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Wetland assessment in Alberta's oil sands mining area

by

Rebecca Campbel Rooney

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Abstract

Oil sands mining in Alberta will destroy tens of thousands of hectares of boreal habitat. This land will need to be reclaimed. Current closure plans call for the construction of shallow open water wetlands to cover about 10-30% of the reclaimed landscape. Already, several trial wetlands have been constructed by mine operators, but no large-scale wetland creation has been attempted. For wetland reclamation to be successful, clear targets and tools for wetland monitoring and assessment are needed. I characterized the local- and landscape-level environmental conditions and aquatic plant communities in naturally occurring, undisturbed shallow open water wetlands to serve as a reference for comparison with reclaimed wetlands. I developed two related tools to evaluate wetland condition; one focusing on levels of abiotic stress, another on biological integrity. Using these tools, I conclude that current constructed wetlands differ from reference sites in terms of aquatic plant community structure, nutrient levels, and exposure to contaminants like naphthenic acids. Using multivariate analyses, I identified seven distinct biotic assemblages, two of which might serve as targets for future reclamation. I modelled the relationship between local- and landscape-level variables and aquatic plant diversity to test hypotheses about the relative importance of relationships between environmental variables and species richness. I conclude that diversity is more strongly related to local variables than surrounding land use, but that land use does play a role, albeit one that changes with the spatial scale considered. My results can inform reclamation practices by setting clear goals for future projects and by providing tools to measure progress towards them.

Acknowledgements

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Finally, my family, friends, and first teachers who instilled in me a love of nature, a sense of justice, and the temerity to believe that one person can make a difference. Brenda, Robert, and Caitlin Rooney, David Presonka, Marie-Lynn Hamilton, Nick Lapointe, Keith Hallgren, Josie Unka, Heather Force, Lianne Reddie, and all the Terra Informers; you fostered my inquisitive instincts – no matter what a mess I made. Look Mom, I caught up and now I'm at the head of the class. Steve Andersen, let's keep getting into mischief:

> "Bring your Swiss-army knife, and a bottle of something/
> and I'll bring some spray paint, and a new deck of cards."
>
> \- John K. Samson.

Contents

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List of Abbreviations

AIC: Akaike's information criterion

ANOVA: Analysis of variance

BPJ: Best professional judgement

CART: Classification and regression tree

CEMA: Cumulative Effects Management Association

Chl-a: Chlorophyll-a

CRR: Continuous reference range method of scoring biological metrics

CWR: Continuous whole range method of scoring biological metrics

DOC: Dissolved organic carbon

DRR: Discrete reference range method of scoring biological metrics

DWR: Discrete whole range method of scoring biological metrics

ERCB: Energy and Resources Conservation Board

EIA: Environmental impact assessment

EPEA: Alberta's Environmental Protection and Enhancement Act

GIS: Geographic information system

GOA: Government of Alberta

IBI: Index of biological integrity

ISA: Indicator species analysis

LOI: Loss on ignition at 550 °C

MRPP: Multi-response permutation procedure

NMS: Non-metric multi-dimensional scaling ordination

OSPA: Oil sands process affected wetlands (reclamation wetlands contaminated by oil sands tailings)

OSREF: Oil sands reference wetland (reclamation wetlands free from tailings contamination)

PCA: Principal components analysis

REF: Reference wetlands (natural wetlands free from oil sands related stress or disturbance)

RMSEA: Root mean square error of approximation

SAV: Submersed aquatic vegetation

SEM: Structural equation modeling

SRP: Soluble reactive phosphorus

TC: Total carbon

TDN: Total dissolved nitrogen

TDP: Total dissolved phosphorus

TDS: Total dissolved solids

TN: Total nitrogen

TOC: Total organic carbon

TP: Total phosphorus

TSS: Total suspended solids

1. *Context of oil sands mining and reclamation in Alberta, Canada.*

Introduction

The overarching goal of my thesis work is to develop and test a scientifically defensible tool for evaluating reclamation wetlands in Alberta's oil sands area and then to use it to assess existing reclamation wetlands. I anticipate that the results of this assessment will help inform future reclamation work by defining appropriate goals for wetland reclamation and by connecting wetland plant communities with local- and landscape-level conditions. Thus, an ancillary objective of my thesis work is to offer guidance to improve wetland reclamation practices in the oil sands area.

To set the context for the following data chapters, I begin with an introduction to oil sands mining and reclamation. It is necessary to understand the spatial scale at which reclamation is occurring and the challenges posed by the reclamation landscape and the materials available for wetland construction, to grasp the urgency with which guidance on wetland reclamation is needed. It is also important to know something of the legal context in which reclamation is occurring and how mine regulators and operators interpret the term "reclamation," as any evaluation of reclamation success must reflect this definition. Therefore, this chapter is devoted to describing the mining and reclamation process and the laws and regulations that govern it.

The resource

The oil sands deposits in Alberta make it the second largest oil reserve in the world (after Saudi Arabia), with 177 billion barrels of established bitumen reserves (ERCB 2010). Less than 4% of the initial established reserves have been exploited, leaving nearly 170 billion barrels of bitumen buried under about 14,020,000 ha of the Boreal

Plains ecoregion (ERCB 2010). About 475,000 ha of those deposits lie in the surface mineable area and will eventually be extracted by open pit mining (ERCB 2010). Already more than 99% of the surface mineable area is leased (GOA 2010a), meaning that an area larger than the state of Rhode Island will eventually be mined and will need to be reclaimed.

Ramping up

Commercial exploitation of oil sands by open pit mining began in 1967 with Suncor Energy Inc. Since then, growth in the industry has generally mirrored the price of synthetic crude oil: the upgraded product made from mined bitumen (Fig. 1-1). Development slowed with the economic downturn in 2008-2009 (Fig. 1-1); however, the price of oil recovered in 2011 and the production of oil from oil sands deposits is forecast to grow at an average annual rate of 5.9%, with capital investments in mining exceeding $11 billion/yr for at least the next decade (ERCB 2010). As of May 2011, approvals had been granted for 10 oil sands mines (Fig. 1-2). Total's Joslyn mine is expected to receive approval to operate in the summer of 2011 and Syncrude's Aurora South mine has been approved in principle (Richens pers. comm.). Another three mine proposals have entered the review process: Shell Canada's Jackpine Expansion and Pierre River mines, and Suncor Energy's Voyageur South mine. Furthermore, a new company has proposed building two additional mines: Tech Resources Ltd./Silver Birch Energy Corp.'s Equinox and Frontiers mines. With new mines on the horizon and new companies entering the field, clearly the oil sands mining industry's growth trend will continue.

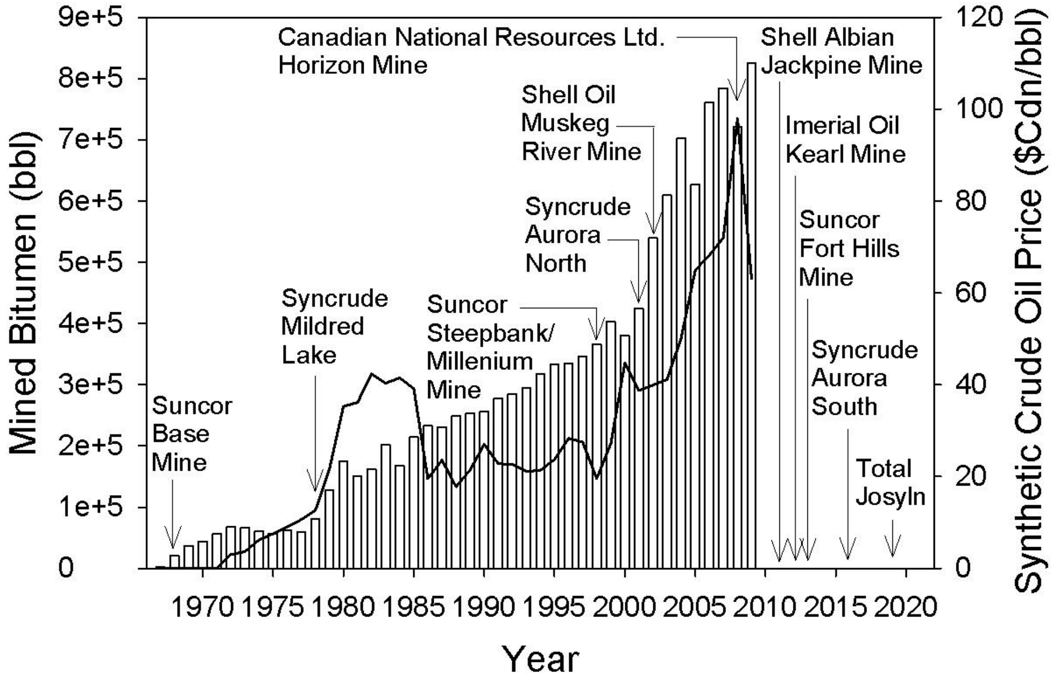

Fig. 1-1. Growth in mined bitumen (bars) from Alberta's oil sands region and the associated price per barrel (line) of synthetic crude oil. Synthetic crude oil is the product created by upgrading mined bitumen. It is generally sold at a premium relative to conventional crude oil. Arrows indicate start dates for mines that hold approvals to operate. Additional mines are currently engaged in the approval process. Data from Alberta's Energy Resources Conservation Board (ERCB 2010).

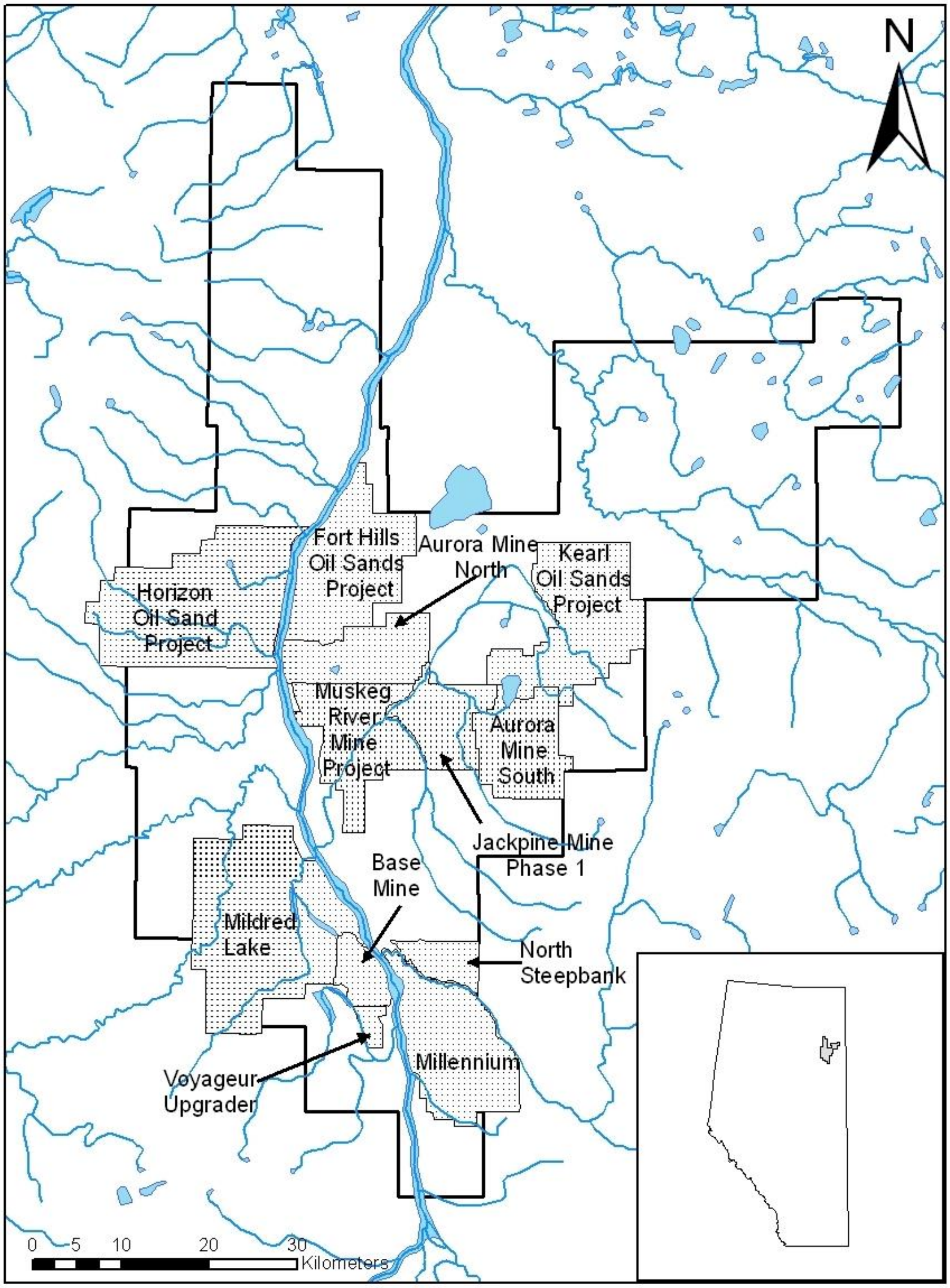

Fig. 1-2. Map of the surface mineable area and the footprints of oil sands mining projects with approval to operate as of March 2011. Data adapted from the ERCB's online Scheme Approval Map Viewer (Keeler pers. comm.).

Mining

Oil sands mining involves first clear cutting and draining the land, then stripping off and stockpiling any peat and topsoil, before digging out the mineral overburden to reach the oil bearing sand deposit (Harris 2007). This creates open pits up to 100 m deep as well as large stockpiles of overburden and topsoil that must be stored until they can be reclaimed. Preventing the mine pits from flooding creates cones of hydrologic depressurization that draw down ground water levels (Woynillowicz et al. 2005). Mining can thereby dewater and destroy entire aquifers (Hackbarth 1980). The oil sand ore is dug up and transported to an on-site facility where the oil is separated from associated sand and clay particles using water and chemical solvents. Approximately 12 barrels of water are used to produce a single barrel of bitumen from oil sands ore (Mikula et al. 2008), but 70% of this is recycled, such that permanent withdrawls from the Athabasca River total two to five barrels of water per barrel of bitumen produced (Schindler et al. 2007).

Tailings

Once the bitumen is removed, the sand, clay, water, residual bitumen, and solvents become tailings materials. Tailings are contaminated with hydrocarbons and heavy metals including arsenic, mercury, lead, benzene, polycyclic aromatic hydrocarbons, and naphthenic acids (EC 2011; MacKinnon et al. 2004). Due to their toxicity, tailings are stored in tailings ponds on company leases, although there is substantial evidence that liquid tailings are leaking into the Athabasca River and surrounding ground water (see Timoney and Lee 2009 for review). Over three to five years, the liquid tailings separate into sand, mature fine tailings, and water that is recycled back into the extraction process (GOA 2011b). The mature fine tailings form a permanent suspension of about 30% fine sediment particles (diameter < 44 µm) in water, which has proven very difficult to treat (GOA 2011b). These tailings behave like quicksand and must be dewatered to create a trafficable surface before reclamation can commence. About 1.5 barrels of mature fine tailings are produced for every barrel of bitumen created by oil sands mining (Mikula et al. 2008).

Formerly, fine tailings were treated with gypsum to encourage particles to flocculate and settle, allowing the water to be pumped off and recycled. By this technique, fine tailings require 30 years or more to dewater (Mamer 2010). Consequently, tailings ponds have accumulated on company leases: they now cover >17,000 ha of land (Lemphers et al. 2010) and present a grave hazard to wildlife and downstream water resources (Grant et al. 2010). For example, in 2008, over 1600 ducks died after landing on a Syncrude Canada Ltd. tailings pond and becoming covered in bitumen (Timoney and Lee 2009).

Reclamation debt

The boom in bitumen production that began around the turn of the millennium (Fig. 1-1) is associated with a similar increase in the amount of land disturbed by oil sands mining. It is estimated that between 0.33 and 0.63 m^2 of boreal Alberta are destroyed for every 1 m^3 of synthetic crude oil produced by mining (Jordaan et al. 2009). Reclamation, however, has lagged behind, and we define the growing gap between the total amount of land disturbed and the amount that has been reclaimed as the reclamation debt (Fig. 1-3). Based on industry annual reports, as of December 31, 2009 over 67,330 ha were disturbed by oil sands mining and only 5609 ha were reported as temporarily or permanently reclaimed (Richens pers. comm.). These numbers are a product of industry self-reporting and do not represent the amount of land recognized by the government as reclaimed.

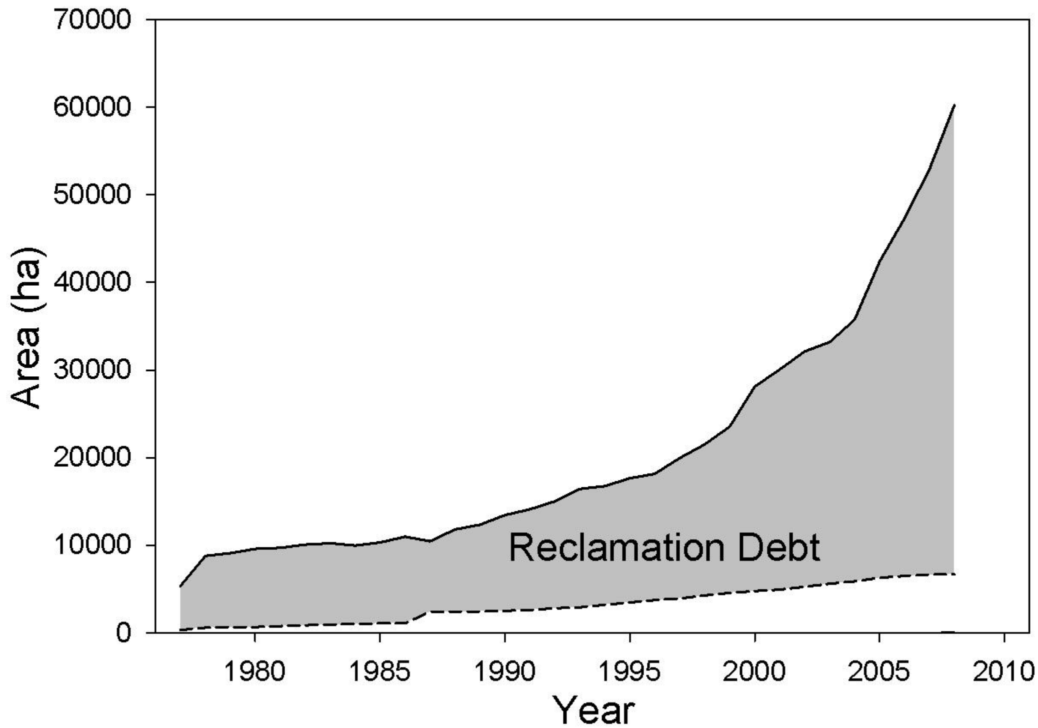

Fig. 1-3. The mounting reclamation debt (grey shading) is the difference between the cumulative hectares of land that have been disturbed by oil sands mining activities (solid line) and the cumulative hectares that the industry reports are reclaimed (dashed line). Note that of the 5609 ha of land reported as reclaimed by industry as of 31 December 2009, only 104 ha have been certified as reclaimed by the Alberta Government. Data provided by Alberta Environment (Richens pers. comm.).

To date, only 104 ha of land are certified as reclaimed: the Gateway Hill area on the lease of Syncrude Canada Ltd. was certified by Alberta Environment in 2008 (Capstick 2008). As the first parcel of land to be certified after over 40 years of oil sands mining, this was a substantial accomplishment, but considering how much land has been disturbed, it represents an insignificant area. Gateway Hill also represented the lowest hanging reclamation fruit. Before reclamation it served as the S4 South overburden placement area: it was never actually mined or exposed to tailings materials and their associated contaminants. In addition, whereas the land had originally been part of a large peatland complex, reclamation converted it into an upland area. No wetland has yet been evaluated for certification by Alberta Environment.

Responsible mining enterprises practice progressive reclamation, whereby reclamation proceeds alongside development so that reclamation liabilities are kept at a minimum. Progressive reclamation also allows reclamation practices to be evaluated and improved by the process of adaptive management, because they are implemented in stages at smaller scales. There are two main reasons why the practice of progressive reclamation has not been adopted by oil sands mining companies. First, there is no staged reclamation certification process administered by the government (Lemphers et al. 2010), so companies are not compelled to reclaim for the life span of their mines (20 to 40 yr) (GOA 2010b). Second, there is a scarcity of land ready to be reclaimed because so much land is needed to store the large volumes of liquid tailings during the decades necessary for them to dewater (Mamer 2010).

Mounting concern over growing tailings stockpiles prompted the Energy and Resources Conservation Board to issue a directive to reduce liquid tailings stockpiles through conversion into dry materials stored in designated disposal areas (ERCB 2009b). New tailings reductions technologies are expected to enable tailings ponds to be reclaimed within a decade (Mamer 2010), which should speed reclamation and reduce the amount of land required to store tailings, freeing it up to be reclaimed. Using new tailings reduction technology, Suncor Energy Ltd. began closure of its first tailings pond

(Wapisiw Lookout, formerly known as Tar Island) in 2007. Wapisiw Lookout is scheduled to become a 220 ha area of mixed-wood forest with a shallow open water wetland situated at the southern end to collect runoff (Suncor 2010a). In addition, the oldest mines (Fig. 1-1) are approaching the end of their life spans, and thus are beginning closure plan implementation. For example, Suncor has published the goal of doubling (100% increase over 2007 amount) the area of land reclaimed by 2015 (Suncor 2010a). With the expectation that reclamation will ramp-up in the coming years, there is an urgent need for standardized, scientifically sound means to evaluate reclaimed landscapes and a streamlined process for reclamation certification.

Legal context of reclamation

The Environmental Protection and Enhancement Act (EPEA) (GOA 1993) and the Water Act (GOA 2000) provide the regulatory framework for mining reclamation in Alberta. Both pieces of provincial legislation require proponents to seek approval from the Energy and Resources Conservation Board (ERCB) to operate a mine. The EPEA requires proponents to produce an environmental impact assessment (EIA) as part of their application to the ERCB, in which baseline environmental conditions are documented and predicted environmental impacts are noted. These EIAs also lay out a plan for reclamation of the site after mine closure. To avoid poorly functioning watersheds, inconsistent landforms, and inappropriate wildlife habitat, mine operators are expected to plan for reclamation and closure at the landscape-level and to integrate their plans with those of neighboring lease holders. The ERCB uses the EIAs to evaluate the economic, social, and environmental consequences of the proposed mine, and to determine whether the development is in the public interest.

If the proponent is granted approval to operate a mine, they must pay a security for land reclamation performance to cover the costs of reclamation should the operator fail to obtain reclamation certification (GOA 2011a). Until the operator has received a reclamation certificate, they retain outstanding reclamation liabilities that may discourage investors. Unfortunately, recent investigations have revealed that the securities held by the province ($820 million as of 2009) are grossly inadequate to cover the projected actual reclamation cost, with an estimated shortfall of $10 to $15 billion (Lemphers et al. 2010).

What is reclamation?

The EPEA's Conservation & Reclamation Regulations (AR 115/93 S2) defines the objective of reclamation as the reinstatement of "equivalent land capability," which it further defines as where "the ability of the land to support various land uses after conservation and reclamation is similar to the ability that existed prior to an activity being conducted on the land, but that the individual land uses will not necessarily be identical," (GOA 1993). This raises the distinction between reclamation and restoration. Generally, the goals of reclamation include stabilizing topography, eliminating hazards to the public, improving aesthetics, and any work required to enable people to put the land to its former uses, whereas the goal of restoration is to return a site to a precise condition (biological, chemical, and physical) typically based on its historic, pre-disturbance condition (SER 2004). Whereas reclamation may require revegetation to stabilize slopes and reduce erosion, restoration strives to recreate self-sustaining vegetation communities that resemble adjacent reference communities and support the same ecosystem functions. The law requires companies to reclaim the land following mine closure, they do not need to restore it.

Prior to 2007, EPEA approvals issued to oil sands mine operators provided no standard for reclamation beyond "equivalent land capability." EPEA approvals granted in 2007, however, redefined the statutory standard for what constitutes successful reclamation as where "the reclaimed soils and landforms are capable of supporting a self-sustaining, locally common boreal forest, regardless of the end land use," (GOA 2007a, 2007b, 2007d, 2007e). Also in 2007, the Oil Sands Multistakeholder Committee called by consensus on the Alberta Government to "define a reclamation standard that describes final certification requirements where site conditions are clearly self-sustaining, and where natural succession to a typical boreal ecosystem would occur"(GOA 2007c p. 22). Subsequently, the Cumulative Effects Management Association (CEMA) drafted the 2009 Framework for Reclamation Certification Criteria and Indicators for Mineable Oil Sands (Poscente 2009). The motivation behind this report was to formalize decision criteria that would be used to evaluate reclaimed landscapes for certification purposes to provide consistent decisions and to increase the transparency of the decision making process. The framework outlines three reclamation objectives: 1) to establish and integrate natural features on the reclaimed landscape, including wetlands, 2) natural functions are occurring on the reclaimed landscapes, 3) the end land use capability is equivalent to that prior to disturbance (Poscente 2009). CEMA has asked Alberta Environment and Alberta Sustainable Resources Development to endorse their framework, suggesting a move towards a more stringent definition of reclamation that more closely resembles restoration (sensu SER 2004).

Conclusion

Growth in the oil sands mining industry is resuming as the price of oil recovers from the slump of 2008-2009 (Fig. 1-1). Concomitant with industry growth is the continued expansion of the reclamation debt as companies fail to practice progressive reclamation (Fig. 1-3). Without large scale testing of proposed reclamation practices, a great deal of uncertainty remains regarding reclamation outcomes in the oil sands area, particularly regarding tailings ponds, which have already swollen to cover more than 17,000 ha of land (Lemphers et al. 2010). A new directive from the ERCB (ERCB 2009b) and pledges from mine operators (e.g., Suncor 2010a) mean that reclamation will soon begin in earnest, but as of yet there are no standardized and scientifically sound tools for evaluating wetland reclamation success. Even the definition of what constitutes reclaimed land is in the process of changing (Poscente 2009). Clearly, a reclamation standard must be set, and the decision criteria used to evaluate reclaimed land must be formalized to ensure consistent and transparent certification assessments and the alleviation of mining company reclamation liabilities. Reclamation assessment tools will not only serve to expedite the reclamation certification process, they will also offer guidance for how reclamation practices can be improved by presenting appropriate biological reclamation targets and by identifying what environmental conditions are correlated with those target community types.

In the following chapters, I employ the reference condition approach (sensu Stoddard et al. 2006) to develop wetland assessment tools for the oil sands surface mineable area. I use a suite of shallow open water wetlands including reclamation wetlands contaminated by tailings (OSPA), reclamation wetlands free from tailings contamination (OSREF), and true reference wetlands situated in parks or other protected areas that are free from oil sands mining related disturbance (REF). These wetlands span a similar range in local-level environmental conditions known to be important to plants, such as salinity, turbidity, and open water area. In chapter 2, I explore gradients in 52 environmental variables and identify those that explain the most variation in local-level conditions among the shallow open water wetlands. I use these variables to build a stress gradient that distinguishes among wetlands on the basis of their local-environmental condition. In chapter 3, I investigate the submersed and floating aquatic plant communities in reclamation and reference wetlands to identify different plant assemblages and I test the hypothesis that the assemblages in reclamation wetlands differ from those in reference wetlands. In chapter 4, I use the floating and submersed plant communities to develop and test an Index of Biological Integrity (sensu Karr 1991) that can be used to evaluate reclamation wetlands for certification purposes. In chapter 5, I tie the biodiversity of the submersed and floating (aquatic) plant communities to local- and landscape-level conditions using a structural equation model. I use this model to test hypotheses about the relative influence of in-lake environmental conditions and surrounding land use as well as the relative influence of direct and indirect effects of surrounding land use on aquatic plant diversity. I explore these questions at a series of nested spatial scales.

I conclude that oil sands mine reclamation has failed to create wetlands resembling appropriate natural analogues, either in terms of their in-lake environmental conditions or the aquatic plant assemblages that they support. To improve reclamation success, practitioners should use the targets I identified in Chapter 3 and their environmental correlates as a model for future wetland construction projects. The stress gradient that I developed in Chapter 2 and the index of biological integrity I developed in Chapter 4 should be incorporated into wetland monitoring and evaluation to measure how close practitioners come to achieving appropriate community-based targets.

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2. Quantifying a stress gradient: An objective approach to variable selection, standardization and weighting in ecosystem assessment[1].

Introduction

Indices of biological integrity (IBIs) are one of the most popular approaches to evaluating ecological condition. A major problem with creating an IBI for a new region is that biotic metric selection and IBI validation are achieved through correlation with some pre-existing measure of habitat quality, often referred to as the human disturbance gradient (U.S.EPA. 2002), more accurately called a stress gradient (sensu Grime 1977). Briefly, IBIs are composed of multiple biological metrics, each sensitive to a range in environmental stresses. These metrics are standardized and weighted such that, when summed, they produce an index score that integrates all of the sources of stress acting on an ecosystem to indicate its relative ecological condition. In IBI development, biological metrics are selected from a pool of candidate metrics such that redundancy is minimized and sensitivity to environmental stress is maximized (Karr 1991). Candidate biological metrics most strongly correlated with the stress gradient are considered most sensitive and are therefore preferred. Once the IBI has been developed, it must be tested by measuring the level of agreement between IBI scores and that initial measure of environmental stress for an independent group of sites. Thus, both IBI development and validation require a method for ranking sites based on their level of impairment that is independent of the biotic community to be used in IBI development. The efficacy and

[1] A version of this chapter has been published. Rooney, R.C. and S.E. Bayley, 2010. Quantifying a stress gradient: An objective approach to variable selection, standardization and weighting in ecosystem assessment. Ecological Indicators, 10: 1174-1183.

credibility of the IBI will hinge on the reliability of the initial stress ranking. Unfortunately, the method used to generate this initial ranking is often ambiguous.

Best professional judgment (BPJ) is often used to rank sites for IBI development and testing (e.g., Rothrock et al. 2008) or to select a combination of variables to use in site ranking (e.g., DeKeyser et al. 2003). While there can be great agreement among experts assessing biological condition (e.g., Davies and Jackson 2006), BPJ lacks objectivity and assessments are dependent on the expertise of the decision makers and the quality and type of information available to them. The rationale for decisions made in index development is often not articulated and the resultant ranking is not repeatable. Monitoring anthropogenic disturbance and evaluating reclamation can be a contentious issue, and any lack of clarity or objectivity in index development makes an IBI more difficult to defend.

It has been recommended that the initial ranking of sites should, like the IBI itself, be tabulated using multiple measurements of stress acting on the biota (U.S.EPA. 2002). This is because integrating multiple measures yields a more precise estimate of ecological condition. Also, the multi-metric approach is better able to detect impairment from different types of disturbance that may act cumulatively and synergistically on an ecosystem (Karr 1993).

Regardless of how it is produced, the stress gradient should quantify the degree of anthropogenic impact at a site on some standardized scale to allow for comparison among sites. Most commonly, researchers use a pre-existing index to develop and test an IBI for a new region or ecosystem, such as a validated rapid assessment method (e.g., Croft and Chow-Fraser 2007; Miller et al. 2006; Reiss 2006). Where no pre-existing validated index exists, an alternative ranking system is required.

Where high-resolution, current spatial data on anthropogenic disturbance is available, it may be used to rank sites along a land-use gradient (e.g., Reavie et al. 2008; Wang et al. 2008). One advantage of this approach is that sites do not need to be visited individually, allowing for broad regional assessments; however, this approach assumes that environmental conditions influencing the biota will be a direct result of surrounding land-use, which is not necessarily true (e.g., Tangen et al. 2003). Presuming that local conditions are tied to surrounding land-use, the spatial scale relevant to the biota remains to be determined (Brazner et al. 2007). Site ranking on the basis of land-use data is well suited to identifying regions with a high probability of biological impairment and where a more detailed site assessment is warranted. It is less suited to developing a stress index that will predict biological integrity at individual sites. Regardless, some objective method is still required for choosing among the near-infinite number of possible landscape metrics. In the absence of remotely sensed data on anthropogenic disturbance or where the link between land-use and local-level conditions is tenuous, local-level physical and chemical variables may be used to rank sites along a stress gradient (e.g., Gernes and Helgen 1999).

Stress index development involves three main steps: variable selection, standardization and weighting (Falcone et al. 2010). I sought to: (1) develop a method of objectively selecting among correlated candidate variables that minimizes redundancy but retains the maximum amount of information possible, and that results in an index that is easy to use and interpret; (2) contrast a standardization method that retains the normal distribution of the data (Z-scoring) with one that produces an even frequency distribution of index scores (percentile binning); and (3) examine the effect of different weighting schemes on index sensitivity. All of this was in an effort to develop an optimized approach to ranking sites based on their relative habitat quality as indicated by local-level physical and chemical variables that minimizes the use of best professional judgment and is explicit when such judgment is necessary. Ultimately, my goal was to produce a sensitive human disturbance gradient (more accurately, a stress index) that can be used as the backbone of IBI development and validation.

