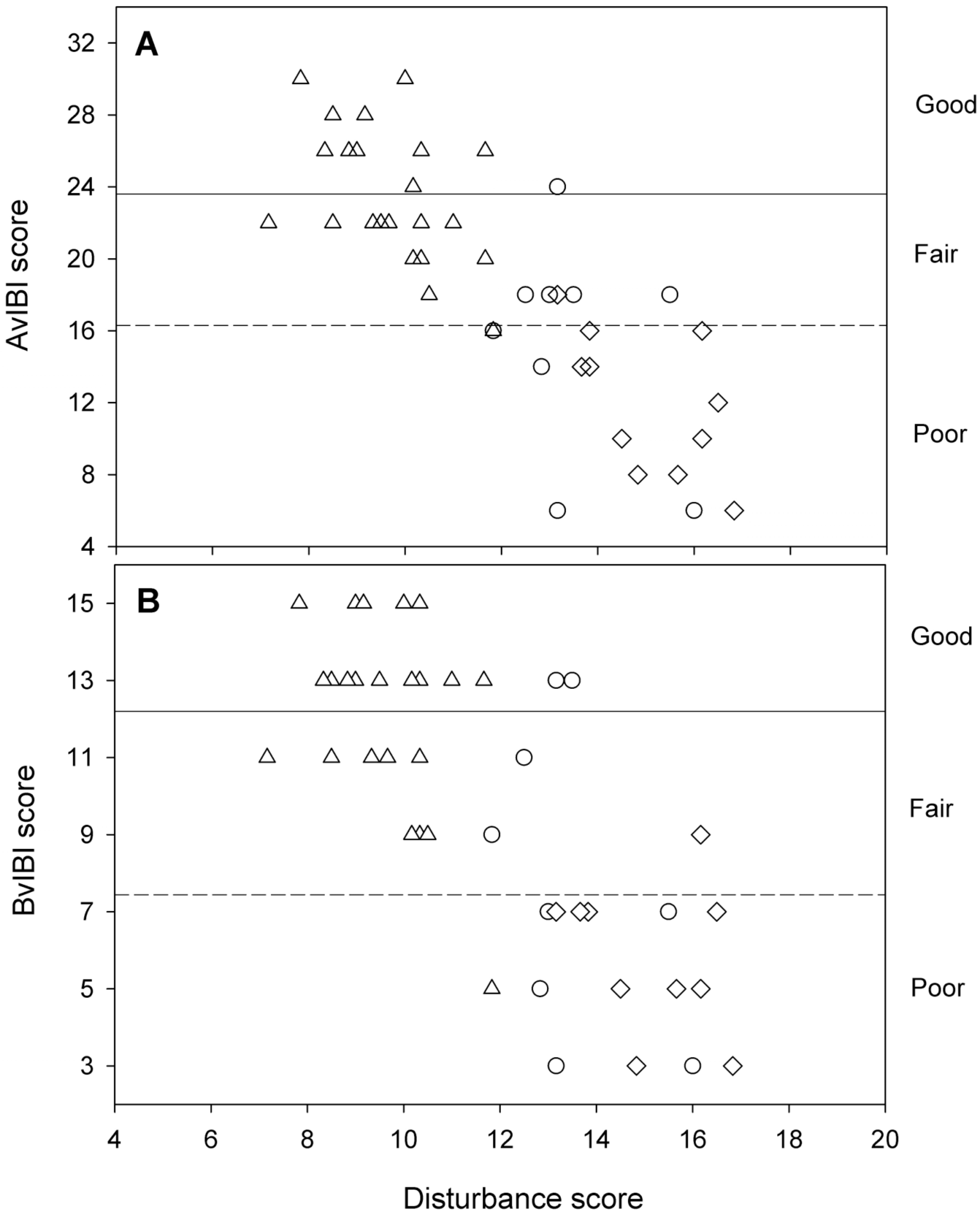

Figure 2.3. vIBI scores plotted against disturbance scores for the AvIBI (A; R^2=0.68, p<0.001), and BvIBI (B; R^2=0.57, p<0.001). Triangle = REF site, circle = OSREF site, diamond = OSPA site. Ranges for poor, fair, and good vIBI scores were calculated by dividing the full range of vIBI scores in 3 using the mean REF site score (solid line) and 2 standard deviations below the mean REF site score (dashed line).

3. *Vegetation and environmental variation among natural boreal and oil sands reclaimed marsh communities.*

Introduction

Mining is an industry that typically destroys intact ecosystems to access a targeted resource, and environmental legislation usually requires that ecosystems be replaced following mine closure. Globally, terrestrial ecosystems tend to dominate the pre-mining landscape, so mined-land reclamation has traditionally attempted to return the land to the pre-existing terrestrial state. Following mining, the coarse mineral, clay, or sandy nature of mine tailings provides a poor initial substrate on which to establish soils and vegetation (Bradshaw 1997, Vetterlein et al. 1999, Wieglieb and Felinks 2001). As a result, the restoration of soil nutrients and vegetation cover is slow, (Nair et al. 2001, Wieglieb and Felinks 2001, Holl 2002, Tischew and Kirmer 2006), particularly when the climate is cool and the growing season is short, as it is in the northern boreal forest of Canada (Strong 2000). Furthermore, soil compaction, erosion, nutrient limitation, and slow dispersal of appropriate vegetation propagules often prevents the rapid establishment of productive vegetation communities (Bradshaw 1997, Vetterlein et al. 1999, Wieglieb and Felinks 2001, Holl 2002). Reestablishment of wetland communities following mining appears to be similarly difficult, with newly created wetlands facing issues of low penetrability of sediment, low nutrients, insufficient or nonexistent seed sources (VivianSmith and Handel 1996, Nair et al. 2001), and unstable or inappropriate hydrology (Johnson and Miyanishi 2008).

Alberta, Canada, holds the world's second largest reserve of crude oil, behind Saudi Arabia. Ninety-nine percent of this reserve is in the form of oil sands deposits (ERCB 2009). Oil sands, a mixture of sand, clay, saline water, and bitumen (a viscous form of heavy crude oil), is extracted by surface mining through layers of marine shale when deposits are within 65 m of the surface, or by in situ extraction using steam injection when deposits lie deeper. The total surface mineable oil sands area in Alberta is 3750 km^2, almost half of which is currently under active lease. Already 520 km^2 of boreal landscape has been destroyed to access this resource and many more projects are slated to begin operations within the next decade (ERCB 2009). The regional landscape overlying the oil sands deposits is greater than 50% wetland by area, 95% of which are fen or bog peatlands (Golder Associates 2002). Following mine closures an area the size of Rhode Island will have been turned into open pit mines and expansive tailings ponds, with toxic elements in water and sediment such as salinity, naphthenic acids, metals, and polycyclic aromatic hydrocarbons, as well as a substrate incapable of supporting higher plants.

Provincial legislation mandates that oil sands companies reclaim the post-mining landscape to "equivalent land capability" (Harris 2007). This is defined as the ability of the land to support various land uses after reclamation that are similar, although not necessarily identical, to those that existed before mining (Alberta Environment 1999). The mined landscape must receive certification as reclaimed before industry is considered to have met this requirement (Harris 2007). Equivalent land capability can be a difficult concept to measure. This is particularly true with respect to wetlands in the reclaimed oil sands landscape. Oil sands wetlands are not expected to return to the same wetland type as were found on the pre-existing landscape, and wetlands are expected to comprise only 20-30% of the reclaimed landscape due to limitations in reclamation engineering and hydrology (C. Qualizza, pers. comm.). The post-mining landscape will, at least initially, be best adapted to the development of marshes and shallow open water wetlands. These have already begun to form on oil sands leases, whether by design for research purposes, or spontaneously when surface and hydrologic conditions have allowed (Trites and Bayley 2009a). Hydrophytic vegetation has naturally colonized these wetlands, and a few wetlands appear to have developed productive emergent marsh and wet meadow zones. Under natural conditions, marshes and shallow open water wetlands are infrequent in the boreal plain of northern Alberta in comparison to fens and bogs. Even less common are wetlands with naturally elevated salinities. However, given the mineral substrate and elevated sediment and water salinities of the post-mining landscape, creation of shallow saline open water and marsh wetlands may represent the best endpoint for reclaimed wetlands (Purdy et al. 2005).

For oil sands reclaimed wetlands to reach equivalent land capability they must replace both the social and ecological functions that pre-industrial wetlands provided. They must restore First Nations land uses, including subsistence hunting, trapping, and harvest of medicinal plants. As well, they must provide corridors and refugia for wildlife, while functioning as a component of the regional water budget, providing flood control, water retention, groundwater recharge, shoreline stabilization, and water treatment (Harris 2007, Johnson and Miyanishi 2008). Wetland vegetation is an integral part of these values and functions (Mitsch and Gosselink 2000, Fennessy et al. 2002), and percent cover, biomass, and species composition are measurable parameters that can be compared among reclaimed wetlands and natural analogues of the expected climax community, to assess the relative success of reclaimed wetlands in reaching equivalent land capability.

Multimetric ecological assessment tools, such as the vegetation-based Index of Biological Integrity (vIBI) have been employed in the U.S. to monitor restoration and reclamation progress, and to enforce policy (Simon 2000). When collapsing vegetation community data into a small number of individual metrics, as is done with the vIBI approach, certain information on the vegetation community as a whole is discarded in order to produce a tool that is cost effective and simple to apply (Reynoldson et al. 1997). The vIBI developed for monitoring oil sands wetland reclamation identifies certain oil sands reclaimed wetlands to be performing poorly, while others score higher, within the range of comparable natural wetlands in the boreal region (Chapter 2). The numerical output of the vIBI is not immediately conducive to interpretation as to why certain sites receive high scores. While individual metrics may be similar among natural and reclaimed wetlands, multivariate statistical techniques can better elucidate important compositional differences between wetland types (Balcombe et al. 2005, Collier 2009), and are useful in detecting trends among sites at the community level, particularly when natural variation is high (Keleher and Rader 2008). An additional benefit of the multivariate statistical approach to community analysis is the potential to assess how physical and chemical variables shift across species compositional gradients, and infer how they may be affecting the compositional gradients (McCune and Grace 2002).

The purpose of this study is to compare reclaimed marsh vegetation communities to undisturbed natural marshes, and to identify potential factors that may allow some reclaimed marshes to perform within the range of natural marshes. Visual differences in vegetation cover and community composition among wetlands within the same treatment indicate a need to group sites on a more biologically meaningful basis than construction history. Specific objectives were to (1) group study sites on the basis of their plant communities to determine how vegetation on reclaimed oil sands marshes compares to natural vegetation, (2) identify underlying physical and chemical features that may be associated with or cause differences among vegetation communities, and (3) identify the factors that limit vegetation development at reclaimed sites to increase the likelihood of reclamation success.

Methods

Study sites

Fieldwork was conducted in 2007 and 2008 in reclaimed oil sands sites and natural reference sites within the North American boreal plains ecoregion (Fig. 3.1, Appendix Table A.1). The boreal plain is characterized by flat topography with surficial glacial deposits of loamy till and gravel-sand glaciofluvial 30-200 m deep overlying Mesozoic- and Cenozoic-age sedimentary bedrock (Johnson and Miyanishi 2008). The subhumid mid to high boreal climate annually averages -2 to +1 ºC, with 400 to 500 mm of wet precipitation and 150 to 200 cm of snow (Zoltai et al. 1988). Seventy percent of the total annual precipitation occurs between May and September (Devito et al. 2000). There is an annual water deficit of 40-60 mm due to higher potential evapotranspiration than precipitation, and the groundwater patterns of this region are complex and poorly documented due to the low topography and deep glacial deposits (Zoltai et al. 1988, Price 2005). The undisturbed landscape is composed of forest, wetlands, lakes and ponds.

