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Use of Indices of Biological Integrity (IBIs) to assess wetland health in dry and wet conditions

by

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Abstract

Indices of Biological Integrity (IBIs) estimate the biological condition of an ecosystem by measuring biological metrics that predict underlying environmental stress. I evaluated the use of IBIs developed from 5 biotic communities at 81 semi-permanent/permanent natural and constructed wetlands. Wet meadow vegetation (R^2 = 0.67) and songbirds (R^2 = 0.59) were consistently sensitive to environmental stress and were strong surrogates of one another (R^2 = 0.56), suggesting that plants can be used to predict songbird integrity and vise versa. Other plant and bird communities were not good indicators of environmental stress. A subset of 45 sites was resampled to evaluate the sensitivity of the wet meadow vegetation IBI to plant community changes between dry and wet conditions. Non-metric multi-dimensional scaling (NMS) revealed that IBI scores were fairly insensitive to plant community changes from relatively dry to wet conditions. These results suggest that plant-based IBIs are in fact effective at measuring ecosystem health in the Aspen Parkland.

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1. General introduction and thesis overview

Wetlands are essential components of our waters that provide ecosystem services such as hydrologic storage, flood control, groundwater recharge and wildlife habitat (Keddy, 2000). Wetlands, however, are one the most threatened ecosystems on Earth (Mitsch and Gosselink, 2007). Approximately 70% of wetlands in the prairie regions of Canada have been converted to upland to support agriculture (Kennedy and Mayer, 2002). As urban and suburban development is altering the remaining wetlands in this region, concerns for both the biodiversity of wetlands and provisioning of wetland functions (i.e. wildlife breeding habitat), and values (i.e. hunting and recreation) has led to a demand for wetland assessment tools to aid management of wetlands in the Aspen Parkland Ecoregion of Canada.

Indices of Biological Integrity

Biological assessment tools can provide a precise estimate of the ecological condition of a wetland. *Bioassessment* is the practice of using organisms to assess ecosystem health (Rader, 2001). *Ecosystem health* refers to both natural processes that occur in all ecosystems (i.e. decomposition, production) and goods and services that benefit society in terms of economic value or societal value (i.e. flood control, wildlife habitat) (Rader, 2001). Indices of Biological Integrity (IBIs) quantify the human influence on an ecosystem by comparing the biological integrity of test sites to the expectations based on reference conditions. The *reference condition* corresponds to the range of conditions that represent the highest level of ecosystem functioning across a suite of functions, or the least-disturbed sites in the region (Brinson and Rheinhardt, 1996). A site's *biological integrity* refers to its ability to support and maintain a healthy biotic community similar to reference conditions (Karr, 1991; Karr and Chu, 1999) and can be used as a surrogate proxy of a site's ecosystem health. Biological communities are considered to be good indicators of ecosystem health because an ecosystem's biota (e.g. plants) and its functions (e.g. production decomposition, transpiration, nutrient cycling) are interdependent (Rader, 2001). IBIs are widely used and are appealing because they focus on attributes of living organisms that reflect underlying physical and chemical conditions. Using the IBI approach, several biological metrics, defined as measurable attributes that are sensitive to a gradient of human disturbance (Karr and Chu, 1999), are integrated into a multi-metric index score. IBIs were first used to monitor fish communities in streams and rivers in Midwestern United States (Karr, 1981) and they have since been adapted to assess the health of wetlands using common indicator taxa such as plants and birds (Adamus and Brandt, 1990).

The comparison of several indicators to represent overall ecosystem health

Both plant and bird communities have been shown to be good indicators of environmental stress in the United States and elsewhere in Canada. Plant-based IBIs have been developed to assess wetlands in the prairie pothole region (DeKeyser et al., 2003; Hargiss et al., 2008), Ohio (Mack, 2007), Pennsylvania (Miller et al., 2006), and the Great Lakes (Rothrock et al., 2008; Wilcox et al., 2002). In Alberta, IBIs were successfully developed from vegetation communities based in the open water zone (Rooney and Bayley, 2011) and the wet meadow zone (Raab and Bayley, 2012) to assess oil sands reclamation marshes in the boreal region. Although birds have been used less to assess wetland health, they could be particularly good indicators because of their societal value (Adamus, 1996) and because high trophic levels can sometimes indicate disturbance at lower levels (DeLuca et al., 2004), whereas lower trophic levels do not necessarily indicate integrity at higher levels. IBIs have been developed from bird communities to assess wetlands in West Virginia (Veselka IV et al., 2010) and tidal wetlands in Chesapeake Bay (DeLuca et al., 2004).

The sensitivity of indicators to underlying environmental conditions can differ among geographical regions as a result of regionally specific evolutionary and biogeographic processes (Karr and Chu, 1999) as well as the type of environmental stress being assessed (e.g. Angermeier and Karr, 1986; Karr et al., 1986). Hence, several taxonomic groups need to be evaluated when developing IBIs in geographically distinct regions. Plant-based IBIs have been shown to be sensitive to local environmental conditions (Mensing et al., 1998), surrounding land-use practices (DeKeyser et al., 2003), and are correlated with other wetland health assessment tools based on rapid assessment methods (Mack, 2007; Miller et al., 2006). IBIs developed from birds have been shown to be sensitive to habitat fragmentation (Canterbury et al., 2000), agricultural impacts (Bradford et al., 1998), and surrounding land use (DeLuca et al., 2004).

Although a single taxonomic group is commonly used in the IBI approach, there has been debate as to whether IBIs that monitor only one assemblage are representative of all other biological communities and components of an ecosystem. Previous studies indicate that some communities can provide unique information about ecosystem health while others provide redundant information (Brazner et al., 2007). Bryce et al. (2002) depicted how using birds as indicators in riparian areas in combination with aquatic indicators would give a more complete understanding of stream health in Oregon. O'Connor (2000) found that bird metrics predicted multiple types of disturbances better than other indicators (not including plants) in New England lakes. Brazner et al (2007) found that birds had the least similar responses to other biotic communities whereas wetland vegetation had similar responses to other biota. Other studies have found that habitat and bird indices are strongly related (Canterbury et al., 2000).

Decisions on whether multiple indicators are needed to assess ecosystem health also depend on wetland management program objectives. For instance, if the program's goal is to assess the overall condition of constructed or restored wetlands, then potentially only one taxonomic group is needed, whereas more detailed goals to monitor multiple organisms, improve construction design, or diagnose specific types of stress could require multiple indicators. All indicators do not respond the same to environmental stresses, and different indicator groups can provide different information (Rader, 2001). A sophisticated scientific evaluation of multiple biological communities should be done to determine which and how many indicators are necessary.

A sound biological monitoring tool has low sensitivity to inter-annual variability

Vegetation in wetlands is closely connected with water levels and hydroperiod (Hutchinson, 1975; Keddy, 2000). Differing functional groups (e.g. emergent species, drawdown species) are physiologically and biologically adapted to specific water levels, making zonations discernable along an elevation or moisture gradient (Hutchinson, 1975; Van der Valk, 2005). Stewart and Kantrud's (1971) wetland classification system classifies wetlands based on their water level permanence and the vegetation communities. Semi-permanent and permanent wetlands contain standing water in their basins for most or all of the year (Stewart and Kantrud, 1971; Van der Valk, 2005), although they still exhibit large fluxes in water levels. Vegetation zonations are dynamic and the species composition changes inter-annually as a result of natural variability in water level and hydroperiod, referred to as wet-dry cycles (Van der Valk, 2005; Van der Valk and Davis, 1978; Wilcox, 2004). Shifts in plant communities have been well documented in numerous studies over variable climate conditions (Swanson et al., 2003; Van der Valk and Davis, 1978; Wilcox, 2004).

Biological monitoring should measure biological attributes that are sensitive to degradation in environmental condition caused by humans while being minimally affected by natural variation (Karr and Chu, 1999). If IBI scores are sensitive to plant community changes, then it makes it difficult to discern whether a change in IBI score is due to human disturbance or natural variation. Wilcox et al. (2002) concluded that their plant-based IBI for Great Lakes' wetlands was sensitive to plant community shifts in response to changing water levels, making IBI scores irreproducible over time. Likewise, an IBI developed for a range of wetland classes (temporary to semi-permanent) in the prairie pothole region in North Dakota (DeKeyser et al., 2003; Hargiss et al., 2008; Kirby and DeKeyser, 2003) was tested over 4 consecutive years by Euliss and Mushet (2011), who found that IBI score improved with rising water levels.