Methods

Study design

Although the approach I developed below can be used for agricultural or urban stresses, I chose to develop the stress index using data collected from reclamation marshes on oil sands mining leases and from appropriate natural analogues. Oil sands mining involves stripping off overlying vegetation, soil and up to 100 m of a marine-shale or glacial till overburden in order to reach the oil bearing sand (Johnson and Miyanishi 2008). This disturbs vast tracts of land: as of 2010, 67,330 ha had been disturbed (Richens pers. comm.) and an additional 99,714 ha are approved to be surface mined (Keeler pers. comm.). Before mining, wetlands constitute roughly 65% of the landscape covering the mineable oil sands deposit, 63% of which is forested fen (Raine et al. 2002). Thus, about 108,000 ha of wetland, mainly peatland, will be lost as a result of already approved mining projects. Whole landscapes, including wetlands, will need to be reclaimed, resulting in one of the largest reclamation projects in the world.

All constructed wetlands will experience some degree of physical stress as a consequence of their construction. In addition, oil sands reclamation wetlands are impacted by contamination from overburden and oil sands process affected (OSPA) tailings. OSPA is contaminated by residual bitumen, naphthenic acids and polycyclic aromatic hydrocarbons. Additionally, OSPA is salty with elevated sodium, sulfate and chloride levels. Even reclamation wetlands free from OSPA will have elevated salt levels because the marine shale and clay overburden has a high ionic content. Thus, wetland reclamation efforts to date have focused on building fresh to sub-saline marshes.

I sampled 20 reclamation marshes located on the leases of the two largest oil sands mining companies: Syncrude Canada Ltd. and Suncor Energy Inc. The wetlands belonged to two classes: those exposed to oil sands process affected materials (OSPA; n = 11) and those that were free of tailings contamination (OSREF; n = 9). I predict that OSPA wetlands will generally be under greater environmental stress than OSREF wetlands as a consequence of the additional toxicants present in oil sands tailings.

To evaluate whether reclamation marshes are healthy, they should be compared to undisturbed marshes spanning the same range in salinity, depth and surface area. Boreal sub-saline marshes are uncommon, but they do occur in Alberta and are the appropriate reference for evaluation of oil sands reclamation marshes (Purdy et al. 2005). I sampled 27 reference (REF) marshes in the boreal ecoregion to represent the least disturbed end of the stress gradient. They were situated in landscapes dominated by forest, with only small amounts of forestry or agriculture within 2 km of their open water boundaries.

All marshes included an open water zone, containing phytoplankton, periphyton and submersed aquatic macrophytes. Maximum depth of the open water ranged from 50 to >200 cm and area ranged from 0.06 to 19.02 ha (median = 0.65 ha). Emergent zones were characterized by saturated sediment and vegetation was typified by *Typha latifolia* or *Schoenoplectus* spp. Wet meadow zones were characterized by water beneath the sediment surface and vegetation was usually dominated by grass or *Carex* spp.

Sampling procedure

In spring, HOBO™ depth data loggers were installed and maximum depth measured. During peak vegetation biomass, vegetation was sampled, HOBO™ loggers were retrieved, sediment samples were taken, phytoplankton chlorophyll-a was collected, water chemistry was sampled, and Secchi depth was measured. All sampling conducted in the open water zone was done by kayak so as not to disturb the sediment.

Sediment was sampled within each vegetative zone: the wet meadow, emergent and open water. Each sample was a composite of three replicate cores taken along radial transects that divided each wetland into thirds. Cores contained about the upper 10 cm of sediment, and were collected using a suction-corer with an inner diameter of 5.72 cm.

Physical variables measured at each site include the width of vegetation zones, daily water depth from HOBO™ depth loggers, and August Secchi depth. Open water area was measured from aerial photographs that were ground-truthed during site visits. Aerial photos ranged in scale from 1:20,000 to 1:30,000, and were taken between 1999 and 2009 during the open water season.

Physicochemical analysis

Conductivity, pH and temperature of the water were measured in situ using a handheld YSI 556 Multiprobe System. Integrated depth water samples underwent preliminary processing in the field. Water was filtered through Whatman GF/F glass microfiber filters for analysis of cations (Na^{+}, K^{+}, Ca^{2+}, Mg^{2+} and NH_4^{+}) and anions (SO_4^{2-}, Cl^{-}, NO_2^{-}+NO_3^{-}, CO_3^{2-} and HCO_3^{-}), soluble reactive phosphorus (SRP), total dissolved nitrogen (TDN), dissolved organic carbon (DOC), total dissolved phosphorus (TDP), Fe, Al, Si, and total dissolved solids (TDS). An unfiltered water subsample was retained for analysis of total nitrogen (TN), total phosphorus (TP), total organic carbon (TOC), alkalinity and total suspended solids (TSS). Because of its contribution to turbidity, phytoplankton Chlorophyll-a was measured using a Shimadzu RF-1501 spectrofluorophotometer at an excitation wavelength of 436 nm and an emission wavelength of 680 nm. Metals (Al, Fe) and some cations (Na^{+}, K^{+}, Ca^{2+}, Mg^{2+}) were determined using an Elan 6000, Perkin Elmer ICP-MS following accepted standard methods (AWWA 1999a). Detection limits for cations by this method were 0.0005 mg Na/L, 0.002 mg Mg/L, 0.03 mg Ca/L, and 0.006 mg K/L. Nitrogen, phosphorus, and Si were measured using a Lachat QuikChem 8500 FIA automated ion analyzer, following standard methods (AWWA 1999b, 1999c, 1999d, 1999e). TN, TDN, TP, and TDP were first digested using potassium persulfate, and then measured following standard methods for NO_2^{-}+NO_3^{-} (AWWA 1999c) and SRP (AWWA 1999d), respectively. TOC and DOC were measured using a Shimadzu 5000A TOC analyzer, following U.S. EPA method 415.1 (U.S.EPA. 1979). Alkalinity, CO_3^{2-}, and HCO_3^{-} were determined by titration with 0.01 M sulfuric acid. Cl^{-} and SO_4^{2-} were determined using a Dionex DX600 Ion

Chromatograph and following U.S.EPA method 300.1 (Pfaff et al. 1997). TDS and TSS were determined gravimetrically from residues obtained by oven drying at 103 °C. An additional water sample was taken for analysis of naphthenic acids following the Fourier-transform infrared spectroscopy method (Jivraj et al. 1996).

Sediment was kept frozen until analysis. A subsample was analysed using a Coulter LS230 laser diffraction particle analyzer that measures solids ranging from 0.04 to 2000 µm. The oil, water and solids content was measured using refluxing toluene in a soxhlet extraction apparatus (Syncrude 2006b). Another sub-sample was oven-dried at 60°C for 48 hours and then ground and further sub-sampled for analysis of TC, TN, TP and LOI. Sub-samples of 4 to 6 mg were measured with a Mettler Toledo XP56 microbalance (± 0.001 mg) before analysis of TC and TN by combustion in an Exeter Analytical CE40 Elemental Analyzer. Sub-samples of 0.2 g were weighed using a Mettler Toledo AT261 Delta Range microbalance (± 0.00001 mg) for analysis of TP using the peroxide/sulfuric acid digestion method (Parkinson and Allen 1975) and a Varian Cary 50 spectrophotometer. A 0.5 g sub-sample was weighed out using a Mettler Toledo AE240 balance (± 0.0001 mg) and then placed in a muffle furnace at 550°C for 4 hours and re-weighed to measure LOI.

Approach to stress index development

Developing a stress index consists of three main steps: 1) variable selection, 2) standardization, and 3) weighting (Fig. 2-1). To select among the 52 environmental variables measured, I used principal components analysis (PCA) to reduce the candidate variables to orthogonal synthetic variables within four a priori defined categories: water chemistry, sediment chemistry, physical variables, and oil sands specific contaminants.

The selection of these four categories was based on my BPJ of what environmental variables would likely influence the plant community and could differ if one examined another source of disturbance or taxonomic group. I then selected the variables most strongly correlated with the resultant axes to represent them in the stress index.

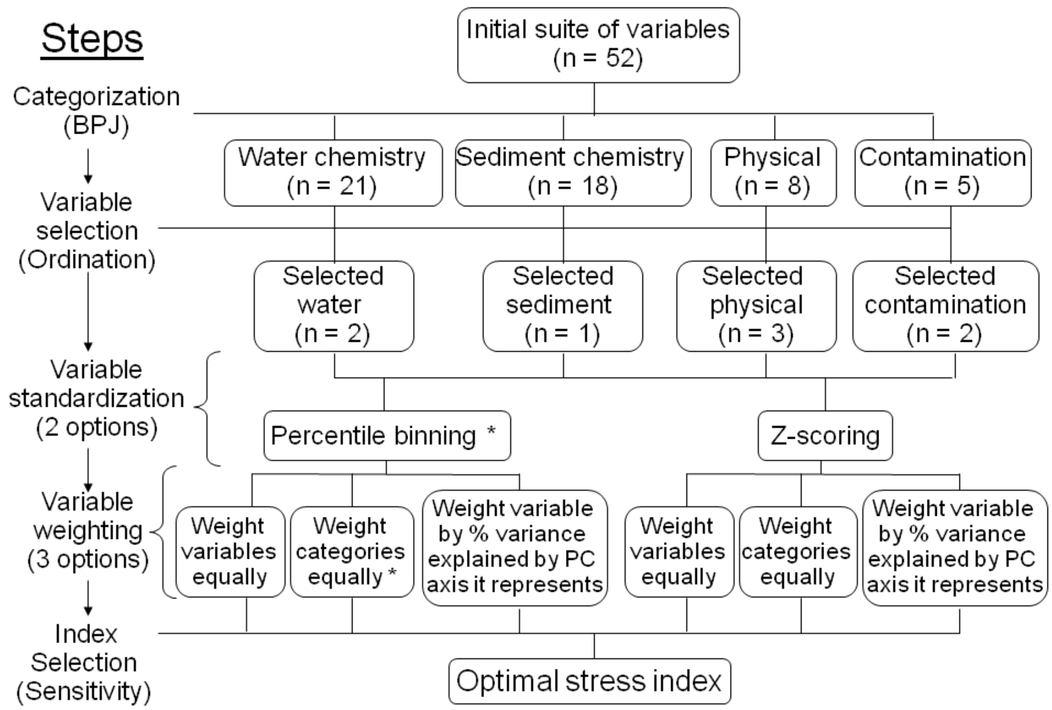

Fig. 2-1. Schematic depicting major steps in the development of the stress index and the options that I explored at each step. * indicates the option identified as providing optimum sensitivity with minimal redundancy among constituent variables.

Next, selected variables were standardized so that the resulting scores could be combined to produce an index value. I contrasted two standardization approaches. Percentile binning variables produces scores with an even distribution that pulls outliers in and disperses values nearer the mean. In contrast, taking the Z-score retains the normal distribution of the original values (Zar 1999).

Third, a weighting scheme was selected as the basis on which standardized scores would be combined to give an index value. I compared three weighting schemes. The first scheme weights each variable equally. The second weights each pre-determined category of variables equally. The third weights each variable by the percent of variance explained by the PC axis it was selected to represent.

Variable selection

Our initial dataset included 52 environmental variables measured at each of 47 marshes. To be clear, variable selection was conducted without reference to wetland type (REF, OSREF, and OSPA). To prepare the dataset for ordination, I replaced values below detection limit with one half of the detection limit, and missing values with marsh type averages to eliminate zeros. Multivariate normality is difficult to assess directly, but this PCA assumption will generally be met if the distribution of each variable is normal (McCune and Grace 2002). I accepted variables with skewness and kurtosis <1.3 (Zar 1999), and performed transformations to improve normality where necessary.

Once the data were ready for ordination, I divided the measured variables into these categories: water variables (n = 21), sediment variables (n = 18), physical variables (n = 8) and contamination variables (n = 5) (Fig. 2-1). I checked the linearity of relationships among variables within these categories with biplots. I performed a PCA on a cross-products correlation matrix for each of these categories to produce synthetic orthogonal variables. Ordinations were performed using PCORD V. 4.36 (McCune and Mefford 1999). I decided how many axes were significant using the heuristic broken-stick model (McCune and Grace 2002). When no eigenvalues exceeded their corresponding broken-stick eigenvalues, I determined which axes were significant by visually identifying cutoffs in scree-plots of unexplained variance.

I then selected the environmental variable with the largest eigenvector on each significant PC axis to represent it in the stress index. I made exceptions where two variable's eigenvector values were within ±0.05 and the variable with the slightly lower eigenvector value was deemed a more direct or more practical variable to measure.

I used model I ANOVAs to evaluate whether site scores on significant axes and representative variables differed significantly among the 3 marsh types. Where a significant difference was detected, I performed Tukey's multiple comparison analysis.

Correlation among the selected variables was evaluated by nonparametric Spearman rank correlation analysis. All univariate statistics were performed using SYSTAT 12 (SYSTAT 2007).

Variable standardization

Variables must be standardized so that they can be summed together to produce a final stress score. First, variables with an expected inverse relationship to stress (e.g., total nitrogen) were multiplied by -1, such that values increase as stress increases. While nutrient enrichment is a form of stress in many aquatic environments, in these marshes nutrient deficiency is a bigger problem. After correcting for the direction of the correlation between selected variables and stress level I standardized using two different approaches: Z-scoring and percentile binning.

A Z-score measures how many standard deviations each observation is from the mean. If the original data is normally distributed, the Z-scores will range from about -4 to +4, with a mean of 0. Negative stress scores are counter intuitive, so I converted negative values by adding 5 to each Z-score. We added 5 rather than the minimum observed value for each variable because Z-scores will not typically range beyond ± 5, so adding new sites to the gradient should never result in a negative value, regardless of how low the stress levels are. This produces scores ranging from 0 to 10.

I contrasted Z-scoring with standardization by the percentile binning approach advocated by Falcone et al. (2010). Values ranging from 1-20th percentile were given a score of 1; 21-40th a score of 2; 41-60th a score of 3; 61-80th a score of 4; and 81-100th a score of 5. This produces scores ranging from 1 to 5.

Variable weighting

I contrast three weighting schemes. First, I give each selected variable equal weight, allowing categories with more significant principal components to exert greater control on the overall stress score than categories with fewer significant components. Second, I average the scores within each category so that each category contributes equally to the overall stress score of a site, and variables in categories with more significant components individually exert less influence than those in categories with fewer components. Third, I weight each variable by the percent of variance explained by the PC axis it was selected to represent. By this scheme, variables that explain a greater proportion of the correlative structure of the data are given greater weight, as recommended by Wang et al. (2008).

Gradient validation

I applied each of the three weighting schemes to both standardization approaches to produce six stress scores for each marsh, which I compared using Spearman rank correlation coefficients to evaluate the concordance among different standardization and weighting approaches. I used a nonparametric approach because percentile binning produces scores with a platykurtic distribution.

I have three classes of marshes and I predict that stress will increase among them as successive sources of stress act as additional environmental filters (sensu van der Valk 1981): REF (typical marsh stress), OSREF (typical marsh stress plus physical and some chemical stress), OSPA (typical marsh stress plus physical and complex chemical stress). Because variables were selected independent of marsh type, some validation of the stress indices can be achieved by comparing the capacity of the six types of stress scores to discriminate among marsh types. I did this using nonparametric Kruskal-Wallis analysis with SYSTAT version 12.02 (SYSTAT 2007). I then used a nonparametric Tukey-type multiple comparisons test to determine how marsh types differed significantly from one another in terms of their stress index scores. In this method Q is calculated as the differences in mean rank between two groups divided by their standard error. This approach is appropriate for uneven sample sizes (Zar 1999).

Results

Variable selection

Eight variables were selected in total, but they were not evenly distributed among the four categories: water chemistry, sediment chemistry, physical variables, and contamination (Fig. 2-1). Rather, categories with more significant axes in their PCAs had more variables representing them in the final score calculation. Below, I present the results for each of the four PCAs. Correlation among selected representative variables was minimal. The most strongly correlated variables were cation and Cl^- concentration (Spearman *rho* = 0.61), TN and % water in the emergent zone sediment were also positively correlated (Spearman *rho* = 0.54). All other variables had Spearman correlation coefficients with an absolute value less than 0.5. Results of ANOVAs and Tukey's tests evaluating differences among the wetland types in terms of their site scores on each PC axis and the variables selected to represent them are listed in Table 2-1.

Table 2-1. Model I ANOVA and Tukey's multiple comparison results testing the ability of significant PC axes and their representative variables to discriminate among the three marsh types: REF, OSREF, and OSPA. Marsh types that lack the same superscript letter are significantly different at $\alpha = 0.05$. The marsh types are listed in ascending order of their mean values. Although predictions that stress increased from REF to OSREF to OSPA were not always upheld, no incongruous trends were observed, e.g, REF was never under greater stress than OSPA. Where ANOVA did not yield a significant result, Tukey's multiple comparisons analysis was not applicable (n.a.).

Category	Dependent variable	*F*-value (d.f.: 2, 44)	*p*-value	Tukey's multiple comparisons
Water	PC1	4.47787	0.017	OSPA^{a}, OSREF^{b}, REF^{b}
	Cations	2.17388	0.126	n.a.
	PC2	14.30348	<0.001	REF^{a}, OSREF^{b}, OSPA^{b}
	TN	5.10453	0.010	OSREF^{a}, OSPA^{a,b}, REF^{b}
Sediment	PC1	57.80407	<0.001	REF^{a}, OSREF^{b}, OSPA^{b}
	% water in emergent zone sediment	64.78054	<0.001	OSPA^{a}, OSREF^{a}, REF^{b}
Physical	PC1	2.57412	0.088	n.a.
	Maximum depth	2.09690	0.135	n.a.
	PC2	2.20579	0.122	n.a.
	Secchi depth / total depth	1.24727	0.297	n.a.
	PC3	16.01413	<0.001	REF^{a}, OSPA^{b}, OSREF^{b}
	Amplitude	3.65734	0.034	REF^{a}, OSPA^{ab}, OSREF^{b}
Contamination	PC1	12.29513	<0.001	OSPA^{a}, OSREF^{b}, REF^{b}
	% oil in emergent zone sediment	5.46904	0.008	REF^{a}, OSREF^{a}, OSPA^{b}
	PC2	8.34289	<0.001	OSPA^{a}, REF^{b}, OSREF^{b}
	Cl^{-}	3.81294	0.030	OSREF^{a}, REF^{ab}, OSPA^{b}

Water

The first 2 axes cumulatively explained 57.1% of total variance in the 21 variables (Fig. 2-2). Axis 1 (eigenvalue = 7.778, explained 37.0% variance) represented the ionic composition of the water, with total cations, total anions, conductivity, TDS, Na^{+}, SO_4^{2-}, alkalinity, Mg^+ and Cl^- as variables with absolute eigenvector values > 0.2. I selected total cations to represent the axis as it had the largest eigenvector (-0.3537). Axis 2 (eigenvalue = 4.208, explained 20.036% variance) represented nutrient levels, with TN, TDN, TP, $NH4^+$, SRP, the ratio of sodium to calcium and magnesium (Na/Ca+Mg), and Si having eigenvectors with absolute values > 0.2. TN had the largest eigenvector value (-0.4135) and was selected to represent axis 2.

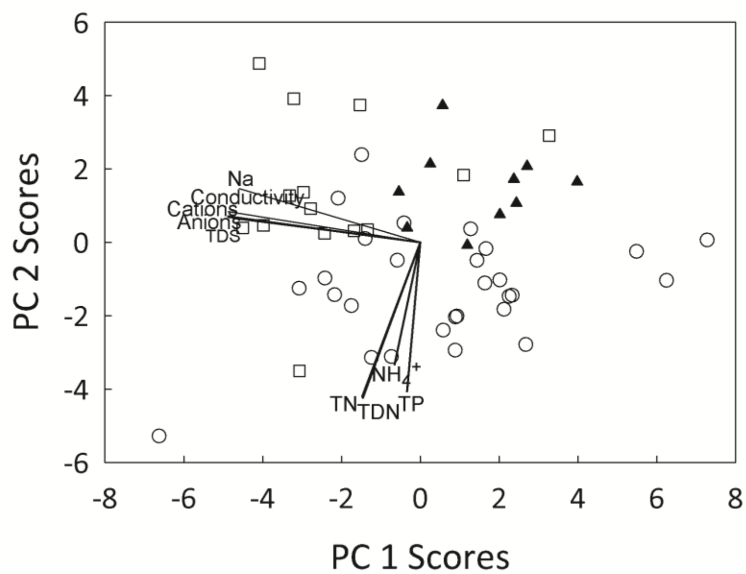

Fig. 2-2. Joint plot of principal components analysis (PCA) on a correlation matrix containing 21 water chemistry variables sampled at three marsh types: OSPA (square), OSREF (triangle), and REF (circle). Vector length is 5 x *r* (correlation coefficient) on each PC axis. Only variables with eigenvalues >0.3 are presented.

Sediment

Only the first PC axis was deemed significant (eigenvalue = 14.189, explained 78.8% variance) and it reflected variance in nutrients and organic content of the sediment from different vegetation zones (Fig. 2-3). All of the 18 sediment variables had absolute eigenvectors > 0.2. Although percent solids of emergent zone sediment had a slightly greater absolute eigenvector value than percent water content (eigenvector =

0.2569 vs. eigenvector = -0.2566), I selected % water content of emergent zone sediment to representative this axis because it is more interpretable than percent solids.

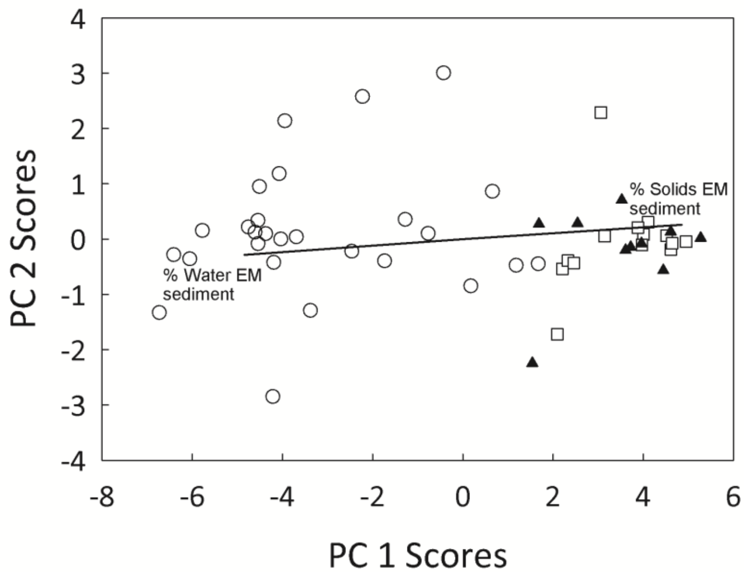

Fig. 2-3. Joint plot of principal components analysis (PCA) results from a correlation matrix containing 18 sediment chemistry variables sampled in 3 types of marshes: OSPA (square), OSREF (triangle), and REF (circle). Vector length is 5 x *r* (correlation coefficient) on each PC axis. Only variables with eigenvalues >0.25 are displayed. N.B. Only the first PC axis is significant.

Physical

None of the axes produced in the PCA had eigenvalues greater than their broken stick eigenvalues so I relied on a scree plot (not shown) to identify the number of significant axes. By this approach, I considered the first three axes significant, which cumulatively explained 62.0% of the variance in the 8 physical variables (Fig. 2-4 a,b). Axis 1 (eigenvalue = 2.229, explained 27.863% variance) reflected bathymetry, and was most strongly related to maximum depth (eigenvector = 0.5120), which I selected to represent axis 1. Axis 2 (eigenvalue = 1.568, explained 19.599% variance) represented transparency and I selected the ratio of Secchi depth to total depth (eigenvector = -0.7221) to represent axis 2. Axis 3 (eigenvalue = 1.166, explained 14.572% variance) reflected hydrologic variation and I selected amplitude as the representative variable (eigenvector = 0.6345).

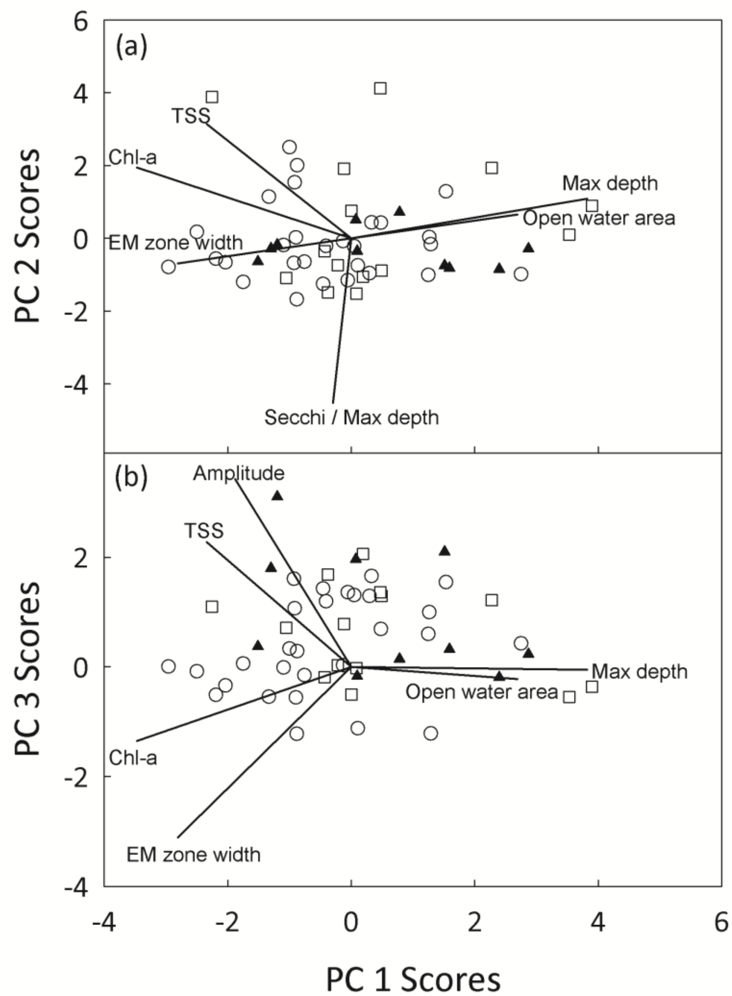

Fig. 2-4. Joint plot of principal components analysis (PCA) results from a correlation matrix containing 8 physical variables. Vector lengths are 5 x *r* (correlation coefficient) between each variable with an eigenvalue >0.3 and significant PC axes: axis 1 and 2 (a), axis 1 and 3 (b). Three marsh types are shown: OSPA (square), OSREF (triangle), and REF (circle).

Contamination

I interpreted the first two principal components axes, which cumulatively explained 84.9% of the variance in the 5 contamination variables (Fig. 2-5). Axis 1 (eigenvalue = 3.038, explained 60.7% variance) was mainly related to percent oil content in the sediment (eigenvector = -0.5510). Although the eigenvector for oil content of sediment from the wet meadow zone was 0.002 greater than the eigenvector for emergent zone sediment, I selected percent oil content in sediment from the emergent zone to represent this axis in the stress gradient. I did this because emergent zone sediment was already selected to represent the sediment variables, so selecting oil content from emergent zone sediment instead of from the wet meadow zone reduced the number of samples required from each marsh. Axis 2 (eigenvalue = 1.209, explained 24.171% variance) was most strongly influenced by chloride and total naphthenic acid concentration in wetland water samples. I selected chloride concentration of the water to represent this axis (eigenvector = -0.8200).

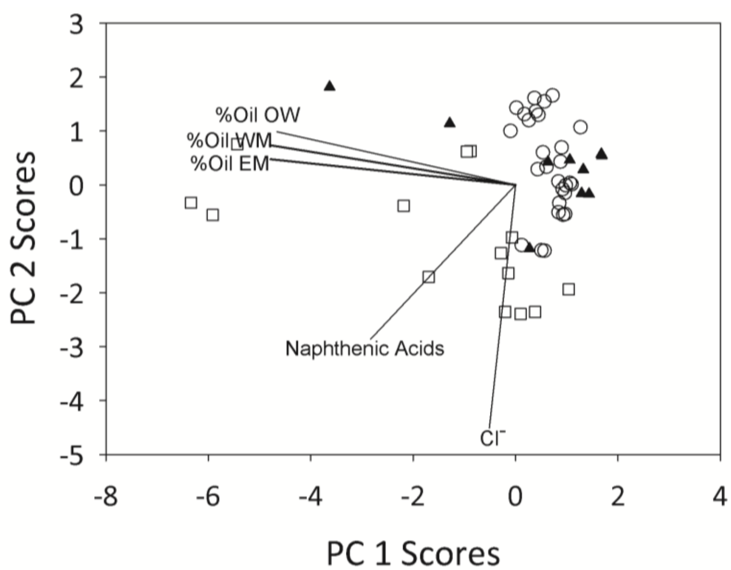

Fig. 2-5. Joint plot of principal components analysis (PCA) results from a correlation matrix containing 5 contamination variables measured at three types of marshes: OSPA (square), OSREF (triangle), and REF (circle). Vector lengths are 5 x *r* (correlation coefficient) between each variable with an eigenvector value >0.3 and significant PC axes.

Variable standardization

Selected variables expected to be negatively correlated with stress included TN, water content of the emergent zone sediment, and Secchi depth as a proportion of maximum depth.

Validation

There was general agreement (positive correlation) among the 6 scoring methods (Table 2-2). Stress scores produced using different weighting techniques but the same standardization approach were more similar than those using different standardization approaches but the same weighting technique: $r < 0.88$ when comparing between standardization methods, but $r > 0.95$ when comparing weighting methods that used the same standardization approach.

Table 2-2. Spearman rank correlation coefficient (*rho*) matrix for the six approaches to calculating stress scores. This includes standardization by Z-scoring (Z) and by percentile binning (P) for each of three weighting methods: 1 is even weighting of variables; 2 is even weighting of categories; and 3 is weighting of each variable by the percent of variance explained by the principal components axis it was selected to represent. (n = 47).

	Z1	P1	Z2	P2	Z3	P3
Z1	1					
P1	0.877	1				
Z2	0.984	0.875	1			
P2	0.853	0.985	0.871	1		
Z3	0.966	0.875	0.982	0.881	1	
P3	0.818	0.951	0.830	0.965	0.859	1

A strongly significant difference among sums of ranks of scores for the three marsh types was found using all six scoring approaches (Table 2-3). Mean ranks, standard error and *Q* values for each of the stress indices that found a significant difference among sums of ranks of the different marsh types are given in Table 2-4. All six indices found that REF marshes were under less environmental stress than reclamation marshes, but only the indices that used percentile binning with evenly weighted categories and percentile binning with weighting in proportion to variance explained in the PCA were able to discriminate between OSREF and OSPA sites. In keeping with predictions, these two indices found OSPA marshes were under greater stress than OSREF marshes.

Table 2-3. Results of Kruskal-Wallis analysis, contrasting the sums of ranks of scores among the three marsh types: REF, OSREF, and OSPA. Scores were generated with standardization by either Z-scoring (Z) or by percentile binning (P) for each of three weighting methods: 1 is even weighting of variables; 2 is even weighting of categories; and 3 is weighting each variable by the percent of variance explained by the principal components axis it was selected to represent, (n = 47).

Scoring Method	Kruskal-Wallis test statistic	*p*-value
Z1	25.5	<0.00001
P1	27.6	<0.00001
Z2	27.6	<0.00001
P2	29.0	<0.00001
Z3	27.7	<0.00001
P3	27.3	<0.00001

Table 2-4. Absolute Q values and consequent inferences about stress scores characteristic of the three marsh types: REF, OSREF, and OSPA. If the absolute Q_{value} exceeds $Q_{(0.05, 3)} = 2.394$, the pair under comparison differ significantly at $\alpha = 0.05$. Scores were generated by either standardization by Z-scoring (Z) and by percentile binning (P) for each of three weighting methods: 1 is even weighting of variables; 2 is even weighting of categories; and 3 is weighting of each variable by the percent of variance explained by the principal components axis it was selected to represent, (n = 47).

Score	REF vs. OSREF	REF vs. OSPA	OSREF vs. OSPA	Inference
Z1	9.611	12.457	0.842	REF<OSREF=OSPA
P1	8.875	13.661	2.144	REF<OSREF=OSPA
Z2	9.722	13.181	1.218	REF<OSREF=OSPA
P2	8.558	14.330	2.799	REF<OSREF<OSPA
Z3	9.313	13.477	1.706	REF<OSREF=OSPA
P3	8.251	13.930	2.771	REF<OSREF<OSPA

Discussion

I sought to develop an approach to ranking sites on the basis of their habitat quality that was independent of measurements of the biota and that minimized reliance on best professional judgment (BPJ). My efforts produced the approach highlighted in

Figure 2-1. Overall, I was successful at restricting the use of BPJ to particular stages of index development; however, I could not eliminate it completely.