Upland forest vegetation in the region is characterized by *Picea glauca* (white spruce), *Abies balsamea* (balsam fir), *Populus tremuloides* (trembling aspen), *Populus balsamifera* (balsam poplar) and *Betula papyrifera* (white birch), with *Picea mariana* (black spruce) occurring in areas with poor drainage. Wetlands in this region are primarily peat-producing fens and bogs, however, marshes and shallow open water wetlands also occur, ranging in salinity from fresh to saline. The predominant fen and bog wetland vegetation in the region is *Picea mariana*, *Larix laricina* (tamarack), ericaceous shrubs, *Sphagnum* and other mosses, sedges (*Carex* spp.), rushes (e.g., *Scirpus* spp.), herbs and forbs. Marshes and shallow open water wetlands are characterized by willows (*Salix* spp.), *Typha latifolia* (broadleaf cattail), *Carex* spp., grasses (e.g., *Calamagrostis* spp., *Scholochloa fetucacea*), rushes, herbs and forbs (Zoltai et al. 1988).

A total of 45 wetlands were initially selected based on their treatment history: 20 reclaimed oil sands wetlands and 25 natural reference (REF) wetlands. Reclaimed wetlands were located on the oil sands leases of Syncrude Canada Limited and Suncor Energy Incorporated near Fort McMurray, Alberta, Canada (56.8531° N, 111.3180° W to 57.1150° N, 111.6833° W), and were predominately fresh to subsaline shallow open water wetlands with marsh fringes. Their age when sampled ranged from 3 to 35 years, although the age of one site could not be determined as it was impacted only by hydrologic alteration, at an unknown date. With the exception of the single youngest site, all sites were 7 years of age or older at the time of sampling. Previous work in the oil sands has identified 8 years as the division between young and mature reclaimed wetlands (Leonhardt 2003), and I sampled all suitable sites this age or older that were available as of 2008.

I classified oil sands reclaimed wetlands a priori into 2 treatments, on the basis of their construction and physicochemical history. This method broadly took into account whether a wetland was the subject of physical disturbance such as gravel extraction or impoundment (oil sands reference, or OSREF treatment), or both physical and chemical disturbance (oil sands process-affected, or OSPA treatment). Chemical disturbance occurs in the form of exposure to oil sands process water or substrate, materials that are often highly saline, and can contain naphthenic acids, polycyclic aromatic hydrocarbons as well as other contaminants. This disturbance could have occurred in a single period in the history of the wetland, or is ongoing, as in the case of some wetlands receiving seepage water from nearby tailings facilities. The natural salinity of the marine shale in the region may lead to increased salinity on OSREF sites as well. Reclaimed oil sands wetlands have been both intentionally constructed, and have formed "opportunistically" where conditions on the post-mining landscape have allowed. The average age of reclaimed wetlands was 16.0 ± 9.8 years. OSREF wetlands were older than OSPA wetlands (19.9 ± 10.8 years vs. 13.2 ± 8.4 years), however this was not statistically significant (One-way ANOVA; $F(1,17)=2.32$, $p=0.15$). The third treatment, REF wetlands, located at 6 loci across the boreal plain (53.6015° N, 105.8957° W to 59.1098° N, 118.0797° W), were selected to be shallow open water marshes, and to span a range in salinity from fresh to sub-saline. The age of natural wetlands was unknown, although it is estimated that they have been on the landscape for >1000 years, and may have experienced periodic alteration by beaver (*Castor canadensis*) activity.

All study sites were considered Class V permanent open water wetlands, as described in Stewart and Kantrud (1971). Three wetland vegetation zones occurred at my study wetlands: the wet meadow zone, the emergent zone, and the shallow open water zone. This project focused on the wet meadow and emergent zones of each site. The wet meadow zone contains sedge and grass species as well as obligate and facultative wetland forb species. It is often flooded in the spring, however, the water level drops below the sediment surface for the majority of the growing season. Adjacent to the wet meadow towards the open water zone, is the emergent (or marsh) zone. This zone contains robust wetland obligate emergent species, and is typically flooded for the duration of the growing season.

Wetland size including area of open water to the upland edge of the wet meadow ranged from 0.4 ha to 25 ha, with a mean of 4.3 ($\pm$ 5.6) ha and median of 2.3 ha. It was my intention to capture similar size ranges among the 3 treatments. This was achieved with REF and OSREF wetlands (0.6 to 24 ha and 0.8 to 25 ha, respectively). However, OSPA wetlands were generally smaller in size (0.4 to 3.2 ha).

Sampling technique

Vegetation data

At each site three transects perpendicular to the edge of open water were established, roughly dividing the wetland into thirds. Along transects I measured zone width, water depths at zone interfaces, and transects were used as a point of reference for quadrat placement. In the centre of the wet meadow, perpendicular and clockwise from each transect I placed 2 community composition quadrats (1 m^2) and 2 biomass clipping quadrats (0.25 m^2). In the emergent zone at each transect I placed 2 community composition quadrats and 1 biomass clipping quadrat. A total of 6 1 m^2 community composition plots per zone, per wetland were assessed. When large differences in community structure among sites were expected, (for instance between oil sands and reference wetlands), 5-10 vegetation plots were recommended based on a multivariate power analysis of wetland vegetation plots (James-Pirri et al. 2007). All macrophyte species were identified within 1 m^2 community composition plots, and percent cover by species was estimated. When it was not possible to identify macrophytes to the species level, genus was used. Nomenclature followed that of the Flora of Alberta (Moss 1983).

To obtain a measure of productivity, macrophyte aboveground growth in the 0.25 m^2 biomass plots was clipped to within 1 cm of the substrate surface. Samples were weighed for total aboveground biomass, recording the dominant species separately from the remainder of the harvested material.

Environmental data

Water and sediment chemistry was sampled in August of each year. Water conductivity and pH were measured in situ using a handheld YSI MPS 556. Water samples were collected using an integrated water sampler to obtain a composite water column sample from the centre of the wetland. Water was analyzed following methods described in Bayley and Prather (2003) for total nitrogen (TN), total dissolved nitrogen (TDN), total phosphorus (TP), total dissolved phosphorus (TDP), soluble reactive phosphorus (SRP), dissolved organic carbon (DOC), total dissolved solids (TDS), total suspended solids (TSS), alkalinity, anions and cations. Naphthenic acids (NAs) in water were measured by the Fourier-transform infrared spectroscopy method (Jivraj et al. 1996).

Sediment cores were taken using a suction-corer at 3 locations per zone to a depth of 10 cm, in the centre of the zone. Sediment samples were composited by zone to account for possible patchiness of the sediment throughout the wetland. Sediment was homogenized, and oven-dried at 60°C for 48 hours to determine water content. Subsamples were analyzed for total carbon (TC) and TN by combustion in an Exeter Analytical CE40 Elemental Analyzer. Total P was determined using the peroxide/sulfuric acid digestion method (Parkinson and Allen 1975) followed by spectrophotometric analysis. Oil content of sediment was determined using refluxing toluene in a soxhlet extraction apparatus as described in Rooney and Bayley (2010a).

HOBO water level loggers were installed at each wetland at the beginning of the growing season to record water level amplitude for the duration of the growing season. Secchi depth and mean maximum water depth were taken in the open water zone at each site. Open-water area was determined from aerial photographs or satellite imagery when recent aerial photographs were unavailable.

Rooney and Bayley (2010b) calculated site disturbance scores to evaluate the relative level of disturbance among all study sites on a synthetic abiotic physicochemical disturbance gradient. The disturbance score was treated as a single physicochemical variable in analyses. The disturbance scores effectively spread sites across a range of disturbance from relatively pristine to highly disturbed, using 8 physical and chemical parameters (open water cations, TN, and chloride, emergent zone % water and oil content of sediment, maximum open water depth, secchi depth as a proportion of total depth, open water amplitude over the growing season) measured at each site. The disturbance gradient did not differentiate between anthropogenic and natural disturbance, and incorporated elements of water chemistry, sediment chemistry, and wetland morphology.

Data screening

I created zone-specific species composition values for each study wetland by combining values from the six 1 m^2 plots from each zone at each site into a single composite plot, representative of the entire zone. I removed rare species from community composition data before multivariate data analyses, to reduce noise and enhance the likelihood of detecting relationships among species communities and environmental variables (McCune and Grace 2002). Rare species were deleted if they were present at less than 5% of sample units, which was presence at less than three sites. This reduced the overall species number by greater than 50%, from 134 to 62 for the wet meadow and from 53 to 24 for the emergent zone (Appendix Table A.2). Percent cover data, as proportions, were arcsin square root transformed before analyses (Sokal and Rohlf 1995).

To assess whether the wet meadow and emergent vegetation communities at my sites were independent of each other, I performed a Mantel Test with the randomization (Monte Carlo) method of interpretation and Bray-Curtis distance (Sokal and Rohlf 1995) to compare the wet meadow and emergent zone distance matrices. The results indicate non-independence of the wet meadow and emergent zone species composition data (r-statistic=0.314, p<0.001). I therefore chose to carry out all subsequent analyses on the wet meadow data, which had a greater number of species.

Statistical analyses

Rather than group sites on the basis of their construction history, which was visibly apparent in the field as an insufficient predictor of wetland vegetation condition, I chose to group them on the basis of their vegetation community composition. I then aimed to identified the species that characterized these vegetation communities, examined differences in productivity and vegetation cover among these communities, and explained community differences by patterns in species and environmental variables. Multivariate techniques such as hierarchical cluster analysis and ordination facilitate comparison among study sites on the basis of their species composition (McCune and Grace 2002), and these techniques can be particularly effective when assessing community similarity or looking for species and environmental trends among highly variable or degraded sites (Reynoldson et al. 1997, Keleher and Rader 2008, Collier 2009).

Site grouping and communities

To group sites by vegetation community type I used hierarchical cluster analysis (distance measure: Bray-Curtis, linkage type: flexible beta -0.25) on wet meadow species composition data. To determine the optimal number of groups I used Indicator Species Analysis (ISA) with a Monte Carlo test of significance (Dufrene and Legendre 1997). The optimal group number was selected on the criteria of low average Monte Carlo test p-value, and higher numbers of significant indicators. I used ISA to identify the characteristic, or indicator species of each community type.

Vegetation cover and biomass

To assess variation in aboveground biomass production and vegetation percent cover among vegetation communities I used one-way ANOVA with Tukey's multiple comparisons in SPSS v. 17 (SPSS Inc. 2008).

Vegetation and environmental patterns

Next, to observe the grouping of sites in multivariate species space, and to examine major species composition and environmental gradients, I performed non-metric multidimensional scaling (NMS) using the Bray-Curtis distance measure in PCOrd v. 4 (McCune and Mefford 1999, McCune and Grace 2002). To determine the optimal dimensionality for the NMS, a scree plot identified 2 axes as the best trade-off between the amount of information displayed on a single plot and stress in the model. The NMS was re-run with this dimensionality, from a random starting configuration. The final stress of the model was near the upper limit of what is considered biologically interpretable (final stress=19.80), however large sample sizes, such as my 45 sites, are known to increase NMS stress while still providing interpretable results.

To examine patterns in species composition and underlying physical and chemical gradients among study sites, species and environmental overlays were displayed on NMS plots. All species used in the vegetation community analysis matrix were included as potential joint plot vectors. The environmental joint plot included the following variables: sediment C, N, P and water content, water depth in vegetation plots, open water conductivity and naphthenic acids, and the abiotic disturbance score of each site. Overlay vectors were plotted only if they had R^2 correlations with the NMS of >0.20 in order to explore trends, though not necessarily infer statistical significance.

Results

Site grouping and communities

The 45 sites clustered into 3 vegetation communities (hierarchical cluster analysis and ISA; 20% information remaining, 2.17% chaining). These communities were designated as a natural community, a reclaimed sedge community, and a disturbed/saline community (Table 3.1). The majority (>75%) of natural sites fall within the natural community group, defined by the presence of *Carex atherodes* (swallow-tail sedge), and *Scutellaria galericulata* (marsh skullcap). Two OSREF sites were included in the natural category, both of which support *C. atherodes* in their wet meadow vegetation communities, a significant indicator species of this community.