The ability to produce IBIs that are robust to natural variation could vary by region and wetland type. Although Euliss and Mushet (2011) found that temporary to semi-permanent wetlands had variable IBI scores in the Prairie Pothole Region, IBIs may be better suited to other regions and wetland types, such as more permanent wetlands in the Aspen Parkland, where moisture deficits are less extreme (0-15mm; Hogg, 1994) than in the Prairie Pothole Region (30mm; Winter, 1989). Semi-permanent and permanent wetlands often have sources of groundwater inputs and as a result are less likely to dry out than temporary and seasonal wetlands (Euliss and Mushet, 1996). That being said, Mack (2007) reported on vegetation-based IBIs that could successfully assess health across a variety of wetland types and regions in Ohio.

Monitoring wetland vegetation by sampling along natural moisture gradients could also affect the ability to produce robust IBIs that sufficiently distinguish human disturbance from natural variation. For example, DeKeyser et al. (2003) developed the IBI for prairie pothole wetlands by incorporating data sampled from all vegetation zonations within a site. In contrast, the IBI to assess emergent wetlands in Ohio (Mack, 2007) focused sampling efforts in the largest emergent zone to account for the varying number and size of vegetation zonations. Sampling a single zone within the wetland could factor out some natural variation associated with sampling along a naturally variable moisture gradient. IBIs were developed for reclamation oil sands marshes in the boreal region of Alberta by monitoring the plant community in the wet meadow zone (Raab and Bayley, 2012) and open-water zone (Rooney and Bayley, 2011) alone, suggesting that measurable and comprehensive metrics can be devised from a single functional or taxonomic group rather than having to monitor the entire community.

Finally, not all studies adhere to similar standard methods and criteria for metric testing and selection (i.e. eliminating metrics based on redundancy analysis). Multi-metric IBIs should incorporate a comprehensive set of metrics that meet a number of criteria listed by Cairns et al. (1993).

Thesis overview

The goal of this study is to evaluate the use of Indices of Biological Integrity (IBIs) to assess wetland health of semi-permanent and permanent wetlands in the Aspen Parkland Ecoregion of Canada under varying weather conditions. The broad hypotheses for my thesis are: 1) IBIs that assess a site's biological condition can be produced from both plants and birds having sensitive indicators to disturbance and 2) IBI scores will be fairly insensitive to plant community changes resulting from variable conditions in weather, thereby making their IBI scores reproducible over time.

In the 2nd chapter of my thesis, I developed, tested and evaluated IBIs produced from several individual taxonomic groups (vegetation from the wet meadow zone, emergent zone and open-water zone, songbirds and waterfowl) and multiple taxonomic groups (i.e. wet meadow vegetation and songbirds) to assess wetland health.

In chapter 3, I focus on the reproducibility of the wet meadow vegetation IBI scores during successive dry and wet conditions to assess whether IBIs are in fact useful to assess more permanent depressional wetlands. I examined whether the wet meadow vegetation community shifted at study sites between dry and wet years and whether the wet meadow vegetation IBI was sensitive to those changes.

In the 4th chapter of my thesis, I provide an overview of the implications of this research for bioassessment practices in the Aspen Parkland and the contributions this research makes to the fields of wetland management and biological indicators.

References

Adamus, P.R., 1996. Bioindicators for Assessing Ecological Integrity of Prairie Wetlands. EPA/600/R-96/082. U.S. E.P.A., National Health and Environmental Effects Research Laboratory, Western Ecology Division, Covallis, OR.

Adamus, P.R., Brandt, K.B., 1990. Impacts on Quality of Inland Wetlands of the United States: a Survey of Indicators, Techniques, and Applications of Community Level Biomonitoring Data. U.S. E.P.A., Environmental Research Laboratory.

Angermeier, P.L., Karr, J.R., 1986. Fish communities along environmental gradients in a system of tropical streams. Environ. Biol. Fishes. 9, 117-135.

Bradford, D.F., Franson, S.E., Neale, A.C., Heggem, D.T., Miller, G.R., Canterbury, G.E., 1998. Bird species assemblages as indicators of biological integrity in Great Basin rangeland. Environ. Monit. Assess. 49, 1-22.

Brazner, J.C., Danz, N.P., Niemi, G.J., Regal, R.R., Trebitz, A.S., Howe, R.W., Hanowski, J.M., Johnson, L.B., Ciborowski, J.J.H., Johnston, C.A., Reavie, E.D., Brady, V.J., Sgro, G.V., 2007. Evaluation of geographic, geomorphic and human influences on Great Lakes wetland indicators: A multi-assemblage approach. Ecol. Indic. 7, 610-635.

Brinson, M.M., Rheinhardt, R., 1996. The Role of reference wetlands in functional assessment and mitigation. Ecol. Appl. 6, 69-76.

Bryce, S.A., Hughes, R.M., Kaufmann, P.R., 2002. Development of a bird integrity index: Using bird assemblages as indicators of riparian condition. Environ. Manage. 30, 294-310.

Cairns, J., McCormick, P.V., Niederlehner, B.R., 1993. A proposed framework for developing indicators of ecosystem health. Hydrobiologia. 263, 1-44.

Canterbury, G.E., Martin, T.E., Petit, D.R., Petit, L.J., Bradford, D.F., 2000. Bird communities and habitat as ecological indicators of forest condition in regional monitoring. Conserv. Biol. 14, 544-558.

DeKeyser, E.S., Kirby, D.R., Ell, M.J., 2003. An index of plant community integrity: Development of methodology for assessing prairie wetland plant communities. Ecol. Indic. 3, 119-133.

DeLuca, W.V., Studds, C.E., Rockwood, L.L., Marra, P.P., 2004. Influence of land use on the integrity of marsh bird communities of Chesapeake Bay, USA. Wetlands. 24, 837-847.

Euliss, N., Mushet, D., 2011. A multi-year comparison of IPCI scores for prairie pothole wetlands: Implications of temporal and spatial variation. Wetlands. 31, 713-723.

Euliss, N.H., Mushet, D.M., 1996. Water-level fluctuation in wetlands as a function of landscape condition in the prairie pothole region. Wetlands. 16, 587-593.

Hargiss, C.L.M., DeKeyser, E.S., Kirby, D.R., Ell, M.J., 2008. Regional assessment of wetland plant communities using the index of plant community integrity. Ecol. Indic. 8, 303-307.

Hogg, E.H., 1994. Climate and the southern limit of the Western Canadian Boreal forest. Canadian Journal of Forest Research. 24, 1835-1845.

Hutchinson, G.E., 1975. A Treatise on Limnology, Volume 3: Limnological Botany, John Wiley, New York, NY, 660 pp.

Karr, J.R., 1981. Assessment of biotic integrity using fish communities. Fisheries. 6, 21-27.

Karr, J.R., 1991. Biological Integrity - a long-neglected aspect of water-resource management. Ecol. Appl. 1, 66-84.

Karr, J.R., Chu, E.W., 1999. Restoring Life in Running Waters: Better biological monitoring, Island Press, Washington, DC, 220 pp.

Karr, J.R., Fausch, K.D., Angermeier, P.L., Yant, P.R., Schlosser, I.J., 1986. Assessment of Biological Integrity in Running Water: A method and its rationale, Illinois Natural History Survey Special Publication No. 5, Champaign, IL, 28 pp.

Keddy, P.A., 2000. Wetland Ecology: Principles and conservation, Cambridge University Press, New York, NY, 614 pp.

Kennedy, G., Mayer, T., 2002. Natural and constructed wetlands in Canada: An overview. Water Qual. Res. J. Can. 37, 295-325.

Kirby, D.R., DeKeyser, E.S., 2003. Index of wetland biological integrity development and assessment of semi-permanent wetlands in the Missouri Coteau Region of North Dakota. Section 104[b], Final Report for North Dakota Department of Health, (3) Wetland Grant Funds,.

Mack, J.J., 2007. Developing a wetland IBI with statewide application after multiple testing iterations. Ecol. Indic. 7, 864-881.