Five distinct steps were involved in obtaining the optimal stress index (Fig. 2-1). The first step was to sort the candidate variables into categories deemed to influence habitat quality from a plant community perspective. BPJ was used to select what variables to measure and to categorize them. The second step was to select among correlated candidate variables a sub-set that retained as much information as possible while minimizing redundancy. BPJ was occasionally used in the variable selection phase when the penultimate variable's eigenvector was within 0.05 of the largest eigenvector value, but was considered a superior choice in terms of efficiency or interpretability. In such cases, it was possible to state explicitly the rationale used to make the decision, such that the process can be judged and repeated. The third and fourth steps were to standardize and weight the variables so they could be summed to get a stress index score for each wetland. I contrasted two standardization approaches and three weighting schemes. I found that both the standardization approach and the weighting scheme influence the sensitivity of the resultant stress index, but weighting is less influential than the standardization approach. The weighting schemes that I applied were not derived from BPJ, but from properties of the dataset. The fifth and final step in index development was to identify which combination of standardization and weighting schemes provided the greatest sensitivity, so that I could select the optimal stress index. I was able to select among standardization and weighting schemes objectively, without resorting to BPJ. Thus I was able to use a pre-determined selection of environmental variables to objectively rank 47 marshes in terms of their level of environmental stress and to create a stress index that could rank novel sites with the greatest efficiency and sensitivity. Discussion regarding each of the major steps in stress index development follows.

Variable selection

Principal components ordination of the variables within each category provided a simple and objective method to minimize redundancy within the index, as PC axes will be independent of each other (McCune and Grace 2002). However, this does not mean that representative variables will be entirely uncorrelated. Generally, limiting acceptable correlation coefficients ≤ 0.7 between all pairs of selected metrics is considered adequate to limit compound errors within a multi-metric index (e.g., Wang et al. 2008). None of my variable pairs had a Spearman *rho* > 0.6, and over 60% of the pairs had Spearman *rho* < 0.2. Thus, I conclude that my approach successfully limited redundancy in the stress index.

Alternative approaches to reducing redundancy include using ordination scores on significant axes as the disturbance scores (e.g., Falcone et al. 2010). However, direct environmental measurements are more easily understood than ordination scores that synthesize correlated variables in multidimensional space (Karr 1993). Additionally, using PC scores as variables in an index is impractical. To calculate the stress score for a new site, all 52 of the original variables would need to be sampled and the ordination would need to be repeated using the entire dataset. Also, because the ordination solution may change with the addition of new sites, the relative position (or PC score) of any site already sampled could shift. Thus, the stress score for a given site could change, even if its physical and chemical condition remains constant.

Another alternative is to select among environmental variables based on their correlation with biological variables of interest (e.g., Wang et al. 2008). Then, iteratively, select among correlated biotic variables those most strongly correlated with the refined suite of environmental variables. This approach reduces redundancy and avoids the challenges presented by using ordination scores, but it is self-referential. Correlation between an IBI and a stress index cannot be used for IBI validation if the constituent environmental variables were chosen based on their correlation with constituent biotic metrics. Additionally, this approach may suffer from path dependency: if the selection process were performed in reverse, i.e., if the suite of biotic variables was refined first, it could theoretically result in the selection of a different set of variables.

In contrast, my approach objectively selects those variables most representative of environmental variability, reduces redundancy and the number of variables that must be sampled at new sites, and retains the interpretability that is the main advantage of multi-metric assessment methods. It also avoids the tautology of selecting environmental variables on the basis of their correlation with biotic variables, when the ultimate goal is to then use the stress gradient to select among candidate biotic variables in order to develop and test an IBI.

My approach suffered some loss of sensitivity by using representative variables in place of PC axis scores, but generally there was very good agreement between ANOVA and Tukey's results obtained from ordination scores and from the variables selected to represent significant PC axes (Table 2-1). Some variables were unable to detect a difference among REF, OSREF, and OSPA sites, including cations, maximum depth and Secchi depth as a proportion of total depth, suggesting that they would make poor performance indicators if used in isolation. My objective, however, was not to discriminate among the three wetland types, but to measure the severity of environmental stress at a site. Rejecting variables that do not differ among marsh types would discount the natural variability among reference sites and deny the possibility of successful reclamation. Variables that do not discriminate among marsh types still reflect environmental stress. It is merely that the three marsh types are not characterized by different levels of that particular stress.

In the case of cations and maximum depth, failure to discriminate among marsh types is unsurprising, as I intentionally selected reference sites to span the same range in these values exhibited by reclamation marshes. Regarding Secchi depth, there was a great range in the transparency of OSPA marshes, with Secchi depth ranging from 3.3 to 100% of maximum depth (total suspended solids ranged from 920 to 0.025 mg/L). Likely this tremendous variability is responsible for the failure to detect a difference among marsh types.

In all cases where ANOVAs were significant, the nature of the differences among wetland types was in accordance with expectations; however, stress did not consistently increase from REF to OSREF to OSPA sites. Percent water content and TN grouped the two types of reclamation marshes together as under greater stress than REF sites, whereas % oil content grouped OSREF marshes together with REF sites as less stressed than OSPA sites (Table 2-1). Cl^- was lower in OSREF than OSPA sites, but REF sites spanned the entire range of Cl^- concentrations; however, this was intentional, as I sought to include natural sites that represented a large gradient in salinity. Amplitude was lower in REF marshes than at OSREF sites, suggesting water levels in REF sites were more stable, but the amplitude of OSPA sites was highly variable (Table 2-1). This, I suspect, is because many REF and some OSPA sites are seepage fed, resulting in steady water levels, whereas OSREF and some OSPA sites have water pumped into them or are surface fed by small drainage basins, increasing the volatility of their water depth.

Standardization

The main advantage of Z-scoring is that the distribution of the values is retained; i.e., outliers of extremely good or poor quality are retained in the tails, rather than binned with more moderate values. Retaining a normal distribution permits the use of parametric statistics in analyzing stress scores. In addition, Z-scores are a continuous variable, and thus produce fewer ties than percentile binning, where scores are integers. The main disadvantage of Z-scoring is that it requires that data be normally distributed, so it is sometimes necessary to transform data prior to calculating Z-scores. This makes interpretation less intuitive. In addition, the resolution is reduced to discriminate among sites with middling stress levels as most sites are given the mean stress score.

Z-scoring was contrasted with standardizing by the percentile binning approach advocated by Falcone et al. (2010). Falcone et al. (2010) found the percentile binning approach optimized sensitivity, as each value contributes to the site score but the influence of outliers is moderated. Transformation of the data prior to binning is unnecessary because calculating the percentile requires no distributional assumptions. However, resultant scores have a platykurtic distribution, so nonparametric statistics are required to analyze them. Additionally, percentile binning produces many tied scores, complicating rank-based comparison. Levelling the frequency distribution of scores does present an advantage in some cases: it increases resolution to detect differences among sites with moderate stress levels. However, if outliers are of interest because they represent the extremes of good and poor habitat quality, that information will be lost.

Based on Kruskal-Wallis and nonparametric multiple comparisons analyses, the choice of standardization approach has a large impact on the capacity of the stress index to discriminate between reference marshes and the two types of reclamation marshes in the oil sands area. In two of the three weighting schemes, percentile binning yielded greater stress scores for OSPA sites than OSREF sites, whereas Z-scores lumped reclamation marshes together regardless of the weighting approach. Likely, the percentile binning approach was able to resolve this difference because of increased resolution at moderate stress levels.

Both standardization approaches have advantages and disadvantages; however, for my purposes, percentile binning is superior. As stated by Fore et al. (1996), "although the condition of the sites near the ends of the spectrum [of disturbance] are easy to judge, moderately degraded sites are not." Percentile binning is superior in this case because it improves resolution in the middle of the stress score range, allowing the detection of slight improvements in condition and to discriminate between sites of intermediate quality.

Weighting

Variable weighting is often undertaken to emphasize variables considered of greater ecological or biological importance. Alternatively, weighting is used as a means of reducing redundancy within an index by ascribing correlated variables lower weights than those that explain a greater amount of variability within the index. Applying equal weight to all variables is usually considered the null model and alternative weighting schemes are compared to it (e.g., Falcone et al. 2010). Even the decision not to apply weightings, however, is to choose a weighting scheme: that of weighting all variables equally. Thus, even in the absence of an ecological or biological rationale for weighting by a given scheme, weighting should be considered explicitly.

In this chapter, the number of variables per category varies based on the dimensionality of the principal components solution. Weighting variables equally within the index (Scheme 1) will mean that categories such as water chemistry, which include two representative variables, will be weighted twice as heavily as categories with one variable, such as sediment chemistry. Conversely, weighting the categories equally (Scheme 2) or the variables in proportion to the amount of variance explained by each variable's PC axis (Scheme 3) gives greater weight to variables belonging to categories with fewer variables, i.e., those with simpler PCA solutions. There is no biological justification for unevenly weighting categories, but nor is there a biological reason to give variables uneven weights. In the absence of a biological basis for ascribing weights, the structure of the environmental data itself can be used. Down-weighting variables that explain less of the overall data structure will reduce redundancy within the index while retaining the maximum amount of explanatory power possible. I evaluated the three weighting schemes based on their concordance and respective sensitivity.

The weighting schemes produced scores that were highly correlated with each other, especially when comparing among scores calculated using the same standardization approach (Table 2-2). However, based on multiple comparisons testing, it appears that equally weighting each variable may produce a score that is less sensitive than either weighting the categories equally or weighting the variables in proportion to percent variance explained by the PC axes they represent (Table 2-4). When percentile binning was used to standardize, the latter two weighting schemes resolved the OSREF marshes from OSPA marshes.

Falcone et al. (2010) found that weighting improved an index's ability to discriminate between high and low quality habitat in proportion to the degree of redundancy remaining in the index. Both weighting the categories equally and weighting the variables in proportion to percent variance explained by the PC axes they represent inherently reduce redundancy by emphasizing the ordination structure. Thus, the increased sensitivity of these two weighting approaches could be due to minimized redundancy in the final stress index. In this chapter, the more sensitive weighting approaches coincidentally down-weighted the importance of the three physical variables. Recall that, when examined singly, two of the physical variables did not detect a significant differences among marsh types (Table 2-1). Likely down-weighting them relative to variables that were able to discriminate among marsh types increased the sensitivity of the whole index.

It should be noted that PCAs with a greater number of variables in the initial correlation matrix will have a larger total amount of variance to be explained, and will, by random chance alone, explain a lesser percentage of the total variance with each significant axis. Thus, variables in categories with fewer variables in total will be ascribed greater weight simply by chance. In this dataset, for example, the water chemistry category contained 21 variables and the first PC axis explained only 37% of the total variance among them, whereas the contamination category contained only 5 variables and the first PC axis explained 61% of the total variance. Both these categories produced two significant PC axes, but the weighting of the contamination variables will be greater than the water chemistry variables, in part because I initially measured more water chemistry variables. Therefore, while weighting variables by the percent of variance explained by the PC axis each represents reduces redundancy by emphasizing the correlative structure of the dataset, it also prejudices the index in favour of categories containing fewer variables. Weighting all categories equally also ascribes greater weight to variables within categories yielding fewer significant PC axes, but not as a consequence of this phenomenon. Additionally, a case can be made in favour of weighting categories equally as the plant community will be directly affected by all four categories of variables. Consequently, I consider weighting categories equally as the optimal weighting scheme.

Conclusion

This chapter outlines a new approach to index development (Fig. 2-1) that minimizes reliance on best professional judgment and results in an effective and sensitive stress index. The method can be followed to generate an index of habitat assessment where no pre-existing validated index is available for validation by correlation. Consequently, the approach is ideal for developing ecosystem assessment indices in new regions and for evaluating impairment caused by novel forms of disturbance or disturbance at unprecedented scales, such as in the Alberta oil sands. Its main advantage is that it is developed independent of biotic variables, so it can later be used to develop and validate biotic indices.

Both standardization and weighting methods influence the sensitivity of the final index, although there is general agreement among the six methods that I tested. In habitat assessment, the goal is often to resolve differences between moderately disturbed sites. Thus, I recommend percentile binning despite the subsequent reliance on nonparametric statistics. In terms of weighting, approaches that further reduce redundancy by emphasizing the correlative structure of the environmental dataset are more sensitive than simply weighting all variables equally, but where categories contain uneven numbers of variables, there is a risk that categories with fewer variables will be unjustifiably weighted as more important. In this chapter, the use of percentile binning with equal weighting among categories resulted in the most sensitive index and produced the largest Kruskal-Wallis test statistic in comparing among marsh types. Thus, this is the combination of standardization and weighting approaches that I recommend.

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3. *SETTING RECLAMATION TARGETS AND EVALUATING PROGRESS: SUBMERSED AQUATIC VEGETATION IN NATURAL AND POST-OIL SANDS MINING WETLANDS IN ALBERTA, CANADA*[2].

Introduction

In a world with diminishing conventional oil supplies, unconventional supplies become economically attractive resources. In oil-producing Alberta, unconventional oil sands bitumen accounted for 72% of all oil production in 2008, and 55% of this (42 x 10^6 m^3) came from surface mining (ERCB 2009a). Unconventional supplies may increase oil reserves, but they are often exploited at greater environmental cost than conventional oil.

To access oil sand ore by surface mining, companies first remove the forest, peat, and overburden that overlie the deposit (Johnson and Miyanishi 2008). Between mining and reclamation, this results in a barren landscape of overburden dumps, toxic tailings impoundments and open mine pits. Vast areas are affected: already more than 67,000 ha have been disturbed (Richens pers. comm.) and an additional 408,000 ha are expected to be mined (ERCB 2010). Wetlands make up about 65% of the region, 63% of that wetland area is wooded fen (Raine et al. 2002). Marshes and shallow open-water wetlands, in contrast, constitute only 3% of the landscape (Raine et al. 2002). Given the predominance of wetlands in the pre-mining landscape, about 108,000 ha of wetland will be lost as a result of mining projects already existing or approved (Keeler pers. comm.). Whole landscapes, including wetlands, will need to be created, resulting in the largest wetland reclamation project in Canadian history, one of the largest globally.

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[2] A version of this chapter has been published. Rooney, R.C. and S.E. Bayley, 2011. Setting reclamation targets and evaluating progress: submersed aquatic vegetation in natural and post-oil sands mining wetlands in Alberta, Canada. Ecological Engineering, 37: 569-579.

Oil sands mines are not required to restore the land they lease; they are only required to reclaim it (sensu SER 2004). Conservation and reclamation regulations of Alberta's Environmental Protection and Enhancement Act require oil sands mining companies to reclaim disturbed land to equivalent land capability, where "the ability of the land to support various land-uses after conservation and reclamation is similar to the ability that existed prior to an activity being conducted on the land, but that the individual land uses will not necessarily be identical," (GOA 1993). The most recent mine-specific approvals require companies to submit wetland reclamation plans to the Government for approval by December 31st, 2011; however, they provide little guidance and no community-based targets for wetland reclamation. Several reclamation wetlands have been constructed in the last 35 years, but none have yet been certified by the Alberta Government as "successfully reclaimed." In part, this is because the Government has yet to set wetland reclamation targets and therefore has no benchmarks for reclamation success. In light of the new requirement that oil sands mining companies submit wetland reclamation plans, the need for science-based targets for wetland reclamation is increasingly urgent.

The major challenge to wetland reclamation in the region is chemical contamination, particularly elevated salinity (Harris 2007). Materials used in constructing the post-mining landscape include mine tailings as well as the peat, surface soil, subsoil, and overburden that are stripped off to reach the oil sand. All tailings materials are exposed to the oil extraction process and contain residual bitumen and associated hydrocarbon toxicants. Tailings water is recycled through the ore many times, and thus accumulates high levels of ions including ammonia (NH_4), chloride (Cl), boron (B), and copper (Cu) (MacKinnon et al. 2004). These contaminants may combine synergistically to increase toxicity to the biota (e.g., Leung et al. 2003; Nero et al. 2006). Reclamation wetlands range from freshwater (< 0.5 ppt) to oligosaline (0.5 to 5 ppt), with elevated concentrations of Cl, Na and SO_4 that hinder re-colonization by historically dominant fen and bog species (e.g., Luong 1999; Renault 2005). Many marsh species, in contrast, are quite salt tolerant (Stewart and Kantrud 1972). Instead of trying to recreate peatlands, oil sands companies plan to construct oligosaline shallow open-water marshes to meet their reclamation obligations (Harris 2007). Marshes may accumulate organic matter if conditions are favourable (Trites and Bayley 2009a) and have the potential to develop into peatlands, but this process typically requires thousands of years (e.g., Bauer et al. 2003) and may be inhibited by environmental conditions such as highly fluctuating water levels or high salinity (Vitt 1994).

Given that oil sands mining companies are not required to return the land to its historical condition (peatlands) (GOA 1993; Harris 2007), what are appropriate targets for reclamation? I suggest that because the pre-mining landscape was dominated by relatively pristine wetlands, "equivalent land capability" must also include healthy wetlands. If one consider a healthy wetland to be one that supports a "species composition, diversity, and functional organization comparable to that of natural habitat of the region" (Karr 1991), then reclamation targets should be based on biotic communities found in naturally occurring wetlands, and reclamation success measured by the similarity between reclamation and reference communities. Mining companies will be constructing oligosaline shallow open-water marshes, thus I suggest that reclamation wetlands be evaluated by comparison to naturally occurring oligosaline shallow open-water marshes of the least-disturbed condition rather than to neighbouring peatlands. Although rare, oligosaline marshes do exist in northern Alberta. This type of wetland represents the "best attainable condition" (sensu Stoddard et al. 2006) for most of the wetlands created in the post-mining landscape and has been previously recommended as the appropriate reference condition for evaluating oil sands reclamation marshes (Purdy et al. 2005; Trites and Bayley 2009b).

Submersed aquatic vegetation (SAV) is an excellent indicator of wetland condition, being sensitive to environmental variables such as nutrient levels, water clarity, hydrology, and salt concentration (see review in Lacoul and Freedman 2006). SAV is particularly useful in indicating nutrient deficiency (Sondergaard et al. 2010), which is an important source of stress in reclamation wetlands (Rooney and Bayley 2010). SAV is functionally important, connecting the sediment to the water – pumping nutrients up from the sediment (Barko et al. 1991), and oxidizing the rooting zone (Flessa 1994). These exchanges have important consequences for the cycling of nutrients and metals within wetlands (Moore et al. 1994). SAV is also biologically important, affecting multiple trophic levels (Norlin et al. 2005). It provides habitat for fish, waterfowl, amphibians, and invertebrates (e.g., Hart and Lovvorn 2005; Hornung and Foote 2006), as well as algae (Lassen et al. 1997), which are an important source of food in wetlands (Hart and Lovvorn 2003). Thus, the structure and composition of the SAV community is not only indicative of, but also exercises influence on, the overall condition and value of shallow open-water wetlands.

I sought to characterize reclamation targets and to evaluate the ability of oil sands reclamation wetlands to achieve them. Reclamation targets are needed not just to provide a benchmark for reclamation certification, but to enable mining companies to practice adaptive management and to encourage progressive, continuous reclamation. I used SAV communities present in reference wetlands to set community-based reclamation targets and compared the SAV in oil sands reclamation marshes to them.

I had three main questions. First, how many SAV assemblages are present in the wetlands I sampled? If more than one distinct assemblage is found in reference wetlands, multiple reclamation targets may be acceptable. By distinct, I mean assemblages dominated by different species or supporting different indicator species with significant fidelity and exclusivity. Second, what species characterize the different SAV assemblages, and what environmental conditions are they associated with? I set reclamation targets by the SAV assemblages typical of reference wetlands, thus those assemblages must be defined. The SAV communities in reclamation wetlands must also be characterized to measure their similarity to reference communities. Environmental conditions associated with SAV assemblages found in reference wetlands could be fostered in reclamation wetlands, whereas conditions associated with impaired SAV assemblages should be considered remediation priorities. Third, is there a difference in SAV community composition among reclamation and reference shallow open-water wetlands? By Karr's (1991) definition of health, a substantial difference in community composition would suggest that, at the time of evaluation, reclamation wetlands were impaired relative to natural analogues.

Materials and methods

Study design

I sampled 63 wetlands in total, including 25 reclamation shallow open-water wetlands that ranged in age from 3 to 35 years (Fig. 3-1). This included every reclamation wetland over 3 years of age located on the leases of the two dominant oil sands mining companies. These were located on company leases, in a landscape heavily influenced by mining activity, and ranged from fresh to oligosaline (total dissolved solids (TDS) = 0.3 to 3.5 g/L), in water depth (0.3 to 3.3 m), and in open-water area (0.08 to 19.02 ha). They were of two types: oil sands reference (OSREF; n=12) and oil sands process affected (OSPA;

n=13). Some OSPA sites are exposed to continuous seepage from tailings ponds, while others had one-time inputs of tailings, whereas OSREF were either remnants of natural wetlands or new wetlands formed on overburden dumps and were free of contaminated tailings water or materials. I anticipated finding a gradient in biological condition (sensu Davies and Jackson 2006) along which reclamation wetlands exposed to tailings materials are under the greatest stress and stress is intermediate in reclamation wetlands free from tailings.

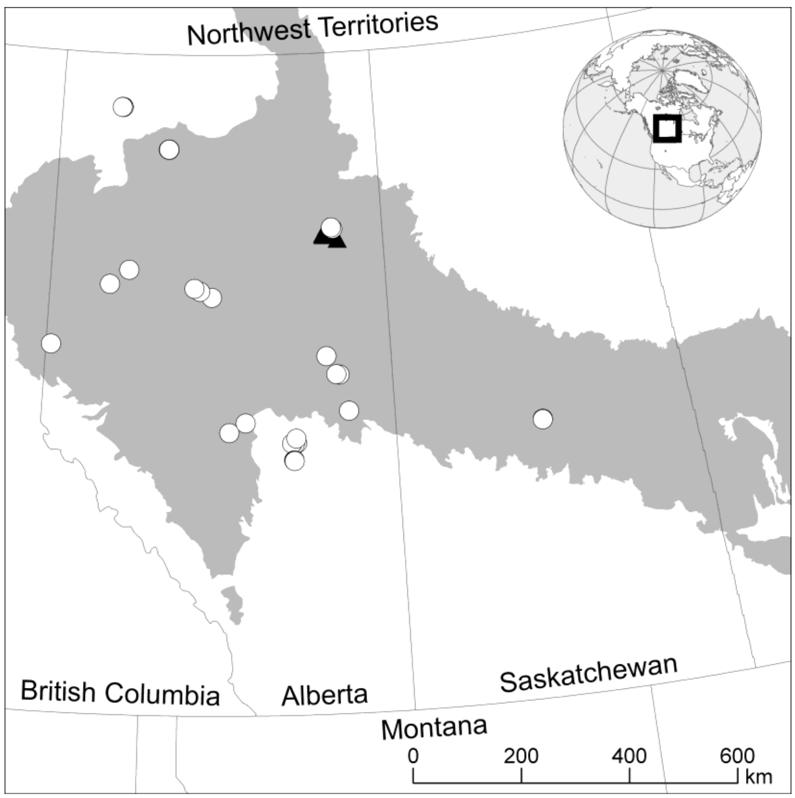

Fig. 3-1. Map depicting the distribution of sampled wetlands. Note that reclamation wetlands (black triangles) all fall on two oil sands company leases whereas reference wetlands (white circles) are distributed across the Boreal ecoregion (grey), both east and west of the cluster of reclamation wetlands as well as north and south of them.

The remaining 38 shallow open-water wetlands were reference wetlands (REF) that represented the least-disturbed condition from which the "best attainable condition" (sensu Stoddard et al. 2006) can be estimated. The reference sites did not include peatlands such as those typical of the oil sands area before mining; the reference sites covered about the same range in TDS (0.1 to 4.9 g/L), water depth (0.2 to 3.3 m), and surface area (0.06 to 18.38 ha) as reclamation wetlands. REF sites were scattered across the northern half of Alberta and into Saskatchewan as I sampled every accessible cluster of undisturbed oilgosaline shallow open-water marshes within the Boreal ecoregion. They ranged north and south as well as east and west of the oil sands area (Fig. 3-1). All were located in landscapes dominated by forest, with only small amounts of agriculture or forestry within 2 km of their open water zones.

Sampling

Sampling occurred during the open-water season in 2007, 2008, and 2009. HOBO™ data loggers recorded changes in water depth. Effective shoreline slope was estimated by the width of emergent and wet meadow vegetation zones. These were measured at three locations surrounding each wetland. Open-water area was measured from aerial photographs where post-1999 imagery was available, otherwise I used Quickbird satellite imagery. All images were imported into a digital file for measurement using GIS software (ESRI 2009), such that the final imagery had pixels ≤3 m in resolution.

During peak biomass, vegetation was sampled from kayaks using the rake technique outlined in Bayley and Prather (2003). In brief, 10 transects were made across each open-water zone. At a random location along each transect, a 1 m^2 quadrat was deployed. SAV was collected from within each quadrat and identified to the lowest practical taxonomic level (usually species) following Moss and Packer (1983). Taxa that I could not identify to species included an aquatic moss, the macroalgae *Chara* spp. and two species of *Myriophyllum*: *M. sibiricum* and *M. verticillatum*, which can only be distinguished when plants are in flower. The relative abundance (volumetric) of each taxon within each quadrat was recorded and total SAV density was estimated on a scale of 1 to 5. Relative abundance numbers from the 10 quadrats were then composited. Simpson's dominance was calculated as the sum of the squared proportional abundance of each species (Simpson 1949). Taxa richness is the sum of all floating and submerged taxa observed at a given wetland. It includes all species observed in a quadrat and any additional species observed along the 10 transects but not captured within one of the 10 quadrats. Secchi depth was measured to estimate transparency. Water samples were taken by kayak using an integrated water sampler and underwent preliminary processing in the field. Water was filtered through Whatman GF/F glass microfiber filters for analysis of cations (Na^+, K^+, Ca^{2+}, Mg^{2+}), anions (SO_4^{2-}, Cl^-, CO_3^{2-}, and HCO_3^-), soluble reactive phosphorus (SRP), total dissolved nitrogen (TDN), dissolved organic carbon (DOC), total dissolved phosphorus (TDP), total dissolved solids (TDS), silicon (Si), boron (B), and metals (Fe, Al). Filters were retained for analysis of phytoplankton chlorophyll-a (Chl-a). An unfiltered subsample was collected for analysis of total nitrogen (TN), total phosphorus (TP), conductivity, alkalinity, total suspended solids (TSS), and naphthenic acids. Laboratory analysis followed standard methods outlined in Rooney and Bayley (2010) and Chapter 2.

Sediment was sampled from the open-water zone at >20 cm water depth using a suction-corer with an inner diameter of 5.7 cm. Each sample was a composite of three replicate cores taken along transects that divided each wetland into thirds. Sediment oil content, TC, TN, TP, and loss on ignition at 550 ˚C (LOI%) were measured following standard analytical methods detailed in Rooney and Bayley (2010) and Chapter 2.

Approach

Clarke (1993) proposed that a unified framework for evaluating anthropogenic impacts on biotic communities should: (1) display community patterns graphically; (2) identify taxa responsible for grouping samples together; (3) test for significant differences among groups of samples; and (4) link community differences to patterns in environmental variables. I employed an approach that meets these criteria to evaluate reclamation using SAV. I displayed patterns in the SAV community with a combination of hierarchical clustering and non-metric multi-dimensional scaling ordination (NMS). I used indicator species analysis (ISA) to identify the optimal number of SAV community groups and to identify which taxa best indicate community group membership. I used a combination of multiple-response permutation procedure (MRPP) and analysis of independence to evaluate whether SAV community composition differs significantly between natural and reclaimed wetlands, and I used joint plots to determine what environmental variables are related to SAV community differences.

Analysis

For ordinations, I eliminated taxa present in less than 5% of the 63 wetlands to reduce noise associated with rare taxa. Three wetlands contained no aquatic vegetation: I gave them 100% abundance of the dummy variable “No Species.” Parallel analyses were run excluding sites without SAV, but results were consistent between analyses, so only those including the dummy variable are considered.

Deviations from univariate normality were assessed for each taxon, using skewness and kurtosis values (Zar 1999). I determined that a log (x+1) transformation yielded the greatest improvements in normality and so used log-transformed SAV data in all subsequent analyses. Multivariate analyses were all performed using PC ORD V4 (McCune and Mefford 1999), whereas univariate analyses were carried out in SYSTAT V12 (SYSTAT 2007).

How many SAV communities are there and what characterizes them?

I used two approaches to characterize SAV assemblages in the wetlands. First, to identify assemblages of SAV and to determine what taxa define them, I used a combination of cluster analysis and ISA. I used Ward's linkage method on relative Euclidean distance calculated from log-transformed SAV relative abundance to cluster the 63 wetlands. I used the *p*-values generated by 1000-permutation Monte Carlo tests in ISA as the basis for deciding the optimal number of SAV assemblages: the number of groups producing the lowest mean *p*-value for all taxa (McCune and Grace 2002). I then characterized the SAV assemblages using ISA to determine what taxa were most discriminatory (highest indicator value) (Dufrene and Legendre 1997). The indicator value of a taxon can range from 0 to 100, with 100 representing perfect indication where the taxon is both faithful and exclusive to the group it indicates (McCune and Grace 2002).

Second, I conducted NMS ordination on a Bray-Curtis distance matrix of SAV relative abundance to position the 63 wetlands in ordination space (McCune and Grace 2002). I identified the optimal dimensionality of the ordination with Monte Carlo testing (40 runs with real data and 50 runs with randomized data for 1 to 6 dimensional solutions). I then overlaid vectors representing the strength and direction of correlations between NMS axes and the relative abundance of each taxon on top of the final ordination solution to create community joint plots. These graphically represent what taxa are driving community differences among the three wetland types: REF, OSREF, and OSPA. To explore correlations between SAV and environmental variables, I overlay 32 environmental variables on the NMS ordinations of SAV relative abundance to produce environmental joint plots. The 32 environmental variables examined included shoreline slope, open-water area, Secchi depth, Na^+, K^+, Ca^{2+}, Mg^{2+}, SO_4^{2-}, Cl^-, CO_3^{2-}, HCO^{3-}, SRP, TDN, DOC, TDP, TDS, Si, B, Fe, Al, Chl-a, TN, TP, conductivity, alkalinity, TSS, and naphthenic acids levels in the water, and oil content, TC, TN, TP, and LOI of the sediment. Only variables with Pearson's correlation coefficients > 0.1 or those considered a priori to be of biological importance, however, are discussed.

Finally, I used model I ANOVAs and Scheffe's tests to evaluate differences among SAV assemblages in terms of total richness, SAV density, and SAV Simpson's dominance. Richness was square-root transformed, and density and Simpson's dominance were arcsine square-root transformed to meet ANOVA assumptions. I performed these tests both including and excluding the community group indicated by "No Species."

Are reclamation wetlands impaired?

I used SAV relative abundance data in reference wetlands to set reclamation targets. If reclamation wetlands fail to support the SAV assemblages typical of reference wetlands, they do not meet these targets and I consider them impaired. I used two approaches to examine whether the SAV community present in a given wetland was dependent on wetland type. First, I rank-transformed the distance matrix used in my NMS ordination and then carried out MRPP (Mielke and Berry 2001), a non-parametric method analogous to a multivariate analysis of variance. MRPP tests for group differences in multivariate data, using weighted mean-within group distances to evaluate the probability of observing, purely by chance, a community difference as large or larger than the one observed. Groups were weighted by $n/\sum(n)$, as recommended by McCune and Grace (2002). I also performed pair-wise MRPP analyses with Bonferroni correction ($\alpha = 0.017$). Second, in a categorical approach to the question, I calculated Pearson's Chi-square statistic to evaluate whether the SAV assemblage at a wetland was independent of its type.

I also compared square-root transformed SAV richness among the wetland types. First, grouping OSREF and OSPA sites together, I used a two-sample t-test to compare REF and reclamation wetlands. Second, taking OSREF and OSPA sites independently, I used a model I ANOVA and a Scheffe's test to compare all three types (Zar 1999).

Results

I observed 26 SAV taxa (Appendix 3-1), although richness ranged from 0 to 12 taxa per wetland and the average wetland had only five taxa. Five of the taxa I observed (c.a. 20%) are considered rare in Alberta; the halophyte *Ruppia cirrhosa*, the carnivorous *Utricularia minor*, and the pondweeds *Potamogeton foliosus*, *P. natans*, and *P. obtusifolius* (Kershaw et al. 2001). Of these rare taxa, only *P. obtusifolius* was observed exclusively in REF wetlands. I observed no invasive species. Comparing reference to reclamation wetlands (OSREF and OSPA taken together), there was no significant difference in richness (t-test, $t_{61} = 0.481$, $p = 0.63235$); however, when reclamation sites were considered separately, OSPA sites were significantly less diverse than either OSREF or REF wetlands (One-way ANOVA, $F_{2,60} = 12.16$, $p = 0.00004$, Scheffe's $p < 0.01$ in both cases) and REF wetlands were less diverse than OSREF sites (Scheffe's $p = 0.02$). Richness thus exhibited a hump-shaped distribution along our biological condition gradient: nutrient-poor OSREF wetlands had significantly greater richness than nutrient-rich REF wetlands and tailings-contaminated OSPA wetlands.