The reclaimed sedge community had even numbers of OSPA and OSREF sites, however 3 REF sites also belonged to this community. These sites had wet meadows primarily characterized by *Carex aquatilis* (water sedge), which was the significant indicator species of this community. While still sedge-dominated like the natural vegetation community, the dominant sedge species in the reclaimed sedge community differed.

The disturbed/saline community was the only community not dominated by sedge species, and was composed primarily of 6 OSPA sites, but did contain 2 OSREF sites, and 3 REF sites. The significant indicator species of this community were *Hordeum jubatum,* a salinity-tolerant grass species, and *Sonchus* spp., a genus of sow thistle considered invasive to Alberta (Alberta Weed Control Act 2001).

Due to the unexpected inclusion of REF sites in the disturbed/saline community group, I examined the 3 communities in greater detail by returning to the cluster analysis and the ISA-determined pruning levels. Beyond the 3 distinct vegetation communities, the ISA indicated that further division of my study sites into 7 vegetation communities (at 45% information remaining) could aid in the interpretation of the vegetation communities. Hierarchical clustering reduces numbers of clusters by merging them, thus the 7 communities can be treated as sub-communities of the 3 communities (Table 3.2). I found both the natural and reclaimed sedge communities each were composed of two sub-communities. The disturbed/saline community divided into three sub-communities: Reedgrass (*Calamagrostis stricta*), disturbed, and natural saline.

The natural saline sub-community describes the vegetation of 2 REF sites. Seven species of halophytic vegetation are associated with the natural saline sub-community as significant indicators (Table 3.2). These REF sites clustered more closely with the disturbed/saline community than other natural REF sites when sites were clustered in 3 communities. The disturbed sub-community was characterized by *Melilotus* spp., a genus of sweet clover considered invasive to Alberta, and was found at 5 OSPA and 2 OSREF wetlands. Two sites, one REF wetland, and one OSREF wetland shared the Reedgrass sub-community, indicated by *C. stricta*.

Biomass and species richness among communities

Of the 3 vegetation communities (Table 3.1), the natural vegetation community was significantly more productive than either the reclaimed sedge or disturbed saline communities (One-way ANOVA with Tukey's multiple comparisons; $F(2,42)=17.42$, $p<0.001$), with nearly 50 percent greater August aboveground biomass (Fig. 3.2). The majority of aboveground biomass could be attributed to *Carex* spp. in the natural and reclaimed sedge communities, whereas *Carex* spp. comprised less than 20 percent of the aboveground biomass in the disturbed/saline community (Fig. 3.3).

Percent cover of vegetation the natural and reclaimed sedge communities were similar, despite the differences in aboveground biomass (Fig. 3.2). The disturbed/saline community had significantly lower percent cover than the natural community, or inversely, a significantly greater area of bare ground within the wet meadow zone (One-way ANOVA with Tukey's multiple comparisons; $F(2,42)=6.26$, $p<0.01$).

Examining the disturbed/saline community in more detail, as 3 sub-communities, shows aboveground biomass in the natural saline sub-community was significantly greater than the disturbed sub-commmunity (One-way ANOVA with Tukey's multiple comparisons; $F(2,8)=11.86$, $p<0.01$). Vegetation percent cover in the natural saline sub-community was also significantly greater than the disturbed sub-commmunity (One-way ANOVA with Tukey's multiple comparisons; $F(2,8)=19.26$, $p<0.01$). The Reedgrass sub-community, however, did not differ significantly from either of the two other disturbed/saline sub-communities, as the high variability of these values combined with low sample sizes limited the detection of statistically significant differences.

Mean species richness was not significantly different among vegetation community groups, nor did it differ among wetland treatment types (Table 3.3). Total species richness among vegetation communities were similar, ranging from 86 in the reclaimed sedge community, to 112 in the natural community. Total species richness for each treatment was highest for REF sites, with 120 species, and equal for both OSREF and OSPA, with 85 species in each.

The average age of oil sands reclaimed sites with the reclaimed sedge community was not significantly different from those with the disturbed/saline community (One-way ANOVA; $F(1,16)=0.063$, $p=0.81$). The average age of sites included in the natural vegetation community group was assumed to be >1000 years, as these were primarily REF sites.

Vegetation and environmental patterns among communities

The three vegetation communities identified by hierarchical clustering formed visually distinct clusters in the NMS plot (Fig. 3.4), which allowed us to use vector overlays to examine species and environmental gradients among them. The 2-dimensional NMS reached low final instability (<0.00001) after 112 of 200 possible iterations. Both Axis 1 and 2 were significant (Monte-Carlo $p<0.05$) and together represented 74% of the variance in the model.

Axis 1 was primarily a *C. atherodes* – *Equisetum* spp./*Sonchus* spp. gradient (Fig. 3.4, inset). This axis separated the natural community from both the reclaimed sedge and disturbed/saline communities. Greater cover of *C. atherodes* was found low on the axis where the natural community sites were grouped, while *Equisetum* spp. (horsetail) and invasive *Sonchus* spp. were found high on the axis.

Axis 2 was a *C. aquatilis* – halophyte species gradient, and separated the reclaimed sedge community from the disturbed/saline community. *C. aquatilis* was found high on the axis, where the reclaimed sedge community was clustered. REF treatment wetlands were clustered low on Axis 1, which was expected as REF wetlands made up the majority of sites with natural vegetation communities. Both OSPA and OSREF treatment wetlands were distributed high on Axis 1 and evenly along Axis 2 indicating that the variation in vegetation communities among these wetlands is due to differing percent cover of *C. aquatilis* and halophyte species (in particular *Hordeum jubatum*).

Environmental vectors varied along Axis 1 only (Fig. 3.5), and thus reflected environmental differences associated with the natural vegetation community versus both the reclaimed sedge and disturbed/saline communities. Sites that were low on Axis 1 tended to have high sediment nutrients (TN and TP), and sediment water content, while those that were high on the axis had higher synthetic disturbance scores, and naphthenic acids in the water.

When the 3 vegetation community groups were divided into their 7 sub-communities, wetlands with the two natural sub-communities overlapped within the natural community cluster, while wetlands with the remaining 5 sub-communities had little overlap among them, with the exception of the Reedgrass subgroup, where sites were spread along Axis 1 (Fig. 3.6). Wetlands supporting the natural saline sub-community were clearly separated from other members of the disturbed/saline community, as well as the other 2 vegetation communities, low on Axis 2. When the 2 REF sites of the natural saline sub-community are considered separately, it is clear that wetlands representing the OSREF and OSPA treatments do not share vegetation characteristics of either the natural community or the natural saline sub-community.

Discussion

Treatment type cannot predict vegetation

Construction history and materials were poor predictors of the vegetation communities that develop in oil sands wetlands during the reclamation process. The sites that clustered together in the 3 communities based on species composition differed from the 3 wetland treatments assigned a priori based on site history [natural reference (REF), oil sands reference (OSREF), and oil sands process-affected (OSPA)]. While natural reference wetland sites were mainly grouped together as the natural vegetation community, grouping of oil sands reclaimed wetlands did not mirror their treatment types (OSREF and OSPA). Eighteen of the 20 reclaimed wetland sites were clustered as either the reclaimed sedge or disturbed/saline wet meadow vegetation communities, with both OSREF and OSPA wetlands found with either reclaimed sedge or disturbed/saline communities. My broad a priori categorization of sites was not a useful predictor of wetland vegetation community, likely as a result of the diversity of reclamation materials, proximity to seed sources, and hydrological settings that include receipt of fresh precipitation, groundwater, or tailings pond seepage water.

The disturbed/saline community had the most variable wet meadow vegetation, and was the least tightly grouped in ordination space of the 3 communities. This is consistent with findings that the multiple ways by which a site could be highly disturbed leads to a higher degree of variability among disturbed sites (Ehrenfeld and Schneider 1993, Keleher and Rader 2008). The inclusion of two naturally saline REF sites within the disturbed/saline community is a problematic aspect of using only 3 community groups to describe all the wetland vegetation encountered in this study. The disturbed/saline community is best understood when its 3 constituent community sub-groups (Reedgrass, disturbed, natural saline) are considered. The 2 natural saline sites clustered separately from other disturbed wetlands, due to their halophytic vegetation community, and contained species also observed by Trites and Bayley (2009a) in a previous survey of naturally saline vegetation in the boreal plain. While the 2 natural saline sites represent a native vegetation community, I did not encounter it in either of the oil sands treatment types, despite high salinity water at certain reclaimed sites.

The reclaimed sedge community was the most tightly clustered group in ordination space, implying a high degree similarity in vegetation community composition among member sites, and less within-group variation than was found in the other wet meadow vegetation communities. This community was rarely encountered in natural wetlands, with only 3 natural REF sites supporting a *C. aquatilis*-dominated wet meadow.

The natural wet meadow community characterized by *C. atherodes* was consistent with natural communities identified in previous studies in the boreal region of Alberta (Bayley and Mewhort 2004, Trites and Bayley 2009a), and with the exception of 2 OSREF sites, this vegetation community was found exclusively at natural REF sites. The OSREF sites with natural *C. atherodes* wet meadow communities consisted of one site that is relatively undisturbed except for hydrologic alteration, although it is found adjacent to a large tailings facility, and a second site that was constructed to study effectiveness of waterfowl habitat creation.

Active revegetation may accelerate recovery

Using reference sites as the benchmark of ecological health allows the assessment of the instantaneous condition of a site to determine if reclamation objectives are being met. Reference sites, which have been selected to represent sustainable conditions in a particular region, also provide a template to guide reclamation practices and outcomes (Brinson and Rheinhardt 1996). Early planting of appropriate desired species may pre-empt the establishment of dominant, less desirable species on longer timelines (Noon 1996, Aronson and Galatowitsch 2008), and overcome the immediate effects of dispersal limitation in newly created wetlands (Seabloom and van der Valk 2003). Trites and Bayley (2009a) hypothesized that the paucity of *C. atherodes* in reclaimed wet meadows was due to slow dispersal of this species in the absence of connected, flowing water or waterfowl transport to reclaimed substrates. Lougheed et al. (2008) found that plant communities in ecologically degraded wetlands in Michigan were heterogeneous at the landscape level, likely due to differential colonization by plant species, so if uniform establishment of a particular community is desired in oil sands reclaimed wetlands, plantings may help overcome dispersal limitation at the local level. Noon (1996) refers to an Arrival and Establishment Phase in newly created wetlands undergoing primary succession, characterized by the random arrival and establishment of species. Annuals dominate this phase, and while they may stabilize sediments and trap seeds, they contribute little to the long-term establishment of the vegetation community since they lack expansive belowground growth. Belowground growth, particularly rhizomatous expansion of vegetative perennials, aids in the development of soil properties such soil moisture, and nutrient storage and cycling capacity that contribute to long-term resilience to perturbation (Noon 1996).

The diversity of substrates, chemistry and hydrology in the post-reclamation landscape may require a flexible revegetation program as only a few species may be suited to particular post-mining conditions, and I have shown the development of vegetation communities to be unpredictable by their treatment type. Mitsch and Wilson (1996) have suggested a multiple-seeding, multiple-transplanting approach, allowing local conditions to select the species from a larger set of ecologically appropriate stock. Seeding or planting a combination of both halophytic species and freshwater boreal species known to be tolerant of moderate levels of salinity would be a good starting point for revegetation. In a salt marsh revegetation study, diverse plantings have been shown to reach higher aboveground biomass and retain more sediment N than single species plantings (Callaway et al. 2003). After the plant community is established, autochthonous mechanisms such as above- and belowground biomass growth, in excess of decomposition, allow the buildup of sediment organic matter, thereby enriching the substrate, lowering bulk density, and increasing water content (Noon 1996, Ballantine and Schneider 2009, Trites and Bayley 2009b). Organic matter accumulation has already been demonstrated as possible in post-reclamation oil sands wetlands (Trites and Bayley 2009b).