Mensing, D.M., Galatowitsch, S.M., Tester, J.R., 1998. Anthropogenic effects on the biodiversity of riparian wetlands of a northern temperate landscape. J. Environ. Manage. 53, 349-377.

Miller, S.J., Wardrop, D.H., Mahaney, W.M., Brooks, R.P., 2006. A plant-based index of biological integrity (IBI) for headwater wetlands in central Pennsylvania. Ecol. Indic. 6, 290-312.

Mitsch, W.J., Gosselink, J.G., 2007. Wetlands 4th ed., John Wiley & Sons, Inc., New York, NY, 582 pp.

O'Connor, R.J., Walls, T.E., Hughes, R.M., 2000. Using multiple taxonomic groups to index the ecological condition of lakes. Environ. Monit. Assess. 61, 207-228.

Raab, D.J., Bayley, S.E., 2012. Reclamation of wetland habitat in the Alberta oil sands: Generating assessment targets using boreal marsh vegetation communities. Ecol. Indic. 15, 43-51.

Rader, R.B.,2001. An introdution to wetland bioassessment and management. In Rader, R.B., Batzer, D.P., Wissinger, S.A. (Eds.), Bioassessment and Management of North American Freshwater Wetlands. Jon Wiley & Sons, New York, NY, pp. 3-9.

Rooney, R.C., Bayley, S.E., 2011. 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. Environ. Monit. Assess. 184, 749-761.

Rothrock, P.E., Simon, T.P., Stewart, P.M., 2008. Development, calibration, and validation of a littoral zone plant index of biotic integrity (PIBI) for lacustrine wetlands. Ecol. Indic. 8, 79-88.

Stewart, R.E., Kantrud, H.A., 1971. Classification of natural ponds and lakes in the glaciated prairie region, U.S. Bureau of Sport Fisheries and Wildlife, Washington, DC, 57 pp.

Swanson, G.A., Euliss, N.H., Hanson, B.A., Mushet, D.M.,2003. Dynamics of a prairie pothole wetland complex: implications for wetland management. In Winter, T.C. (Ed.) Hydrological, Chemical, and Biological Characteristics of a Prairie Pothole Wetland Complex Under Highly Variable Climate Conditions - The Cottonwood Lake Area, East-central North Dakota. USGS Professional Paper 1675, Denver, CO, pp. 55-94.

Van der Valk, A.G., 2005. Water-level fluctuations in North American prairie wetlands. Hydrobiologia. 539, 171-188.

Van der Valk, A.G., Davis, C.A., 1978. The role of seed banks in the vegetation dynamics of prairie glacial marshes. Ecology. 59, 322-335.

Veselka IV, W., Anderson, J.T., Kordek, W.S., 2010. Using dual classifications in the development of avian wetland indices of biological integrity for wetlands in West Virginia, USA. Environ. Monit. Assess. 164, 533-548.

Wilcox, D.A., 2004. Implications of hydrologic variability on the succession of plants in the Great Lakes wetlands. Aquat. Ecosyst. Health Manage. 7, 223-231.

Wilcox, D.A., Meeker, J.E., Hudson, P.L., Armitage, B.J., Black, M.G., Uzarski, D.G., 2002. Hydrologic variability and the application of index of biotic integrity metrics to wetlands: A Great Lakes evaluation. Wetlands. 22, 588-615.

Winter, T.C.,1989. Hydrologic studies of wetlands in the northern prairie. In Van der Valk, A.G. (Ed.) Northern Prairie Wetlands. Iowa State University Press, pp. 16-54.

2. Use of several individual and multiple taxonomic groups to assess wetland health in the Aspen Parkland Ecoregion of Canada

2.1 Introduction

Agricultural and urban development have led to the loss of as much as 70% of wetlands in Canada since European settlement (Kennedy and Mayer, 2002; Mitsch and Gosselink, 2007). The Aspen Parkland Ecoregion in western Canada has experienced persistent loss of wetlands that has significantly altered the region and compromised its overall ecosystem functioning (Dahl and Watmough, 2007). The deterioration of ecosystem health and function has presently shifted the focus to conservation of wetlands (Kennedy and Mayer, 2002) and today wetlands are highly valued and recognized for their complex hydrological, biogeochemical and ecological functions. Wetlands provide critical breeding habitat for waterfowl and wetland-dependent songbirds and are unique ecosystems that hold a high diversity of plants specifically adapted to living in the ecotone between aquatic and terrestrial habitats.

Notwithstanding the need to protect, conserve and manage water resources, a national wetland monitoring program does not exist in Canada (Dahl and Watmough, 2007) and a standardized approach to monitoring ecological health of compensation wetlands is lacking (Rubec and Hanson, 2009). Even in the United States, where wetland assessment tools are more advanced, wetland-monitoring programs are often difficult to implement and defend. In the United States, wetland assessment tools such as Indices of Biotic Integrity (IBIs) have been supported by sophisticated validation processes that include multiple testing iterations with multiple data sets across a range of spatial and temporal variability (Mack, 2007).

IBIs are biological monitoring and assessment tools that measure the biological integrity of biotic communities as a surrogate proxy of ecosystem health. The first IBI was developed by Karr (1981) to evaluate stream condition by monitoring fish assemblages in the Midwestern US. The approach has been adapted to assess wetlands using several common taxa including plants and birds. In this approach, potential biological metrics from an individual taxon or taxonomic group are tested for a relationship to a gradient of human influence. Human influence can be estimated by measuring environmental conditions based on local-level measurements of physical and chemical stressors across a range of sites spanning the gradient of human influence (Rooney and Bayley, 2010). Several biological metrics that exhibit the strongest relationship to degradation in environmental stress are then selected, standardized and combined into an IBI. IBIs need to be rigorously tested by verifying their relationship to the stress gradient on an independent suite of sites as well as in years with varying water levels (Wilcox et al., 2002). IBIs can be valuable tools for management agencies to report on the condition of a wetland, target wetlands for protection, prioritize and design mitigation projects, set baseline criteria for compensation wetlands and monitor compensation success. Moreover, wetlands can be parceled into health categories that are more practical for managers than an index score. Typically, studies have recommended preliminary wetland health categories that are often based on best professional judgment in order to satisfy management requirements, although Wardrop et al. (2007) have used an objective approach to define thresholds for wetland health categories.

Several potential taxonomic groups are often contrasted to determine suitable indicators that best predict human influence (i.e. Bryce et al., 2002; Cvetkovic and Chow-Fraser, 2011; Seilheimer et al., 2009). Multiple taxonomic groups need to be rigorously evaluated because the sensitivity of biota to human influence varies by region, wetland type, scale, and differing kinds of stressors. Plants are perhaps the most widely used biological indicator for wetland monitoring. Advantages of using plants as indicators as summarized by the USEPA (Adamus and Brandt, 1990; Nevel et al., 2004) include their presence in almost all wetlands, low cost of sampling and known sensitivity to specific stressors, while disadvantages include their limited social value, laborious sampling requirements, and lagged response to some stressors, which may allow plants to survive in poor conditions for many years. In addition, wetlands exhibit wet-dry cycles whereby plant communities change with water fluctuations (Stewart and Kantrud, 1971; Van der Valk, 2005). Some scientists have contended that this inter-annual variation in species composition would make IBI's based on plant metrics inconsistent (Wilcox et al., 2002).

The USEPA summarizes that the advantages of using birds as indicators include their high social value and the relative ease of surveying methods (Adamus and Brandt, 1990; Nevel et al., 2004). Furthermore, birds are known to respond to changes in habitat quality and are sensitive to landscape-scale disturbances (Mensing et al., 1998). Their differing home range areas and migratory behavior, however, could introduce uncertainties as to whether changes in species populations are due to local factors or events occurring elsewhere (Nevel et al., 2004). Other disadvantages of birds as indicators include the high level of training required for the auditory identification of passerines as well as sampling errors in point count methods due to differing detection probabilities among species, variability in observer detection and uncontrolled weather variables such as noise, time of day and weather. After rigorously testing for the best indicator, selection of a taxonomic group is ultimately based on policy, scientific considerations and biomonitoring objectives (Adamus and Brandt, 1990).