How many SAV assemblages are there?

After clustering on the basis of relative Euclidean distance, I identified two levels of community grouping in the SAV data (Fig. 3-2). At the coarser level, there are four community Groups (mean ISA p-value = 0.23 ±0.07 SE), achieved by pruning the dendrogram to retain about 45% of the information in the original community dissimilarity matrix. Nested within those Groups are seven Sub-groups (mean ISA p-value = 0.14 ±0.04 SE) (Table 3-1), achieved by pruning the dendrogram to retain about 60% of the information (Fig. 3-2). The dendrogram had low (1.71%) chaining.

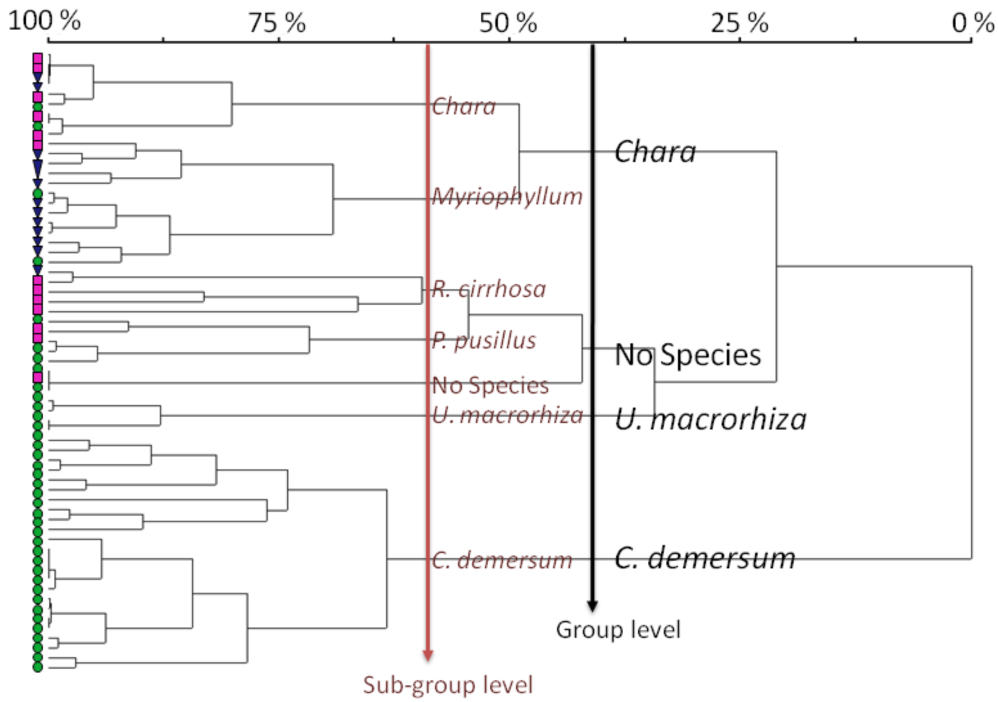

Fig. 3-2. Dendrogram produced by cluster analysis on relative Euclidean distances calculated from the log-transformed SAV relative abundance at 63 wetlands. Wetland type is indicated on the left: REF = circles, OSREF = triangles, and OSPA = squares. Percent of information remaining is indicated along the top. Vertical arrows prune the dendrogram at the Group and Sub-group levels. Note the concentration of REF wetlands in the *C. demersum* and *U. macrorhiza* Groups and the restriction of reclamation wetlands to the *Chara* and No Species Groups.

Table 3-1. Results of indicator species analysis with four Groups and seven Sub-groups. Only the taxa with significant ($p < 0.05$) indicator values (IV) are represented. Taxa in bold are used to name SAV assemblages.

Group Indicator	IV	*p*-value	Sub-group Indicator	IV	*p*-value
***Chara* spp.**	74.5	0.001	***Chara* spp.**	48.7	0.001
Myriophyllum spp.	59.5	0.003	***Myriophyllum* spp.**	88.1	0.001
Stuckenia pectinata	41.6	0.025	*Hippirus vulgaris*	38.5	0.015
No species	23.1	0.016	**No species**	100.0	0.001
			Ruppia cirrhosa	35.8	0.017
			Potamogeton pusillus	66.5	0.001
			Sagittaria cuneata	35.0	0.027
Ceratophyllum demersum	77.6	0.001	***Ceratophyllum demersum***	61.2	0.001

Utricularia macrorhiza	81.5	0.001	***Utricularia macrorhiza***	76.7	0.001
Aquatic moss	42.1	0.008	Aquatic moss	42.1	0.024

The SAV Sub-groups differed in terms of their total richness (model I ANOVA: $F_{6,56} = 14.54$, $p < 0.00001$), their macrophyte density ($F_{6,56} = 9.08$, $p < 0.00001$) and their Simpson's dominance ($F_{6,56} = 5.10$, $p = 0.00031$) (Fig. 3-3). However, in general SAV assemblages were species poor and strongly dominated by a single taxon. It appeared as though these differences might be due primarily to the inclusion of the No Species Sub-group, and so we repeated the analysis excluding the three wetlands with no aquatic vegetation growing in them. Despite the reduction in degrees of freedom for the F-tests, the differences in total richness, density, and dominance remained significant ($p < 0.004$ in all cases). However, the Scheffe's results did change as a result of the elimination of the No Species Sub-group and the consequent increase in statistical power due to the reduction in the number of comparisons being made. Generally, existing differences became better resolved. In the case of dominance, when the No Species Sub-group was excluded, no two community types were significantly different at $\alpha = 0.05$, but *Myriophyllum* did possess marginally lesser dominance than the *Chara* and *R. cirrhosa* sub-groups with $p = 0.06$.

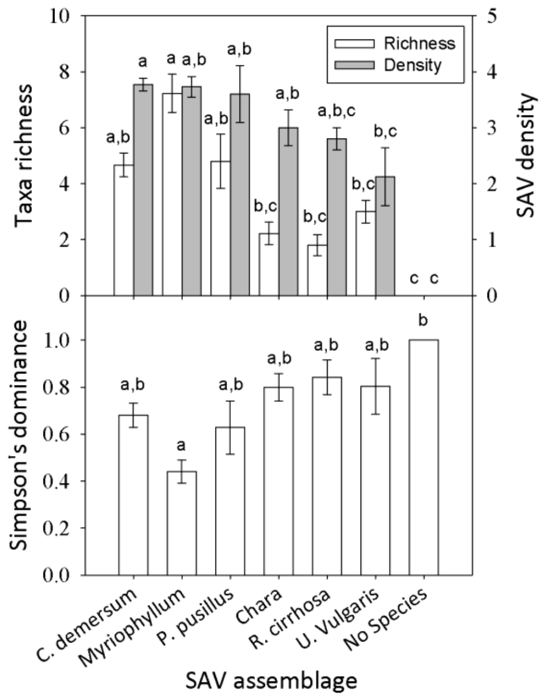

Fig. 3-3. Comparison of community traits among SAV Sub-groups. Top: Mean richness (white bars) and SAV density (grey bars). Bottom: Simpson's dominance (white bars). Error bars are standard error. Letters above bars indicate group membership based on Scheffe's multiple comparison testing.

Which taxa best characterize the SAV assemblages?

At the Group-level, 7 of the 26 SAV taxa observed in this study had significant indicator value (Table 3-1), and I named each Group after its indicator with the greatest indicator value. At the Sub-group-level there were 10 significant indicators if the "No species" dummy variable is included. Sub-groups were also named after their primary indicator. There was little cross-over of species between Sub-groups. For example, the species *Ceratophyllum demersum* was present in several community groups, but its relative abundance was always <25%, whereas in sites belonging to the *C. demersum* Sub-group, its mean relative abundance was 75%. Similarly, *Utricularia macrorhiza* plants were occasionally observed in other community groups, but always at <10% relative abundance, whereas its mean relative abundance in wetlands with the *U. macrorhiza* Sub-group was 85%.

The optimal solution to our NMS ordination of community composition was 3-dimensional with reasonably low stress (15.54) and low instability (0.0001) (Fig. 3-4). Cumulatively, the ordination explained 83.5% of the variance in the original dissimilarity matrix: 22.2% by the first, 32.8% by the second, and 28.5% by the third axis. Our interpretation of the NMS is in agreement with the results of cluster analysis.

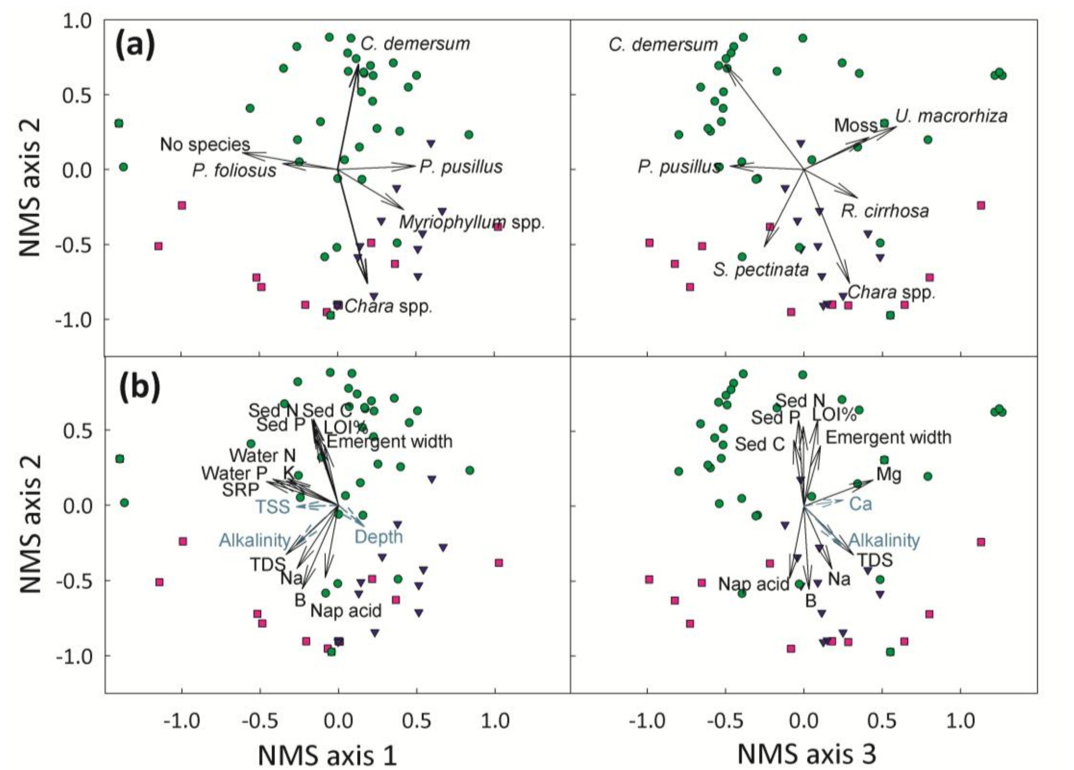

Fig. 3-4. Joint plot of wetlands, coded by the three wetland types, positioned in species space by non-metric multi-dimensional scaling ordination. Circles are reference wetlands (REF); triangles are oil sands reference wetlands (OSREF); and squares are oil sands process affected wetlands (OSPA). (a) Vectors represent the correlation (*r*-values) between SAV taxa and ordination axes. (b) Vectors represent correlation (*r*-values) between environmental variables and ordination axes. Black vectors have Pearson's correlation coefficients > 0.1 with at least one NMS axis. Grey dashed vectors are presented because of their biological importance.

Grouping sites in the ordination based on their wetland type, the second NMS axis separated REF wetlands from reclamation wetlands, whereas OSREF and OSPA separated along the first NMS axis (Fig. 3-4a). There was little separation of wetland types along the third NMS axis. REF sites are characterized by higher relative abundance of nutrient-loving *C. demersum* or, in some cases, by the carnivorous *U. macrorhiza* and aquatic moss, whereas OSREF sites have greater relative abundance of the alkali-loving macroalgae *Chara* spp., *Myriophyllym* spp., or *Potamogeton pusillus* (Fig. 3-4a). OSPA sites are less tightly grouped. They are associated with greater relative abundance of salt-tolerant *Stuckenia pectinata*, *R. cirrhosa,* or *Chara* spp*.* The relative abundance of *Chara* spp. is negatively correlated with *C. demersum* (Fig. 3-4a).

Are there environmental variables associated with different wetland types or SAV assemblages?

Unsurprisingly, REF wetlands have low concentrations of contaminants associated with tailings water (naphthenic acids, B, Na, and TDS) (Fig. 3-4b). Instead, they are characterized by high levels of nutrients in their sediment (LOI%, TN, TP, and TC) and broad zones of emergent vegetation surrounding the open-water zone, yet they span a range in water nutrient levels (TN, TP, and SRP) and cations (Mg, Ca, and K). They also vary in terms of depth and water transparency (TSS). Typically, they have lower Na than either OSREF or OSPA wetlands.

The two SAV assemblages characteristic of REF wetlands (*C. demersum* and *U. macrorhiza*) can be distinguished on the basis of water chemistry (Table 3-2).

Table 3-2. Summary of SAV assemblages, their diagnostic species, associated environmental conditions, and the types of wetland where they are most often found.

Sub-group	Indicator taxa	Environmental conditions	Dominant type
Chara	*Chara* spp.	Clear, deep, alkaline, impenetrable substrates, with tailings contamination.	OSPA & OSREF
Myriophyllum	*Myriophyllum* spp. , *H. vulgaris*	Low nutrients and TDS. Moderate alkalinity, Ca and Mg.	OSREF
No species	No species	Uninhabitable. Very high ionic levels or very turbid. Recovering from drought or newly created.	REF & OSPA
R. cirrhosa	*R. cirrhosa*	Saline and sodic, with tailings contamination.	OSPA
P. pusillus	*P. pusillus, S. cuneata*	Dilute. Low water nutrients, TSS, and alkalinity. Sediment nutrients variable.	OSPA & OSREF
C. demersum	*C. demersum*	Reference marsh. High nutrients, wide emergent zone, higher TSS, lower Ca and Mg.	REF
U. macrorhiza	*U. macrorhiza,* aquatic moss	Reference fen-marsh intermediary. High sediment nutrients, wide emergent zone, low water nutrients, clear water, high Ca and Mg.	REF

C. demersum dominates where water nutrients and TSS are higher, and Ca and Mg levels are low, particularly with respect to K. The *U. macrorhiza* Sub-group is less common overall, and occurs where water nutrients are low, water is clear, and Ca and Mg are high (Fig. 3-5).

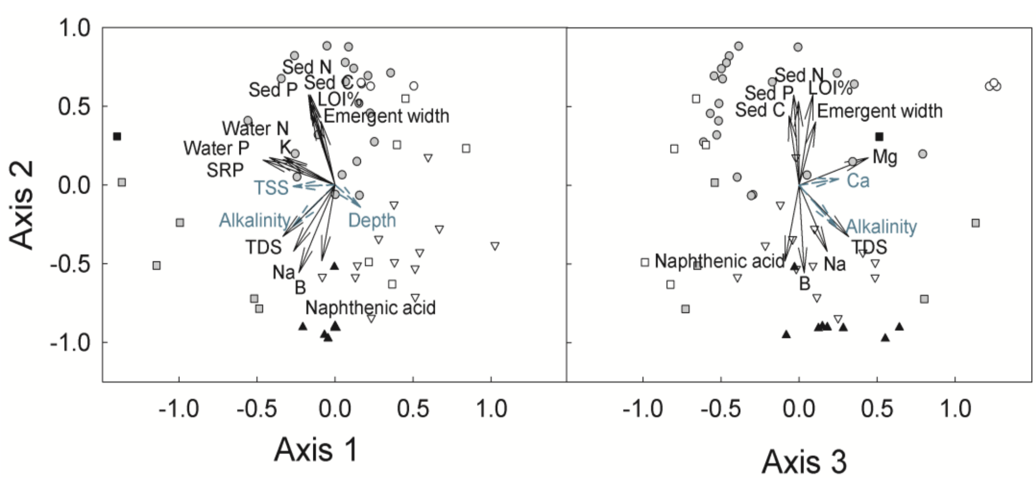

Fig. 3-5. Joint plot of wetlands, coded by the seven SAV assemblages, positioned in species space by non-metric multi-dimensional scaling ordination. Upwards triangles are the *Chara* Sub-group; downward triangles the *Myriophyllum* Sub-group; black squares the No Species Sub-group; grey squares the *R. cirrhosa* Sub-group; white squares the *P. pusillus* Sub-group; grey circles the *C. demersum* Sub-group; and white circles the *U. macrorhiza* Sub-group. Vectors represent the correlation (*r*-values) between environmental variables and ordination axes. Solid black vectors are environmental variables with Pearson's correlation coefficients > 0.1 with at least one axis, whereas dashed grey vectors are included because they are considered biologically important.

Reclamation wetlands are characterized by elevated salinity (Na and TDS) and low sediment nutrients (Fig. 3-4b). Like REF wetlands, they too range in water nutrient levels, depth, and transparency. They also vary in terms of Mg, Ca, and K concentration, but their dominant cation is typically Na. Contrasting the two types of reclamation wetland, OSREF sites have significantly lower TP (one-tailed t-test t_{23} = 1.96, p = 0.03), naphthenic acids (one-tailed t-test t_{23} = 8.16, $p < 0.001$), and B (one-tailed *t*-test t_{23} = 11.57, $p < 0.001$) than OSPA sites. There is also a trend that water in OSREF wetlands has lower SRP, TN, and higher Ca than OSPA wetlands (Fig. 3-4b), but this was not significant at $\alpha = 0.05$.

Within the *Chara* Group, wetlands with the *Myriophyllum* Sub-group (mainly OSREF) are typified by low nutrient levels in both sediment and water. Some exhibit evidence of contamination from tailings, and some exhibit elevated alkalinity, Ca, and Mg (Fig. 3-5, Table 3-2). The *Chara* Sub-group (mainly OSPA sites) is tightly clustered low on axis 2 as the assemblage occurs under the greatest levels of oil sands tailings contaminants, the narrowest emergent zone widths, and the lowest levels of sediment nutrients (Fig. 3-5, Table 3-2).

Only three of the wetlands belonging to the "No Species" Group actually contained no SAV. The other members of this Group did not include the "No Species" dummy variable, but were strongly dominated by a single species and therefore ISA identified the dummy variable as the best indicator of the Group. At the Sub-group level, however, the three wetlands belonging to the No Species Sub-group had no aquatic vegetation growing in their open water (richness = 0). The principal commonalities among wetlands with the No Species Sub-group (REF and OSPA) are high levels of alkalinity, nutrients in the water, cations, TSS, and TDS (Fig. 3-5, Table 3-2). Wetlands with the *R. cirrhosa* Sub-group (mainly

OSPA wetlands) are more dispersed in ordination space, but consistently have low levels of sediment nutrients and are contaminated by tailings (Fig. 3-5, Table 3-2). *R. cirrhosa* is found under higher concentrations of water nutrients, TSS, TDS, and in shallower water than the *Chara* Sub-group that populates most other OSPA wetlands. The *P. pusillus* Sub-group (OSPA and REF), occupies wetlands under a range of sediment nutrients and tailings contamination levels, but consistently occurred under lower K, Ca, Mg, and TDS concentrations that *R. cirrhosa* or *Chara* (Fig. 3-5, Table 3-2).

Are reclamation wetlands impaired?

The composition of the SAV community in reference sites was significantly different from that of either OSREF or OSPA wetlands (Table 3-3). Additionally, despite a smaller sample size for the comparison (n = 25 vs. about 50), SAV communities in OSREF and OSPA wetlands also differed significantly (Table 3-3). The analysis of independence confirmed this: the SAV assemblage present at a given wetland is not independent of wetland type at both the Group- (χ^2 = 49.78, d.f. = 6, p < 0.00001) and the Sub-group-levels (χ^2 = 67.75, d.f. = 12, p < 0.00001).

Table 3-3. Multi-response permutation procedure results. The test statistic (*T*) and its associated *p*-value refer to the null hypothesis that community composition within the groups being compared is no more different than expected to occur by chance. When examining pair-wise comparisons with Bonferroni correction, α = 0.017. The chance-corrected within group agreement (*A*) is a measure of effect size. If species composition within the groups is identical, *A* = 1, whereas if the heterogeneity within groups equals that expected by chance, *A* = 0. If there is more heterogeneity within groups than expected by chance, *A* < 0.

Comparison	*T*-value	*p*-value	*A*
All n = 63	-13.009	<0.000001	0.184
REF vs OSPA n = 51	-12.345	<0.000001	0.158
REF vs OSREF n = 50	-10.055	<0.000001	0.119
OSPA vs OSREF n = 25	-4.521	0.0012809	0.092

In general, the *Chara* and No Species Groups are found in reclamation wetlands, whereas only REF wetlands have members in the *C. demersum* and *U. macrorhiza* Groups.

OSREF wetlands belonged mainly to the *Myriophyllum* Sub-group with two members in the *Chara* Sub-group, whereas OSPA wetlands included examples of the *Myriophyllum*, No Species, and *P. pusillus* Sub-groups, but mainly supported the *Chara* and *R. cirrhosa* Sub-groups (Table 3-4).

Table 3-4. The number of OSPA, OSREF, and REF wetlands sampled that belong to each community Group and Sub-group, as defined by agglomerative cluster analysis based on relative Euclidean distance calculated from log (x+1) transformed SAV relative abundance and linked using Ward's method.

	Community	OSPA	OSREF	REF
Group	*Chara*	6	12	4
	No Species	7	0	6
	C. demersum	0	0	24
	U. macrorhiza	0	0	4
Sub-group	*Chara*	5	2	2
	Myriophyllum	1	10	2
	No Species	1	0	2
	R. cirrhosa	4	0	1
	P. pusillus	2	0	3
	C. demersum	0	0	24
	U. macrorhiza	0	0	4

Discussion

This work represents the first efforts to establish a reference condition for the evaluation of submersed aquatic vegetation (SAV) communities in freshwater to oligosaline boreal shallow open-water wetlands. These results identify plant-assemblage targets for post-mining reclamation and indicate under what environmental conditions both reference and atypical SAV assemblages occur. I identified seven SAV Sub-groups occupying the reference and reclamation wetlands, each possessing significant indicator taxa. These Sub-groups differed in terms of richness, SAV density, and dominance. I determined what environmental conditions were associated with the seven Sub-groups. I also determined that reclamation wetlands do not support the same Sub-groups as reference wetlands, and thus I consider reclamation wetlands impaired. I note that impairment is a status at a point in time, and that a current evaluation of impairment does not preclude a wetland from being assessed as healthy in the future, should its community composition shift to better reflect a community type characteristic of reference wetlands. For example, this might result in response to remediation actions.

I observed seven SAV assemblages

The SAV taxa fell into seven distinct assemblages or Sub-groups that formed obvious clusters in ordination space (Fig. 3-5). This clustering indicates that the SAV assemblages are cohesive units consisting of diagnostic species. This assertion is supported by indicator species analysis, which identified diagnostic taxa belonging to each Sub-group (Table 3-1). Thus, I conclude that the 63 wetlands I sampled contained seven distinct SAV assemblages.

The strong dominance by single species and the low richness typical of most of the Sub-groups contributed to their distinctive nature. Finding distinct SAV assemblages rather than loose intergrading communities is not uncommon in boreal shallow-open water marshes. For example, examining wetland plants in 5 reference and 10 oil sands reclamation wetlands with open-water zones, Trites and Bayley (2009b) identified five SAV assemblages. Their assemblages included four that I also observed: both the reference assemblages (*C. demersum* and *U. macrorhiza*), as well as the reclamation assemblages indicated by *Chara* spp. and *P. pusillus*. The fifth community that they observed was indicated by *S. pectinata*. This species was not a significant indicator of any of my seven Sub-groups, but was an indicator of the *Chara* Group and was associated with the Sub-group indicated by *Myriophyllum* spp. and *H. vulgaris*.

This study likely had a greater power of detection than Trites and Bayley (2009b) because I sampled more sites. I sampled 63 shallow open-water wetlands, whereas they sampled 15. I also sampled more intensively: ten 1 m^2 quadrats per wetland vs. three composited circular quadrats 1 m in radius. I observed two species in REF sites that they only found in reclamation wetlands and another three species that Trites and Bayley (2009b) never observed. I also found six species in reclamation wetlands that Trites and Bayley (2009b) observed only in reference wetlands. Despite the difference in sample size and sampling intensity, there was a great deal of similarity in our results and conclusions, supporting the interpretation that the SAV Sub-groups that I observed form consistent and distinct assemblages in boreal Alberta.

It has been speculated that part of the reason biodiversity is lower in OSPA wetlands than in reference wetlands could be their young age and the time required for colonization (Wong et al. 2008). As our field work was conducted three to five years after the work of Trites and Bayley (2009b), some of the differences in species observed might be attributable to continued colonization. However, I found no significant relationship between richness or community composition and wetland age (Appendix 3-2) despite a range from 3 to 35 yr (mean = 16 yr). More likely I observed a greater number of aquatic species because my power of detection was greater.

Biotic and environmental characterization of SAV assemblages

It should not be surprising that 20% of the species I observed are considered regionally rare (Kershaw et al. 2001), as I was examining a rare habitat type. In general, all the species I observed are characteristic of oligosaline shallow open-water habitat (Stewart and Kantrud 1972), but represent a range in salinity tolerance.

To characterize the seven SAV assemblages in terms of their biota and the environmental conditions with which they are associated, I begin with the two groups dominating REF wetlands (Table 3-2). In general, the *C. demersum* and *U. macrorhiza* Sub-groups are found where sediment nutrients are abundant, emergent vegetation zones are wide, and contamination from tailings is low. While they represent a range in ionic concentrations, suggesting that natural SAV assemblages are possible even in the presence of elevated salinity, they are dominated by the cations K, Mg, and Ca rather than Na. The richness and Simpson's dominance of these two reference assemblages were equivalent; however, density of SAV was significantly higher in the *C. demersum* Sub-group, indicating that SAV productivity is naturally variable.

Differentiating between moderate-rich fens and marshes in the Boreal Forest Natural Region has historically proven difficult, although they have important ecological and functional differences (Zoltai and Vitt 1995). For example, compared to fens, marshes typically have higher rates of primary production and decomposition (Thormann et al. 1999; Wray and Bayley 2007) and lower plant diversity (Whitehouse and Bayley 2005). Marshes and fens form a continuum with end types that can be distinguished on the basis of environmental variables and, at least in the wet meadow zone, possess characteristic plant assemblages (Bayley and Mewhort 2004). For example, the presence of *Drepanocladus* spp. is considered diagnostic of fen habitat (Vitt 1994). The wet meadow zones of the four wetlands containing the *U. macrorhiza* Sub-group included *Drepanocladus* spp. along with other species characteristic of fens, such as *Triglochin maritima* and *Equisetum fluvatile*. However, they were not classic fens, and were dominated by *Carex* spp. characteristic of marsh habitat (Raab 2010). Based on the plant community, nutrient levels, and ionic composition, I consider the *U. macrorhiza* Sub-group as occupying an intermediate point between fen and marsh habitat, whereas the *C. demersum* Sub-group is typical of classic marshes (Bayley and Mewhort 2004).

OSPA wetlands, particularly the more saline ones, mainly have SAV belonging to the *R. cirrhosa* or the *Chara* Sub-groups; two of the lowest richness and highest dominance assemblages that I observed. Likely, these assemblages possess adaptations that enable them to tolerate the stress imposed by wetland construction and tailings contamination. *R. cirrhosa* is one of the few submergent species that can tolerate hypersaline conditions (> 50 ppt TDS) (Stewart and Kantrud 1972). This tolerance may be responsible for the occurrence of *R. cirrhosa* in saline wetlands such as those affected by tailings water. *Chara* spp. form thick monospecific stands: in wetlands containing the *Chara* Sub-group its average relative abundance was >80%. It occurs in deeper water than most other SAV (e.g., Sand-Jensen and Madsen 1991), although it is intolerant of shading (Lacoul and Freedman 2006) and is normally associated with reduced turbidity, where it may be a superior competitor (Van den Berg et al. 1999). One advantage *Chara* spp. has in reclamation wetlands with consolidated tailings bottoms is that its holdfasts do not require it to root in the relatively impenetrable tailings material. Other studies have found that *Chara* spp. are advantaged on impenetrable bottoms (Van den Berg et al. 1999) and Cooper (2004) identified *Chara* spp. as an early colonist on consolidated oil sands tailings substrates. Thus, I consider the *R. cirrhosa* Sub-group a saline/sodic community, whereas the *Chara* Sub-group is a clear, deep, alkaline community often found on impenetrable substrates. Let it be noted, however, that wetlands constructed in the future are expected to cap consolidated tailings with at least 1 m of tailings sand and peat mix (Harris 2007). This capping may not alter the chemical composition of interstitial water, but will likely reduce the impenetrability of the substrate. Thus, *Chara* spp. may be less advantaged in future reclamation wetlands.

Another five wetlands clustered to form the *P. pusillus* Sub-group, indicated by *P. pusillus* and *S. cuneata*. The OSPA wetlands supporting this Sub-group had some of the lowest salt and cation levels of all OSPA wetlands: TDS in these two wetlands was 1.26 and 1.66 g/L, levels typical of *P. pusillus* populations (Stewart and Kantrud 1972). Trites and Bayley (2009b) noted that *P. pusillus* was found under similar nutrient and ionic concentrations; however, they did not observe *S. cuneata*.

Two OSREF wetlands contained the *Chara* Sub-group, but the majority (n = 10) contained the *Myriophyllum* Sub-group. In OSREF wetlands, *Chara* spp. dominated where alkalinity, TDS, and water nutrients were higher. Where these values were lower, *Myriophyllum* spp. and *H. vulgaris* were found. Therefore, like the OSPA wetlands, OSREF wetlands have different assemblages depending on how saline the water is.

Three of the wetlands contained no aquatic plants, forming the No Species Sub-group. The two REF sites had conductivities and water nutrients about 2 x, and TDS about 3 x levels found in wetlands containing the *C. demersum* or *U. macrorhiza* Sub-groups. Perhaps elevated ionic levels in these two wetlands precluded the survival of SAV. Annual drawdown in the years I sampled them was not much above average: mean amplitude among REF sites was 0.18 (±SE = 0.01 m), whereas these two wetlands dropped 0.21 and 0.23 m; however, the region where these two wetlands are situated experienced a drought from 2002 to 2004 that reduced them to salt pans (Trites and Bayley 2009b), and likely the sites were still recovering. The OSPA wetland included in this group was very turbid (TSS = 920 mg/L). It was also the youngest constructed wetland that we sampled (3 years since construction) and was situated adjacent to an unvegetated tailings storage dump. Work in prairie potholes suggests that a minimum 3-5 years are required for a plant community to assemble in a restored wetland without active planting (Kellogg and Bridgham 2002), although colonization of SAV is understood to occur rapidly relative to other plant guilds (Galatowitsch and van der Valk 1996), even on oil sands tailings material (Cooper 2004). Perhaps as the surrounding watershed becomes vegetated and the suspended materials settle over time, the transparency of this wetland will improve and aquatic plants will be able to survive. I consider wetlands with the No Species Sub-group to be uninhabitable, but recognize that this condition is likely temporary and might result from a variety of environmental constraints, including natural stresses such as those associated with wet-dry cycles (van der Valk 2005).

Reclamation wetlands are impaired

I set out to evaluate wetland reclamation by the benchmark of SAV assemblages prevalent in reference wetlands. I found that reclamation wetlands supported SAV with significantly different community composition than reference wetlands, and that the SAV assemblage growing in a wetland depended on its type: REF, OSREF, or OSPA. These SAV community differences were significant, despite substantial overlap in environmental conditions: reference wetlands spanned the same range in open-water area, depth, water nutrient levels, and salinity as those formed in the wake of oil sands mining, and yet they support totally different SAV assemblages. I therefore consider reclamation wetlands impaired.

Most REF wetlands (74%) contained one of two SAV assemblages, neither of which occurred in a reclamation wetland. The species *U. macrorhiza* and *C. demersum* were never observed in an OSPA wetland. Although they were occasionally observed in OSREF wetlands, they were always a minor component of the community, unlike the dominant role they occupied in REF wetlands. Recall that I sampled every accessible cluster of naturally occurring oligosaline marshes in the Boreal ecoregion of Alberta and even ventured into Saskatchewan (Fig. 3-1). Despite my comprehensive sampling efforts, I found only rare examples of the SAV assemblages populating OSPA and OSREF wetlands among the REF wetlands (Table 3-4).

SAV richness was lower in OSPA wetlands than in OSREF and REF sites. However, with a mean richness of <5 taxa per site, diversity in REF wetlands was not high. The greatest SAV richness was found in OSREF wetlands, supporting a hump-shaped relationship between richness and disturbance. I suspect that SAV richness was depressed in the naturally nutrient-rich REF sites by competition and in OSPA sites by physiological stress imposed by tailings contamination (Grime 1973). The moderate to low richness present in reference communities suggests that aiming for high alpha diversity in reclamation wetlands is not necessary, although abnormally low richness is a possibility in tailings contaminated wetlands.