The reclamation timeline, or speed at which successful reclamation is expected or required, is an important factor when deciding whether to plant species at newly constructed wetlands, as short timelines may require propagule introduction at a faster rate than would occur naturally (Seabloom and van der Valk 2003, Balcombe et al.

2005). If managers are seeking to meet reclamation goals of healthy, functioning wetlands within the first decade following mine closure, strong initial efforts to establish the vegetation community in reclaimed wetlands, including plantings and seed bank enhancement could be the best method to increase success (Gutrich et al. 2009).

Sediment fertilization may increase biomass production

Studies of reclaimed wetland vegetation community development have typically focused on percent cover as a measure of productivity (e.g., Seabloom and van der Valk 2003, Aronson and Galatowitsch 2008). My results have shown that greater vegetation cover does not necessarily mean higher aboveground biomass, and if I were to examine only the percent cover data among the natural and reclaimed sedge community groups I could falsely conclude that there is equal productivity in both. Clipped aboveground biomass, a more direct measure of aboveground production, demonstrated significantly higher production in the natural community than the reclaimed or disturbed communities.

The high percent cover, but relatively low aboveground biomass of the reclaimed sedge community indicates that while the vegetation community may become established with respect to species composition and percent cover, other factors, such as physical and chemical stress, may limit aboveground biomass production from reaching the same level as undisturbed natural wetlands. It is possible that wetland age is a factor limiting biomass production, as the average age of sites with natural vegetation communities was >1000 years, while the average age of sites with reclaimed sedge vegetation communities, or disturbed/saline vegetation communities (excluding 4 natural REF sites) was approximately 16 years. Much of the early spring growth of *Carex* spp. is supplied by nutrients stored in belowground rhizomes (Bernard et al. 1988), and age can limit the extent of belowground rhizomatous biomass. Belowground biomass growth in both oil sands reclaimed wetlands and natural fresh to sub-saline boreal marshes is quite low, comprising less than 10% of total annual production (Trites and Bayley 2010b), although the ingrowth technique used in their study may represent a low estimate. Shaver and Billings (1975) found that active root development of the sedge *C. aquatilis* in Alaska only occurred in the initial 2-3 years of growth, however roots had a lifespan of up to 10 years. It is possible that age is restricting biomass production in reclaimed wetlands, with younger reclaimed sedge meadows having a limited rhizome network with which to exchange nutrients.

In this study, sediment nutrients are higher at the natural sites, as is water content of the sediment (inversely related to bulk density). In examining the environmental vector overlays, NMS Axis 1 can be seen as a gradient from high sediment organic matter to mineral sediment. A comparable gradient was found to account for greater than 50% of variation in aboveground biomass of graminoids in European wet meadows (Olde Venterink et al. 2001). While percent cover was not different among the natural vegetation community and the reclaimed sedge community, lower aboveground biomass in the latter community could be a result of nutrient deficiency.

For natural communities to develop in reclaimed oil sands wetlands the nutrient content of the sediment may need to be artificially enhanced, as indicated by high sediment nutrient levels in the natural wetland community. Nutrient additions to mineral dune soils in North Dakota were found to increase the overall biomass of graminoid species, as well as decrease the extent of exposed sediment (Willis 1963). Fertilization with N, P and K increased production in dune slacks, when water was not a limiting factor, a situation that may share characteristics with the post-reclamation landscape where mineral tailings will be used as a substrate for wetlands. The enrichment of substrates with fertilizer is likely to accelerate the ecosystem processes that lead to healthy, viable, and resilient vegetation communities.

Fertilization of European wet grasslands with N, P and K effectively increased aboveground biomass production, however graminoid species contributed proportionally more to overall production than other species following fertilization (Vermeer 1987). The majority of boreal plain marshes are naturally eutrophic or hypereutrophic (Trites and Bayley 2009b), and in the boreal plain, the low diversity wetlands characterized by dominant graminoids such as *C. atherodes* may in fact be the optimal marsh climax community.

Salinity is not an impossible barrier to reclamation

The issues that challenge reclamation in the oil sands centre on the elevated salinity in water and sediment from weathering of saline, sodic rock, the presence of hydrocarbons and naphthenic acids resulting from the extraction process, fine clay particles in the tailings, heavy metal contamination, as well as initially low nutrient content of sediment (Harris 2007, Johnson and Miyanishi 2008, Kelly et al. 2009). These factors can limit the establishment of healthy vegetation communities.

Salinity is known to affect wetland vegetation (e.g., Lissner and Schierup 1997, Houle et al. 2001, Lewis and Weber 2002), and at natural wetlands in the boreal plain salinity has been shown to shape the vegetation community towards halophytedominance (Fairbarns 1990, Purdy et al. 2005, Trites and Bayley 2009a). I did not see a clear salinity gradient among vegetation communities, and with the exception of the 2 natural sites with halophyte-dominated wet meadow communities, it did not appear that salinity was affecting the vegetation communities I observed, in spite of the salinity gradient along which the natural REF and oil sands OSPA and OSREF sites were sampled. Thus, matching vegetation to salinity may not be critical to reclamation success, if success is measured by the re-establishment of vegetation communities similar to those found naturally in shallow open water marshes in the boreal region. The introduction of species such as *C. atherodes* and *C. aquatilis*, however, which are tolerant of the salinity levels found at oil sands reclaimed sites (Trites and Bayley 2009b), may supplant weedy or invasive species and contribute to substrate stabilization and organic matter accumulation.

Both the abiotic disturbance score and naphthenic acids increased from REF wetlands towards the OSREF and OSPA wetlands grouped high on NMS Axis 1 (Fig. 3.5). Kamaluddin and Zwiazek (2002) found a direct decrease in root respiration following the exposure of aspen (*Populus tremuloides*) seedlings to naphthenic acids, and proposed that this mechanism was responsible for decreased aboveground biomass growth in exposed seedlings. I did not analyze pore water of wet meadow sediment, however naphthenic acids are a known constituent of process affected tailings materials, and their presence in the water is likely indicative of presence in the sediment of reclaimed wetlands. Leung et al. (2003) examined the effects of naphthenic acids on phytoplankton species composition in oil sands reclaimed wetland mesocosms, however, they were unable to separate the effects of increased salinity from that of naphthenic acids on phytoplankton. As naphthenic acids increase along NMS Axis 1, and the reclaimed sedge and disturbed/saline vegetation communities are separated along Axis 2, it does not seem likely that they are affecting species composition in reclaimed wet meadows.

Axis 2 may be affected by environmental variables not directly measured in this study. Individual site history is difficult to determine, beyond the broad OSPA or OSREF categorization, and sites may have been subject to additions of sediment amendments, or specific design criteria at some point since creation. These factors may partially explain vegetation community differences among reclaimed wetlands. The distance of reclaimed wetlands from propagule sources could also determine initial revegetation success, especially in the absence of a planting or seeding program (Houlahan et al. 2006, Aronson and Galatowitsch 2008). The nearer to propagule sources such as remnant wetlands on the periphery of oil sands leases, the more likely viable propagules are to arrive on reclaimed substrates from nearby sources via environmental and faunal dispersal mechanisms. This may represent the dominant pathway to establishing the reclaimed sedge community currently found at certain reclaimed oil sands wetlands.

Conclusion

Oil sands reclaimed wetland vegetation communities tend towards either a sedge-dominated wet meadow community (reclaimed sedge), or an invasive species and halophyte dominated wet meadow community (disturbed/saline). In the absence of clear physical or chemical differences between these two reclaimed communities, I should consider that that early intervention with plantings, fertilization, construction of wetlands near propagule sources when possible may be the best opportunity to guide reclaimed wetland vegetation communities towards those of natural boreal marshes.

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Table 3.1. Wet meadow vegetation community groups and indicator species.

Group	Community	Indicator species	Membership
1	Natural	*Carex atherodes, Scutellaria galericulata*	21 (19 REF, 2 OSREF)
2	Reclaimed sedge	*Carex aquatilis*	13 (3 REF, 5 OSREF, 5 OSPA)
3	Disturbed/saline	*Hordeum jubatum, Sonchus* spp.	11 (3 REF, 2 OSREF, 6 OSPA)

Indicator species: maximum indicator value in ISA > 0.5, $p < 0.01$

Table 3.2. Wet meadow vegetation community sub-groups and indicator species.

Group	Sub-community	Indicator species	Membership
1a	Natural I	*Scutellaria galericulata***	14 (12 REF, 2 OSREF)
1b	Natural II	*Carex atherodes***	7 (7 REF)
2a	Reclaimed sedge I	*Carex aquatilis***	6 (3 REF, 3 OSPA)
2b	Reclaimed sedge II	*Carex utriculata***	7 (5 OSREF, 2 OSPA)
3a	Reedgrass	*Calamagrostis stricta***, *Achillea sibirica***, *Achillea millefolium***, *Typha latifolia**	2 (1 REF, 1 OSPA)
3b	Disturbed	*Melilotus* spp.*	7 (2 OSREF, 5 OSPA)
3c	Natural saline	*Glaux maritima***, *Hordeum jubatum***, *Puccinellia nuttalliana***, *Scirpus paludosus***, *Agropyron trachycaulum***, *Scolochloa festucacea**, *Triglochin maritima**	2 (2 REF)

Indicator species: maximum indicator value in ISA > 0.5, $^{**}p < 0.01$, $^{*}p < 0.05$

Table 3.3. Mean and total wet meadow species for wetland vegetation communities, and a priori wetland treatment types.

Community/treatment	n	Mean (±SE) species per site	Total species per community/treatment
Natural	21	25.8 (1.18)	112
Reclaimed sedge	13	21.4 (2.67)	86
Disturbed/saline	11	26.6 (2.24)	103
REF	25	24.9 (1.51)	120
OSREF	9	25.3 (2.24)	85
OSPA	11	23.8 (2.57)	85

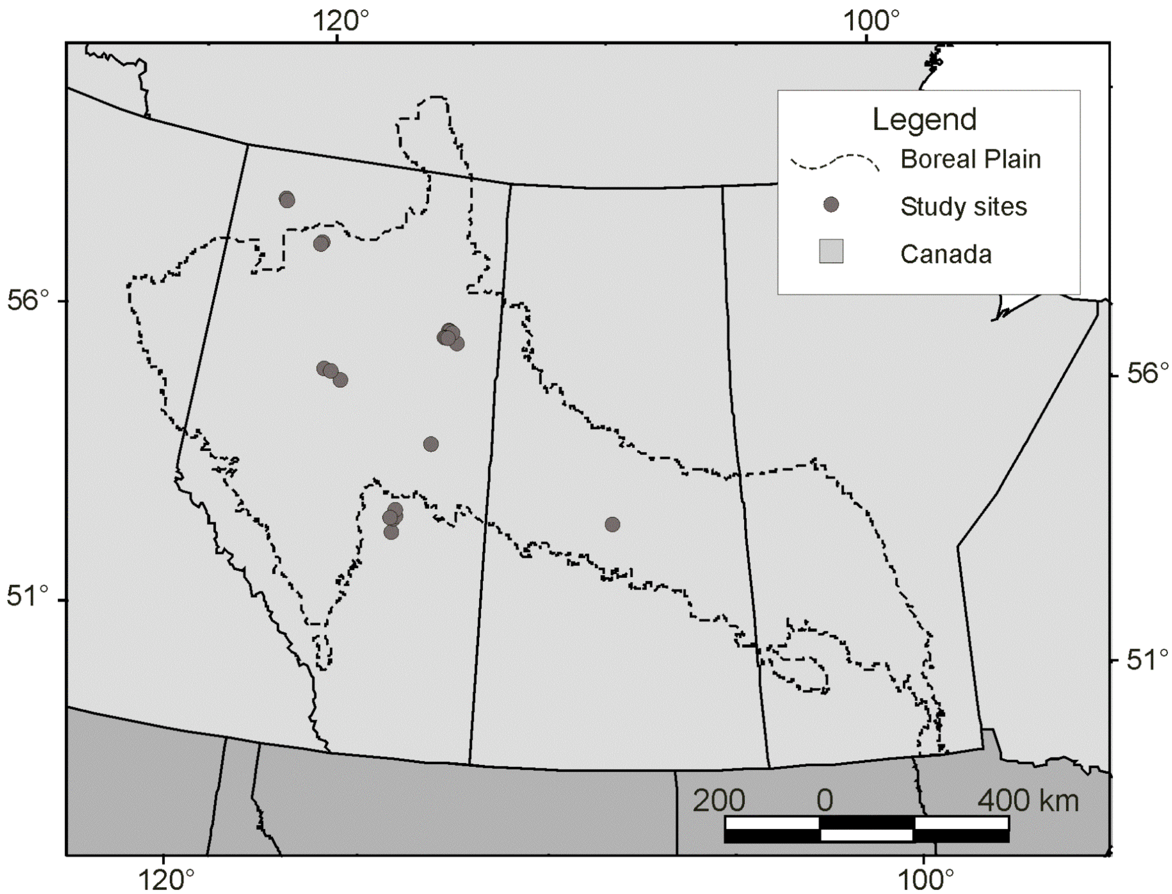

Figure 3.1. Study site locations within the boreal plain ecozone of northern Alberta and Saskatchewan. Sites located to the south of the zone in central Alberta are found within a small pocket of boreal forest, discontinuous with the rest of the zone. Sites to the north of the boreal plain in northern Alberta are within a zone of transition to the taiga plains.