Although IBIs use indicators as a surrogate of ecosystem health, there is concern whether a single taxonomic group adequately reflects the biological integrity of other biota. For example, Brazner et al.'s study (2007) evaluating natural and anthropogenic variation associated with multiple taxonomic groups in the Great Lakes coastal wetlands found that birds did not have similar responses to other communities whereas wetland vegetation did have similar responses to other communities. Other studies have also found evidence that individual taxonomic groups have differing sensitivities to specific kinds of stress at differing spatial scales (Mensing et al., 1998; O'Connor et al., 2000; Yates and Bailey, 2011), resulting in congruence among taxonomic groups that is stress- and scale-dependent (Brazner et al., 2007). In Rooney and Bayley's (in review) community structure and concordance study, which examined a subset of the sites used in this study, 6 taxonomic groups responded to similar environmental variables although they were not strong surrogates of one another. A multi-assemblage approach could be necessary for a comprehensive monitoring program (Cairns et al., 1993), particularly if managers are interested in assessing the condition of more than a single taxonomic group (Borja and Dauer, 2008). However, the trade-off between increased cost and sampling effort and the amount of information gained may not be worthwhile, especially if taxonomic groups have similar responses to stress and are redundant. Furthermore, it may not be practical to monitor multiple taxonomic groups if the goal of monitoring is to reduce complex ecological information into a simple and applicable tool for policy and management objectives. Hence, a single taxonomic group could suffice to detect general improvement or degradation in site condition, while monitoring multiple taxonomic groups could provide a more comprehensive evaluation.

The research for this study involved sampling 5 taxonomic groups (vegetation from the wet meadow, emergent and open water zones, waterbirds and songbirds) at 81 sites made up of semipermanent and permanent natural and compensatory sites spanning a range of environmental stress. The first goal was to use the IBI approach to evaluate the potential of several individual taxonomic groups to assess wetland health in Alberta's Central Parkland Subregion. The second goal was to test whether the IBIs developed from a single taxonomic group were adequate surrogates of ecosystem health by evaluating whether a two-taxon IBI developed from plants and birds provided a more precise estimate of environmental stress than any single taxonomic group. I specifically hypothesized that 1) a gradient of biological condition would reflect underlying environmental conditions; 2) IBIs developed from several individual taxonomic groups would have varying sensitivities to underlying environmental conditions; and 3) combining IBIs developed from different taxonomic groups would be more precise than any single taxonomic group alone.

This is the first development and testing iteration for a wetland assessment tool in the Parkland Region of Canada.

2.2 Methods

Study area and sampling design

My study area was primarily located in Beaverhills Subwatershed of the Central Parkland Subregion of Alberta, which is part of the Aspen Parkland Ecoregion of Canada (Figure 2.1). The Aspen Parkland Ecoregion is a transition zone between the boreal region to the north and the prairie grasslands to the south, characterized by aspen and mixedwood forests and numerous freshwater wetlands in remnant natural areas. The landscape is dominated by agriculture as well as some urban and suburban areas (GOA, 2011). Climate in this region has a moisture-deficit regime where potential evapotranspiration exceeds annual precipitation (Hogg, 1994). Mean daily temperature between May and September was 14.1° C in 2008 and 12.8° C in 2009 (Environment Canada, 2011). Annual precipitation was 214.5 mm in 2008 and 172.4 mm in 2009, which was considerably lower than the mean of 338.9 mm between 1971 and 2000 (Environment Canada, 2011).

Out of a total of 81 semi-permanent and permanent sites sampled in 2008 and 2009, 45 were natural sites and 36 were compensation sites. Sites were between 1 and 13 ha, including the open water and the emergent vegetation zones. The wet meadow zone was not included in total wetland area measurements because it was too difficult to distinguish from surrounding upland in aerial imagery. Of the 45 natural wetlands, 27 were reference sites buffered by > 50% undisturbed forest within a 500 m radius, representing the region's least-disturbed range of natural variability. The remaining 18 natural wetlands were agricultural wetlands surrounded by > 50% cultivated or grazed land within a 500 m radius. Permission to access all sites on private land was obtained. The 36 compensation wetlands included 9 restored sites, 16 naturalized constructed sites and 11 stormwater management ponds. Naturalized constructed sites differ from older stormwater management ponds in that they are designed to mimic some appearances and functions of natural wetlands. Mean age of compensation wetlands was 17 years, all constructed sites were > 3 years old and only 5 naturalized constructed sites were < 7 years.

Biological sampling

Macrophytes were sampled between late July and August, when peak biomass is expected. Natural wetlands typically had 3-4 distinct zonations with differing vegetation communities: An open water zone with submersed and floating aquatic plants; a drawdown zone with annuals and re-colonizing emergent plants; an emergent zone with robust emergents (mainly *Typha latifolia*); and a wet meadow zone with a mixture of grasses, sedges and other herbaceous plants. Sampling from the drawdown zone was not included in analysis. Due to an extensive drought over the past several years, which eliminated the emergent vegetation at some sites, the sample size in the emergent zone was reduced to 57 rather than 81.

To sample macrophytes in the emergent and wet meadow zones, sites were divided into thirds by three radial transects. At each transect, two 1x1 m quadrats were sampled in the middle of every zone with quadrat pairs spaced 5 m apart, totaling 6 quadrats per zone (Raab and Bayley, 2012). An *a priori* single factor ANOVA power analysis found that a sample size of 4 quadrats in the wet meadow was needed to detect differences in richness among sites (N = 12, alpha = 0.05, power = 0.95, effect size = 0.72). In each quadrat, all herbaceous plants were identified to species following Moss and Packer (1983) and their percent cover was estimated to the nearest 5%. Width of each zone and average vegetation height per zone were also measured at each transect. In addition, the Robel technique was used as a proxy for above-ground biomass in the wet meadow zone following Raab and Bayley (2012). This technique, however, was not suitable for the taller emergent zone so a stem count of *Typha latifolia* was used instead.

Submersed aquatic vegetation (SAV) in the open water zone was sampled using the rake technique described by Rooney and Bayley (2011b). In this technique, the sampler navigated by kayak to 10 stratified-random locations in the open water zone >50 cm deep and made a 1m vertical sweep of the water column with a rake. SAV species collected on the rake were identified following Moss and Packer (1983) and its relative cover on the rake was estimated to the nearest 5%. At some restored and agricultural sites, SAV was sampled at depths <50 cm because of the drawdown that occurred in 2009. Species names were updated according to the International Code of Botanical Nomenclature.

Bird surveys were repeated at each site three times during the breeding season (May-June) between sunrise and 11:00 am. Two observers conducted a 10-minute visual survey from a vantage point of the entire open water zone, recording species abundance of waterfowl and other waterbirds (hereafter referred to as waterbirds). The two observers also performed two 8-minute surveys, recording the species of all auditory and visual detections of wetland-associated passerines and secretive waders (hereafter referred to as songbirds) within a 50 m fixed-radius point count. Point counts were located at the interface of the wet meadow and emergent zones. Time of visit was rotated during repeated visits (Forrest, 2010). Species names were updated according to the North American Classification Committee of the American Ornithologist's Union.

Physicochemical sampling

Water samples were collected at each wetland in late July and analyzed for nitrogen (ammonia ($NH4^+$), nitrate (NO_2NO_3), total nitrogen (TN), total dissolved nitrogen (TDN)), phosphorous (soluble reactive phosphorous (SRP), total phosphorous (TP), total dissolved phosphorous (TDP)), major ions (chloride, sulfate, sodium, potassium, calcium, magnesium, iron, aluminum), and others (dissolved organic carbon (DOC), silicone dioxide, non-filterable residue, chlorophyll-f). All water analysis methods were performed as described by

Bayley and Prather (2003). In addition, pH and conductivity were measured *in situ* with a Hach meter and turbidity was estimated with a secchi disk at 10 points in the open water zone.

Sediment cores were taken in July at the wet meadow-emergent zone interface and immediately frozen for further laboratory analysis. Samples consisted of a homogenized composite of three 10 cm cores with a 5.72 cm diameter suction corer. Total phosphorous (TP) was measured using the peroxide/sulfuric acid digestion method (Parkinson and Allen, 1975) and a Varian Cary 50 spectrophotometer. Homogenized composite samples were weighed to calculate bulk density. The wet and dry mass (drying oven set to 60°C for 48 h) of a 50g ± 1g sample (wet mass) was weighed to calculate the percent water content in the sediment. I used a Mettler Toledo AE240 balance (±0.0001 mg) to weigh a 0.5g sample that was placed in a muffle furnace at 550°C for 4 h and re-weighed to determine loss on ignition (Rooney and Bayley, 2010). Samples from sediment cores were also analyzed for percent nitrogen (% N) and carbon (% C).