It might be argued that, providing the plant assemblages found in reclamation wetlands are represented in the reference condition (even at only 5 to 10%), reclamation should be considered successful, particularly if no non-native species were observed. However, invasive SAV species are not a problem in northern Alberta (Allen pers. com.). More importantly, the total reclaimed landscape will eventually cover more than 167,000 ha with an estimated 30% of that area expected to be wetlands. At this spatial extent, major changes in plant assemblage prevalence will have ecological consequences. On the basis of current reclamation plans, I anticipate a large-scale conversion from peatland habitat to oligosaline marshes and upland forest (Harris 2007). Recent work by Walker et al. (2009) notes the fundamental non-interchangeability of ecosystems and the potential impacts of such conversions on regional biodiversity. I can only speculate on the effects of creating a large island of oligosaline marsh habitat in the midst of the Boreal forest, but I can conclude that, with exclusive coverage by atypical assemblages, five SAV taxa would be lost from the region: including the regionally rare *P. obtusifolius*, as well as aquatic moss, *Zanichella palustris*, *P. fresii*, and *Najas flexilis*, as these were only found in *C. demersum* or *U. macrorhiza* assemblages. I chose to study SAV not only because of its indicator potential, but also because of the important ecological functions it serves in shallow open-water wetlands. Fish, waterfowl, amphibians, invertebrates, and epiphyton dependent on SAV will be influenced by a change in SAV assemblage, particularly one that reduces SAV density or diversity. Not only are there differences in richness and density among the Sub-groups, but there are also structural and functional differences. For example, *P. pusillus*, *S. cuneata*, *R. cirrhosa*, *Myriophyllum* spp., and *H. vulgaris* are rooted plants, whereas *C. demersum* and *U. macrorhiza* are unrooted. *C. demersum* and *U. macrorhiza* usually fill-in the water column, producing shade and shelter for invertebrates, whereas *Chara* spp. form a thick mat on the sediment overlain by open water. Certainly, pelagic community changes must be anticipated if the dominant plant goes from a carnivorous one (*U. macrorhiza*) to an autotroph. Thus, even if the atypical assemblages were of equivalent richness and density to reference assemblages, important ecological differences between reclamation wetlands and the reference condition would persist.

Exclusive coverage by atypical SAV assemblages across the whole reclamation landscape will not yield a "balanced, integrated, adaptive community having a species composition, diversity, and functional organization comparable to that of natural habitat of the region" (Karr 1991). However, it would be ill-advised to create only wetlands containing the *C. demersum* and *U. macrorhiza* Sub-groups. Gamma-level biodiversity should be encouraged by retaining examples of the atypical SAV assemblages within the reclamation landscape (Trites and Bayley 2009b). I recommend that reclamation efforts aim to maintain the ratio of SAV Sub-groups present among the REF wetlands (Table 3-4).

Not only do the SAV communities typical of OSREF and OSPA wetlands differ significantly from those typical of REF wetlands, but they differ from each other. A transplant experiment by Cooper (2004) found that the use of consolidated tailings as a sub-soil reduced emergence from wetland seedbanks and that transferring sediment/plant mesocosms among wetlands resulted in plant community changes. For example, sediment plots from OSREF and REF wetlands transplanted into an OSPA wetland underwent community shifts from aquatic moss or *Myriophyllum* assemblages to those dominated by *S. pectinata* or *P. pusillus* (Cooper 2004). In the absence of other possible causal mechanisms, I suggest that tailings water and materials have impacts on the SAV community above and beyond the effects of physical disturbance and increased sulfate and sodium attributable to the marine-shale overburden.

It bears reiterating that despite the absence of tailings water or materials, the SAV assemblages found in OSREF wetlands differ from those typical of REF ones. This is important, as several researchers have used OSREF wetlands as reference systems for the evaluation of OSPA wetlands (e.g., Barr 2009; Wytrykush et al. 2008). In my opinion, the appropriate reference systems for the evaluation of OSPA wetlands are true reference wetlands (REF), free not only from tailings contamination, but also from the stresses associated with wetland construction, the presence of the saline-sodic overburden, and a surrounding landscape fragmented and heavily disturbed by mining. We aim absurdly low if we treat OSREF wetlands as representing the "best attainable condition" (sensu Stoddard et al. 2006) expected for all oil sands reclamation wetlands.

On the basis of currently approved projects and the best available information, over 100,000 ha of natural wetland habitat will be destroyed (Raine et al. 2002; Richens pers. comm.), and given that the Government of Alberta maintains a 100% approval rate for proposed mining projects, this number will only grow larger in the future. This extensive loss of habitat is occurring despite scarce evidence that successful wetland reclamation is possible. At present, mine closure plans call for the creation of a mosaic of upland forest and shallow open-water wetlands where once the land was dominated by peatlands. Given that the Government has already accepted the large-scale conversion of peatlands to shallow open-water wetlands ranging from fresh to oligosaline, I argue that, at the very least, what wetland reclamation does occur must be held to a rigorous standard; one based on sound scientific principles.

Approach

I developed a unified framework (sensu Clarke 1993) for the evaluation of reclamation success based on the community structure of vegetation. This framework will be particularly useful in ecosystems where the natural range of variability in biotic condition is high and where more than one reference condition might be appropriate. By defining multiple reference states, I acknowledge that multiple restoration trajectories are possible (Matthews et al. 2009b). Brooks et al. (2005) noted that reclamation wetlands often fail to reflect the level of heterogeneity typical of natural wetlands. In part, homogeneity of reclaimed wetland communities results because reclamation wetlands are designed to meet a single target, rather than to provide different sets of functions and values. Currently, no community-based targets for reclamation are in place. On the basis of this chapter, I recommend that at least two community-based targets be adopted.

Given the spatial extent of reclamation necessary in the surface mineable oil sands area, Alberta is at risk of creating a very large, homogenous landscape with low gamma diversity if all reclamation wetlands were required to aim for a single target.

Conclusion

It is possible to support healthy, natural aquatic plant communities under environmental conditions such as those created in oil sands reclamation wetlands. However, oil sands wetland reclamation to date has not succeeded in producing aquatic plant assemblages that resemble those that typify the reference condition: communities dominated by *C. demersum* or *U. macrorhiza* and aquatic moss should be accepted. These species are present in the oil sands mining region, so it is doubtful that dispersal limitation prevents their occurrence in reclamation wetlands. Seeding and planting species typical of reference wetlands may improve reclamation outcomes; however, unless environmental conditions favoring their persistence are in place, they will likely die off. Increasing sediment nutrients and organic matter content and reducing basin slopes, total dissolved solids and sodium concentrations may help foster reference plant assemblages. The level of alkalinity, nutrients in the water, and turbidity appear to be far less important in terms of supporting reference assemblages, and will not usually act as impediments to developing healthy aquatic plant communities in reclamation wetlands.

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4. Development and testing of an index of biotic integrity based on submersed and floating vegetation and its application to assess reclamation wetlands in Alberta's oil sands area, Canada[3].

Introduction

Since its inception in 1967, oil sands mining in north eastern Alberta has disturbed over 67,000 ha, and an additional 408,000 ha are expected to be mined (Richens pers. comm.). Reclamation work has been carried out for over 35 years; however, it wasn't until 2008 that the first parcel of land was certified as reclaimed by the Alberta government. The 104 ha of reclaimed land now consists of upland forest, but before mining it was part of a large wetland complex.

Wetlands constitute about 65% of the oil sands mining region, 63% of which is wooded fen (Raine et al. 2002). Shrubby fens are the next most common wetland type, constituting 10% of the total landscape. Reclamation of mined land will necessarily include some wetland construction (GOA 1993). Current plans aim to create a reclamation landscape containing 33% wetlands (Harris 2007), or approximately 22,000 ha. Already, about 25 shallow open water wetlands have spontaneously developed or been constructed on the leases of Suncor Energy Inc. and Syncrude Canada Ltd, the two companies collectively responsible for 81% of all Albertan oil sand mined in 2007 (ERCB 2009a). Yet appropriate reclamation targets for wetlands have not yet been adopted and to date there is no means of evaluating reclamation progress. The lack of a method to monitor and evaluate wetland reclamation is part of the reason that despite 35 years of effort, wetland

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reclamation remains in preliminary stages and no reclamation wetland has been assessed for certification.

The multi-metric index of biological integrity (IBI) is the bioassessment method favoured in North America. Over 90% of US watershed management agencies use some form of multi-metric assessment (Barbour and Yoder 2000). First developed in the 1980s to assess rivers using the fish community (Karr 1987), the IBI approach has since been adapted to evaluate numerous ecosystems using various taxa. Plant-based IBIs are a popular means of evaluating wetland condition (e.g., Croft and Chow-Fraser 2007; DeKeyser et al. 2003; Miller et al. 2006; Simon et al. 2001).

Although IBIs are more commonly derived from the emergent vegetation, the floating and submersed aquatic vegetation (SAV) has great potential to serve as a source of metrics for IBI development, especially for the open water portion of the wetland complex. SAV is sensitive to environmental conditions including nutrient availability, transparency, and salinity (e.g., Lacoul and Freedman 2006) among other factors (Weisner et al. 1997). In addition, SAV exerts influence on the wetlands it inhabits. It connects the sediment and the water, acting as a nutrient pump (Barko et al. 1991) and oxygenating the rhizosphere (Flessa 1994). It can influence light availability by reducing sediment resuspension (James et al. 2004) or by creating a canopy. Furthermore, it provides food and habitat for numerous taxa including invertebrates, amphibians, fish, and waterfowl. Thus, SAV is not only affected by, but also exerts an influence on wetland condition. Finally, it is relatively quick, easy, and inexpensive to sample and has well established taxonomy and widely available keys. This makes it an excellent subject for IBI development. SAV can be used to characterize open-water habitat (e.g., Sondergaard et al. 2010), but I wanted to determine whether SAV alone (i.e., excluding emergent vegetation) could adequately characterize wetland condition, and so I undertook to develop and test an IBI based on floating and submersed vegetation.

My aim was to develop an IBI based on the SAV community and to use it to evaluate reclamation wetlands in the oil sands mining region of Alberta. I wanted the IBI scores to be based on scientifically sound and defensible criteria, including a scoring method that maximizes its sensitivity to environmental stress. In addition, I wanted to determine whether I could improve the IBI by broadening the pool of available metrics to include those with non-linear relationships to disturbance. I therefore made three major adaptations to published IBI methods: 1) I used an independent and objectively constructed stress gradient (Chapter 2, Rooney and Bayley 2010) calculated from physical and chemical data to select among candidate metrics and to test the IBI; 2) I contrasted four different scoring methods that are representative of the range of scoring methods used in existing IBIs; and 3) I used piecewise quantile regression to seek metrics with non-linear but nonetheless significant relationships to environmental stress. I then used the IBI scores to carry out an assessment of the biological integrity of existing reclamation wetlands in the oil sands region of Alberta, Canada.

Materials and methods

Design

To develop the vegetation-based index of biotic integrity, I sampled in-lake environmental conditions and the submersed and floating aquatic vegetation community in a suite of 62 wetlands including both reclamation and reference wetlands (sensu Stoddard et al. 2006), i.e., those representing least-disturbed conditions. I divided the resultant dataset into thirds with equal numbers of each wetland type. I used two thirds (n = 42) to develop the IBI and retained the remaining third (n = 20) to test the IBI. I used the environmental data to construct a stress gradient (see Chapter 2 or Rooney and Bayley 2010 for details). I then calculated 60 metrics using the plant community data (Appendix 4-1) and assessed the relationships between metric values and stress scores at each wetland using linear and piecewise quantile regression.

Sites

The 62 wetlands I sampled included 25 reclamation wetlands and 37 reference wetlands. Reclamation wetlands were located on oil sands mining property leased by Suncor Energy Inc. and Syncrude Canada Ltd. in a landscape heavily influenced by mining activity and tailings storage (Fig. 4-1). Reference wetlands were located mainly in protected parks scattered across the Boreal Plains ecoregion of Alberta (Fig. 4-1). They were situated in landscapes dominated by forest and had low levels of forestry or agricultural activity within 2 km of their open water zones. Both types of wetland covered a broad range in TDS (reference = 0.1 to 4.9 g/L; reclamation = 0.3 to 3.5 g/L), water depth (reference = 0.2 to 3.3 m; reclamation = 0.3 to 3.3 m), and surface area (reference = 0.06 to 18.38 ha; reclamation = 0.08 to 19.02 ha). Reclamation wetlands included both those exposed to oil sands tailings materials (n = 13) and those free from tailings contamination (n = 12). Reclamation wetlands ranged in age from 3 to 30 years, with a mean age of 14 years.

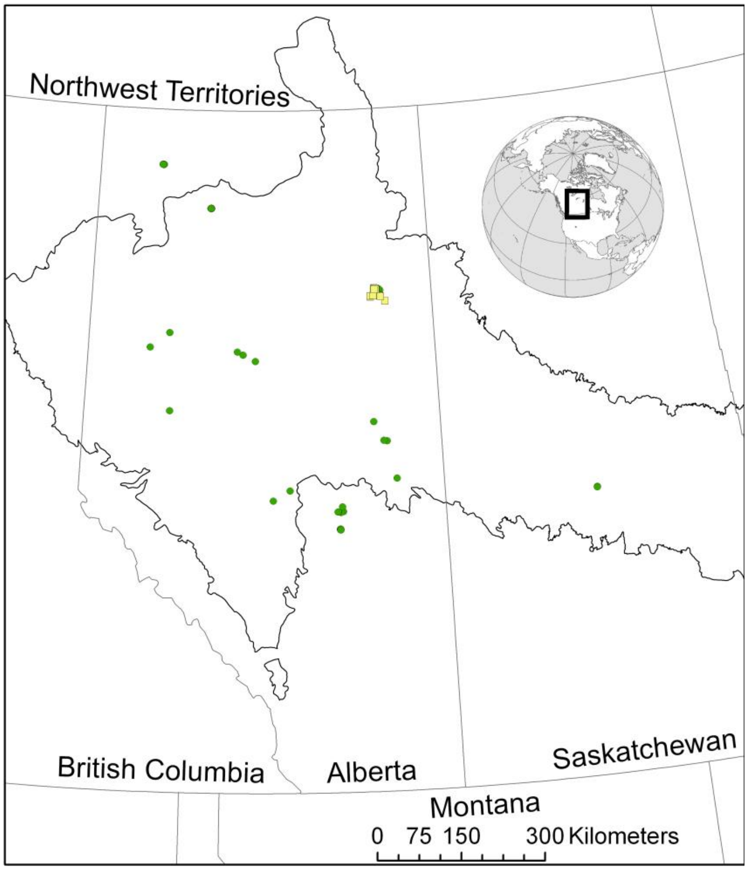

Fig. 4-1. Map depicting the location of study sites. Reclamation wetlands (squares) are clustered on the leases of the two dominant oil sands mining companies, whereas reference wetlands (circles) are scattered across the Boreal Plains ecoregion. Not all sites are visible because of symbol overlap (n = 62).

Environmental data

The stress gradient that I used to develop and test the IBI is calculated from eight abiotic variables which were found to be most representative of environmental stress from an initial suite of 52 physical and chemical variables following the method outlined in Rooney and Bayley (2010) and Chapter 2. These included chloride (Cl) and nitrogen (TN) levels as well as conductivity of the water; water and oil content of sediment; water depth and amplitude; and Secchi depth/total depth. For details about stress score calculations, see Rooney and Bayley (2010) or Chapter 2.

Vegetation

Floating and submersed vegetation were sampled from kayaks during the period of peak biomass using the rake technique outlined in Rooney and Bayley (2011b) and Chapter 3. In short, I traversed each open water zone with 10 transects and deployed a 1 m^2 quadrat at a random location along each transect. I collected SAV from within each quadrat and identified plants to the lowest practical taxonomic level (usually species) following Moss and Packer (1983) with taxonomy updated according to the Integrated Taxonomic Information System (ITIS 2010). I was unable to identify a few taxa to the species-level reliably in the field: an aquatic moss, the macroalgae *Chara* spp., and *Myriophyllum sibiricum* and *M. verticillatum,* which I therefore considered jointly as *Myriophyllum* spp. I recorded the relative abundance (as percent infestation) of each taxon within each quadrat and estimated total SAV biomass using a rating system with a scale of 1 to 5. This SAV rating system yielded scores well correlated with biomass per square metre in a previous study (Bayley et al. 2007), suggesting that rating is a reasonable surrogate for SAV biomass. I took the median of the ratings from the 10 quadrats to yield a median SAV rating for every wetland. I also estimated "overall" SAV biomass in the open water zone, giving a single rating to reflect the SAV biomass across the wetland as a whole. Relative abundance numbers from the 10 quadrats were averaged. All metrics calculated from relative abundance numbers include only species found in at least one quadrat. Any species observed along transects but not included in a quadrat were included in total richness and all metrics based on total richness.

Metric selection

I performed simple linear regression for each of the 60 metrics on stress gradient scores using Maximum Likelihood estimation in the program SYSTAT (2007) in order to identify metrics with positive or negative linear relationships to environmental stress. I included any metrics with significant ($\alpha = 0.05$) relationships to stress scores in further analyses.

Metrics with non-linear relationships to disturbance may still convey useful information about a wetland's condition. Based on the "wedge shaped" pattern between several of the candidate metrics and stress gradient scores, I tested for thresholds using a quantile piecewise linear regression approach (Brenden et al. 2008). This is a regression tree method that uses quantiles to partition the metric values into groups based on differences in the conditional relationship between the metric and stress scores at each tree. Wedge shaped relationships are common in ecology (McCune and Grace 2002; Wang et al. 2003), and result when the upper limit of the response variable is potentially set by one of several determining factors, not just the measured predictor variable. The quantile piecewise linear approach is a means of threshold detection well suited to dealing with the challenges of wedge shaped relationships (Chaudhuri and Loh 2002) but is also robust with other patterns that are common among ecological variables (Brenden et al. 2008). I used the program GUIDE to carry out piecewise quantile regression (Loh 2010) using the 90th percentile. Regression trees were pruned using CART's cost-complexity model with 10-fold cross-validation (Loh 2009). I pruned the resulting tree using the threshold of 0.01 standard errors. This value can range from 0 to 1000 and controls the ultimate size of the pruned classification tree, with smaller numbers yielding larger trees. Only a single metric had a significant non-linear relationship to disturbance: H'_ED/G_Simp_ED is a measure of dominance, where the greater the difference between H'_ED and G_Simp_ED, the more the community is dominated by a few common taxa (Jost et al. 2010). To evaluate the contribution of this non-linear metric, I recalculated IBI scores for the development and test sets using only metrics with significant linear relationships to stress scores. Testing the IBI including versus excluding this non-linear metric revealed that its inclusion did not improve the overall sensitivity of the IBI and I therefore excluded it from further analyses (details in Appendix 4-2).

I evaluated redundancy among the selected metrics by calculating Pearson's correlation coefficients in SYSTAT (2007). When two metrics had a correlation coefficient ≥ 0.6, I retained the metric with the stronger relationship to stress scores and excluded the other.

Metric scoring

There are two factors that characterize any scoring method (Fig. 4-1). First, scoring can be discrete, dividing the range of observed metric values into bins and then assigning scores based on bin membership, or it may be continuous, assigning scores as a linear interpolation between the maximum and minimum metric values. Second, the range used for scoring can include all observed values or scoring can be relative to the range of values from the reference sites, thereby situating scores relative to the reference condition. Combining these two factors yields four classes of scoring method and I used examples of each scoring method to score the selected metrics (Fig. 4-2). Binning for discrete scoring was achieved using percentiles (sensu Rooney and Bayley 2010). For Method 1(the discrete – whole range scoring or DWR) I divided the entire range of metric values observed in the development set into 5 bins, each 20 percentiles wide. Thus, for example, if a metric exhibited a negative relationship to stress, values > 80^{th} percentile of its range would be scored a 5, values in the 60-80^{th} percentile range would score a 4, the 40-60^{th} percentile would score a 3, etc. For Method 2 (the discrete – reference range scoring or DRR) I applied the same approach, but using only the range of values observed among reference wetlands in the development set. For Method 3 (continuous – whole range scoring or CWR) and Method 4 (continuous – reference range scoring or CRR) I followed the methods used in Blocksom (2003), converting metric values into scores using the equations indicated in Figure 4-2. For metrics that were positively correlated with stress scores, the metric score was inverted: 100 – the value calculated following equations in Figure 4-2. An example calculation of a wetland's IBI score using the CRR method is presented in Appendix 4-3. For each method, I then summed the metric scores together to obtain an IBI score for each wetland in the development set. Thus, the minimum score possible by discrete scoring is simply the number of metrics and the maximum is 5 times the number of metrics included in the IBI. In contrast, by the continuous scoring method, the minimum possible score is 0% times the number of metrics and the maximum possible score is 100% times the number of metrics included in the IBI.

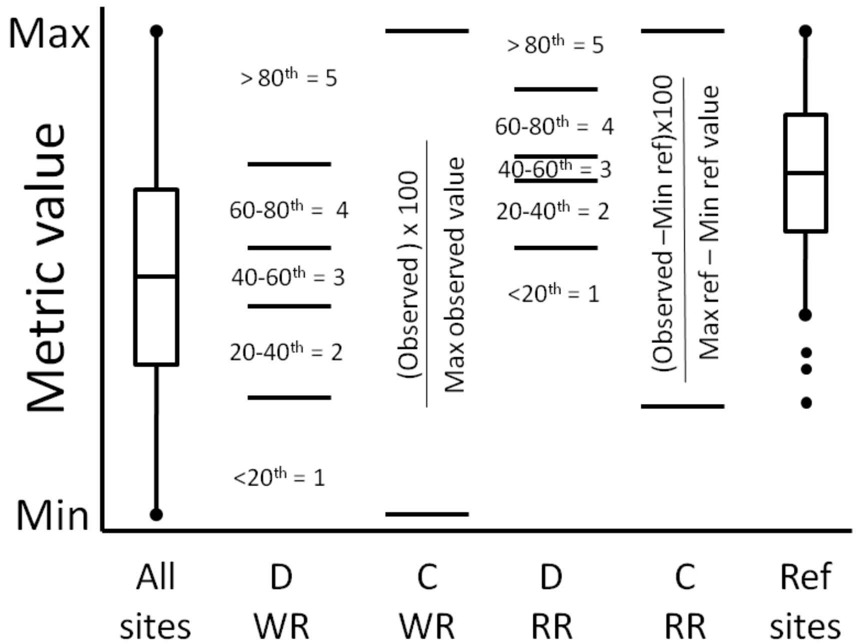

Fig. 4-2. Overview of the four methods used to score metrics. D refers to discrete methods, C to continuous methods, WR refers to scoring relative to the range of all observed values, and RR to scoring relative only to the range of reference wetlands. Discrete scoring divides the range into 20 percentile wide bins to give scores ranging from 1 to 5. Continuous scoring is based on percentages rather than percentiles with equations from Blocksom (2003).

The relative ability of the four scoring methods to yield an IBI that is strongly predictive of stress scores was evaluated for the development and test sets by comparing the Spearman rank correlation coefficients for each IBI and the appropriate stress scores. I used Spearman rank correlation because Andersen-Darling normality tests found that residuals from regressing IBI scores on stress scores were not normally distributed for the test data set. I assessed the significance of Spearman coefficients with a bootstrapping method. I created 999 samples with n = 42 for the development set and n = 20 for the test set in order to determine 95% confidence intervals on the Spearman coefficients. Bootstrapping and confidence interval generation was done using the percentile method in

SYSTAT (2007). If the 95% confidence intervals included 0, the IBI generated by that scoring method was deemed not significantly correlated with stress scores.

The scoring method producing the IBI with the strongest significant relationship (largest Spearman *rho* value) to stress scores was considered most sensitive. However, other factors were considered in deciding which method was best, including the relative width of 95% confidence intervals, breadth of the range of possible IBI scores, and theoretical grounds.

IBI testing

To test the IBI, I calculated the IBI scores for the third of wetlands reserved in the testing set. The wetlands in the testing set were independent of those in the development set, allowing me to evaluate the ability of the IBI to predict stress scores and thus to infer environmental condition.

Reclamation wetland assessment

To assess the integrity of reclamation wetlands in the oil sands area, I compared the IBI scores of tailings-contaminated reclamation wetlands, reclamation wetlands free of mine tailings, and reference wetlands using a Model I ANOVA and a Tukey's HSD multiple comparison test. I only considered scores from the IBI that I identified as superior during IBI testing.

Results

Metric selection

Of the 60 metrics tested, only eleven had significant linear relationships to stress scores (Table 4-1). The mean r^2 value for these regressions was 0.22 with a maximum of 0.32 for the relative abundance of *Ceratophyllum demersum* (%C_Cdemersum) and a minimum of 0.12 for the % of all taxa belonging to the genus *Potamogeton* (%_Potamogeton). *C. demersum* is a species that typically dominates reference wetlands, whereas the potamogetons are a species-rich genus found more commonly in disturbed wetlands (Chapter 3, Rooney and Bayley 2011b).

Table 4-1. Regression results for metrics with significant linear relationships to environmental stress, represented by stress scores. For all *F*-tests, degrees of freedom were 1, 40.

Metric	Equation	r^2	*F*	*p*-value
Richness of floating vegetation (FLT_S)	= 12.98773 - 1.29956x	0.20	10.10	0.00286
Relative abundance of halophytes (%C_Halophyte)	= 10.73205 + 0.03441x	0.27	14.55	0.00046
Relative abundance of alkali-loving species (%C_Alkali)	= 10.60872 + 0.03567x	0.27	15.14	0.00037
% cover of floating leafed species (%C_Float_Leaf)	= 12.51859 - 0.05133x	0.17	8.08	0.00703
% of total richness constituted by halophytes (%_Halophyte)	= 10.84229 + 0.03978x	0.22	11.09	0.00188
% of total richness constituted by alkali-loving species (%_Alkali)	= 10.54412 + 0.04190x	0.25	13.63	0.00067
% of total richness constituted by floating leafed species (%_Float_Leaf)	= 13.09690 - 0.06570x	0.23	12.25	0.00116
Relative abundance of *Chara* spp. (%C_Chara)	= 11.21151 + 0.02832x	0.14	6.45	0.01507
Relative abundance of *C. demersum* (%C_C_demersum)	= 13.07610 - 0.04062x	0.32	18.99	0.00009
% of total richness constituted by *C. demersum* (%C_demersum)	= 12.89425 - 0.07561x	0.26	14.29	0.00051
% of total richness constituted by *Potamogeton* spp. (%_Potamogetons)	= 11.13201 + 0.03904x	0.12	5.69	0.02186

Redundancy analysis

After discarding metrics that were redundant (Pearson $r \geq 0.6$), five metrics remained (Table 4-2). These included one measure of diversity: the richness of floating species (FLT_S); two measures of functional group abundance: the relative abundance of alkali-tolerant species (%C_Alkali), and the percent cover of floating leaf species (%C_Float_Leaf); and two measures of taxa of interest: %C_Cdemersum and %_Potamogetons.

Table 4-2. Pearson's correlation coefficients among selected metrics. Pairs of metrics with coefficients with absolute values ≥ 0.6 are redundant. Among redundant metrics, the metric that explains a greater percent of the variation in stress scores (larger r^2 value in Table 4-1) was retained for the IBI.

	FLT_S	%C_Halophyte	%C_Alkali	%C_Float_Leaf	%_Halophyte	%_Alkali	%_Float _Leaf	%C_Chara	%C_C_demersum	%C_demersum	%_Potamogetons
FLT_S	1										
%C_Halophyte	-0.4707	1									
%C_Alkali	-0.4527	0.84163	1								
%C_Float_ Leaf	0.43919	-0.4373	-0.4187	1							
%_Halophyte	-0.4724	0.87145	0.76183	-0.4319	1						
%_Alkali	-0.5002	0.59678	0.7807	-0.4406	0.70558	1					
%_Float_ Leaf	0.78552	-0.4709	-0.5987	0.56584	-0.4959	-0.6211	1				
%C_Chara	-0.471	0.86478	0.84473	-0.3651	0.82076	0.6417	-0.5268	1			
%C_ Cdemersum	0.51644	-0.5305	-0.5197	0.50832	-0.4556	-0.4814	0.6273	-0.4372	1		
%C_demersum	0.18994	-0.3958	-0.3942	0.34402	-0.3906	-0.4086	0.29421	-0.3171	0.75859	1	
%_ Potamogetons	-0.0725	-0.2858	-0.0344	-0.1455	-0.2797	0.28609	-0.2005	-0.3226	-0.0994	-0.1846	1

Scoring method comparison

Both the discrete and continuous scoring methods generated IBIs with significant correlations with stress scores (Table 4-3). Using the development set, the four methods yielded Spearman *rho* values that were roughly equivalent. Spearman *rho* values were more variable among methods when the test set was considered, with continuous scoring producing slightly stronger correlations. Discrete scoring produced narrower confidence intervals on Spearman *rho* values than continuous scoring methods. Whether scoring was discrete or continuous had a greater impact on Spearman *rho* values and confidence intervals than the range of values used for scoring (whole-range vs. reference range).

Table 4-3. Spearman rank correlation (*rho*) results for stress scores and the IBIs generated by the four scoring methods: discrete – whole range (DWR), discrete – reference range (DRR), continuous – whole range (CWR), and continuous – reference range (CRR). Confidence intervals were generated by bootstrapping using the percentile method in SYSTAT software.

		Spearman	95% C.I.	
		rho	Lower	Upper
Development suite (n = 42)	DWR	-0.66325	-0.78297	-0.49921
	DRR	-0.65913	-0.78557	-0.47035
	CWR	-0.68809	-0.79805	-0.52350
	CRR	-0.67040	-0.80433	-0.49764
Testing suite (n = 20)	DWR	-0.59176	-0.87301	-0.11112
	DRR	-0.60584	-0.86739	-0.19025
	CWR	-0.68423	-0.93579	-0.25067
	CRR	-0.63079	-0.88612	-0.18908

IBI scores calculated using the discrete approach had a potential range of 5 to 25, whereas those calculated using the continuous approach could have ranged from 0 to 500. Actual metric and IBI score ranges are presented in Appendix 4-3, along with an example of how the IBI score of a wetland is calculated following the continuous-reference range (CRR) method.

Reclamation wetland assessment

The mean CRR-derived IBI scores for reference wetlands, tailings-free, and tailings-contaminated reclamation wetlands were 252, 112, and 94.8, respectively. There are strongly significant differences among these means (Model I ANOVA: $F_{2,59} = 34.7$, $p < 0.00001$) and Tukey's HSD Test indicates that IBI scores are significantly higher for the reference wetlands ($p < 0.00001$, in both cases) than for either type of reclamation wetland, which group together ($p = 0.81156$) (Fig. 4-3).

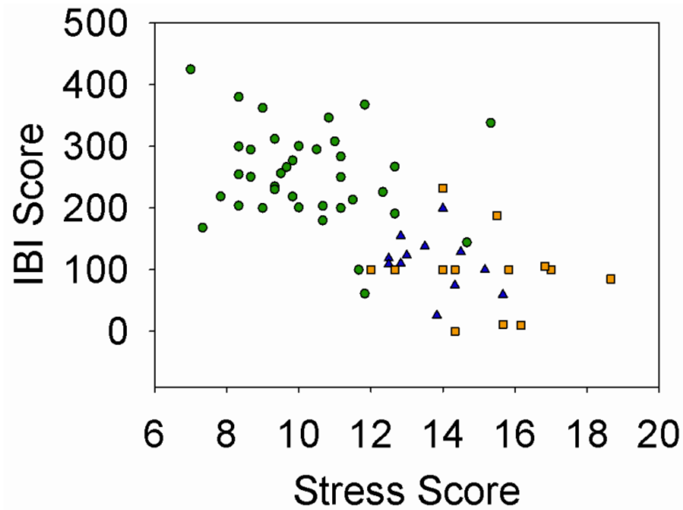

Fig. 4-3. Plot of IBI scores tallied using the continuous – reference range scoring method against stress scores with symbols representing wetland type. Reference wetlands (circle) have significantly higher IBI scores and lower stress scores, on average, than tailings-free (triangles) and tailings-contaminated (squares) reclamation wetlands: (n = 62).

Discussion

I developed and tested an IBI using metrics derived from SAV that was suitable for evaluating reclamation wetlands in Alberta's oil sands region. We evaluated four different scoring methods representative of those used in existing IBIs and used the superior method to conduct a preliminary assessment of existing reclamation wetlands.

Selected metrics

Researchers developing indices of biotic integrity (IBIs) have consistently found that it is possible to quantify ecological condition using only a handful of simple metrics (e.g., Mack 2007). We found that of 60 candidate metrics calculated from the submersed aquatic vegetation (SAV) community, only 5 were needed to yield strong and significant predictions of wetland condition for both the suite of wetlands used to develop the IBI and an independent suite of wetlands used for testing it. Thus, we conclude that the SAV community is a good indicator of shallow open-water wetland condition in northern

Alberta and that a small number of simply acquired measurements are adequate to quantify it.