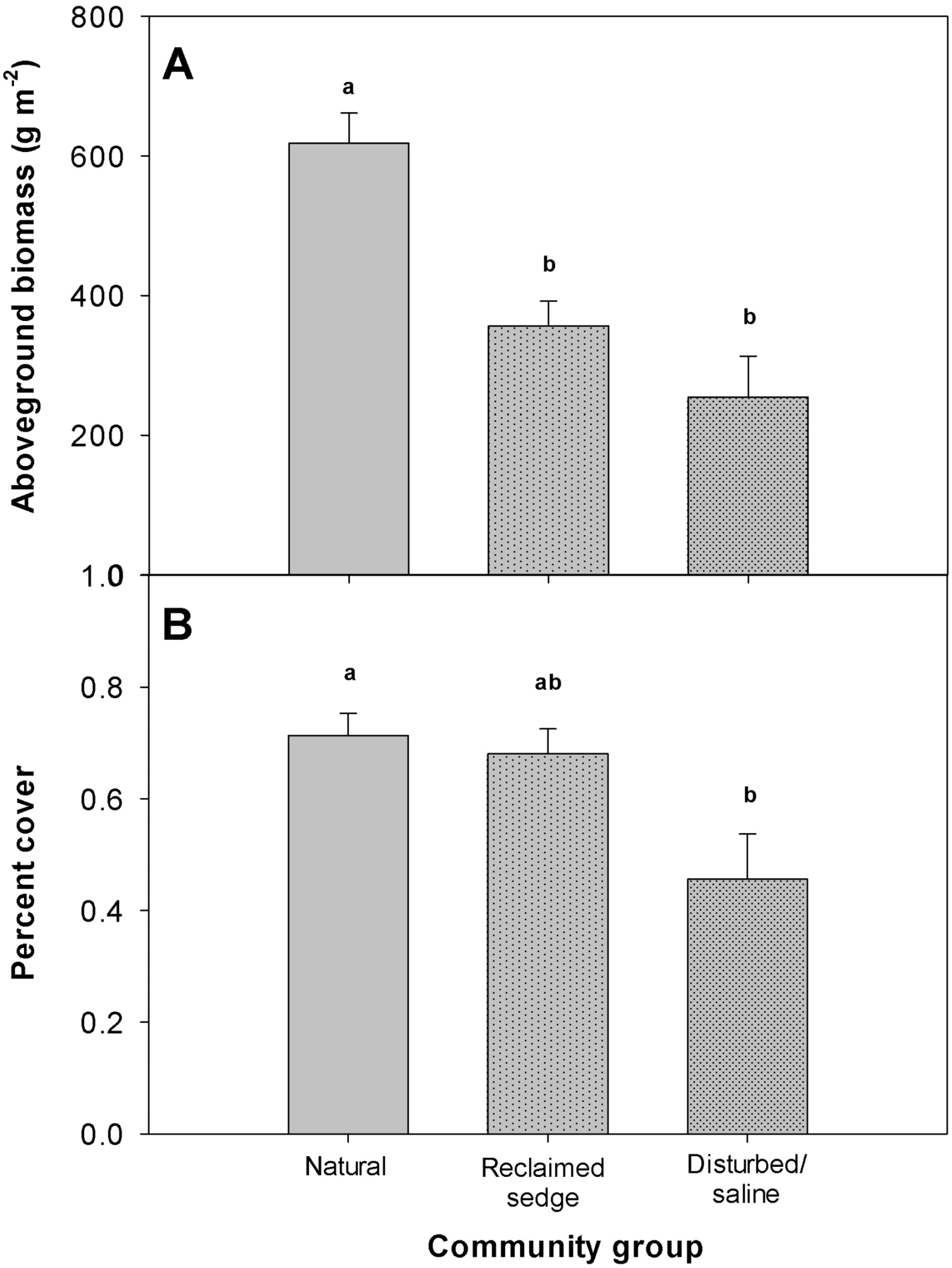

Figure 3.2. A) Wet meadow aboveground biomass by community group and B) Wet meadow percent cover as a proportion from 0-1, by community group. Bars with different lower case letters are significantly different (p<0.01). Vertical bars correspond to standard errors of the mean (n=45 sites).

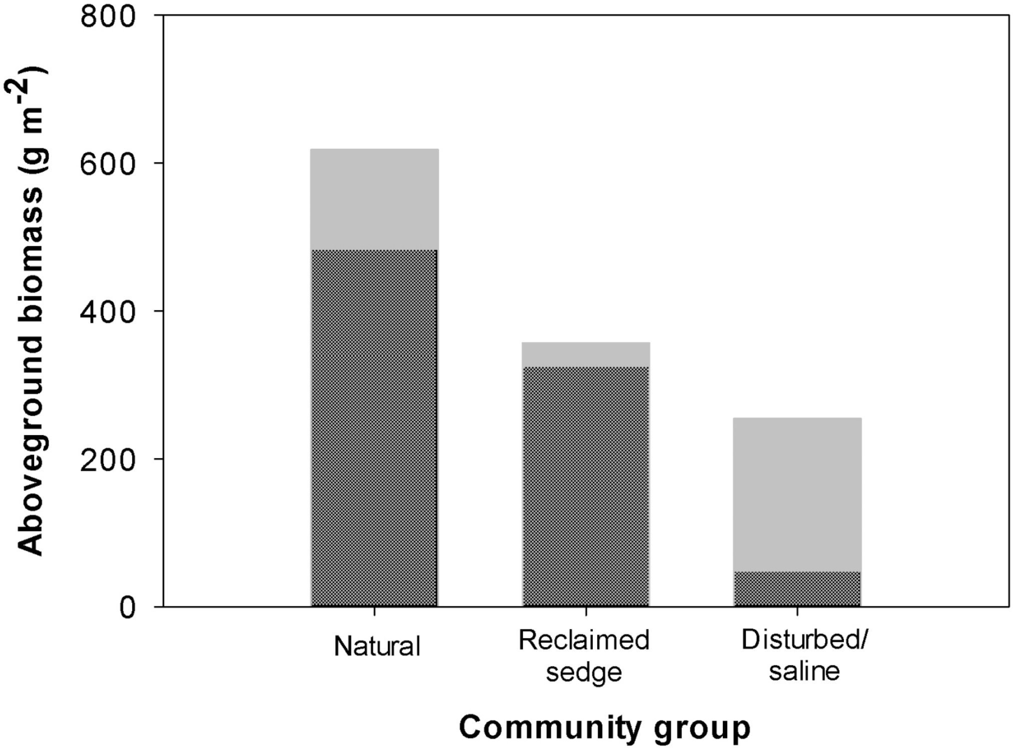

Figure 3.3. *Carex* spp. contribution to wet meadow aboveground biomass by community group. Dark grey represents aboveground biomass of *Carex* spp.; light grey represents the portion of aboveground biomass contributed by other species (n=45 sites).

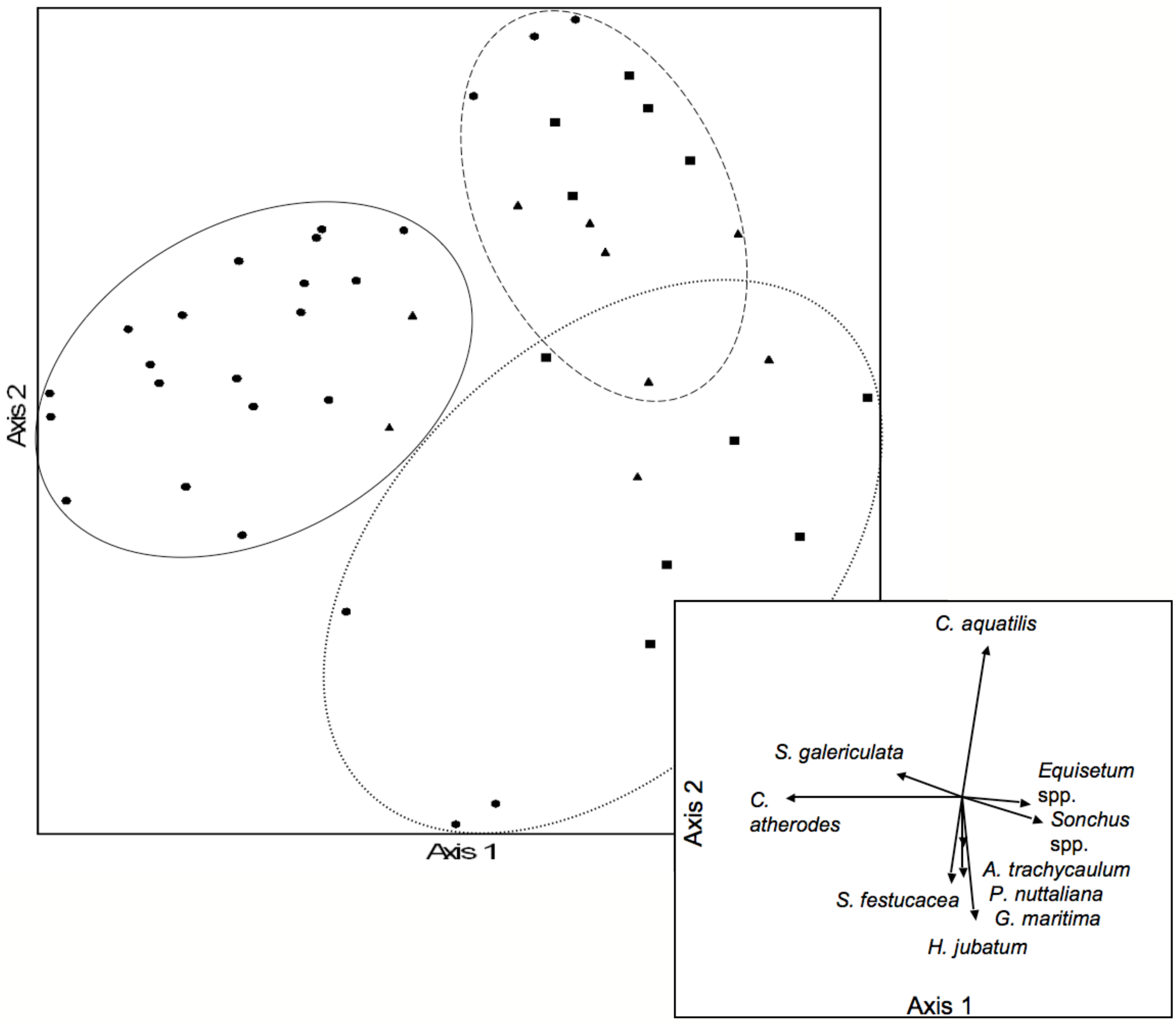

Figure 3.4. Wet meadow vegetation community cover data NMS ordination with inset of species vector overlay joint plot. Ellipses indicate the 3 community groups identified by cluster analysis: natural (solid line), reclaimed sedge (dashed line), and disturbed/saline (dotted line) with 3 site treatments identified by symbols: REF (circle), OSREF (triangle), and OSPA (square). Overlay vectors are included at R^2>0.20.