Shoreline slope was determined at each vegetation transect by measuring the height of a horizontal laser beam 10 meters away from the edge of the open water and calculating the rise over run. Area of the emergent zone and the open water zone were estimated by digitizing aerial imagery in ESRI's ArcGIS 9.0 (2008).

IBI approach

The IBI approach was used to develop IBIs for several individual taxonomic groups for assessing wetland health. Three steps were involved in developing the

IBIs: 1) Define an environmental stress gradient based on physicochemical parameters; 2) Test, select, and score metrics; and 3) Validate the IBIs on independent test sites.

Development of an environmental stress gradient

The environmental stress gradient was developed based on each site's physical and chemical conditions using methodologies described in detail by Rooney and Bayley (2010). Out of 41 variables from 3 abiotic categories (physical structure, water chemistry and sediment chemistry) included in Principal Components Analysis (PCA), eight final variables most correlated with the first two ordination axes were: shoreline slope and proportion secchi depth (physical structure), NO_2NO_3, TN and conductivity (water chemistry), and % N, TP and % water content (sediment chemistry). Percentile binning was used to score each parameter between 1 and 5 based on its expected response to disturbance, following methods described by Rooney and Bayley (2010). Each of the 8 metric scores were then added together and rescaled to a score between 0 and 10.

Metric selection, standardization and scoring

Stratified-random selection was used to choose which sites were used for development and testing groups. The development bin comprised 54 sites and the testing bin comprised 27 sites. For each taxonomic group, I examined a large array of metrics for linear relationships to the stress gradient using scatterplots and linear regression. Metrics measured a variety of attributes (i.e. richness and composition, habitat quality) respresenting taxonomic/functional guild structure and community structure (Table 2.1). Transformations were done on proportion-based metrics to normalize the distribution and reduce heteroscedasticity of residuals. A log transformation was performed on the width of the wet meadow zone to reduce the variability of the residuals of higher values (Zarr, 1999). The Floristic Quality Assessment Index (FQAI) score was also calculated for all three vegetation zones based on previous work done in the Aspen Parkland by Forrest (2010). A site's FQAI is calculated by multiplying the mean coefficient of conservatism (C-value) by the square root of number of native species, where the C-value ranks each plant species based on its likelihood of being encountered in disturbed habitats (Lopez and Fennessy, 2002; Miller and Wardrop, 2006).

Approximately 65 metrics were tested for each of the 5 taxonomic groups (Table 2.1). Metrics with R^2-values > 0.2 were considered candidate metrics (Mack, 2007). The final selection of metrics was determined by performing redundancy analysis of candidate metrics using Pearson's correlations. In cases where two metrics had correlation coefficients > 0.7 (Rooney and Bayley, 2011a), only the metric with the largest R^2-value was retained for use in the IBI.

To score metrics, a continuous reference range approach was used that Blocksom (2003) describes in detail. In this approach, percentile binning is used to standardize metric values for each site relative to the reference condition, which is calculated as the difference between the 75^{th} percentile of reference sites (upper bound) and the 25^{th} percentile of constructed sites (lower bound) (Table 2.2). IBI scores were calculated by summing metric scores and rescaling them between 0 and 100.

Wetland health categories

Classification and regression tree analysis (CART) was used to group sites into health categories following methods described by Wardrop et al. (2007). CART builds a regression tree by recursively partitioning data into two mutually exclusive groups by choosing the predictor variable that best describes the response variable and providing a threshold value that splits the data into two groups. The stress gradient was used as the response variable and IBI score was used as the predictor variable. A separate analysis was run for each taxonomic group. Each model was pruned to yield 4 health categories: Exceptional, Good, Fair and Poor. This technique was used to evaluate IBIs based on the conservativeness of their health category thresholds.

Evaluation of IBIs developed from individual and multiple taxonomic groups

The IBIs were developed for individual taxonomic groups that had more than one sensitive metric. In addition, IBIs developed from wet meadow vegetation and songbirds were combined into a two-taxon IBI. The linear relationship between IBI and stress scores at test sites was used to verify that IBI scores were in fact good predictors of environmental stress. After verification, development and test sites were combined to determine the overall sensitivity between IBI and stress scores.

The IBIs developed from several individual and multiple taxonomic groups were evaluated based on the following criteria:

1) Consistency of the IBI's relationship to the stress gradient on the independent test sites
2) overall IBI sensitivity to the stress gradient
3) Correlation between IBIs developed from differing taxonomic groups
4) Conservativeness of wetland health categories.

The first criteria compared how consistent each IBI was at predicting stress scores across the environmental gradient while the second criteria compared the overall strength of relationship to the stress gradient. The third criterion examined the correlation among IBIs to evaluate whether certain taxonomic groups were good surrogates of one another. The fourth criteria contrasted differing health category thresholds, whereby more stringent thresholds were considered to be better because they represented a higher degree of conservativeness.

2.3 Results

A total of 139 macrophytes were found in the emergent and wet meadow zones (Appendix A) and 23 SAV species were found in the open water zone (Appendix B). Of the 87 wetland-associated birds encountered, 43 species were included as wetland-dependent songbirds (see Appendix C) and 44 species were included as waterbirds (see Appendix D). IBIs were developed for each taxonomic group by selecting several metrics that had a significant linear relationship to the environmental stress gradient.

Significant metrics

Out of 23 candidate wet meadow vegetation metrics, 4 non-redundant metrics were selected for IBI development (Table 2.2): the average width of the wet meadow zone ($R^2 = 0.65$, $p < 0.001$); the % of total cover of *Carex* species ($R^2 = 0.44$, $p < 0.001$); % of total cover of native perennials ($R^2 = 0.42$, $p < 0.001$); and the FQAI ($R^2 = 0.39$, $p < 0.001$). Two non-redundant metrics were chosen out of 16 candidate metrics from the emergent vegetation zone (Table 2.2): the FQAI ($R^2 = 0.43$, $p < 0.001$) and % cover of native species ($R^2 = 0.42$). No metrics from the SAV community were significant enough to produce an IBI that reflected underlying environmental conditions.

Five non-redundant songbird metrics were chosen from 19 candidate metrics (Table 2.2): % of total richness of insectivores/granivores ($R^2 = 0.47$, $p < 0.001$), % of total richness of ground nesting species ($R^2 = 0.44$, $p < 0.001$), number of temperate migratory species ($R^2 = 0.37$, $p < 0.001$), relative abundance of canopy foraging species ($R^2 = 0.33$, $p < 0.001$) and number of passerine species ($R^2 = 0.31$, $p < 0.001$). No metrics from the waterbird community were significant enough to produce an IBI.

Evaluation of IBIs developed from several individual taxonomic groups

Wet meadow vegetation, emergent vegetation and songbirds produced IBIs with multiple metrics that were sensitive to the stress gradient. The wet meadow vegetation IBI had the most consistent relationship to environmental stress on test sites ($R^2 = 0.72$, $p < 0.001$, Table 2.3). The songbird IBI also retained a strong relationship to the stress gradient on test sites ($R^2 = 0.50$, $p < 0.001$ Table 2.3).

The emergent vegetation IBI, however, had a weaker relationship to the stress gradient on test sites ($R^2 = 0.30$, $p < 0.05$, Table 2.3). The wet meadow vegetation IBI had the strongest overall relationship to environmental stress ($R^2 = 0.67$, $p < 0.001$, Table 2.3, Figure 2.2a), followed by the songbird IBI ($R^2 = 0.59$, $p < 0.001$, Table 2.3, Figure 2.2b) and then the emergent vegetation IBI ($R^2 = 0.40$, $p < 0.001$). There was a strong relationship between the songbird IBI and the wet meadow vegetation IBI ($R^2 = 0.56$, $p < 0.001$, Figure 2.3), indicating that songbirds and wet meadow vegetation are strong surrogates of each other.