Although, as recommended by Karr (1987), our initial suite of candidate metrics included those from the population-, community-, and ecosystem-level, we found that taking measurements of specific plant populations, functional groups, and community structure was adequate to assess wetland condition. Of five metrics included in the final IBI, two are measures of richness and three are measures of relative abundance. Richness and abundance metrics frequently dominate IBIs because of their greater sensitivity to low levels of environmental stress as compared to ecological processes (Karr and Chu 1997). Other plant-based IBIs are also dominated by measures of richness and abundance (e.g., DeKeyser et al. 2003; Hargiss et al. 2008; Miller et al. 2006).

The relationships between individual metrics and environmental stress help illustrate how the stresses associated with oil sands mining influence the submersed and floating plant communities. For example, the significant negative relationship between the richness of floating species and stress scores indicates that healthier wetlands support a greater diversity of floating species. One or two species of the floating genus *Lemna* are often the first to arrive at newly created wetlands, likely because they cling and are readily transported between sites (Keddy 1976). Thus, less established wetlands may have a low diversity of floating species due to dominance by a few strong dispersers. Not only do stressed wetlands have fewer floating species, but the percent cover of floating vegetation is also lower in wetlands with high stress scores. This reduction in % cover of floating vegetation is likely an effect of reduced productivity, perhaps due to lower nutrient levels (Rooney and Bayley 2010). The positive association between environmental stress and the dominance of alkali-tolerant species is probably due to the presence of alkaline tailings water and cation-rich mineral sediments in reclamation wetlands (Rooney and Bayley 2010). An earlier investigation into the submersed plant communities across a gradient in oil sands disturbance found that that *C. demersum* was characteristic of reference wetlands where nutrient levels were higher, whereas a diverse collection of *Potamogeton* spp. was characteristic of reclaimed sites, especially the tailings-free variety (Rooney and Bayley 2011b).

Scoring methods

The metric scoring method may influence IBI performance because it alters the distribution of metric values, which in turn may affect the IBI's relationship to disturbance. It may also influence the IBI's sensitivity to random sampling variation. Despite its importance, we are not aware of any other study comparing the effect of scoring method on plant-based IBIs or on IBIs geared towards wetland assessment.

Blocksom (2003) examined the effects of scoring methods on an invertebrate-based IBI developed for streams. She, too, compared continuous and discrete methods of scoring. She found that the correlation between IBI scores and an index of habitat quality was greater, with slightly (< 0.1) larger Pearson correlation coefficients, when continuous scoring was used. Our results support her conclusion that continuous scoring yields a more sensitive IBI (Table 4-3). In the discrete methods, binning acts like a smoothing feature to reduce some of the variability among metric values. Conversely, continuous scoring propagates or may even enhance small differences in metric values, resulting in slightly greater sensitivity. Furthermore, continuous scoring provided a much broader range of IBI values than discrete scoring: IBI scores ranged from 0 to 425 for both continuous methods vs. 5 to 22 or 5 to 24 for the DRR and DWR methods, respectively. This greater range of values permits enhanced resolution when discriminating among wetlands. Consequently, we consider continuous scoring methods superior to discrete scoring methods.

We also examined the effect of whether the range of values used to score metrics included values from all sites or was restricted to the range of values observed in reference sites. Both methods yielded roughly equivalent Spearman *rho* values when resultant IBI scores were correlated with stress scores (Table 4-3). Blocksom (2003) did not attempt a DWR scoring method. Of the methods she did contrast, she concluded that a CWR method was superior. Her conclusion, however, was not based on differences in the strength of the relationship between IBI scores and environmental condition (Pearson $r = 0.638$ for the method based on the reference range vs. 0.640 and 0.585 for methods based on the whole range). We therefore find that the range of values used to assign metric scores had comparatively little effect on IBI sensitivity or resolution. However, we argue that scoring relative to the reference range rather than the whole range of values has greater theoretical validity. When scores are assigned relative to the range of values found in reference sites we are in effect giving a site a score that positions it relative to the reference condition. In this case, the reference condition for a given metric is defined by the distribution of metric values measured at the reference sites and thus incorporates the natural range of variability among relatively undisturbed wetlands (Fig. 4-3) while excluding the values found in highly degraded sites. Given that our intent in reference condition-based assessment approaches like the IBI is to score a wetland relative to the reference condition (Bowman and Somers 2005), we argue that the reference range approach has greater validity.

Consequently, we conclude that the CRR method of metric scoring produces the best IBI, as it yields scores that are more sensitive to environmental stress and it possesses greater resolution to discriminate between individual wetlands than discrete scoring methods. It also has greater theoretical validity than scoring based on the whole range of observed values. However, we note that the influence of scoring method was slight and that all four commonly used metric scoring methods produced IBI scores that were strongly and significantly correlated with stress scores.

Evaluation of reclamation wetlands

Using the scoring method identified as superior in the previous section, we assessed reclamation wetlands in the Athabasca oil sands region and found that their biological integrity was significantly below that typical of reference wetlands. Both tailings-contaminated and tailings-free reclamation wetlands scored, on average, below reference wetlands, suggesting that the presence of tailings is not the only source of stress negatively impacting the floating and submersed plant communities in reclamation wetlands. Although, on average, reclamation wetlands proved of lesser biological integrity than reference wetlands, there was great variability in their IBI scores. Tailings-free and tailings-contaminated reclamation wetlands ranged from 26 to 199 and 0 to 233, respectively. No reclamation wetlands reached the upper portion of the range of scores calculated for reference sites, but there was some intergrading with the lower portion of the reference condition range (Fig. 4-3). This suggests that reclamation wetlands can achieve levels of biological integrity comparable to those found in reference wetlands, but that many are failing to do so. We consider this a preliminary assessment because reclamation practices and guidelines continue to evolve and no company has yet sought to have a reclamation wetland certified by the Alberta government. Hence, it is not unexpected that so many reclamation wetlands have low biological integrity.

The five metrics included in the IBI also serve to indicate how reclamation outcomes can be improved. Reclamation practitioners in the oil sands region should aim for a greater diversity of floating species and a greater relative abundance of *C. demersum*. By identifying which metrics are performing poorly in a reclamation wetland, practitioners will be able to determine what aspects of the plant community are under-performing and target these with an adaptive management strategy to improve reclamation outcomes.

Non-linear metrics

Threshold detection methods have found numerous applications in ecological research, including forecasting biodiversity loss (Hilderbrand et al. 2010) and tracking restoration progress (Clements et al. 2010). Quantile piecewise linear regression has proven to be one of the more robust techniques of detecting thresholds and is particularly well suited to the wedge-shaped patterns common in ecological data (Brenden et al. 2008). We sought to determine whether incorporating quantile piecewise linear regression would enhance IBI development by widening the pool of available metrics to encompass those with non-linear relationships to disturbance. Of the 60 SAV metrics we examined, quantile piecewise linear regression produced a superior fit to simple linear regression for only one metric: H'_ED/G_Simp_ED. Miller et al. (2006) also found that most (49 of 50) candidate plant-based metrics possessed linear relationships to disturbance; however, they did not test this finding statistically. Although including quantile piecewise regression allowed us to identify a metric that would not otherwise have been noted as a significant predictor of stress scores, including H'_ED/G_Simp_ED in the IBI did not improve the strength or significance of the correlation between IBI and stress scores for the four scoring methods (Appendix 4-2). Whereas the objective in IBI development is to select only as many metrics as necessary to produce the best possible estimates of ecological condition, we rejected H'_ED/G_Simp_ED. However, we suggest that the application of non-linear regression approaches to metric selection in IBI development warrants further exploration.

Conclusion

We produced an IBI using metrics derived from floating and submersed plant communities that was strongly and significantly correlated with wetland condition and environmental stress. We compared the performance of four scoring alternatives representative of methods used in existing IBIs. Although all four metric scoring methods yielded IBI scores that were correlated with environmental stress, continuous scoring relative to the range of values found in reference sites produced a superior IBI. It offers a good balance of resolution and sensitivity and is firmly grounded in reference condition theory. Our final index will assist regulators and mining companies in evaluating the condition of wetlands situated on oil sands mining leases. Currently, the biological integrity of most reclamation wetlands is significantly below the bar set by reference wetlands. However, a few reclamation wetlands scored within the range of reference site scores. An examination of individual metric scores for each under-performing site will provide insight into what site-specific factors are responsible for community impairment. Based on our comparison of tailings-contaminated and tailings-free reclamation wetlands, we conclude that isolation from tailings materials and process-affected water will be insufficient to guarantee adequate biological integrity. Other sources of stress are also acting to impair the floating and submersed vegetation.

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5. Relative influence of local- and landscape-level habitat quality on aquatic plant diversity in shallow open-water wetlands in Alberta's boreal zone: direct and indirect effects[4].

Introduction

Submersed and floating (aquatic) vegetation are sensitive indicators of local-environmental conditions (e.g., Dennison et al. 1993; Lacoul and Freedman 2006; Sondergaard et al. 2010). Aquatic vegetation also exerts influence on abiotic conditions (Barko et al. 1991; Flessa 1994; Moore et al. 1994), and influences wetland biota across multiple trophic levels (e.g., Norlin et al. 2005) by providing both habitat and food. Consequently, I used the aquatic plant community to evaluate the success of reclamation efforts in Alberta's oil sands mining area (Rooney and Bayley 2011a). I found that despite substantial overlap in in-lake habitat conditions, constructed wetlands support entirely different aquatic plant communities than those found in reference (sensu Stoddard et al. 2006) wetlands (Rooney and Bayley 2011b). This suggests that some other, unmeasured factor is also influencing the diversity of aquatic plants in the reclamation wetlands.

One possibility is that landscape-level factors have previously unidentified influences on the aquatic plants in constructed wetlands. This issue would likely be of global significance to conservation, though here I focus on oil sands mining reclamation, where wetlands are situated in a landscape that is heavily fragmented and disturbed by human activity. According to industry, a block of boreal Alberta over 67,000 ha in area is already disturbed by oil sands mining (Richens pers. comm.), and reclamation wetlands

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are being constructed in its midst. I use the term “reclamation wetlands” because although they are situated on land that oil sands mining companies self-report as reclaimed, they have not undergone provincial assessment to certify them as reclaimed wetlands. Oil sands mining is expected to eventually cover the entire surface mineable area, approximately 475,000 ha (ERCB 2010), and thus understanding the landscape-level impacts has direct conservation implications over a large spatial extent.

Landscape composition can potentially influence aquatic plant diversity in two ways. First, directly, if the surrounding land-use reduces the number of potential propagule sources or limits propagule dispersal such that the rate of propagule inputs into the wetland is affected. For example, the dominance of different aquatic plant species may be related to dispersal constraints between wetlands (e.g., Flinn et al. 2010; Gledhill et al. 2008; Lopez et al. 2002), and the distance a propagule must travel to reach a newly constructed wetland will influence dispersal rates (van der Valk 1981). Second, landscape condition may influence aquatic plant diversity indirectly, if landscape composition or configuration affects local-habitat quality by influencing the flux of energy or materials into a wetland. Several authors have attributed correlations between landscape condition and wetland vegetation to this mechanism (e.g., Lougheed et al. 2001). Reclamation and restoration activities usually involve manipulating the local environment in order to achieve some biological target, such as increased diversity. If the primary influences on aquatic plant diversity act at the landscape-level, however, we may fail to meet these targets, despite our best efforts at local-level reclamation.

Typically, the relative importance of local- and landscape-level factors is evaluated by competing models containing only measurements made at the local-level with those including both local- and landscape-level variables or by variance partitioning

(Johnson and Host 2010 for review). These approaches allow the contrast of independent and combined effects of local- and landscape-level variables; however, they are generally incapable of evaluating the indirect effects of landscape condition, which are attributed instead to local-level variables. E.g., high intensity agriculture may result in elevated nutrient inputs that influence the plant community, but this indirect effect of agriculture will be attributed to water chemistry.

An alternative approach is to use structural equation modeling (SEM), which permits the quantification of both direct and indirect effects. SEM combines attributes of path analysis and confirmatory factor analysis (Arhonditsis et al. 2006), making it well suited to disentangling complex multivariate relationships (Grace et al. 2010). The path analysis aspects allow us to quantify direct and indirect multivariate relationships using structural equations. The factor analysis aspects allow us to incorporate concepts such as local and landscape condition explicitly through latent and composite variables. Latent variables reflect unmeasured causal variables. They exert an influence that is detectable in highly correlated indicator variables. Wetland trophic status is an example of a latent variable: it is a concept that lacks a singular measure, but can be quantified using numerous redundant measurements of nutrient levels and plant biomass. In contrast, composite variables represent the collective effects of a group of variables that do not necessarily covary (Grace and Bollen 2008). Reviews of the application of SEM in ecology are available (see Arhonditsis et al. 2006; Grace et al. 2010).

To ensure that efforts to create diverse aquatic plant communities are directed appropriately, we require the answers to the following three questions: 1) what is the relative importance of local- and landscape-level habitat condition in determining aquatic plant diversity in reference and reclamation wetlands? 2) If landscape condition is influential, are direct or indirect effects of greater importance? 3) Is there an optimal spatial scale at which we should consider landscape-level variables?

Methods

To address these research questions I developed and tested a conceptual model relating landscape- and local-level habitat conditions to submersed and floating vegetation diversity (Fig. 5-1). The model is a tool that can be used to test predictions about the relative importance of local- and landscape-level habitat variables, and the relative importance of direct and indirect landscape-level effects through the process of confirmatory modeling (Grace 2006). To evaluate the effects of spatial scale, I adopted a multi-model approach and ran the conceptual model five times, using data on landscape composition extracted from a different buffer width each time. This enabled me to evaluate at which spatial scale model-data agreement was best.

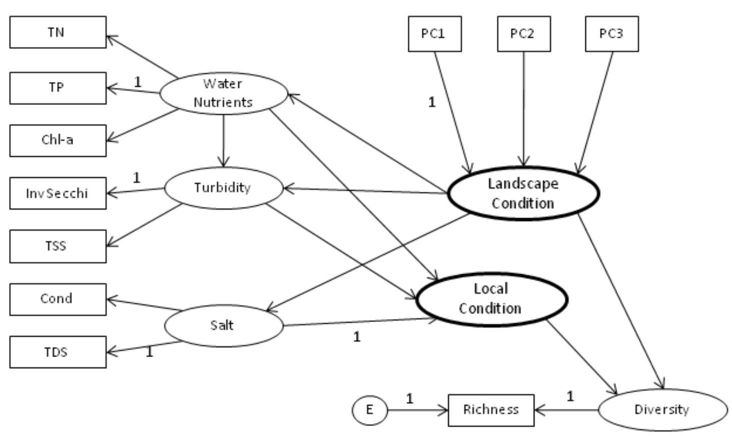


Fig. 5-1. Schema of the model showing latent and composite (bold) variables as ovals and measured variables as squares. One-way arrows imply a directional relationship. Pathways marked with the number 1 were fixed to make the model identifiable. N.B. that errors (E) are associated with all measured variables, but to simplify the schema, E is only depicted for Richness, where the measurement error was specified as 0.17 based on triplicate measures of floating and submersed vegetation richness made at 12 wetlands.

In the conceptual model, local-level habitat conditions are called Local Condition, which is a composite defined as the product of three latent variables: Water Nutrients, Turbidity, and Salt. Water Nutrients are indicated by the concentration of chlorophyll-a (chl-a), total nitrogen (TN), and total phosphorus (TP). Turbidity is indicated by the ratio of Secchi depth to total depth and the concentration of total suspended solids (TSS). Salt is indicated by conductivity and the concentration of total dissolved solids (TDS). Turbidity is also affected by the variable Water Nutrients, as algal growth in response to increased nutrients should reduce light penetration. Landscape Condition is a composite variable defined as the product of three orthogonal variables derived from a principal components analysis of land-cover data collected from each of five spatial scales (described below). Previous work demonstrated that wetland isolation has no detectable relationship to the richness of aquatic plants (Rooney unpublished), and this finding supports the conclusion of previous studies that landscape composition is of greater importance than configuration (e.g., Fahrig 1997). Variables such as soil type or topography were not considered explicitly, but are assumed to be rolled into land cover. For example, the wetlands we sampled were typically situated in flat landscapes; the exception being those positioned adjacent to mine pits or tailings piles, which can present sharp elevation gradients. However, as in any form of modeling, variation due to variables not included in the model will be attributed to correlates that are included in the model. Therefore, such changes in topography would be captured by an analysis of the coverage of mine-related land-cover types that includes the presence of tailings or mine pits.

Site sampling

To test the model, I used data collected from 45 shallow open-water wetlands situated in Alberta's boreal natural region. I sampled both reclamation wetlands situated on oil sands mining leases near Fort McMurray, Alberta (n = 20), as well as reference wetlands located in parks or other protected habitat and representing the reference condition (sensu Stoddard et al. 2006) (n = 25). The wetlands spanned a wide range in open-water area (0.06 to 19.03 ha), nutrient levels (TN: 0.85 – 14.30 mg/L; TP: 0.01 - 1.97 mg/L), salinity (TDS: 0.1 – 3.5 ppt), and turbidity (TSS: 0.025 – 920 mg/L). Reclamation wetlands ranged from 3 to 35 years in age.

Each wetland was sampled in late July or early August of 2007 or 2008, when peak biomass was expected. To sample vegetation, I crossed each wetland 10 times in a kayak. At a random location along each transect, I used a rake to sample the submersed and floating plants within a 1 m^2 quadrat. I also recorded the presence of submersed and floating species observed along the 10 transects but not captured in a quadrat. Aquatic plants were identified to the lowest practical taxonomic level following Moss and Packer (1983) with names updated according to the Integrated Taxonomic Information System Database (ITIS. 2010). Taxa that I could not identify to species included an aquatic moss, the macroalgae *Chara* spp. and two species of *Myriophyllum* that could not be differentiated from one another in the field: *M*. *sibiricum* and *M*. *verticillatum*. Taxa richness was the sum of all floating and submersed taxa observed at a given wetland. Using only the data collected from within quadrats, I also calculated Shannon's diversity transformed according to Jost (2006). Voucher specimens were collected and are housed at the ALTA Vascular Plant Herbarium at the University of Alberta.

In terms of in-lake variables, I recorded water depth and Secchi depth within each quadrat and calculated the average Secchi depth as a proportion of water depth to measure turbidity. I kayaked to the deepest point in each wetland where I used a YSI Model 556 multi-probe to measure conductivity. At the same location I collected water for TN, TP, chl-a, TSS, and TDS with an integrated-depth water sampler. I filtered sub-samples of known volume through Whatman GF/F glass microfiber filters with 0.45 µm pores in order to measure TSS and chl-a. Analyses were carried out following standard methods detailed in Rooney and Bayley (2010)

Landscape data

I obtained recent (<5 yrs) aerial photographs or Quickbird Pansharpened 3-Band satellite imagery of each of the 45 wetlands. I imported all images into a digital file for measurement using GIS software (ESRI 2009) such that the final imagery had pixels ≤3 m in resolution. Those images missing spatial reference data were geo-referenced to existing digitized topographical maps from the Canadian National Topographic Database. I identified 13 major land-cover types within a 2 km buffer surrounding the open water at each wetland: lentic water, lotic water, wetland, forest, open-green (lawn), agriculture (row crop and pasture), roads (paved and unpaved), residential, industrial-commercial, oil and gas, tailings water, solid tailings, and bare ground. I created a shapefile of polygons representing the 13 land-cover types by digitizing at a visual scale between 1:1000 and 1:2500 with ESRI® ArcMap™ 9.3.1 software (ESRI 2009). I created a series of nested buffers of different widths surrounding each open-water zone: 300 m, 500 m, 1000 m, 1500 m, and 2000 m, to yield 5 landscape sizes for each wetland. I then extracted the percent cover of each land-cover type within each of the five buffer widths. To the best of my knowledge, no previous research has assessed the relationship between wetland plants and surrounding land-cover in northern Alberta; however, research has been done on this subject in other regions (e.g., Akasaka et al. 2010; Houlahan et al. 2006; Matthews et al. 2009a) and I selected buffer sizes to cover the range over which land-cover was found to be correlated with wetland plant communities in previous studies. I selected a span that would include wider buffers than are typically correlated with wetland plants because prior research has demonstrated that the spatial scales over which land-cover is correlated with in-lake variables can be larger than the scale at which it is correlated with wetland plants (e.g., Houlahan and Findlay 2004; Houlahan et al. 2006), and I wanted to ensure that I included a sufficient range in spatial scale to detect such differences if they occurred in my system.

I carried out a principal components analysis on arcsine square-root transformed percent cover data for the 45 wetlands, separately for each buffer size, using PC-ORD 4.0 (McCune and Mefford 1999). These ordinations reduced the 13 land-cover types into three orthogonal synthetic axes that cumulatively explained between 55 and 75% of the variance in the cross-products correlation matrix, depending on landscape size. For all five landscape sizes, the first ordination axes (PC1) explained between 30 and 45% of the variance in land-cover, and reflected an increase in land-covers associated with oil sands mining (bare ground, paved and gravel roads, industrial land, and the presence of liquid and solid oil sands mine tailings) and a reduction in natural land-covers (wetland, lentic water, lotic water). I therefore interpreted PC1 to be a reasonable measure of the amount of mine-related disturbance within the landscape. The second axis explained between 15.0 and 15.4% of the total variance in land-cover for all 5 buffer widths, and was correlated with an increase in residential land-cover, and often also an increase in agricultural land-cover. The third axis explained between 9.4 and 13.7% of the total variance in land-cover data. It was consistently correlated with a decrease in residential cover and an increase in the coverage and density of other oil and gas development (i.e., well pads, seismic lines, and pump stations). I examined the land-cover correlates of higher order PC axes for trends that might have bearing on aquatic plant Diversity, but found none.

Modeling

The model was tested using landscape-level data extracted from each of the five spatial scales: 300 m, 500 m, 1000 m, 1500 m, and 2000 m. In order for the model to be identifiable, it was necessary to constrain some variables (Grace et al. 2010). To set the scale of each composite variable, one of its incoming paths was fixed to equal one. Similarly, to set the scale of every latent variable, a path coefficient associated with one of its indicators was fixed to equal one. Although the choice of which variable to fix influences unstandardized path coefficients, it has no effect on standardized path coefficients (Grace 2006), and it is the standardized coefficients that are used to compare the relative magnitude of different pathways in the model. Standardized coefficients measure the slope of the relationship in units of standard deviations. Using standardized coefficients enables me to make comparisons, even though my variables are in different units. For latent variables with ≤2 indicators, I constrained their error variances to be equal (Grace 2006). For Diversity, with its single indicator, this would have meant that it was perfectly indicated by Richness with an error variance of zero. However, I knew there would be error associated with my measurement of Richness, and that this could bias model path coefficients (Grace 2006). I therefore incorporated a measure of reliability derived from triplicate measurements of SAV richness taken at 12 wetlands (Rooney unpublished). I calculated the average correlation among the three measurements (r = 0.83) to determine measurement reliability, and therefore used E = 0.17 to specify the error for Richness (Fig. 5-1). I ran parallel analyses using Shannon's diversity in place of Richness, but reached the same conclusions regarding local- and landscape-level effects and regarding direct and indirect landscape effects. For brevity, only the results obtained using Richness are presented.

I estimated parameters using maximum likelihood procedures and the program AMOS 17.0 (2009). It was therefore necessary to transform the indicator variables to meet the distributional assumption of multivariate normality (Table 5-1). In addition, I had to multiply the Secchi depth indicator of Turbidity by -1 so that larger values would correspond with greater Turbidity (Table 5-1).

Table 5-1. The 11 indicator variables investigated in the model and the transformations applied to achieve normal distributions.

Code	Definition	Transformation
TP	Total phosphorus (µg/L)	Log
TN	Total nitrogen (µg/L)	Log
Chl-a	Chlorophyll-a (µg/L)	Log
TSS	Total suspended solids (mg/L)	Log
Inverse Secchi	-1 times average of Secchi depth divided by maximum depth of each quadrat	Arc-sine square root
TDS	Total dissolved solids (ppt)	Log
Conductivity	Conductivity (µS/cm)	Log
PC1, PC2, and PC3	Principal components scores on axes 1, 2, and 3 for landscape composition ordination	None
Rich	Number of floating and submersed plant taxa observed	Square root

I performed chi-square tests to evaluate the model-data agreement for each spatial scale, with the null hypothesis that the data provides an adequate fit to the model. To compare model fit among the spatial scales, I calculated the root mean square error of approximation (RMSEA) and Akaike's Information Criterion (AIC) metrics. Where models yielded adequate model-data fit, I looked at the proportion of variance in aquatic plant Diversity explained by each model (R^2 values).

Results

The number of taxa inventoried within the entire suite of wetlands (n = 45) was 25. Richness of individual wetlands ranged from 1 to 12 (mean 4 ± 2.66 SD). All species observed are native to the region (Moss and Packer 1983).

Model fit

The data fit the hypothesized model for all five spatial scales, explaining between 50 and 64% of the variance in aquatic plant diversity among wetlands (Table 5-2), but as landscape size increases, both the relative model fit and the percentage of variance in Diversity explained by the model decrease (Table 5-2). The notable exception to this trend is the 1000 m scale, which had the best model-data agreement, but explained the least variance in Diversity.

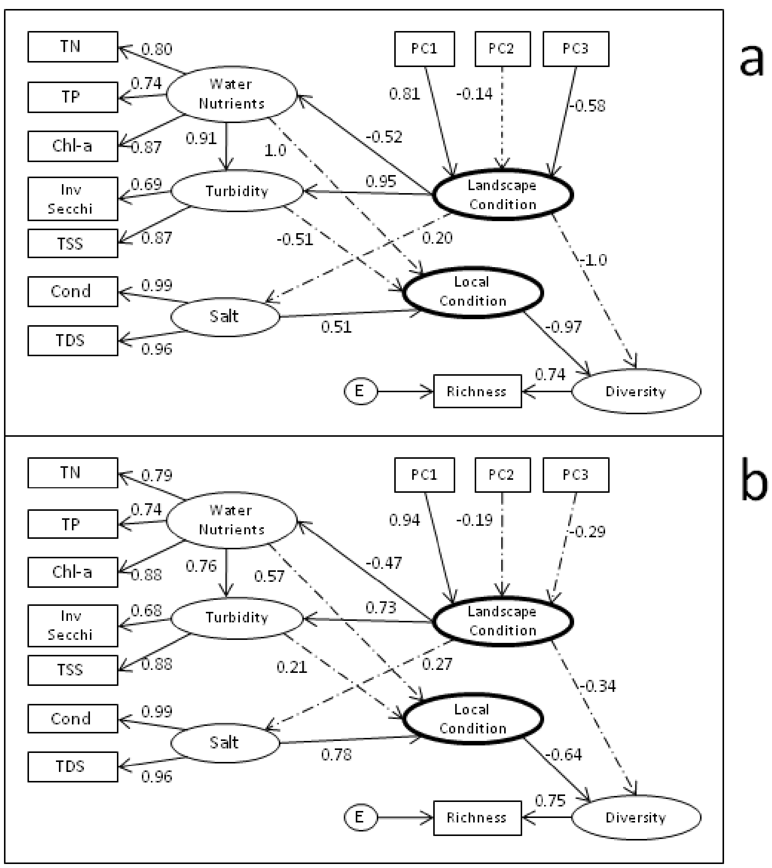
Table 5-2. Goodness-of-fit test results and r^2 values for the five models using Richness as the indicator of Diversity.

Model scale	Chi-square (d.f., *p*-value)	RMSEA (*p*-value)	AIC	r^2 Diversity
300 m	43.03 (40, 0.343)	0.041 (0.527)	95.026	0.639
500 m	44.05 (40, 0.304)	0.048 (0.485)	96.050	0.531
1000 m	41.37 (40, 0.411)	0.028 (0.595)	93.373	0.502
1500 m	54.36 (40, 0.065)	0.090 (0.155)	106.355	0.539
2000 m	55.33 (40, 0.054)	0.093 (0.135)	107.33	0.543

I explored modification indices and residual error matrices to see if model fit could be significantly improved; however, such modifications run the risk of over-fitting the data. In confirmatory model testing (as opposed to exploratory modeling) any modifications must be theoretically sound and any modified model must be tested using independent data (Grace 2006). The suggested modifications made relatively small improvements to model-data agreement and were not generally supported by theory. For example, at 2000 m, modification index values suggest a significant improvement in model fit could be achieved by creating a pathway from the variable Conductivity to the variable TN, but there is no theoretical ground on which to base such a pathway. The modification indices suggested no biologically plausible changes to the model, and thus I do not recommend that paths be added to the model to improve model-data agreement.

Local Condition vs. Landscape Condition

I estimated standardized coefficient values for the models using data from the five spatial scales (Fig. 5-2 a-e).



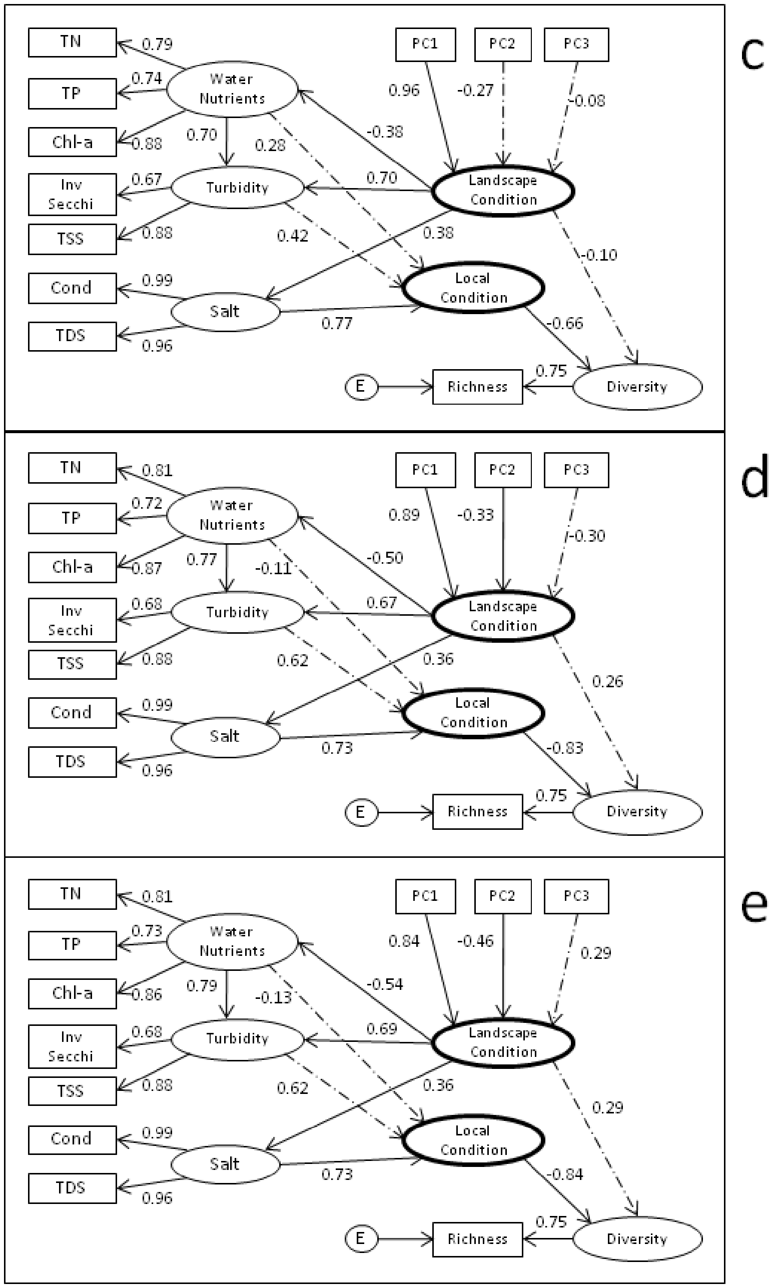


Fig. 5-2. Final modeling results for the five different spatial scales: 300 m (a); 500 m (b); 1000 m (c); 1500 m (d); and 2000 m (e). Results are based on the analysis of 45 wetlands, including reclamation wetlands situated on oil sands mining leases. Values associated with the arrows are standardized path coefficients and indicate the relative magnitude of effects. All five models had adequate model-data agreement. Pathways that are non-significant at α = 0.05 are indicated using dashed lines, whereas solid lines indicate significant paths. Net and indirect effects of landscape condition are reported in Table 5-3.

The net effect of Local Condition on Diversity is due to direct effects, whereas the net effect of Landscape Condition is the sum of direct and indirect paths (Table 5-3). Local Condition is always more important than Landscape Condition, and its influence varies slightly among spatial scales; from -0.64 to -0.96 (Table 5-3). In contrast, the net influence of Landscape Condition is always smaller in absolute magnitude and diminishes as the size of the landscape increases, from -0.37 to -0.13 (Table 5-3).

Table 5-3. Standardized coefficients representing the direct, indirect, and net effects of landscape condition and local condition on Diversity at the five spatial scales using Richness as the indicator of Diversity.