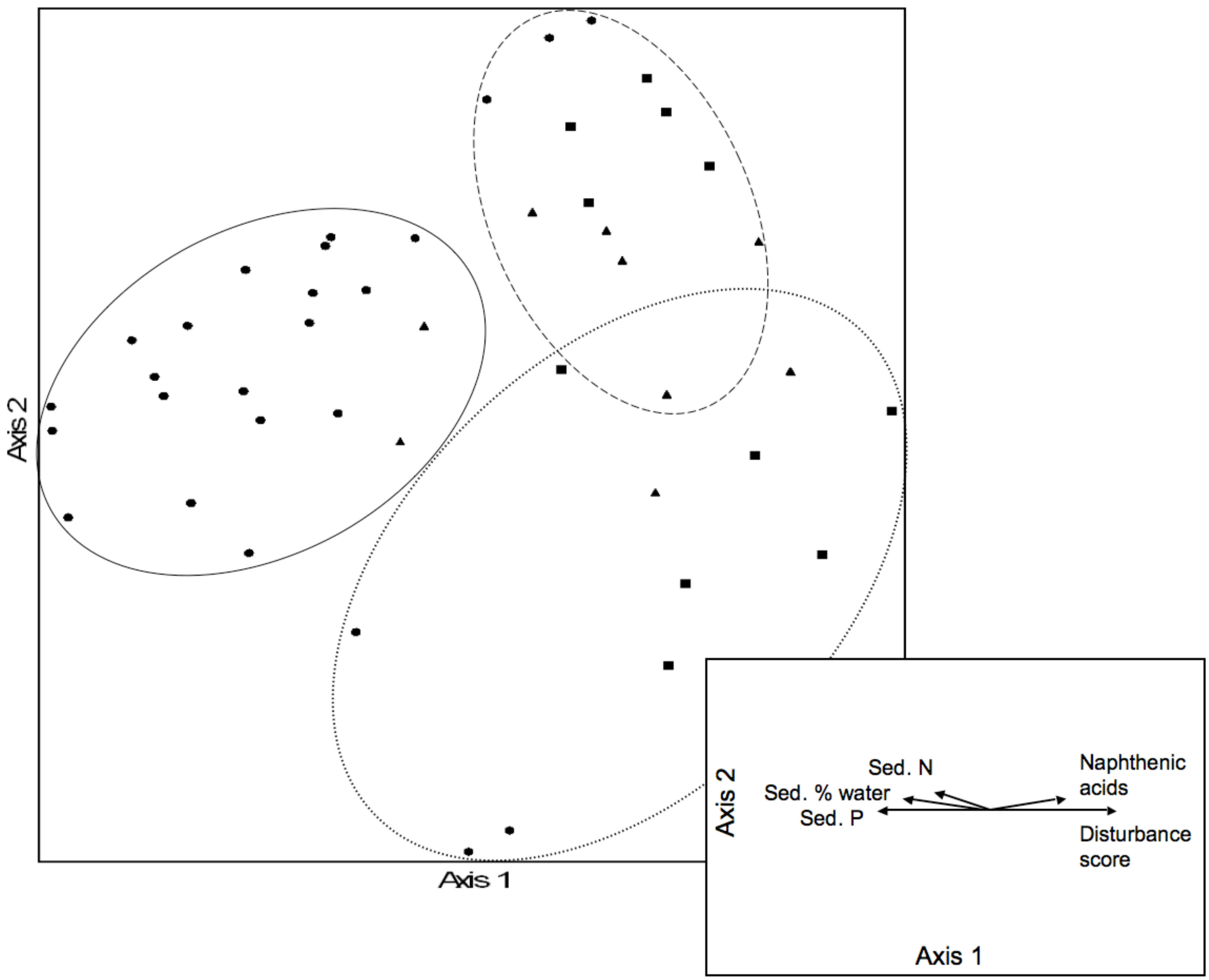

Figure 3.5. Wet meadow vegetation community cover data NMS ordination with inset of environmental vector overlay joint plot. Ellipses indicate the 3 community groups identified by cluster analysis: natural (solid line), reclaimed sedge (dashed line), and disturbed/saline (dotted line) with 3 site treatments identified by symbols: REF (circle), OSREF (triangle), and OSPA (square). Overlay vectors are included at R^2>0.20.

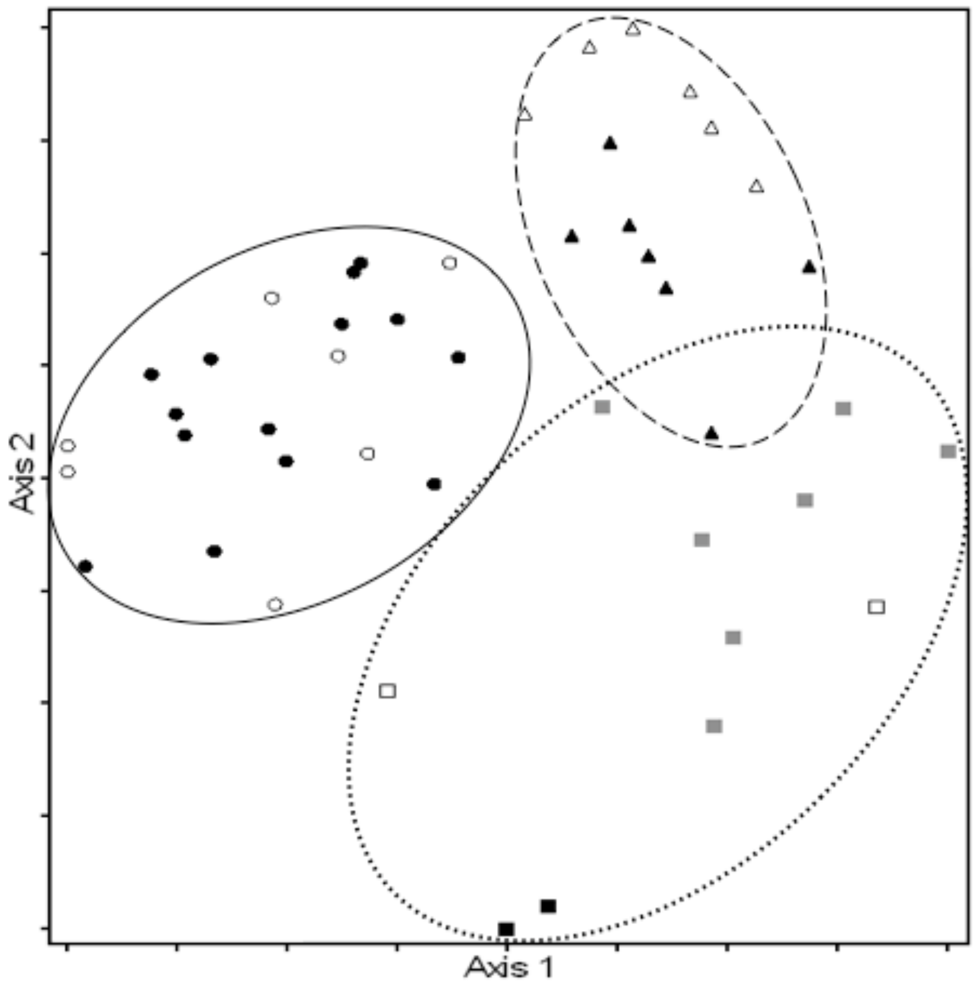

Figure 3.6. Wet meadow community cover data NMS ordination showing 7 sub-communities: natural I (black circle), natural II (open circle), reclaimed sedge I (open triangle), reclaimed sedge II (black triangle), disturbed (grey square), Reedgrass (open square) and natural saline (black square). Ellipses indicate the 3 community groups of which the sub-communities are members: natural (solid line), reclaimed sedge (dashed line), and disturbed/saline (dotted line).

4. General discussion

Assessment of the reconstruction of ecosystems following oil sands mining, which is poised to be one of the largest reclamation projects ever undertaken, occurs rarely in scientific literature. The western boreal forest, which includes the boreal plain ecoregion where this study was conducted, has also been under-studied in comparison to other regions (Foote and Krogman 2006). Thus, this project contributes to both the knowledge on the region and the unique problems associated with oil sands reclamation, as well as to the development of tools to assess success in the reclamation process.

To address the need for monitoring and assessment of oil sands reclaimed wetlands, Chapter 2 developed a wetland assessment tool, the vIBI, which employed the vegetation community as an indicator of ecological health, i.e., similarity to appropriate natural analogues. Similar tools have been developed and successfully implemented elsewhere to enforce environmental policy (Simon 2000). If integrated into a clearly mandated reclamation policy for oil sands industrial operations, this tool could be used to monitor success, and reliably determine if minimum reclamation requirements are met. The vIBI assessment confirms that some of the reclaimed wetlands that exhibit healthy looking vegetation in their wet meadow zone are reaching levels of ecological health within the range of natural reference sites. However, many reclaimed wetlands still perform below this level and it is clear that reclamation managers and engineers will need to determine the barriers preventing the poorest performing sites from achieving ecological health.

Chapter 3 identified some of these barriers by undertaking a detailed examination of the vegetation communities at the study sites with respect to community composition, productivity, and underlying physical and chemical factors that may be affecting vegetation. Three vegetation communities were apparent: a natural sedge-dominated community, a reclaimed sedge-dominated community, and a reclaimed community of disturbance tolerant, invasive, and halophyte plant species. Clear patterns of environmental variables emerged between natural marshes of the boreal region and oil sands reclaimed wetlands, with higher nutrients and sediment water content found in the natural wetlands. However, patterns among oil sands reclaimed wetlands were less obvious and the communities may be more affected by initial reclamation activity than the local environmental factors measured in this study. Neither salinity nor nutrient availability, generally thought of as two of the greatest barriers to the establishment of ecologically healthy wetlands in the post-mining landscape (Harris 2007), appeared to vary among oil sands reclaimed wetlands along a gradient from disturbed to healthy. We hypothesize that instead, once appropriate hydrologic conditions exist, colonization by vegetation propagules may be the greatest determinant of vegetation community on newly reclaimed oil sands surfaces. The vegetation community, once established, will then contribute biomass and litterfall to the sediment, thus increasing organic matter, sediment nutrients, and lowering bulk density by increasing water content (Noon 1996).

As part of the large-scale reclamation that will begin in the next decade, reclamation engineers will need to recreate landscape on a scale that is matched by few other projects historically. Reclaimed wetlands in the oil sands have developed highly variable vegetation communities, likely resulting from the high degree of variability in physical and chemical characteristics and species colonization. Only a small number of these reclaimed wetlands demonstrate levels of ecological health comparable to appropriate natural analogues, and it does not appear that age is a factor in determining their performance. Thus, reclamation engineers and managers must achieve something more than reclamation at a local level; they must determine how to achieve what has been referred to as 'landscape success' (Kentula 2000).

Many uncertainties remain in the reclamation process with respect to the types of wetlands that will be possible to construct and the materials available to do so. A relatively tiny area of land has been reclaimed thus far, and the constant evolution of the oil sands extraction process means that tailings materials may change considerably before full-scale reclamation begins. Future research should focus on determining the optimal vegetation reclamation strategies for reclaimed wetlands as they have been constructed, while understanding the materials being used and the effects they will have on the biota.