CART analysis provided thresholds that delineated wetland health categories (Table 2.4). Although the wet meadow vegetation IBI had the most conservative threshold for sites in "Good" health (Figure 2.2a), the songbird IBI had the most conservative threshold for sites in “Exceptional” health (Figure 2.2b). The wet meadow vegetation IBI ranked nearly all reference sites as “Exceptional” while the songbird IBI ranked some reference sites as “Exceptional” and others as “Good” (Table 2.5). The wet meadow vegetation and songbird IBI had comparable thresholds for "Poor" and "Fair" categories. The emergent vegetation IBI had the least conservative thresholds for all health categories.

Evaluation of the IBI developed from two taxonomic groups

The two-taxon IBI combining IBIs developed from wet meadow vegetation and songbirds retained a consistent relationship to the stress gradient on test sites ($R^2 = 0.70$, $p < 0.001$, Table 2.3). The two-taxon IBI had a slightly stronger relationship ($R^2 = 0.73$, $p < 0.001$, Table 2.3, Figure 2.2c) than the wet meadow IBI ($R^2 = 0.67$) and songbird IBI ($R^2 = 0.59$) alone. In the CART analysis of wetland health categories, the two-taxon IBI had similar thresholds to the songbird IBI (Table 2.4, Figure 2.2c).

2.4 Discussion

Regional variability in the effectiveness of a biological community to reflect human disturbance will influence a manager's choice of which taxonomic group to monitor. As no biomonitoring study had yet been conducted for wetlands in the Central Parkland Subregion, I made no presuppositions as to which taxonomic group held the highest potential as an indicator of ecosystem health. Thus, a comprehensive study that evaluated the potential of several taxonomic groups was required for the first development and testing iteration of the IBI approach. Out of the 5 taxonomic groups tested, IBIs were developed from wet meadow vegetation, emergent vegetation and songbirds. The wet meadow vegetation IBI holds the highest indicator potential, as it had the most sensitive and consistent relationship to environmental stress, and can predict songbird IBI scores with relative success. This finding suggests that the songbird community is responding to changes in habitat quality captured by wet meadow vegetation IBI. Although the two-taxon IBI combining IBIs produced from wet meadow vegetation and songbirds had a slightly stronger relationship to environmental stress than any individual taxonomic group alone, the added information would not likely be worthwhile in consideration of the increased cost and effort of sampling both vegetation and birds.

Significant metrics

The width of the wet meadow zone, which decreased with environmental stress as suspected, has been used in previous IBIs developed in Alberta (Raab and Bayley, 2012). It is simple and straightforward to measure in the field and is advantageous because it can be measured by remote sensing. It is also a surrogate for the amount of habitat available to other biotic communities representing higher trophic levels (i.e. songbirds). As expected FQAI scores decreased with environmental stress in both the wet meadow and emergent zones. A previous study by Forrest (2010) provided coefficients of conservatism (C-values) for every wetland plant species in the Parkland Region of Alberta. C-values ranged between 0 to 10 and were assigned by a group of botanists (Forrest, 2010) based on a species fidelity to specific habitat types and tolerance to environmental stress (Lopez and Fennessy, 2002). FQAIs have been used as wetland indicators in previous IBI assessments (DeKeyser et al., 2003; Mack, 2007; Raab and Bayley, 2012), and although it required intensive sampling effort and identification skills, it is useful because it estimates habitat quality (Miller and Wardrop, 2006) that can subsequently influence higher trophic levels like songbirds. Metrics related to *Carex* and native species were also negatively correlated with the stress gradient as expected, since they are known to be sensitive to both agricultural and urban impacts (Galatowitsch et al., 1998) and have been used as indicators in other regions (DeKeyser et al., 2003; Simon et al., 2001). *Carex* species play a vital role in wetland functioning via nutrient uptake, cycling and primary production (Bernard et al., 1988) and are a dominant genus in marshes in the Aspen Parkland.

For the songbird metrics, richness and composition of insectivores/granivores, ground nesting species, temperate migratory species, canopy foraging species and Passeriformes all had negative linear relationships to environmental stress as expected. These metrics are likely correlated to the stress gradient because they are both influenced by corresponding changes in surrounding habitat and land use. Other bird community indices have been found to be strongly correlated with habitat (Canterbury et al., 2000). Nevertheless, all 5 metrics strongly reflected underlying environmental conditions.

Evaluation of IBIs developed from several individual taxonomic groups

Plant-based indices have been developed to assess wetland health in many jurisdictions in the United States (DeKeyser et al., 2003; Hargiss et al., 2008; Mack, 2007; Mack et al., 2008; Miller et al., 2006) as well as elsewhere in Alberta (Raab and Bayley, 2012; Rooney and Bayley, 2011a). Raab and Bayley (2012) similarly developed a successful IBI using wet meadow vegetation for natural and oil sands reclamation marshes in northern regions of Alberta. Out of the 4 metrics included in the wet meadow vegetation IBI, 3 metrics were either identical or similar to the boreal vegetation IBI (width of the wet meadow zone, FQAI and % of total cover of *Carex sp.*). This suggests potential for a single IBI to be applicable at larger scales and across different types of compensation wetlands, including reclamation and restored wetlands. The 9 restored sites included in this study shows preliminary evidence that they can be evaluated using the wet meadow vegetation IBI, although this finding needs to be verified with a larger sample size and variable water levels.

The wet meadow vegetation IBI had the highest overall sensitivity to environmental stress and retained the most consistent relationship on test sites. The wet meadow vegetation IBI also produced fairly conservative health category thresholds that managers could use to monitor wetland compliance and compensation success, prioritize sites for restoration, set benchmarks for compensation, and provide design criteria for wetland construction and restoration. At low stress ranges, the wet meadow IBI discriminated health categories among agricultural and reference sites (Table 2.5), suggesting that agricultural influence lowers a site's biological integrity. The emergent vegetation IBI had only 2 metrics after redundancy analysis, which is likely not sufficient to produce a robust IBI that is reliable over time in various water levels. Emergent vegetation structure is known to change in response to hydrologic fluctuation, whereby emergent vegetation dies off during periods of high water levels and germinates in new drawdown areas during dry phases (Van der Valk, 2005). During the sampling period, much of the vegetation in the emergent zone had died off as a result of a prolonged drought, and new emergent were germinating in drawdown zones. The logistical issues I encountered in the emergent zone support Wilcox et al.'s (2002) argument that hydrologic fluctuation would make plant metrics inconsistent from year to year. Moreover, the emergent vegetation IBI produced the least conservative thresholds for all health categories, confirming its inadequacy for wetland health assessment in this region.

An IBI could not be produced with the SAV community for the Central Parkland Subregion even though it was successfully developed for northern natural and reclamation marshes in Alberta (Rooney and Bayley, 2011a). SAV species likely were sensitive to hydrocarbon and salt-related toxicity in northern reclamation marshes, suggesting that differences in types of stressors likely led to differing results between studies. In my study, similar species of SAV were found at constructed sites and natural sites, potentially because SAV seeds and propagules are easily transported to constructed wetlands by waterfowl (Galatowitsch and Van der Valk, 1996). Furthermore, the number of total species found in the SAV community was low (23 species) in comparison to the number of emergent species found in the emergent and wet meadow zones (139 species).

The IBI developed from songbirds had the second highest sensitivity to the stress gradient after the wet meadow vegetation IBI. The songbird IBI had the most conservative threshold for sites in "Exceptional" health, suggesting that managers could use the songbird IBI to target high quality wetlands for protection and identify reference condition sites. Perhaps most interesting, however, is that the wet meadow IBI could predict songbird IBI scores and vise versa, supporting findings by Canterbury et al. (2000) that habitat indices are good surrogates of bird integrity. In contrast, Rooney and Bayley's (in review) related study on community composition and concordance determined that the composition of multiple assemblages reflected a similar set of environmental conditions, although multiple assemblages were only mediocre surrogates for each other. The constituent metrics in the wet meadow vegetation IBI (i.e. width of the wet meadow vegetation, FQAI, % of total species of ground nester) are likely good indicators of habitat quality and subsequently reflect other biotic communities.

This finding indicates that monitoring multiple taxonomic groups is not necessarily required when community concordance is low, as long as the IBI and its component metrics reflect the biological integrity of other taxonomic groups.