Spatial scale	Net Local	Net Landscape	Direct Landscape	Indirect Landscape
300 m	-0.97	-0.37	-1.03	0.66
500 m	-0.64	-0.35	-0.34	-0.01
1000 m	-0.66	-0.35	-0.10	-0.25
1500 m	-0.83	-0.15	0.26	-0.41
2000 m	-0.84	-0.13	0.29	-0.42

Direct vs. indirect effects

The net influence of Landscape Condition on Diversity can be parsed into its direct and indirect components (Table 5-3), but the relative importance of direct and indirect effects varies with spatial scale. At 300 and 500 m, the direct pathway coefficient is of greater magnitude, whereas at 1000, 1500, and 2000 m, indirect effects coefficients exceed direct effects coefficients in absolute value (Fig. 5-3).

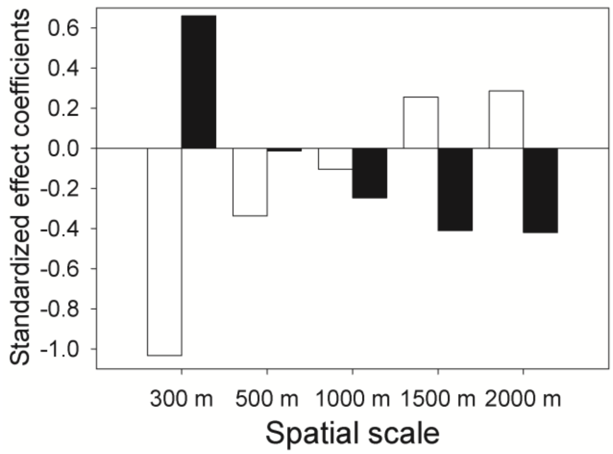


Fig. 5-3. Standardized path coefficients denoting the relative importance of landscape condition broken into its direct (white) and indirect (black) components for the five spatial scales.

Structural model

In terms of the structural model, not all pathways are statistically significant at $\alpha = 0.05$ (Fig. 5-2 a-e), but in SEM it is not required practice to eliminate non-significant paths from a model yielding adequate model-data agreement (Grace 2006). Despite the increase in degrees of freedom, eliminating paths with p-values > 0.05 results in a reduction in model fit and in the proportion of variance in Diversity that the model explains at all five spatial scales. I therefore retained all paths present in the initial model, regardless of their associated *p*-values.

Landscape Condition is primarily the result of PC1 scores, which mainly represented an increase in the amount of mine-related disturbance in the landscape. Local Condition was mainly a product of salinity, and it is Local Condition that primarily determines Diversity. The direct effect of Landscape Condition on Diversity, and the influence of Water Nutrients and Turbidity on Local Condition are consistently non-significant (Fig. 5-2 a-e). The effect of Landscape Condition on Salt only becomes significant at spatial scales ≥1000 m, and is associated with the increase in the loading from PC2 onto Landscape Condition, which is only significant at spatial scales >1000 m (Fig. 5-2 a-e). Recall that PC2 represents an increase in the amount of residential and agricultural land-cover. In contrast, the loading of PC3 on Landscape Condition diminishes with increasing landscape size, and is only significant at 300 m. Recall that PC3 represents an increase in the amount of non-mining oil and gas development in the landscape.

Discussion

The results demonstrate the complexity of the relationships between local- and landscape-level factors and aquatic plant diversity. I make three important contributions. First, I demonstrate that although Local Condition is consistently of greater importance in determining the diversity of submersed and floating plants in these wetlands, Landscape Condition also plays a role; albeit one that diminishes with distance. Second, I highlight the importance of considering indirect effects when contrasting the relative importance of local- and landscape-level variables, especially where direct and indirect effects counteract each other (i.e., they exhibit suppression). Third, I emphasize the need to consider spatial scale explicitly in this type of analysis by demonstrating how the direct, indirect, and net effects of Landscape Condition are scale dependent.

Local Condition vs. Landscape Condition

My first objective was to evaluate the relative influence of local- and landscape-level habitat conditions on aquatic plant diversity. This is a topic of great interest in ecology (Johnson and Host 2010) although there is little consensus on the subject. In some cases the surrounding land-use appears to control wetland plant diversity (e.g., Lougheed et al. 2001), in other cases the local environment is more influential (e.g., Wright et al. 2003), and in yet other studies local- and landscape-level factors were found to play an equal role (e.g., Capers et al. 2010; Matthews et al. 2009a). The relative importance of local- and landscape-level factors may even depend on the dominant land-cover type. For example, Herault and Thoem (2009), studying wetlands in western Europe, found that landscape factors became more important in open landscapes whereas local variables were more important in forested landscapes. Seemingly, the relative role played by local- and landscape-level conditions is system specific.

In these wetlands local-level habitat conditions were of greater importance in determining Diversity at all spatial scales considered. I suspect that the net effect of Landscape Condition was smaller in these wetlands partly because of the relatively large contribution of ground water to their hydrologic budget and the complexity of surface-ground water interactions (Devito et al. 2005). In contrast, where run-off inputs dominate, chemicals and materials are carried into the wetland from the surrounding landscape, connecting Local Condition to Landscape Condition (e.g., Crosbie and Chow-Fraser 1999; Houlahan and Findlay 2004; Lougheed et al. 2001).

Regardless, Landscape Condition had a net negative influence on aquatic plant Diversity at all five spatial scales. In other words, aquatic plant Diversity was lower in landscapes with more mine-related disturbance, greater coverage of residential land cover, or greater coverage or density of non-mining oil and gas development. My conclusion that Diversity was lower in landscapes with more mine-related disturbance (and thus less wetland cover) echoes work on ponds in the U.K. that revealed a positive association between the abundance of pond habitat and plant species richness (Gledhill et al. 2008). Thus, while reclamation focused solely on in-lake conditions may achieve some success, the probability of creating a diverse plant community is increased when surrounding land use is considered.

Direct vs. indirect effects of Landscape Condition on Diversity

The answer to my second question (are direct or indirect effects of greater importance?) depends on spatial scale. In landscapes ≤500 m, the absolute magnitude of direct effects of Landscape Condition on Diversity exceeds the absolute magnitude of indirect effects, but in spatial scales ≥1000 m, the situation is reversed and indirect effects dominate.

This suggests that in smaller landscapes, the primary mechanism by which land-cover affects aquatic plant Diversity is dispersal limitation. Although different species may be differentially subject to dispersal limitation, depending on their mode of dispersal (Flinn et al. 2010), previous work comparing reference and oil sands reclamation wetlands did not find a difference in the dominance of different plant dispersal modes (Trites and Bayley 2009b). Capers et al. (2010) found that aquatic plants, as predominantly passive dispersers, are constrained by dispersal limitation, despite the production of turions and the ability to reproduce vegetatively. My results support this conclusion. Wetlands in landscapes where wetland habitat is scarce will experience reduced colonization as fewer propagules are being released within the region (Houlahan et al. 2006). Such wetlands may also be subject to increased occurrence of genetic bottlenecks and inbreeding (Young et al. 1996) resulting in increased extinction risk (Kery et al. 2000). Reduced colonization coupled with increased local-extinction risk is likely responsible for the direct negative influence of Landscape Condition on Diversity at landscapes ≤500 m.

At larger spatial scales, the dispersal effect is diluted and the indirect influence of land-cover on Diversity begins to dominate. The indirect effect of Landscape Condition is attributed to the transport of materials (salts, suspended sediment, and nutrients) from the surrounding landscape into a wetland by surface runoff, ground water transport (seepage), or wind-driven transport. Likely, the complexity of surface-groundwater interactions in the region (Devito et al. 2005) are responsible for the large spatial scales over which this transport is conducted.

The dominance of dispersal-related land-cover effects at smaller spatial scales is mirrored in results of other studies exploring the relationship between land use and wetland plant diversity. For example, looking at 58 Ontario wetlands, Houlahan et al. (2006) found that plant species richness was most strongly correlated with land-cover within 250-400 m of the wetland and that this correlation diminished steadily with increasing landscape size, even though the relationship between land-cover and local-level habitat variables was strongest at spatial scales between 2000-3000 m (Houlahan and Findlay 2004). In light of my results, it may be that Houlahan et al. (2006) were detecting the direct, dispersal-related effects of Landscape Condition operating at small spatial scales, but that their approach failed to identify the indirect effects of Landscape Condition, which operated at larger spatial scales.

As noted in the Methods section, complications in model interpretation can arise if an influential variable is excluded from the model. I see two ways in which this could result in an overestimation of the direct, dispersal-related effects of Landscape Condition on aquatic plant Diversity. First, any indirect influence of Landscape Condition on aquatic plant Diversity that is mediated by a local-level variable not included in the model will be considered a direct effect by my modeling approach. Second, if I excluded an in-lake variable from the model that both affects aquatic plant Diversity and differs between reclamation and reference wetlands, its effects would be confounded with those of Landscape Condition. I am confident, however, that I did not exclude relevant local-level variables from the conceptual model, as I based their selection on a review of pertinent literature and a prior analysis of 52 physical and chemical variables that might differ between reference and reclamation wetlands (Rooney and Bayley 2010), and included all those correlated with aquatic plant Diversity. Given that the dispersal-related effects of Landscape Condition were minor except at small spatial scales, any overestimation of its effects would be due to the exclusion of an in-lake variable that is only correlated with Landscape Condition at small spatial scales. Future research could reveal that I have overestimated the direct effects of Landscape Condition by missing an important in-lake variable, but given the agreement between my results and other published studies and my extensive initial list of in-lake variables, I consider this unlikely.

One very important finding is that whereas the direct effects of Landscape Condition on Diversity go from negative in small landscapes to positive in larger landscapes, indirect effects go from positive to negative (Fig. 5-3). In other words, as the buffer width increases, increased coverage of mine-related, residential, and oil and gas-related land-covers goes from an association with dispersal limitation to an association with mild dispersal facilitation. In contrast, the influence of these land-covers as mediated through their effects on Local Condition goes from increasing aquatic plant Diversity to its reduction. Thus, at nearly every spatial scale we see suppression with the direct and indirect effects counteracting each other, such that the net effect of Landscape Condition is much smaller than the sum of the absolute direct and indirect effects. This has grave implications for alternative methods of comparing local- and landscape-level influences on diversity. If only the direct effects of Landscape Condition are considered, then its influence at small spatial scales would be vastly overestimated. At 300 m, Landscape Condition would appear to be as influential as Local Condition, an overestimate of 270%. At 1000 m, the net effect of Landscape Condition would be underestimated by about 70%, and at larger scales, the net effect would not only be overestimated by 170-220%, but would be assumed to be positive when, in fact, it is negative. Clearly, the failure to consider indirect effects can lead to erroneous conclusions about the role of Landscape Condition.

Spatial scale

My final aim was to identify whether there was some optimal landscape size at which to consider Landscape Condition. The importance of spatial scale in ecological studies has been noted for more than two decades (e.g., Levin 1992), although multiple spatial scales are seldom evaluated unless spatial scale is explicitly the object of study. Clearly, based on the answer to our second question, the optimum landscape size will depend in part on the study's objectives. The net effect of Landscape Condition diminishes with distance and the capacity to explain variance in Diversity was greatest when I considered landscape composition within 300 m. At this spatial scale, the net effect of Landscape Condition appears to be mostly due to its direct influence on Diversity. Other studies that examined the correlation between wetland plant diversity and land-cover at multiple spatial scales also find the strongest relationships within 300-500 m (e.g., Akasaka et al. 2010; Galatowitsch et al. 2000; Houlahan et al. 2006). Therefore, if it is the net influence of landscape condition that is of concern, such as in any conservation or restoration planning context, or if it is the direct (dispersal-related) effect of Landscape Condition that is of concern, such as in studies of metapopulations or the spread of invasive species, then 300 m will yield the strongest predictions of aquatic plant Diversity.

The indirect or mediated effects of Landscape Condition, however, dominated direct effects in landscapes ≥1000 m, meaning that at larger spatial scales the net effect of landscape composition is primarily the result of its effects on in-lake salinity, turbidity, or nutrient levels. Thus, if study objectives are to detect a net effect of land use mediated by land use's influence on local variables, e.g., in studies attempting to detect the signal of land use on local habitat variables that are important to plant communities (e.g., Crosbie and Chow-Fraser 1999; Houlahan and Findlay 2004), the optimal landscape size might be 1500 to 2000 m.

Implications for practice

This investigation was driven by the concern that the effects of Landscape Condition may outweigh the influence of Local Condition on aquatic plant Diversity, resulting in the failure of reclamation wetlands to achieve appropriately diverse plant communities even where best practices centred on in-lake conditions are adopted. Generally, my modeling results allay this concern and suggest that actions taken at the local-level are likely to enhance plant diversity regardless of the wetland's context. However, Landscape Condition did exert an influence on Diversity, and given the difficulty of constructing well designed wetlands and the uncertainty of success with wetland creation projects (Turner et al. 2001), it is necessary to implement all possible recommendations. Aquatic plant diversity may be further enhanced by planning at the landscape-level to construct reclamation landscapes dominated by natural land-covers or with natural buffers surrounding constructed wetlands. However, even 500 m wide buffers will do little to protect constructed wetlands from the indirect effects of

Landscape Condition. Where protective buffers are not feasible, planting and seeding may mitigate the direct influence of disturbed land-covers on aquatic plant Diversity by compensating for reduced dispersal. To mitigate the indirect influence of disturbed land-covers will require actions that reduce the transport of salt, nutrients, and sediments into wetland basins from distances as great as 2000 m.

This work represents a step towards determining why, despite substantial overlap in relevant in-lake conditions, the community composition of aquatic plants in reclamation wetlands differs from that of reference wetlands. We determined that although the composition of the surrounding landscape has a detectable relationship with aquatic plant diversity, it is slight compared to the effects of in-lake variables like turbidity, salinity, and nutrient levels. Future research should involve an examination of the role of variables not included in our model, such as the relative contribution of surface- vs. ground-water to the hydrologic budget.

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6. General discussion and conclusions

Oil sands mines are being developed at an unprecedented pace and scale and large sections of boreal wetland habitat are being destroyed. An estimated 0.33 to 0.63 m^2 of boreal Alberta are destroyed for every 1 m^3 of synthetic crude oil produced by mining (Jordaan et al. 2009). Already 67,330 ha have been disturbed by oil sands mining (Richens pers. comm.), and another 99,714 ha are approved to be disturbed (Keeler pers. comm.). With 99% of the 475,000 ha surface mineable area already leased by mining companies (GOA 2010a), the spatial footprint of oil sands mining is sure to grow larger.

All this land must eventually be reclaimed (GOA 1993), including the wetland habitat slated for creation in mine Reclamation and Closure plans (CNRL 2006; Imperial 2006; Shell 2007; Suncor 2010b; Syncrude 2006a). The Government of Alberta will need to assess reclaimed land to determine if it should be certified and whether the companies should be released from their reclamation liabilities. With the definition of reclamation success shifting towards a restoration approach (Poscente 2009), this will require scientifically sound standards and formally adopted criteria capable of evaluating the biological integrity of reclaimed lands.

This thesis constitutes some of the first efforts to assess the wetlands being built on oil sands leaseholds to meet reclamation obligations by comparing them to appropriate natural analogues situated in the same ecoregion. Such an assessment was urgently needed to inform the reclamation certification process and to provide suitable goals and recommendations for future wetland reclamation projects.

In Chapter 1 of this thesis, I set the context for wetland reclamation in the oil sands area. I described the mining and reclamation process, and the regulatory environment in which it takes place. Predominantly, I outlined the need for tools that can assess reclaimed wetlands, for appropriate wetland reclamation targets, and for guidance regarding how wetland reclamation practices might be improved. The remainder of my thesis goes on to meet these needs.

The physical and chemical condition of reclamation wetlands was the subject of Chapter 2. In this chapter, I identified the abiotic variables that were able to explain the most variance among the sampled wetlands using an objective approach that minimized reliance on best professional judgement. I then used these variables to develop a stress gradient that could rank wetlands based on the degree to which they are subjected to environmental stress. To do this, I evaluated six combinations of variable standardization and weighting approaches. I found that the weighting method had little impact on final stress score and merely served to reduce the effect of redundancy among the selected variables. In contrast, the standardization method had an important effect on the resolution of the final stress gradient. When variables were standardized following the percentile binning approach, they were much better able to resolve differences among wetlands subject to moderate amounts of stress, whereas when standardized by the Z-scoring method, most sites were lumped into the middle range of stress scores. The stress gradient tool can be used to evaluate future reclamation wetlands on its own merits, but it can also serve as the independent and objective ranking system required to develop bioassessment methods like the index of biological integrity (IBI). Objectivity is critical in wetland assessment where the conclusions must be both scientifically defensible and interpretable to decision makers. Thus, minimizing the role of best professional judgement was a significant improvement over established methods.

A community analysis of aquatic plants in reclamation and reference wetlands was the subject of Chapter 3. I carried out cluster analysis to identify community assemblages, and used indicator species analysis to determine what species were both faithful and exclusive to each assemblage. These species could be used to determine assemblage membership. I then used ordination to position the sampled wetlands in species-space and to identify environmental correlates of the various assemblages. My community analyses revealed that reclamation wetlands supported aquatic vegetation with significantly different community composition than reference wetlands, and that the assemblage growing in a wetland depended on whether it was a reference wetland, a reclamation wetland free of tailings, or a tailings-contaminated wetland. The differences in community composition were stark, with nearly three quarters of all reference wetlands possessing one of two community assemblages, neither of which appeared in a single reclamation wetland. Thus, I conclude that reclamation wetlands are impaired relative to reference ones. Furthermore, I recommend that oil sands reclamation practitioners adopt at least two community-based targets: practitioners should strive to create aquatic plant communities either dominated by high productivity stands of *Ceratophyllum demersum*, or by lower productivity stands of *Utricularia macrorhiza* and aquatic moss. In practical terms, this might require increased sediment nutrients and organic matter content and the reduction of basin slopes, total dissolved solids and sodium concentrations to establish environmental conditions that were correlated with the desired plant assemblages.

One aspect of ecological health that my approach does not address explicitly is the natural temporal variability of reference wetlands. Natural wetlands are highly dynamic systems, especially with regards to flood-drawdown cycles (van der Valk 2005). We sampled multiple reference wetlands (n = 38) and spread our sampling over three consecutive years in an attempt to capture some of the natural variability among wetlands. Thus, natural variability will to some degree be incorporated in our characterization of the reference condition. The importance of these natural cycles, however, is not made obvious in our selection of two reference states represented by the *C. demersum* and *U. macrorhiza* assemblages. While reclamation practitioners should aim for one of these two states, some acknowledgement that natural wetlands go through successional stages and are dynamic ecosystems is required. Truly successful wetland creation should produce a wetland that resembles reference wetlands in terms of community structure and environmental conditions, but that is also resilient in the face of normal periodic stress and capable of adapting to changes in environmental conditions (SER 2004).

Having identified that reclamation wetlands do not resemble reference wetlands in either their physical and chemical conditions or their aquatic plant community composition, in Chapter 4 I developed and tested an index of biological integrity (IBI). I then used this IBI to evaluate 25 reclamation wetlands in Alberta's oil sands mining region. I began with 60 different candidate biological metrics representing the submersed and floating aquatic plant communities, but found that 5 were all that were needed to quantify biological integrity: two diversity-based metrics - species richness of floating vegetation and percent of total richness contributed by *Potamogeton* spp.; and three metrics based the relative abundance - of *Ceratophyllum demersum*, floating leafed species, and alkali-tolerant species. I evaluated the contribution of non-linear metrics to IBI performance, but concluded that the correlation between IBI scores and wetland condition was not improved and so excluded them from the final IBI. I tested two different scoring approaches, and concluded that continuous scoring was superior to discrete scoring. I also evaluated two approaches to relativizing scores, and concluded that scoring relative to the range of values observed among reference wetlands was preferable on theoretical grounds. Using the resulting IBI, I confirmed what was apparent in Chapter 3, tailings-contaminated and tailings-free reclamation wetlands have significantly lower average biological integrity than reference wetlands (ANOVA: $F_{2,59} = 34.7$, $p < 0.00001$).

This raised an interesting question: given that the range of in-lake environmental conditions known to be important to aquatic plants overlapped substantially between reclamation and reference wetlands, why were the communities within them so consistently different? As the subject of Chapter 5, I explored whether the surrounding landscape might hold the answer. I used structural equation modeling to test the hypothesis that landscape condition influenced the diversity of aquatic plants in my wetlands, either directly via dispersal limitations or indirectly through exerting influence on three in-lake (local) environmental variables known to influence aquatic plants: turbidity, salinity, and nutrient levels. I explored the relative strength of these relationships across multiple spatial scales (300-2000 m) in order to identify whether there existed an optimal spatial scale at which land cover should be considered in reference to aquatic plants. My model provided an adequate fit to the data at every spatial scale, but the results yielded more questions than answers. The relative strength of the relationship between landscape condition and aquatic plant diversity declined as the spatial scale of the landscape increased, and this change was accompanied by a trade-off between the direct and indirect pathways of landscape condition's influence on aquatic plant diversity. At ≤ 500 m direct landscape effects were of greater importance than indirect effects, whereas indirect effects of landscape condition became more important at ≥1500 m. Thus, I conclude that the dominant mechanism by which land cover influences aquatic plant diversity depends on the spatial extent of the landscape considered. From these results, I extrapolate that reclamation designs should incorporate abundant wetland habitat and that planting prescriptions should involve seeding/planting to foster diverse aquatic plant communities in reclamation wetlands. Because the influence of the landscape was strongest at the smallest spatial scale that I considered (300 m), I further concluded that the focus of reclamation efforts should remain at the site-level, as this is where efforts to produce diverse wetland plant communities will achieve the greatest effect.

Taken together, these chapters set appropriate goals for wetland reclamation in the oil sands area, including physical, chemical, and biological targets that are based on the reference condition approach to ecosystem assessment (sensu Stoddard et al. 2006). I produced two tools that can be used to evaluate reclamation progress as a part of an adaptive management strategy during reclamation and monitoring and as part of reclamation certification during mine closure: the stress gradient and the IBI. While these tools are not independent, they can be used together to yield a complete assessment of wetland conditions and biological integrity. Whereas the stress gradient can be diagnostic of the environmental causes of past biological change or anticipatory of impending biological change, the IBI provides an integrative assessment by quantifying current biological integrity. The modeling exercise provided valuable information about where to focus reclamation efforts to promote diverse aquatic plant communities and suggested strategies that might be employed (e.g., seeding/planting to overcome dispersal limitations) to counter some of the limitations imposed by the fragmented and disturbed landscape in which reclamation activities are carried out.

After having conducted this research, I conclude that while it may be impossible to restore the post-mining landscape to its historical condition, it should be possible to build shallow open water wetlands with self-sustaining vegetation communities that resemble appropriate reference communities. However, wetland reclamation practices to date have not achieved this objective and existing reclamation wetlands are not suitable for reclamation certification as they represent a major departure from commonly occurring natural wetlands in the ecoregion. Successful wetland reclamation will require the deliberate manipulation of local environmental conditions to control for sodium, chloride, toxicity, turbidity, and nutrient levels, and the careful design of reclamation landscapes to ensure abundant and well connected wetland habitat. However, if these considerations are made, it should be possible to produce functional wetlands that are well integrated with their surrounding boreal landscape.

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Appendix 1

I turned the raw data used in this thesis into a data paper to ensure that is adequately archived and that it and all relevant metadata are available for future studies. The citation is: Rooney, R.C., Bayley, S.E., and Raab, D. 2011. Plant community, environment, and land-use data from oil sands reclamation and reference wetlands, Alberta, 2007-2009. Ecology (data paper). It can be found at the following url: http://esapubs.org/archive/. The meta data for this paper follows.

Meta data

Class I. Dataset Descriptors

A. Data set identity:

Title: Plant community composition, environmental, and land-use data from oil sands reclamation and reference wetlands, AB Canada, 2007-2009.

B. Data set identification code:

Suggested Data Set Identity Code: OSIBI_2007-2009

C. Data set description

Principal Investigators:

Rebecca C. Rooney, Department of Biological Sciences, University of Alberta, Edmonton, Alberta, Canada.

Suzanne E. Bayley, Department of Biological Sciences, University of Alberta, Edmonton, Alberta, Canada

Class II. Research Origin Descriptors

A. Overall project description

Identity: Plant community composition, environmental, and land-use data from oil sands reclamation and reference wetlands, AB Canada, 2007-2009.

Originators:

R. C. Rooney, S. E. Bayley, D. Raab

Period of Study: Plant composition and environmental data from the open-water season (May to September) of 2007, 2008, and 2009. GIS data from aerial photos and Quickbird imagery taken between 2002 and 2008.

Objectives: To evaluate wetland reclamation success on oil sands mining leases using the reference condition approach with wetland plants as bioindicators of wetland condition.

Sources of funding: The National Sciences and Engineering Research Council of Canada, Alberta Ingenuity Grants now part of Alberta Innovates Technology Futures, the Killam Trusts, and the Cumulative Environmental Management Association. In kind support was provided by Suncor Energy Inc. and Syncrude Canada Ltd.

B. Specific subproject description

Study region:

The 74 wetlands sampled in this study are located within the Boreal Plains ecoregion of Alberta and Saskatchewan (Fig. A1-1).

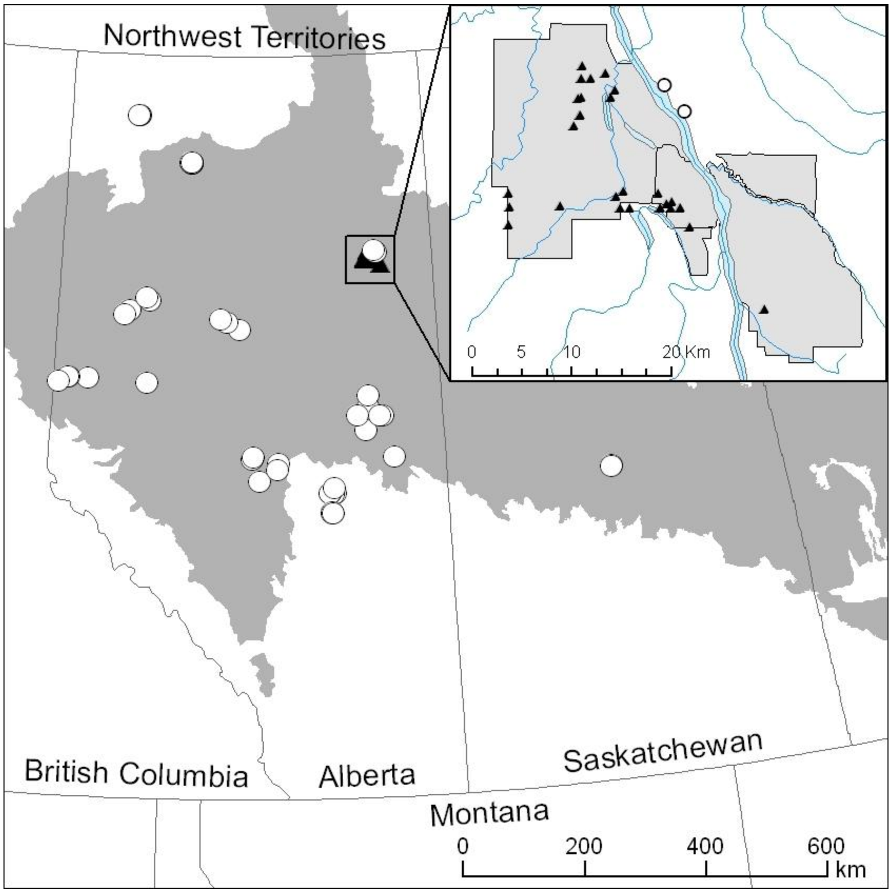

Fig. A1-1 Geographic distribution of the 74 sampled wetlands. White circles represent natural wetlands and are spread across the Boreal Plains ecoregion (grey area). Black triangles represent reclamation wetlands and are clustered in the oil sands surface mineable area. Not all sites are visible because of overlapping symbols. Note that the southernmost cluster of natural wetlands is situated in a patch of boreal habitat in Elk Island National Park. Inset depicts location of sites on or adjacent to oil sands company leaseholds (grey area).

Experimental design:

The 74 wetlands belonged to one of four types. There were 13 oil sands process affected (OSPA) wetlands. They are considered oil sands process affected as they were known to be contaminated with oil sands tailings water and/or solid tailings. Tailings from oil sands processing are known to contain elevated levels of hydrocarbons, ions, and toxic metals (e.g., MacKinnon et al. 2004). There were 12 oil sands reference (OSREF) wetlands, which were situated on oil sands company leases but were not directly contaminated with oil sands tailings, although they may have been constructed using stockpiled overburden or peat. There were 12 natural wetlands that were situated in an agricultural landscape (AG), and another 37 natural wetlands that were situated in relatively undisturbed landscapes (REF), usually in parks or other protected areas.

Methods:

General sampling scheme

Each wetland was visited in spring (May or June) and again in late summer (July or August), when peak vegetation biomass was expected. Generally, each wetland was divided into thirds by radial transects that passed from the upland vegetation through the wet meadow and emergent zones, and ended about 1 m into the open-water zone. Sampling of sediment, wet meadow and emergent zone vegetation, and estimates of zone width were conducted along these transects (Fig. A1-2). Transects were marked, to ensure that the same transects were used in both spring and late summer sampling. Note, however, that the wetland is the unit of measure for all data, i.e., quadrat data was composited to give one value per wetland.

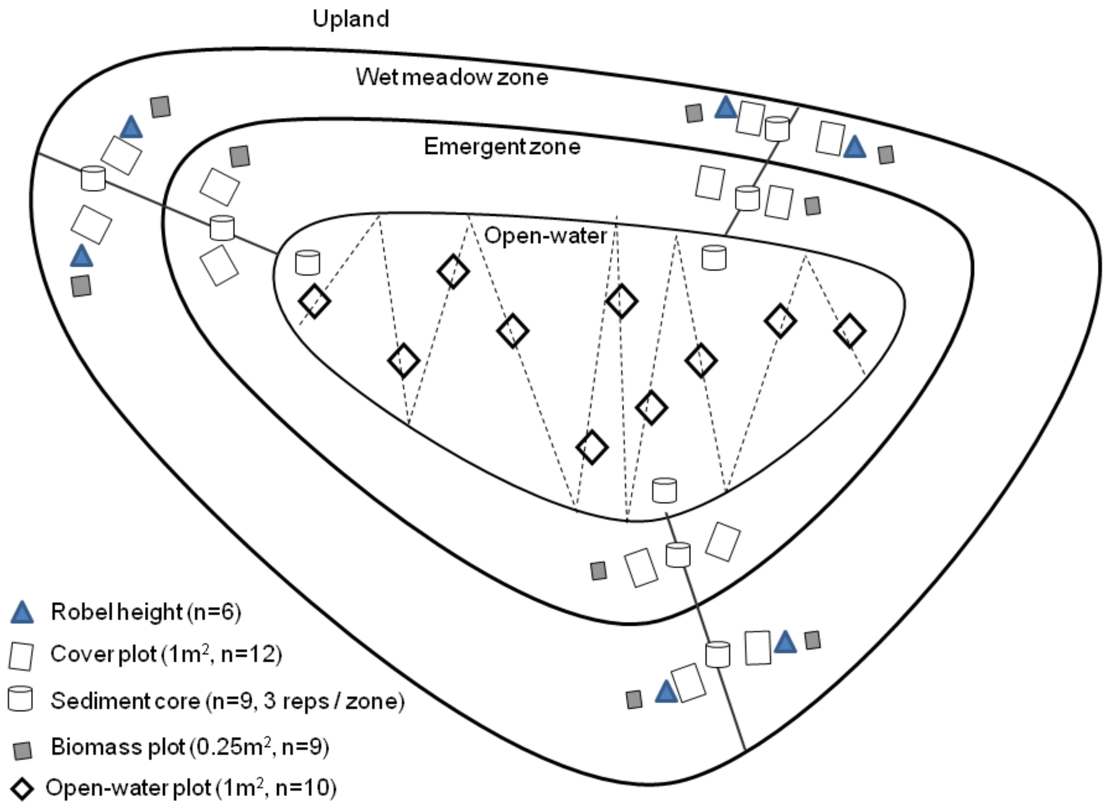

Fig. A1-2. Schematic example of the layout of sampling plots at each wetland. Solid lines represent transects through the marsh zones, whereas dashed (zig-zagging) lines represent the ten transects made through the open water zone.

Water sampling

Sampling was carried out during both the spring and late summer (peak biomass) visits. The deepest point of each wetland was accessed by kayak, so as not to disturb the sediment. If the water was >50 cm deep, an integrated-depth sampler was used to collect a 2 L water sample. Otherwise, water samples were collected by immersing the 2 L acid washed sample bottle directly, about 25 cm beneath the surface. A second sample was taken in a 1 L foil-wrapped glass jar for analysis of naphthenic acids. After collecting the water samples, Secchi depth was measured with a 20 cm diameter Secchi disk and conductivity was measured with a handheld YSI 556

Multiprobe System. Water samples were kept in coolers at about 4˚C until preliminary processing, which took place within 6 hours of collection.