The scope of the vIBI developed in this thesis, which is specific to oil sands marsh reclamation monitoring, is narrower than vIBIs developed to assess the wetlands of entire states, such as Ohio (Mack et al. 2000), North Dakota (DeKeyser et al. 2003), Michigan (Simon et al. 2001), and Pennsylvania (Miller et al. 2006). If this vIBI is to be applied to other regions, or to other wetland types, it must be tested first for its effectiveness in diagnosing ecological health in these new situations and metrics must be adjusted accordingly (Karr and Chu 1999).

The future climate scenario that will affect the oil sands region in northern Alberta is also uncertain. Currently accepted models project temperatures in the northern boreal plains to rise 2-3 ˚C over the next 30-40 years (CCCma 2010), which would increase evapotranspiration, and dramatically decrease the amount of water on the landscape, making the difficult task of reclaiming wetlands even more complex. For example, decreased sediment water content, which has been identified as a stress on the vegetation community in oil sands reclaimed wetlands, would be exacerbated by increases in surface temperature and require special attention in reclamation planning.

Recently Wray and Bayley (2006) prepared a report for Alberta Environment reviewing possible indicators relevant to ecological assessment of central and southern Alberta wetlands, and the Alberta Water Research Institute has funded a project that is developing assessment techniques for wetlands in the prairie parkland region. These projects, along with the work we have done in oil sands reclamation assessment, signals the beginning of IBI development for use in monitoring the restoration, reclamation, and construction of wetlands in Alberta. This is an issue of increasing importance as agricultural, forestry, and oil and gas development expand over the landscape. There are currently 4 major oil sands mining operations in the Athabasca oil sands region, however as of 2009 an additional 10 projects were approved for operation, or are already under construction, with 8 more either announced or under application for approval (ERCB 2009). With the oil sands poised for rapid operational growth, the ability to inform the practice of reconstructing ecosystems from the ground up, and to assess the health of the ecosystems left in their wake, will be critical in mitigating long-term damage to the boreal landscape.

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Table A.1. Site locations and vIBI scores.

Site ID code	Site name	Site location (lat., long.; dec. deg.)	Site treatment type	AvIBI score	BvIBI score
GOLDEN	Golden Pond	56.98356, -111.61679	OSREF	6	3
S_PIT	S-Pit	57.10532, -111.64202	OSREF	6	3
TESTPOND9	Test Pond 9	57.08335, -111.68345	OSPA	6	3
ETOEBERM	East Toe Berm	57.08977, -111.62656	OSPA	8	5
MILLSEEP	Millenium seepage wetland	56.89300, -111.37576	OSPA	8	3
JANS	Jan's Pond	56.98349, -111.51691	OSPA	10	5
MIKES	Mike's Pond	57.11179, -111.37272	OSPA	10	5
SEEPAGE	Seepage control pond	57.08360, -111.63343	OSPA	12	7
CELL44	SWSS Cell 44	56.96676, -111.80002	OSPA	14	7
HISULPH	Hi Sulphate wetland	56.99718, -111.55285	OSPA	14	7
NWINTER	Northwest Interceptor Ditch	57.10020, -111.68347	OSREF	14	5
4MCT	4m CT wetland	56.98349, -111.51691	OSPA	16	9
CELL46	SWSS Cell 46	56.99552, -111.80147	OSPA	16	7
CRANELK	Crane Lake	56.98354, -111.55003	OSREF	16	9
MCPHAIL	McPhail pond	53.60109, -105.89838	REF	16	5
DEEP	Deep Wetland	57.06690, -111.68340	OSREF	18	11
MIQ03	Miquelon site 3	53.24945, -112.87636	REF	18	9
SALTMARSH	Saltmarsh	56.98351, -111.53338	OSREF	18	7
SHALLOW	Shallow wetland	57.06691, -111.68345	OSREF	18	7
SUNCORNAT	Suncor Natural Wetland	56.96689, -111.50015	OSPA	18	7

Site ID code	Site name	Site location (lat., long.; dec. deg.)	Site treatment type	AvIBI score	BvIBI score
SWSSBEAV	SWSS Beaver Wetland	56.98464, -111.71557	OSREF	18	13
19UTIK	Utikuma pond 19	56.06691, -115.53341	REF	20	9
BLACKFT01	Blackfoot site 1	53.52870, -112.78900	REF	20	9
MIQ01	Miquelon site 1	53.24553, -112.88478	REF	20	13
MIQ36	Miquelon site 36	53.23380, -112.86980	REF	20	13
1440LLB	Lac La Biche site 1440	54.95855, -111.86422	REF	22	13
BIRCHBAY	Birch Bay	53.61137, -105.89491	REF	22	11
CL1	Child Lake site 1	58.42637, -116.54110	REF	22	11
ELK2	Elk Island site 2	53.52349, -112.92499	REF	22	11
ELKSOAP	Elk Island soap holes	53.60580, -112.80777	REF	22	11
HAY2	Hay site 2	59.10428, -118.05735	REF	22	13
MIQ23	Miquelon site 23	53.23534, -112.87925	REF	22	13
BILLS	Bill's Lake	56.99890, -111.61212	OSREF	24	13
MIQ02	Miquelon site 2	53.24451, -112.88114	REF	24	13
BLACKFT02	Blackfoot site 2	53.51468, -112.85032	REF	26	15
CL4D	Child Lake site 4D	58.42405, -116.55186	REF	26	13
CL5	Child Lake site 5	58.42045, -116.54059	REF	26	15
ELK2B	Elk Island site 2B	53.52290, -112.93150	REF	26	13
HAYRIV01	Hay River site 1	59.10767, -118.04728	REF	26	13
HAYRIV02	Hay River site 2	59.11012, -118.07887	REF	26	13
HAYRIV03	Hay River site 3	59.10845, -118.08132	REF	26	11
CLSOUTH	Child Lake site south	58.42083, -116.54518	REF	28	15

Site ID code	Site name	Site location (lat., long.; dec. deg.)	Site treatment type	AvIBI score	BvIBI score
CLWEST	Child Lake site west	58.42427, -116.55934	REF	28	13
CL4C	Child Lake site 4C	58.42255, -116.55118	REF	30	15
CLWP68	Child Lake site 68	58.42481, -116.55084	REF	30	15

Table A.2. Full list of species encountered during this study. Certain species were not identified beyond genus-level. Composite species were created when identification between two species was not possible, and are denoted by species codes ending in -spa.

Species name	Zone	Species code	USDA wetland indicator status
Achillea millefolium	**WM***	Achimil	FACU
Achillea sibirica	**WM***	Achisib	UPL
Acorus americanus	**WM/EM**	Acorame	OBL
Actaea rubra	**WM**/EM	Actarub	UPL
Agrimonia striata	WM	Agristr	FACU
Agropyron trachycaulum	**WM***	Agrotra	FACU
Agrostis scabra	**WM***/EM	Agrosca	FAC
Agrostis stolonifera	**WM**	Agrosto	FACW
Alisma plantago-aquatica	**WM**/EM	Alispla	OBL
Alopecurus aequilis	**WM/EM**	Alopaeq	OBL
Amelanchier alnifolia	WM	Amelaln	FACU
Anemone riparia	WM	Anemrip	NI
Arctostaphylos uva-ursi	**WM**	Arctuva	UPL
Arenaria spp.	**WM**	Arenspe	NI
Artemisia biennis	WM/EM	Artebie	FAC
Aster borealis	**WM**	Astebor	OBL
Aster brachyactis	**WM*/EM**	Astebra	FACW
Aster ciliolatus	**WM**	Astecil	UPL
Aster falcatus	**WM**	Astefal	FACU
Aster hesperius	**WM*/EM**	Astehes	OBL
Aster pauciflorus	WM	Astepau	FACW
Aster puniceus	**WM**/EM	Astepun	OBL
Astragalus canadensis	**WM**	Astrcan	FAC
Atriplex prostrata	WM	Atripro	NI
Atriplex subspicata	**WM**	Atrisub	NI
Beckmannia syzigachne	**WM*/EM**	Becksyz	OBL
Betula glandulosa	**WM**	Betugla	OBL
Betula spp.	**WM**	Betuspe	NA
Bidens cernua	**WM*/EM***	Bidecer	OBL
Bromus ciliatus	**WM**	Bromcil	FACW
Bromus inermis	WM	Bromine	UPL
Calamagrostis canadensis	**WM***/EM	Calacan	FACW
Calamagrostis spp. *(stricta + inexpansa)*	**WM*/EM***	Calaspa	FACW
Calla palustris	**WM**	Callpal	OBL
Carex aquatilis	**WM*/EM***	Careaqu	OBL
Carex atherodes	**WM*/EM***	Careath	OBL
Carex aurea	**WM**	Careaur	FACW

Species name	Zone	Species code	USDA wetland indicator status
Carex bebbii	**WM***	Carebeb	OBL
Carex canescens	**WM**/EM	Carecan	OBL
Carex chordorrhiza	**WM**	Carecho	OBL
Carex crawfordii	**WM**/EM	Carecra	FAC
Carex diandra	**WM**/EM	Caredia	OBL
Carex disperma	WM/EM	Caredis	FACW
Carex lanuginosa	**WM**	Carelan	OBL
Carex lasiocarpa	**WM**	Carelas	OBL
Carex praegracilis	WM	Carepra	FACW
Carex pseudo-cyperus	**WM**/EM	Carepse	OBL
Carex retrorsa	**WM**	Careret	OBL
Carex sartwellii	WM	Caresar	FACW
Carex utriculata	**WM*/EM***	Careutr	OBL
Carex vaginata	WM	Carevag	OBL
Castilleja raupii	**WM**	Castrau	FAC
Chenopodium album	WM/**EM**	Chenalb	FAC
Chenopodium capitatum	WM	Chencap	UPL
Chenopodium glaucum	WM	Chengla	FACW
Chenopodium rubrum	**WM**	Chenrub	OBL
Cicuta bulbifera	**WM*/EM**	Cicubul	OBL
Cicuta maculata	**WM*/EM***	Cicumac	OBL
Cirsium arvense	**WM*/EM**	Cirsarv	FACU
Coptis trifolia	**WM**	Copttri	FACW
Deschampsia caespitosa	**WM*/EM**	Desccae	FACW
Distichlis spicata	**WM**	Distspi	FACW
Drepanocladus aduncus	WM	Drepadu	OBL
Eleocharis acicularis	**WM*/EM**	Eleoaci	OBL
Eleocharis palustris	**WM*/EM***	Eleopal	OBL
Epilobium angustifolium	**WM**/EM	Epilang	FACU
Epilobium glandulosum	**WM*/EM**	Epilgla	FACW
Epilobium palustre	**WM*/EM***	Epilpal	OBL
Equisetum spp. *(arvense + fluviatile)*	**WM*/EM***	Equispa	FAC
Erigeron philadelphicus	**WM**/EM	Erigphi	FACW
Eriophorum spp.	WM	Eriospe	OBL
Erysimum cheiranthoides	WM	Erysche	FACU
Fragaria vesca	**WM**	Fragves	NI
Fragaria virginiana	**WM***	Fragvir	FACU
Galeopsis tetrahit	**WM**/EM	Galetet	NI
Galium labridoricum	**WM**	Galilab	OBL
Galium trifidum	**WM*/EM***	Galitri	FACW