Although songbirds had multiple metrics that were sensitive to the stress gradient, an IBI could not be produced using waterbirds. Waterfowl are known to respond to broad-scale changes in habitat and food resources while the environmental stress gradient used in this study exclusively measured local physical and chemical conditions. Waterbird IBIs have been successfully developed to predict alterations in land-use (Glennon and Porter, 2005), which should be taken into consideration if managers are interested in monitoring waterfowl populations.

Evaluation of the IBI developed from two taxonomic groups

The two-taxon IBI that combined IBIs developed from wet meadow vegetation and songbirds had a marginally stronger relationship to the stress gradient than any single taxonomic group alone. However, monitoring multiple taxonomic groups would substantially raise the cost and effort of the wetland bioassessment program and could reduce the utility of the IBI. Different sampling methods and field training skills would be required to monitor breeding birds and vegetation, and differing times of year required for sampling would also induce logistical problems. The wet meadow IBI seems to hold the most potential for wetland assessment in the Aspen Parkland Ecoregion because it had a strong and very consistent relationship to environmental stress and had fairly conservative health category thresholds. The songbird IBI also had a strong relationship to the stress gradient and could be used interchangeably with the wet meadow vegetation IBI when sampling vegetation is not possible. Although the wet meadow and songbird IBIs do not need to be used in combination to predict local environmental conditions, they should both be available so that bioassessment can be done at different times during the spring and summer. Multiple taxonomic groups may still be necessary to provide more comprehensive assessments that predict biological condition in response to specific stressors at different spatial scales.

Conclusions

Although choosing a taxonomic group is contingent on management objectives, the results of this study indicate that indicator potential varies widely among taxonomic groups within the Central Parkland. I determined that wet meadow vegetation and songbirds are good indicators of environmental stress while emergent vegetation is a relatively weak indicator and SAV and waterbirds are poor indicators. Both the wet meadow and songbird IBIs revealed that constructed sites were in poorer biological health than most natural sites, reflecting unhealthy underlying environmental conditions. This finding suggests that wetland ecosystem health is deteriorating as constructed sites are replacing natural wetlands on the landscape. The wet meadow vegetation was the most sensitive taxonomic group to environmental stress and had the most consistent relationship on test sites. The wet meadow vegetation IBI and songbird IBI were strong surrogates of each other, even though they were found to have low community concordance. Contrary to my hypothesis that the two-taxon IBI would improve the strength of the relationship to the stress gradient, it added only slightly more information than the wet meadow vegetation IBI alone. This result probably occurred because wet meadow vegetation and songbird communities are strong surrogates of one another and contained redundant metrics. I argue that using multiple taxonomic groups would not be worthwhile because the added information would not offset the increase in cost and effort of sampling plants and birds. However, the wet meadow vegetation and songbird IBIs could be used interchangeably to extend the field-sampling period for bioassessment. Although multiple taxonomic groups may be needed to measure multiple types of stressors at different scales or to predict community concordance, I found that IBIs developed from either wet meadow vegetation and songbirds alone sufficiently reflected underlying environmental conditions and were fairly good predictors of each other

2.5 References

Adamus, P.R., Brandt, K.B., 1990. Impacts on Quality of Inland Wetlands of the United States: a Survey of Indicators, Techniques, and Applications of Community Level Biomonitoring Data. U.S. E.P.A., Environmental Research Laboratory.

Bayley, S.E., Prather, C.M., 2003. Do wetland lakes exhibit alternative stable states? Submersed aquatic vegetation and chlorophyll in western boreal shallow lakes. Limnol. Oceanogr. 48, 2335-2345.

Bernard, J.M., Solander, D., Kvet, J., 1988. Production and nutrient dynamics in Carex wetlands. Aquat. Bot. 30, 125-147.

Blocksom, K.A., 2003. A performance comparison of metric scoring methods for a multimetric index for Mid-Atlantic highlands streams. Environ. Manage. 31, 670-682.

Borja, A., Dauer, D.M., 2008. Assessing the environmental quality status in estuarine and coastal systems: Comparing methodologies and indices. Ecol. Indic. 8, - 337.

Brazner, J.C., Danz, N.P., Niemi, G.J., Regal, R.R., Trebitz, A.S., Howe, R.W., Hanowski, J.M., Johnson, L.B., Ciborowski, J.J.H., Johnston, C.A., Reavie, E.D., Brady, V.J., Sgro, G.V., 2007. Evaluation of geographic, geomorphic and human influences on Great Lakes wetland indicators: A multi-assemblage approach. Ecol. Indic. 7, 610-635.

Bryce, S.A., Hughes, R.M., Kaufmann, P.R., 2002. Development of a bird integrity index: Using bird assemblages as indicators of riparian condition. Environ. Manage. 30, 294-310.

Cairns, J., McCormick, P.V., Niederlehner, B.R., 1993. A proposed framework for developing indicators of ecosystem health. Hydrobiologia. 263, 1-44.

Canterbury, G.E., Martin, T.E., Petit, D.R., Petit, L.J., Bradford, D.F., 2000. Bird communities and habitat as ecological indicators of forest condition in regional monitoring. Conserv. Biol. 14, 544-558.

Cvetkovic, M., Chow-Fraser, P., 2011. Use of ecological indicators to assess the quality of Great Lakes coastal wetlands. Ecol. Indic. 11, 1609-1622.

Dahl, T.E., Watmough, M.D., 2007. Current approaches to wetland status and trends monitoring in prairie Canada and the continental United States of America. Can. J. Remote Sensing. 33, S17-S27.

DeKeyser, E.S., Kirby, D.R., Ell, M.J., 2003. An index of plant community integrity: Development of methodology for assessing prairie wetland plant communities. Ecol. Indic. 3, 119-133.

Environment Canada, 2011, The Weather Office, retrieved from http://www.theweatheroffice.gc.ca, November 01, 2011.

ESRI, 2008. ArcGIS 9.3. ESRI, Redlands, CA.

Forrest, A.S., 2010. Created stormwater wetlands as wetland compensation and a floristic quality approach to wetland condition assesssment in central Alberta. Biological Sciences, University of Alberta, M.Sc., 117 pp.

Galatowitsch, S.M., Van der Valk, A.G., 1996. The vegetation of restored and natural prairie wetlands. Ecol. Appl. 6, 102-112.

Galatowitsch, S.M., Whited, D.C., Lehtinen, R., Husveth, J., Schik, K., 1998. The vegetation of wet meadows in relation to their land-use. Environ. Monit. Assess. 60, 121-144.

Glennon, M.J., Porter, W.F., 2005. Effects of land use management of biotic integrity: An investigation of bird communities. Biol. Conserv. 126, 499-511.

GOA, 2011, retrieved from http://www.tpr.alberta.ca, November 2011.

Hargiss, C.L.M., DeKeyser, E.S., Kirby, D.R., Ell, M.J., 2008. Regional assessment of wetland plant communities using the index of plant community integrity. Ecol. Indic. 8, 303-307.

Hogg, E.H., 1994. Climate and the southern limit of the Western Canadian Boreal forest. Canadian Journal of Forest Research. 24, 1835-1845.

Karr, J.R., 1981. Assessment of biotic integrity using fish communities. Fisheries. 6, 21-27.

Kennedy, G., Mayer, T., 2002. Natural and constructed wetlands in Canada: An overview. Water Qual. Res. J. Can. 37, 295-325.

Lopez, R.D., Fennessy, M.S., 2002. Testing the floristic quality assessment index as an indicator of wetland condition. Ecol. Appl. 12, 487-497.

Mack, J.J., 2007. Developing a wetland IBI with statewide application after multiple testing iterations. Ecol. Indic. 7, 864-881.

Mack, J.J., Avdis, N.H., Braig, E.C., Johnson, D.L., 2008. Application of a vegetation-based index of biotic Integrity for Lake Erie coastal marshes in Ohio. Aquat. Ecosyst. Health Manage. 11, 91-104.

Mensing, D.M., Galatowitsch, S.M., Tester, J.R., 1998. Anthropogenic effects on the biodiversity of riparian wetlands of a northern temperate landscape. J. Environ. Manage. 53, 349-377.