Water analysis

Water samples underwent preliminary processing in the field. Water was filtered through Whatman GF/F glass microfiber filters for analysis of cations (Na^+, K^+, Ca^{2+}, Mg^{2+} and NH_4^+) and anions (SO_4^{2-}, Cl^-, NO_2^-+NO_3^-, CO_3^{2-} and HCO_3^-), soluble reactive phosphorus, total dissolved nitrogen, dissolved organic carbon, total dissolved phosphorus, Fe, Al, Si, and total dissolved solids. The sub-samples collected for NO_2^-, NO_3^-, and NH_4^+ were acidified with sulfuric acid and then refrigerated prior to analysis. Sub-samples for SRP and TDN were frozen until analysis, whereas sub-samples for DOC, anions, and cations were refrigerated. For chlorophyll-a (chl-a), batches of water of known volume were filtered in the shade until the filter paper was evenly green-tinged, then the filter paper was wrapped in foil to keep out sunlight and kept frozen until analysis. The volume filtered was used to back-calculated the chl-a concentration. An unfiltered water subsample was retained for analysis of total nitrogen, total phosphorus, total organic carbon, alkalinity and total suspended solids. This sample was refrigerated until analysis.

Analysis occurred within two months of sample collection. Phytoplankton chl-a was measured using a Shimadzu RF-1501 spectrofluorophotometer at an excitation wavelength of 436 nm and an emission wavelength of 680 nm. Metals (Al, Fe) and some cations (Na^+, K^+, Ca^{2+}, Mg^{2+}) were determined using an Elan 6000, Perkin Elmer ICP-MS following accepted standard methods (AWWA 1999a). Nitrogen, phosphorus, and Si were measured using a Lachat QuikChem 8500 FIA automated ion analyzer, following standard methods (AWWA 1999b, c, d, e). TN, TDN, TP, and TDP were first digested using potassium persulfate, and then measured following standard methods for NO_2^- $+NO_3^-$ (AWWA 1999c) and SRP (AWWA 1999d), respectively. TOC and DOC were measured using a Shimadzu 5000A TOC analyzer, following U.S. EPA method 415.1 (U.S.EPA. 1979). Alkalinity, CO_3^{2-}, and HCO_3^- were determined by titration with 0.01 M sulfuric acid. Cl^- and SO_4^{2-} were determined using a Dionex DX600 Ion Chromatograph and following U.S.EPA method 300.1 (Pfaff et al. 1997). TDS and TSS were determined gravimetrically from residues obtained by oven drying at 103°C. The water sample taken for analysis of naphthenic acids was processed following the Fourier-transform infrared spectroscopy method (Jivraj et al. 1996). Detection limits are reported in the data file OSIBI_2007-2009_headings.txt. Values that were below detection limits were replaced with half the detection limit value. The data are saved in the tab-delimited text file OSIBI_2007-2009_water.txt.

Sediment sampling

Sediment sampling was carried out during late summer sampling, when peak biomass was expected. There were three sediment samples collected from each wetland: one from within each of the wet meadow, emergent and open-water vegetation zones. Each sample was a composite of three replicate cores taken along transects that divided each wetland into thirds (Fig.A1-2). Each core was about 10 cm deep and was collected using a suction-corer with an inner diameter of 5.72 cm. Sediment samples were extruded into zip lock bags and frozen until analysis.

Sediment analysis

A subsample of each sediment sample was analysed using a Coulter LS230 laser diffraction particle analyzer that measures solids ranging from 0.04 to 2000 µm. Samples that contained too much organic matter for the analysis are designated by the symbol MR. The oil, water, and solids content was measured using refluxing toluene in a soxhlet extraction apparatus (Syncrude 2006). Another sub-sample was oven-dried at 60°C for 48 hours and then ground and further sub-sampled for analysis of particulate carbon (PC), particulate nitrogen (PN), particulate phosphorus (PP) and loss on ignition (LOI). Sub-samples of 4 to 6 mg were measured with a Mettler Toledo XP56 microbalance (± 0.001 mg) before analysis of PC and PN by combustion in an Exeter Analytical CE40 Elemental Analyzer. Sub-samples of 0.2 g were weighed using a Mettler Toledo AT261 Delta Range microbalance (± 0.00001 mg) for analysis of PP using the peroxide/sulfuric acid digestion method (Parkinson and Allen 1975) and a Varian Cary 50 spectrophotometer. A 0.5 g sub-sample was weighed out using a Mettler Toledo AE240 balance (± 0.0001 mg) and then placed in a muffle furnace at 550°C for 4 hours and re-weighed to measure LOI. The data are saved in the tab-delimited text file OSIBI_2007-2009_sediment.txt.

Water level

During spring sampling, HOBO depth data loggers were installed either at the point of maximum depth or at an adjacent location and the difference between logger depth and maximum depth was recorded. Loggers recorded water depth every 6 hours until they were retrieved during late summer sampling, when peak vegetation biomass was expected. Maximum depth was determined as the deepest conditions the wetland experienced. Amplitude was calculated as the maximum depth minus the minimum depth recorded by the loggers. Where loggers failed, maximum depth was depth at the deepest point during the spring sampling period and amplitude is recorded as NA. The data are saved in the tab-delimited text file OSIBI_2007-2009_water_level.txt.

Zone width

During late summer sampling, when peak biomass is expected, a surveyor's tape was used to measure the width of the emergent and wet meadow vegetation zones along each of the three transects (Fig. A1-2). The three measurements were then averaged to give a mean zone width for each zone. The data are saved in the tab-delimited text file OSIBI_2007-2009_zone_width.txt.

Open-water vegetation

Sampling occurred during late summer visits and was conducted by kayak. Ten zig-zagging transects were made across each open-water zone, and along each transect a 1 m^2 quadrat was deployed (Fig. A1-2).

First, the water depth in the centre point of the quadrat was measured. Second, the Secchi depth within the quadrat was measured. Third, the percent cover of floating vegetation was assessed within the boundaries of the quadrat. Fourth, the rake technique was employed to estimate relative abundance of each species of submersed vegetation present in each quadrat as outlined in Bayley and Prather (2003). Relative abundance data was averaged across the ten quadrats to give mean percent abundance at each wetland. Fifth, the vegetation was assigned biomass score of between 1 and 5, where 1 = 0%, 2 = < 5%, 3 = 5-25%, 4 = 25-75%, and 5 >75% infestation.   This biomass scoring technique has shown good correlation with biomass measurements (Bayley et al. 2007). We took the median biomass scores to provide a single score per wetland. At most sites we also estimated the overall biomass of vegetation in the open-water zone.

For relative abundance estimates, plants were identified to the lowest practical taxonomic level (usually species) following Moss and Packer (1983) with taxonomy updated according to the Integrated Taxonomic Information System (ITIS 2010). Voucher specimens of hard to identify species were collected, pressed, and are housed at the ALTA Vascular Plant Herbarium, at the University of Alberta.

Total richness was the sum of all species observed in the quadrats and any additional rare species observed while conducting the ten zig-zagging transects.

The data are saved in the tab-delimited text file OSIBI_2007-2009_open-water_veg.txt.

Emergent vegetation

Emergent vegetation sampling was carried out in late summer, when peak biomass was expected. It consisted of relative abundance estimates from six 1 m^2 quadrats, height measurements within each 1 m^2 quadrat, biomass clippings from three 0.25 m^2 quadrats, and a 15 min time restricted diversity walk-about to increase the probability of encountering rare species (Fig. A1-2).

The relative abundance of species in the emergent zone was estimated within six 1 m^2 quadrats. Quadrats were deployed in about the middle of the emergent vegetation zone, one on either side of each of the three transects, such that quadrats were spaced at least 5 m apart. Within each of these quadrats, plants were identified to the lowest practical taxonomic level (usually species) following Moss and Packer (1983) with taxonomy updated according to the Integrated Taxonomic Information System (ITIS

2010). The percent cover of each species was recorded, and an average of the percent cover of each species in the six 1 m^2 quadrats was taken to yield mean value for each species for each wetland.

The mean height of vegetation within each 1 m^2 quadrat was estimated using a meter stick, and similarly averaged to give a single measure per wetland.

Biomass clippings were taken from 0.25 m^2 quadrats adjacent to each of the three transects in the emergent zone. Samples were oven dried at 60˚C to constant mass and then weighed to give total aboveground biomass. The three measures were averaged to give a single value per wetland.

Total richness was the sum of all species observed in quadrats and any additional species encountered during the 15 min time restricted walk-about.

Voucher specimens of many species were collected and are housed at the ALTA Vascular Plant Herbarium, at the University of Alberta.

The data are saved in the tab-delimited text file OSIBI_2007-2009_emergent_veg.txt.

Wet meadow vegetation

Wet meadow vegetation sampling was carried out in the fall, when peak biomass was expected. It consisted of relative abundance estimates from six 1 m^2 quadrats, height measurements within each 1 m^2 quadrat, Robel height estimates, biomass clippings from six 0.25 m^2 quadrats, and a 15 min time restricted diversity walk-about to increase the probability of encountering rare species (Fig. A1-2).

The relative abundance of species in the wet meadow zone was estimated within six 1 m^2 quadrats. Quadrats were deployed in about the middle of the zone, one on either side of each of the three transects, such that quadrats were spaced at least 5 m apart. Within each of these quadrats, plants were identified to the lowest practical taxonomic level (usually species) following Moss and Packer (1983) with taxonomy updated according to the Integrated Taxonomic Information System (ITIS 2010). The percent cover of each species was recorded, and an average of the percent cover of each species in the six 1 m^2 quadrats was taken to yield mean value for each species for each wetland.

The mean height of vegetation within each 1 m^2 quadrat was estimated, and similarly averaged to give a single measure per wetland.

Robel height is a biomass estimate derived from visual obstruction. It is a non-destructive surrogate for biomass clippings in grasslands (Robel et al. 1970). The visual obstruction observation was made from 4 m distant at either 1 m or 1.5 m above ground. One Robel height estimate was taken at each 1 m^2 quadrat, and these six measures were then averaged to yield a single value per wetland.

Biomass clippings were taken from six 0.25 m^2 quadrats, two per transects in the wet meadow zone, positioned at least 5 m apart. Samples were oven dried at 60˚C to constant mass and then weighed to give total aboveground biomass. The three measures were averaged to give a single value per wetland.

Total richness was the sum of all species observed in quadrats and any additional species encountered during the 15 min time restricted walk-about.

Voucher specimens of many species were collected and are housed at the ALTA Vascular Plant Herbarium, at the University of Alberta.

The data are saved in the tab-delimited text file OSIBI_2007-2009_wet_meadow_veg.txt.

GIS data

A hand-held GPS was used in the field to gather latitude and longitude data for all 74 wetlands.

We obtained recent (<5 yrs) aerial photographs or Quickbird Pansharpened 3-Band satellite imagery of a 45 wetland subset of the entire dataset. All images were imported into a digital file for measurement using GIS software (ESRI 2009) such that the final imagery had pixels ≤3 m in resolution. Those images missing spatial reference data were geo-referenced to existing digitized topographical maps from the Canadian National Topographic Database.

Using this imagery, we measured the open-water and marsh (emergent zone plus wet meadow zone) area of each wetland.

We identified 13 major land-cover types within a 2 km buffer surrounding the open water at each wetland: lentic water, lotic water, wetland, forest, open-green (lawn), agriculture (row crop and pasture), roads (paved and unpaved), residential, industrial-commercial, oil and gas, tailings water, solid tailings, and bare ground. We created a shapefile of polygons representing the 13 land-cover types by digitizing at a visual scale between 1:1000 and 1:2500 with ESRI® ArcMap™ 9.3.1 software (ESRI 2009).

We then created a series of nested buffers of different widths surrounding each open-water zone: 300 m, 500 m, 1000 m, 1500 m, and 2000 m, giving us 5 landscape sizes for each wetland. The buffer sizes were selected to reflect landscape sizes used in similar studies (e.g., Houlahan et al. 2006, Matthews et al. 2009, Akasaka et al. 2010).

We then extracted the percent cover of each land-cover type and the linear density of roads and seismic lines within each of the five buffer widths.

The data are saved in the tab-delimited text file OSIBI_2007-2009_GIS.txt.

Quality control

Data entry from raw data sheets was confirmed by independent review. For plant relative abundance and land-use percent cover data, row tallies were used to further confirm the absence of data entry errors. Sediment and water chemistry data was screened for outliers or implausible values by multiple experienced team members. Dorothy Fabijan, assistant curator of the University of Alberta Vascular Plant Herbarium, verified the plant identifications. GIS digitizing was conducted by two team members. To confirm equivalence in their digitizing, they digitized an overlap of 20% of the total area and we measured cross-digitizer agreement using KAPPA analysis (Jensen 1996).

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Taxonomy and systematics: Plant identifications were generally to species level and followed Moss and Packer (1983).

Permit history: Permits for collection of samples from Elk Island National Park and from Alberta Sustainable Resources and Development are retained by Rebecca Rooney.

Legal/organizational requirements: None

Project personnel: Rebecca Rooney, Suzanne Bayley, Dustin Raab, Allison Bil, Andrea Badger, Cassidy Van Rensen, Sean Coogan, Robbi Bonnin, Katherine Svreck, Shantel Koenig and Matt Bolding

Class III. Data Set Status and Accessibility

A. Status

Latest update: September 2011

Metadata status: Metadata are complete.

Data verification: See Class II, Section B, Methods, Quality control

B. Accessibility

Storage location and medium: Ecological Society of America data archives [Ecological Archives], URL published in each issue of its journals. Raw datasheets are housed by the dataset owner and a digital version is housed on the owner's personal computer and on CD.

Contact person: Rebecca Rooney, CW405 Biological Sciences Bldg., University of Alberta, Edmonton, Alberta, T6G 2E9, Canada, rrooney@ualberta.ca.

Copyright restrictions: None for teaching or research purposes, with attribution to this data paper.

Proprietary restrictions: None for teaching or research purposes, with attribution to this data paper.

Costs: None.

Class VI. Data Structural Descriptors

A. Data set files

Identity and size: OSIBI_2007-2009.zip, 117,000 bytes.

Format and storage media:

OSIBI_2007-2009.zip is a zip file containing nine tab-delimited text files, including one tab-delimited metadata file, OSIBI_2007-2009_headings.txt, which contains header information for the other eight files. The folder also contains a copy of this text file.

Header information:

The metadata file OSIBI_2007-2009_headings.txt contains header information for all the main data files. The header fields in the OSIBI_2007-2009_headings.txt file are as follows:

Variable type: indicates the file containing the variable

Variable name: indicates the header field title being referred to

Description: a text description of the variable

Units: the units, where applicable, that the variable is expressed in

Detection limit: the detection limit for the variable, where applicable

Alphanumeric attributes: mixed

Special characters: NA indicates that data is missing, either because it was never collected or because the sample was destroyed. NZ indicates that the particular wetland in question lacked that vegetation zone and thus vegetation in that zone could not be sampled. E.g., the open-water abutted the wet meadow zone directly. MR indicates that the sediment sample contained too much organic matter for effective size fraction determination.

Authentification procedure: N/A

B. Variable information

Description: See Class VI, section A, Header information

Class V. Supplemental Descriptors

A. Data acquisition

Data forms: N/A

Location of completed data forms: The raw data sheets are retained by Rebecca Rooney, rrooney@ualberta.ca. Copies are held in the office of Dr. Suzanne Bayley at the University of Alberta.

B. Quality assurance/quality control procedures: See Class II, Section B, Methods, Quality control

C. Related material: N/A

D. Computer programs and data processing algorithims: See Class II, Section B, Methods

E. Archiving: N/A

F. Publications using the data set:

These data were used in the following publications: Rooney and Bayley (2010), Rooney and Bayley (2011a, b) and Raab (2010). Rebecca Rooney also intends to use the data in completing her Ph.D. thesis and a subset of the data were used in one manuscript that is currently under review. Additional manuscripts are in preparation. Details can be obtained by contacting Rebecca Rooney.

G. Publications using the same sites: See Class V, Section F. In addition, some of the same sites were included in works by Trites and Bayley (2009a,b), and Purdy et al. (2005).

H. History of data set usage:

Data request history: N/A

Data set update history: N/A

Acknowledgements:

We gratefully acknowledge the hard work of field assistants Andrea Badger, Allison Bil, Matt Bolding, Robbi Bonnin, Sean Coogan, and Cassidy Van Rensen, and of GIS technicians Katherine Svreck, Shantel Koenig. In addition we thank Charlene Nielsen for assistance with GIS data management and QA/QC and Dorothy Fabijan for verification of plant identifications and assistance with herbarium submissions. Dr. Lee Foote, Dr. Rolf Vinebrook, and Dr. Cynthia Paszkowski helped with study design. For funding sources, see Class II, Section A, Sources of funding.

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Appendix 3-1

Note that *Myriophyllum sibiricum* and *M. verticillatum* are only distinguishable when the plant is in flower. Thus, I was frequently unable to distinguish them in the field and considered them jointly in my analyses.

Table A3-1-1. List of submersed and floating aquatic macrophyte taxa observed in the study and the number of wetlands in which each was observed. Rarity as based on Kershaw et al. (2001) and Stewart and Kantrud (1972).

Latin Name	Rarity	Number of wetlands
Potamogeton natans L.	Rare	4
Potamogeton zosteriformis Fern.	Common	12
Potamogeton foliosus Raf.	Rare	3
Potamogeton fresii Rupr.	Common	1
Potamogeton pusillus L.	Common	22
Potamogeton obtusifolius Mert. & Koch	Rare	1
Potamogeton richardsonii (Benn.) Rydb.	Common	6
Stuckenia pectinata (L.) Börner	Common	32
Ruppia cirrhosa (Petag.) Grande	Rare	3
Zannichellia palustris L.	Common	3
Sagittaria cuneata Sheldon	Common	3
Chara spp. L.	Common	31
Polygonum amphibium var. stipulaceum Coleman	Common	4
Ceratophyllum demersum L.	Common	37
Ranunculus aquatilis L.	Common	2
Myriophyllum sp. (either sibiricum or verticillatum) L.	Common	4
Myriophyllum sibiricum Komarov	Common	14
Myriophyllum verticillatum L.	Common	4
Hippuris vulgaris L.	Common	5
Utricularia macrorhiza Le Conte	Common	16
Utricularia minor L.	Rare	5
Aquatic moss	Common	4
Caltha palustris L.	Common	1
Lemna turionifera Landolt	Common	4
Lemna minor L.	Common	32
Lemna trisulca L.	Common	13
Nuphar lutea ssp. *variegata* (Dur.) E.O. Beal	Common	2

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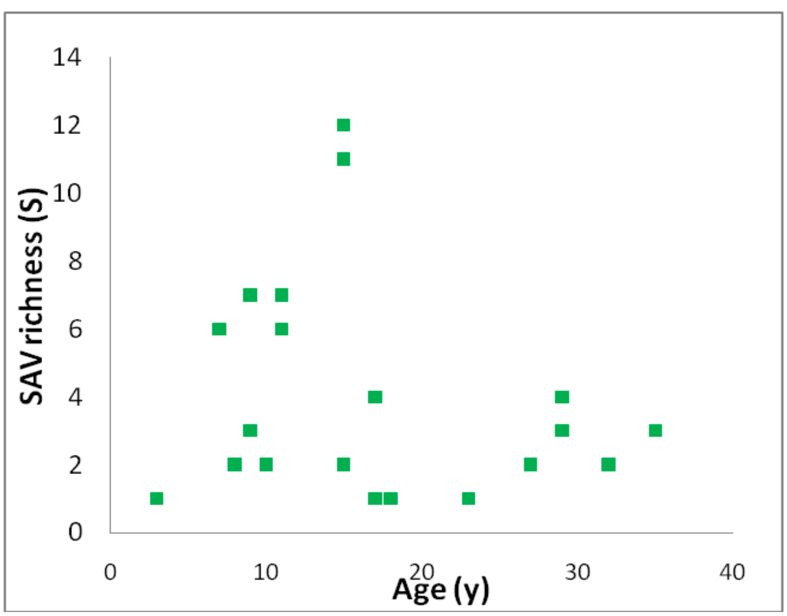
Appendix 3-2

Although the literature suggests that it takes several years for plant communities to assemble in a newly constructed or restored wetland (Kellogg and Bridgham 2002, Gutrich et al. 2009), age was not a good predictor of submersed aquatic plant taxa richness in reclamation wetlands (ANOVA: $F_{1,20}$ = 0.01959, p = 0.89, r^2 = 0.00098). This might have been, in part, because of the reduced sample size when considering only reclamation wetlands with a construction date. An additional source of noise likely obscuring a trend between wetland age and richness is the constant evolution of reclamation practices since they began in 1972. In fact, with the exception of two unusually diverse wetlands, there appears to be a negative relationship between wetland age and taxa richness (Fig. A3-2-1). This apparent trend supports the hypothesis that wetland reclamation practices have improved over the last 35 years.

Fig. A3-2-1. Richness of submersed aquatic vegetation plotted against age for 21 reclamation wetlands in the oil sands mining region of Alberta.



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Appendix 4-1

Table A4-1-1. Definitions of candidate metrics calculated from the submersed and floating vegetation.

Metric	Definition
FLT_S	Richness of floating species
SAV_S	Richness of submersed aquatic species
TOT_S	Richness of floating and submersed aquatic species
H'	Shannon entropy = $-\sum (p \ln p)$ for each species at the pond level
H'_ED	The "true diversity" according to the Shannon entropy index (weighs species according to their frequencies)
S_Quad	The number of species observed in all quadrats summed together
Hmax	ln(S) within each pond, where S is the sum of species used to calculate H'
EH	H' / Hmax; the evenness of plant species abundances within the pond
G_Simp	Gini-Simpson diversity index: 1 - (Sum of squares of species frequencies)
G_Simp_ED	The "true diversity" according to the Gini-Simpson index
No._Rare_taxa	Number of regionally rare taxa[a]: *Ruppia cirrhosa*, *Utricularia minor*, *Potamogeton foliosus*, *P. natans*, and *P. obtusifolius*
%_Rare_taxa	Percent of total species richness comprised of regionally rare species
%C_Rare_taxa	Relative abundance of regionally rare species
%_Common_taxa	Percent of total species richness comprised of common species
%C_Common_taxa	Relative abundance of common species
H'_ED/G_Simp_ED	A measure of dominance
FQAI[b]	R/SQRT(N), where R is Sum_C and N is the number of native vascular plant species
Sum_C	Sum of coefficients of conservatism values on a presence/absence basis
Med_C	Median coefficient of conservatism value
Mean_C	Mean coefficient of conservatism value
%C_Annual	Relative abundance of annual vascular plants
%C_Perennial	Relative abundance of perennial vascular plants
%C_Non-native	Relative abundance of non-native plants
%C_Native	Relative abundance native plants
%C_Carnivorous	Relative abundance of carnivorous plants
%C_Halophyte	Relative abundance of salt tolerant plants: *R. cirrhosa*, *Z. palustris*, *S. pectinata*, *Chara* spp.
%C_Alkali	Relative abundance of alkali-loving plants: *P. alpinus*, *P. foliosus*, *P. zosteriformis*, *S. pectinata*, *Z. palustris*, *Chara* spp.
%C_Monocots	Relative abundance of monocots
%C_Dicots	Relative abundance of dicots
%C_Float_Leaf	Percent cover of floating leafed plants
%C_Sparse_Leaf	Relative abundance of sparse leaved submersed plants: *R. cirrhosa*, *Z. palustris*, *P. natans*, *P. nodosus*, *P. alpinus*, *Ranunculus* spp.
%_Annual	Percent of TOT_S comprised of annual species
%_Perennial	Percent of TOT_S comprised of perennials
%_Non-native	Percent of TOT_S comprised of non-native plants
%_Native	Percent of TOT_S comprised of native plant species
%_Carnivorous	Percent of TOT_S comprised of carnivorous plants
%_Halophyte	Percent of TOT_S comprised of salt tolerant species

%_Alkali	Percent of TOT_S comprised of alkali-loving plants
%_Monocots	Percent of TOT_S comprised of monocots
%_Dicots	Percent of TOT_S comprised of dicots
%_Float_Leaf	Percent of TOT_S comprised of floating leafed plants
%_Sparse_Leaf	Percent of TOT_S comprised of sparse leaved submersed species
P-A_Rcirrhosa[c]	Presence/absence of *R. cirrhosa* (a disturbed community)
P-A_aqua_moss	Presence/absence of aquatic moss (a reference community)
P-A_Cdemersum	Presence/absence of *C. demersum* (a reference community)
P-A_Chara	Presence/absence of *Chara* spp. (a disturbed community)
%C_Umacrorhiza	Relative abundance of *U. macrorhiza* (a reference community)
%C_Cdemersum	Relative abundance of *C. demersum*
%C_Chara	Relative abundance of *Chara* spp.
%C_Myriophyllum	Relative abundance of *Myriophyllum* spp. (a disturbed community)
%C_Ppusillus	Relative abundance of *P. pusillus* (a disturbed community)
%C_Potamogetons	Relative abundance of *Potamogeton* spp. (a disturbed community)
%_Umacrorhiza	Percent of TOT_S comprised of *U. macrorhiza*
%_C.demersum	Percent of TOT_S comprised of *C. demersum*
%_Chara	Percent of TOT_S comprised of *Chara* spp.
%_Myriophyllum	Percent of TOT_S comprised of *Myriophyllum* spp.
%_Ppusillus	Percent of TOT_S comprised of *P. pusillus*
%_Potamogetons	Percent of TOT_S comprised of *Potamogeton* spp.
Med_SAV_Dens	Median SAV biomass rating calculated from the 10 quadrats
Overall_SAV_dens	Estimate of overall SAV biomass rating

[a] Rarity is as defined by Kershaw et al. (2001)
[b] FQAI is following the method described in (Forrest, 2010)
[c] contribution of species to reference versus degraded communities is derived from Rooney and Bayley (2011)

References

Forrest, A., 2010. Created stormwater wetlands as wetland compensation and a floristic quality approach to wetland condition assessment in central Alberta. Biological Sciences, Vol. MSc. Science, University of Alberta, Edmonton, AB

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Appendix 4-2

Details of the quantile piecewise regression of H'_ED/G_Simp_ED on stress scores and the subsequent comparison of the final IBI including versus excluding this non-linear metric.

H'_ED is the true diversity according to the Shannon entropy index (H'). It is a first order measure of diversity that weighs species on the basis of their frequencies (Jost et al. 2010). G_Simp_ED is a second order measure of diversity, based on the sum of squared frequencies of species occurrence (Jost et al. 2010). It is the true diversity according to the Gini-Simpson index. This metric ranged from 1 to 1.42 and produced a significant regression tree with three terminal nodes, one for values less than or equal to 1.15, one for values between 1.15 and 1.25, and a third for values greater than 1.25. Relationships between the metric and stress scores were positive within each of these bins. The mean cross-validation loss for the regression tree was 0.6348, with a naive estimate of standard error of 0.1037. H'_ED/G_Simp_ED was poorly correlated (and therefore non-redundant) with the 11 metrics that had significant linear relationships with stress scores (Maximum Pearson r = 0.32).

In general, excluding the non-linear metric H'_ED/G_Simp_ED had little effect on the strength of the correlation between IBI scores and stress scores for both the development and test data sets (Table A4-2-1). In the testing set, including H'_ED/G_Simp_ED occasionally reduced the Spearman rank correlation coefficients. The effect of including H'_ED/G_Simp_ED on the width of 95% confidence intervals for the correlation between IBI scores and stress scores was small and variable (Table A4-2-1). Because including H'_ED/G_Simp_ED did not improve the relationship between IBI and stress scores, we excluded it from our final IBI, leaving just five metrics, all linearly related to environmental stress.

Table A4-2-1. Spearman rank correlation results for stress scores and IBIs generated by the four scoring methods. Confidence intervals were generated by bootstrapping using the percentile method in SYSTAT software. Results are presented for IBIs created with and without including the non-linear metric H'_ED/G_Simp_ED. Note that in nearly all cases including H'_ED/G_Simp_ED reduced the Spearman *rho* value and yielded wider 95% C.I.s.

		Including H'_ED/G_Simp_ED			Excluding H'_ED/G_Simp_ED		
	Method	Spearman *rho*	95% C.I. Lower	95% C.I. Upper	Spearman *rho*	95% C.I. Lower	95% C.I. Upper
Development suite (n = 42)	DWR	-0.67387	-0.80396	-0.46299	-0.66325	-0.78297	-0.49921
	DRR	-0.64260	-0.77886	-0.43491	-0.65913	-0.78557	-0.47035
	CWR	-0.72761	-0.81879	-0.56713	-0.68809	-0.79805	-0.52350
	CRR	-0.70217	-0.80432	-0.53085	-0.67040	-0.80433	-0.49764
Testing suite (n = 20)	DWR	-0.60847	-0.85575	-0.19831	-0.59176	-0.87301	-0.11112
	DRR	-0.59472	-0.84818	-0.20287	-0.60584	-0.86739	-0.19025
	CWR	-0.64184	-0.88459	-0.20072	-0.68423	-0.93579	-0.25067
	CRR	-0.54477	-0.83396	-0.10606	-0.63079	-0.88612	-0.18908

References

Jost L., DeVries P., Walla T., Greeney H., Chao A. and Ricotta C. 2010. Partitioning diversity for conservation analyses. Diversity and Distributions 16: 65-76.

Appendix 4-3

Calculation of an IBI score and table depicting ranges of metric and IBI scores observed in the 62 wetlands sampled in this study.

Ranges for each metric scored by each of the four methods are presented in Table A4-3-1, along with the final range of IBI scores produced by each method. Based on the strength of its correlation with stress scores, the breadth of its range of values, and its theoretical basis, we determined that the continuous-reference range (CRR) approach to metric scoring was optimal.

Table A4-3-1. Range of metric scores by the four scoring methods: discrete-whole range (DWR), discrete-reference range (DRR), continuous-whole range (CWR), continuous-reference range (CRR). The IBI observed range provides the highest and lowest IBI scores observed in the study (n = 62 wetlands).

Metric	DWR	DRR	CWR	CRR
FLT_S	1 – 5	1 – 5	0 - 100	0 - 100
%C_ALK	1 - 5	1 – 5	0 - 100	0 - 100
%C_FLOAT	1 - 5	1 – 5	0 - 100	0 - 100
%C_ Cdemersum	1 - 5	1 – 5	0 - 100	0 - 100
%_ Potamogetons	1 - 5	1 – 5	0 - 100	0 - 100
IBI observed range	5 - 24	5 – 22	0 - 425	0 - 425

Below follows an example of how an IBI score is calculated following the CRR method. The wetland selected to serve as exemplar is named CLSOUTH. It is a reference wetland located in the Child Lake Natural Area, near High Level, Alberta (Lattitude: 58.42083 N, Longitude: 116.54518 W). It's stress score, calculated following Rooney and Bayley (2010), was 9.00 out of an observed maximum of 18.67 in our suite of 62 wetlands.

CLSOUTH supported two floating plant species, *Lemna minor* and *Lemna trisulca*. Combined, these two species covered 19% of the open water zone. The wetland contained no alkali-loving plants or *Potamogeton* spp. Rather, the wetland was heavily dominated by *Ceratophyllum demersum*, with a relative abundance of 92%.

Combined with the max and min values observed in the development set, the above is all the information required to calculate the IBI score of CLSOUTH (IBI score = 362) using the following three steps (Table A4-3-2):

1) Determine metric values for the five metrics included in the IBI: FLT_S, %C_ALK, %C_FLOAT, %C_ Cdemersum, and %_ Potamogetons.
2) Convert metric values to scores based on the range of values observed in the development data set using equation 1 if the metric is negatively correlated with stress scores or equation 2 if the metric is positively correlated with stress scores:

$$\text{equation 1:} \quad \frac{\text{CLSOUTH's metric value} - \text{minimumobserved value}}{\text{maximumobserved value} - \text{minimum observed value}}$$

$$\text{equation 2:} \quad 100 - \frac{\text{CLSOUTH's metric value} - \text{minimum observed value}}{\text{maximum observed value} - \text{minimum observed value}}$$

   Where only reference wetlands in the development set are included in the observed values.

3) Sum metric scores to obtain IBI score: 50 + 100 + 20 + 92 + 100 = 362.

Table A4-3-2. The metric values and calculated scores for CLSOUTH along with the range of values observed among reference sites in the development data set (n = 25).

Metric	Metric value	Min observed value	Max observed value	Metric score
FLT_S	2	0	4	50
%C_ALK*	0	0	100	100
%C_FLOAT	19	0	93	20
%C_ Cdemersum	92	0	100	92
%_ Potamogetons*	0	0	40	100
IBI score				362

* indicates that the metric is positively correlated with stress scores, and so the scoring is inversed.

References

Rooney R.C. and Bayley S.E. 2010. Quantifying a stress gradient: An objective approach to variable selection, standardization and weighting in ecosystem assessment. Ecological Indicators 10: 1174-1183.