Species name	Zone	Species code	USDA wetland indicator status
Galium triflorum	**WM**	Galitrf	FACU
Geum aleppicum	**WM**/EM	Geumale	FAC
Geum macrophylum	**WM***	Geummac	FACW
Geum rivale	**WM**	Geumriv	FACW
Glaux maritima	**WM***/EM	Glaumar	OBL
Glyceria grandis	**WM/EM**	Glycgra	NI
Glyceria striata	**WM**/EM	Glycstri	OBL
Habenaria hyperborea	**WM**	Habehyp	FACW
Habenaria viridis	WM	Habevir	FAC
Hieracium umbellatum	**WM***	Hierumb	NI
Hierochloe odorata	WM	Hierodo	FACW
Hippurus vulgaris	**WM*/EM***	Hippvul	OBL
Hordeum jubatum	**WM*/EM**	Hordjub	FAC
Impatiens capensis	**WM**/EM	Impacap	FACW
Juncus alpinus	**WM***/EM	Juncalp	OBL
Juncus balticus	**WM***/EM	Juncbal	OBL
Juncus bufonius	**WM*/EM***	Juncbuf	OBL
Juncus filiformis	**WM**	Juncfil	FACW
Ledum groenlandicum	**WM**	Ledugro	OBL
Lotus corniculatus	**WM**	Lotucor	FAC
Lycopus asper	**WM*/EM**	Lycoasp	OBL
Lysimachia thyrsiflora	**WM**	Lysithy	OBL
Marchantia polymorpha	**WM**	Marcpol	NI
Melilotus spp. *(alba + officinalis)*	**WM**	Melispa	FACU
Mentha arvensis	**WM*/EM***	Mentarv	FACW
Menyanthes trifoliata	WM	Menytri	OBL
Myrica gale	**WM**	Myrigal	OBL
Parnassia palustris	**WM***	Parnpal	OBL
Petasites frigidus	**WM**/EM	Petafri	FACW
Petasites sagitattus	WM	Petasag	FACW
Phalaris arundinacea	**WM*/EM**	Phalaru	FACW
Phragmites australis	WM/**EM**	Phraaus	FACW
Plantago eriopoda	**WM**	Planeri	FAC
Poa palustris	**WM***/EM	Poapalu	FAC
Poa pratensis	**WM**	Poaprat	FACU
Polygonum amphibum	**WM*/EM**	Polyamp	OBL
Polygonum lapathifolium	**WM/EM**	Polylap	FACW
Populus spp.	**WM**/EM	Popuspe	NA
Potentilla anserina	**WM***/EM	Poteans	FACW
Potentilla gracilis	WM	Potegra	FAC

Species name	Zone	Species code	USDA wetland indicator status
Potentilla norvegica	**WM***/EM	Potenor	FAC
Potentilla palutstris	**WM***/EM	Potepal	OBL
Potentilla rivalis	**WM**/EM	Poteriv	FACW
Puccinellia nuttalliana	**WM*/EM**	Puccnat	FACW
Ranunculus abortivus	**WM**	Ranuabo	FACW
Ranunculus cymbalaria	**WM*/EM**	Ranucym	OBL
Ranunculus macounii	WM	Ranumac	OBL
Ranunculus reptans	**WM**	Ranurep	FACW
Ranunculus sceleratus	**WM/EM***	Ranusce	OBL
Rhinanthus borealis	**WM**	Rhinbor	FACU
Ribes spp.	**WM**	Ribespe	NA
Ricciocarpos natans	WM	Riccnat	NI
Rorippa islandica	**WM/EM***	Roriisl	OBL
Rosa acicularis	WM	Rosaaci	FACU
Rubus idaeus	**WM**	Rubuida	FACU
Rubus spp.	**WM**	Rubuspe	NA
Rumex maritima	**WM/EM***	Rumemar	FACW
Rumex occidentalis	**WM***/EM	Rumeocc	OBL
Salicornia rubra	**WM/EM**	Salirub	OBL
Salix spp.	**WM*/EM**	Salispe	NA
Scholochloa fetucacea	**WM*/EM***	Schofes	OBL
Scirpus cyperinus	WM	Scircyp	OBL
Scirpus microcarpus	**WM***/EM	Scirmic	OBL
Scirpus paludosus	**WM*/EM***	Scirpal	OBL
Scirpus pungens	WM	Scirpun	OBL
Scirpus spp. (validus + acutus)	**WM*/EM***	Scirspa	OBL
Scutellaria galericulata	**WM*/EM**	Scutgal	OBL
Senecio congestus	WM/**EM**	Senecon	FACW
Senecio eremophilus	WM/EM	Seneere	FACU
Sium suave	**WM*/EM***	Siumsua	OBL
Smilacina stellata	WM	Smilste	FAC
Solidago canadensis	**WM**	Solican	FACU
Solidago gigantea	**WM**	Soligig	FACW
Sonchus spp. (arvensis + ugilinosus)	**WM*/EM***	Soncspa	FAC
Sparganium angustifolium	**WM/EM***	Sparang	OBL
Spartina pectinata	**WM**	Sparpec	FACW
Sphagnum spp.	**WM**	Sphaspe	OBL
Stachys palustris	**WM***	Stacpal	OBL
Stellaria calycantha	**WM***/EM	Stelcal	FACW
Stellaria longifolia	**WM**	Stellon	FACW

Species name	Zone	Species code	USDA wetland indicator status
Suaeda calceoformis	**WM/EM**	Suaecal	FACW
Taraxacum officinalis	**WM***	Taraoff	FACU
Trifolium hybridum	**WM**	Trifhyb	FAC
Triglochin maritima	**WM*/EM**	Trigmar	OBL
Triglochin palustris	**WM*/EM***	Trigpal	OBL
Typha latifolia	**WM*/EM***	Typhlat	OBL
Urtica dioica	**WM*/EM**	Urtidio	FAC
Vaccinium vitis-idaea	**WM**	Vaccvit	FAC
Vicia americana	WM	Viciame	FAC

Zone legend

WM	Found only in wet meadow zone walkaround
WM	Found in wet meadow zone 1 m^2 quadrat
WM*	Found in wet meadow zone 1 m^2 quadrat and included in multivariate analysis
EM	Found only in emergent zone walkaround
EM	Found in emergent zone 1 m^2 quadrat
EM*	Found in emergent zone 1 m^2 quadrat and included in multivariate analysis

USDA Wetland indicator status legend (USDA, NRCS 2010)

OBL	Obligate wetland	Occurs almost always under natural conditions in wetlands.
FACW	Facultative wetland	Usually occurs in wetlands, but occasionally found in non-wetlands.
FAC	Facultative	Equally likely to occur in wetlands or non-wetlands.
FACU	Facultative upland	Usually occurs in non-wetlands.
UPL	Obligate upland	Almost always occurs under natural conditions in non-wetlands.
NI	No information available	
NA	Not applicable	

Table A.3. Species considered invasive to Alberta and provincially regulated as Restricted or Noxious.

Species name	Common name
Bromus tectorum	downy chess/brome
Caragana spp.	caragana
Cardaria chalepensis	hoary cress
Cardaria pubescens	globe-podded hoary cress
Carduus nutans	nodding thistle
Centaurea diffusa	diffuse knapweed
Centaurea maculosa	spotted knapweed
Centaurea repens	Russian knapweed
Centaurea solstitialis	yellow star thistle
Chrysanthemum leucanthemum	ox-eye daisy
Cirsium arvense	Canada thistle
Convolvulus arvensis	field bindweed
Cuscuta gronovii	common dodder
Cynoglossum officinale	hound's tongue
Echium vulgare	viper's-bugloss; blueweed
Elaeagnus angustifolia	Russian olive
Erodium cicutarium	stork's bill
Euphorbia cyparissias	cypress spurge
Euphorbia esula	leafy spurge
Galium aparine	cleavers
Galium spurium	false cleavers
Knautia arvensis	blue buttons, field scabious
Linaria dalmatica	broad-leaved/Dalmatian toadflax
Linaria vulgaris	butter-and-eggs/toadflax
Lolium persicum	Persian darnel
Lychnis alba	white cockle
Lythrum salicaria	purple loosestrife
Matricaria perforata	scentless chamomile
Myriophyllum spicatum	Eurasian water milfoil
Odontites serotina	late-flowering eyebright
Ranunculus acris	tall buttercup
Rhamnus catharticus	European (common) buckthorn
Silene cucubalus	bladder campion
Sonchus arvensis	perennial sow thistle
Tamarix spp.	tamarisk/salt cedar
Tanacetum vulgare	common tansy

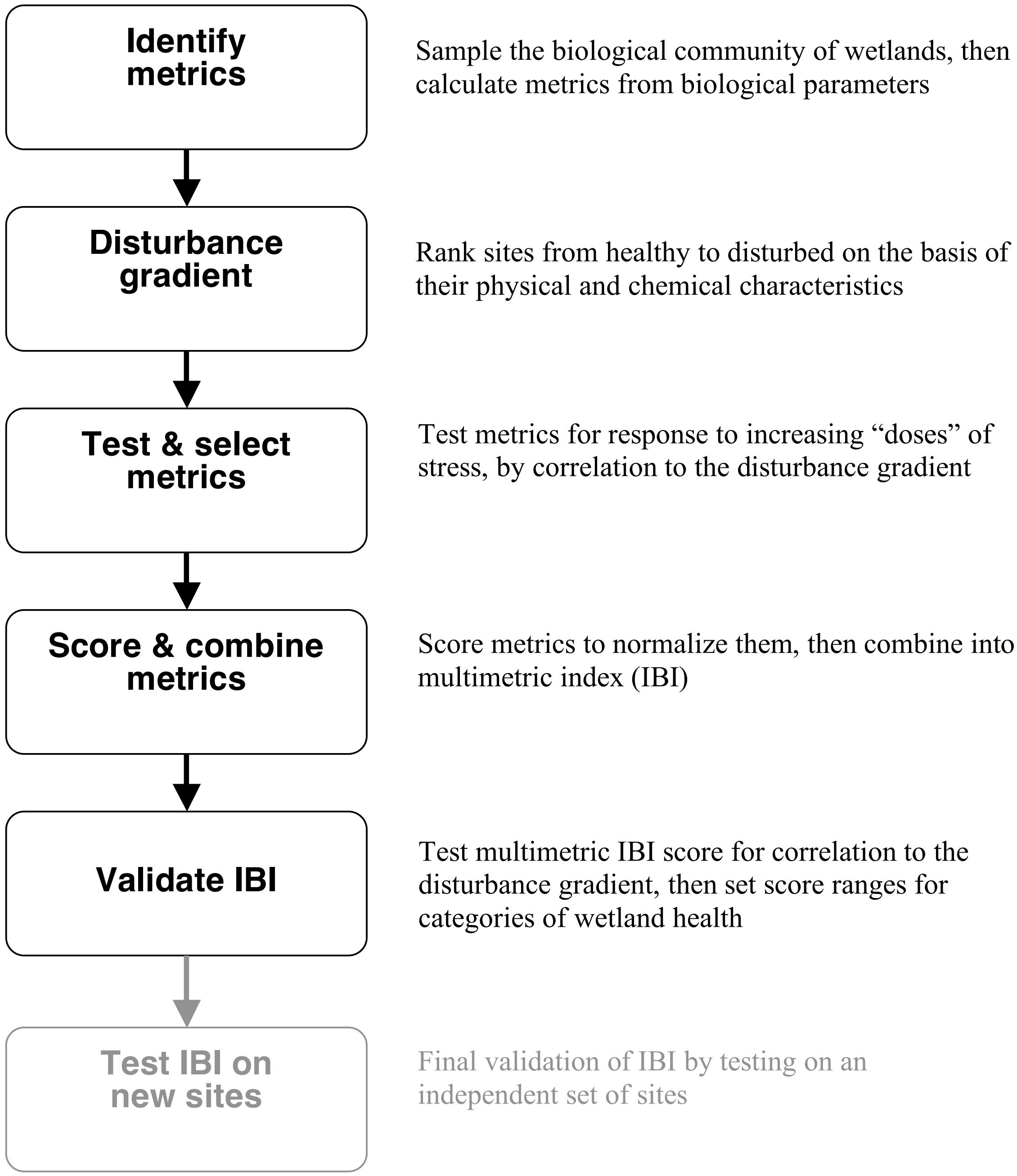

Figure A.1. Development process for the oil sands wetland vegetation-based Index of Biological Integrity. Grey text indicates the final step in IBI validation before implementation, which was not possible due to lack of new oil sands sites at the time of this study.