Miller, S.J., Wardrop, D.H., 2006. Adapting the floristic quality assessment index to indicate anthropogenic disturbance in central Pennsylvania wetlands. Ecol. Indic. 6, 313-326.

Miller, S.J., Wardrop, D.H., Mahaney, W.M., Brooks, R.P., 2006. A plant-based index of biological integrity (IBI) for headwater wetlands in central Pennsylvania. Ecol. Indic. 6, 290-312.

Mitsch, W.J., Gosselink, J.G., 2007. Wetlands 4th ed., John Wiley & Sons, Inc., New York, NY, 582 pp.

Moss, E.H., Packer, J.G., 1983. Flora of Alberta, University of Toronto Press, Toronto, 687 pp.

Nevel, B., Milam, J., Arnold, G., Harris, R., 2004. Measuring Mitigation: A review of the science for compensatory mitigation performance standards. Prepared for the U.S. E.P.A., Environmental Law Institute.

O'Connor, R.J., Walls, T.E., Hughes, R.M., 2000. Using multiple taxonomic groups to index the ecological condition of lakes. Environ. Monit. Assess. 61, 207-228.

Parkinson, J.A., Allen, S.E., 1975. A wet oxidation procedure suitable for the determination of nitrogen and mineral nutrients in biological materials. Commun. Soil Sci. Plant Annal. 6, 1-11.

Raab, D.J., Bayley, S.E., 2012. Reclamation of wetland habitat in the Alberta oil sands: Generating assessment targets using boreal marsh vegetation communities. Ecol. Indic. 15, 43-51.

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

Rooney, R.C., Bayley, S.E., 2011a. 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. Environ. Monit. Assess. 184, 749-761.

Rooney, R.C., Bayley, S.E., 2011b. Setting reclamation targets and evaluating progress: Submersed aquatic vegetation in natural and post-oil sands mining wetlands in Alberta, Canada. Ecol. Eng. 37, 569-579.

Rooney, R.C., Bayley, S.E., in review. Community congruence of plants, invertebrates and birds in natural and constructed shallow open-water wetlands: do we need to monitor multiple assemblages? Ecol. Indic.

Rubec, C.D.A., Hanson, A.R., 2009. Wetland mitigation and compensation: Canadian experience. Wetlands Ecol. Manage. 17, 3-14.

Seilheimer, T.S., Mahoney, T.P., Chow-Fraser, P., 2009. Comparative study of ecological indices for assessing human-induced disturbance in coastal wetlands of the Laurentian Great Lakes. Ecol. Indic. 9, 81-91.

Simon, T.P., Stewart, P.M., Rothrock, P.E., 2001. Development of multimetric indices of biotic integrity for riverine and palustrine wetland plant communities along Southern Lake Michigan. Aquat. Ecosyst. Health Manage. 4, 293-309.

Stewart, R.E., Kantrud, H.A., 1971. Classification of natural ponds and lakes in the glaciated prairie region, U.S. Bureau of Sport Fisheries and Wildlife, Washington, DC, 57 pp.

Van der Valk, A.G., 2005. Water-level fluctuations in North American prairie wetlands. Hydrobiologia. 539, 171-188.

Wardrop, D.H., Kentula, M.E., Stevens Jr., D.L., Jensen, S.F., Brooks, R.P., 2007. Assessment of wetland condition: an example from the Upper Juniata watershed in Pennsylvania, USA. Wetlands. 27, 416-431.

Wilcox, D.A., Meeker, J.E., Hudson, P.L., Armitage, B.J., Black, M.G., Uzarski, D.G., 2002. Hydrologic variability and the application of index of biotic integrity metrics to wetlands: A Great Lakes evaluation. Wetlands. 22, 588-615.

Yates, A.G., Bailey, R.C., 2011. Effects of taxonomic group, spatial scale and descriptor on the relationship between human activity and stream biota. Ecol. Indic. 11, 759-771.

Zarr, 1999. Biostatistical Analysis, Prentice-Hall Inc., Upper Saddle River, NJ, 663 pp.

Table 2.1. Types of metrics that were tested for a linear relationship to the environmental stress gradient. Taxonomic groups with candidate metrics ($R^2 > 0.2$) in each category are underlined. WM = wet meadow zone vegetation, EM = emergent zone vegetation, SAV = submersed aquatic vegetation, SB = songbirds, WB = waterbirds

Type of metric	Taxonomic group
Biological productivity	
Robel height	WM
Typha latifolia stem count	EM
Community richness and composition	
Diversity indices	WM, EM, SB, WF, SAV
Total species richness	WM, EM, SB, WF, SAV
Total species abundance/percent cover	WM, EM, MB, WF, SAV
Native sp.	WM, EM, SAV
Invasive sp.	WM, EM
Taxonomic/functional guilds	
Group (i.e. monocot, dicot)	WM, EM, SAV
Life cycle (i.e. annual, perennial)	WM, EM, SAV
Growth habit (i.e. forb, graminoid)	WM, EM, SAV
Dietary need (e.g. carnivore)	SB, WB
Foraging mode (e.g. aerial)	SB, WB
Nesting location (e.g. ground)	SB, WB
Carex sp.	WM, EM
Sensitivity/tolerance	
Width of vegetation zone	WM, EM
Facultative wetland or obligate species	WM, EM, SB, WB, SAV
Sensitive/tolerant species	WM, EM, SAV
Species of concern	SB, WB

Table 2.2. Individual metrics selected for use in IBIs and their linear relationship to the environmental stress gradient. Upper and lower bounds are untransformed values used to calculate metrics scores. Upper bounds represent the 75th percentile of reference sites and lower bounds represent the 25th percentile of all constructed sites. WM = wet meadow zone vegetation, EM = emergent zone vegetation, SB = songbirds. *** $p < 0.001$.

Metric	Taxonomic group IBI	R^2	Upper bound	Lower bound
Width of wet meadow zone (meters)	WM	0.65***	31.2	1.8
% of total species of insectivores/granivores	SB	0.47***	85.7	33.3
% of total cover of *Carex* spp.	WM	0.44***	41.5	0.0
% of total species of ground nesting spp.	SB	0.44***	88.8	37.3
FQAI	EM	0.43***	11.3	2.4
% cover of native species	EM	0.42***	60.4	3.3
FQAI	WM	0.39***	13.3	4.0
No. of temperate migratory spp.	SB	0.37***	2	0
% of total cover of native perennials	WM	0.35***	90.7	17.6
% abundance of canopy foraging spp.	SB	0.35***	20.9	0.0
No. of Passeriformes spp.	SB	0.31***	7	3

Table 2.3. Linear relationship (R^2 values) between IBIs and the environmental stress gradient. *** $p < 0.001$, ** $p < .001$, * $p < 0.05$. Two-taxon IBI = meadow vegetation + songbirds

Taxonomic group IBI	Development	Testing	All sites
Wet meadow zone vegetation	0.66***	0.73***	0.67***
Emergent zone vegetation	0.46***	0.30*	0.41***
Wetland-dependent songbirds	0.65***	0.50***	0.59***
Two-taxon IBI	0.72***	0.70***	0.73***

Table 2.4. Wetland health category scoring ranges using IBI scores (scale between 0 and 100). Thresholds were determined by classification and regression tree analysis (CART). WM = wet meadow zone vegetation, EM = emergent zone vegetation, SB = songbirds, two-taxon = WM + SB.

Health Category	WM-IBI	EM-IBI	SB-IBI	Two-taxon IBI
Exceptional	> 69	> 62	> 81	> 80
Good	> 51 and < 70	> 21 and < 63	> 37 and < 82	> 40 and < 81
Fair	> 25 and < 52	> 11 and < 22	> 21 and < 38	> 29 and < 41
Poor	< 26	< 12	< 22	< 30

Table 2.5. Percentage of sites in each wetland health category. Reference sites made up 33% of total sites, agricultural sites made up 22% of total sites, restored sites made up 11% of total sites, naturalized constructed ponds made up 20% of total sites and stormwater ponds made up 14% of total sites. WM = wet meadow zone vegetation, EM = emergent zone vegetation, SB = songbirds, Two-taxon = WM + SB.

Health Category	WM-IBI	EM-IBI	SB-IBI	Two-taxon IBI
Exceptional	37	37	16	19
Good	19	42	47	43
Fair	23	8	19	12
Poor	21	12	19	26

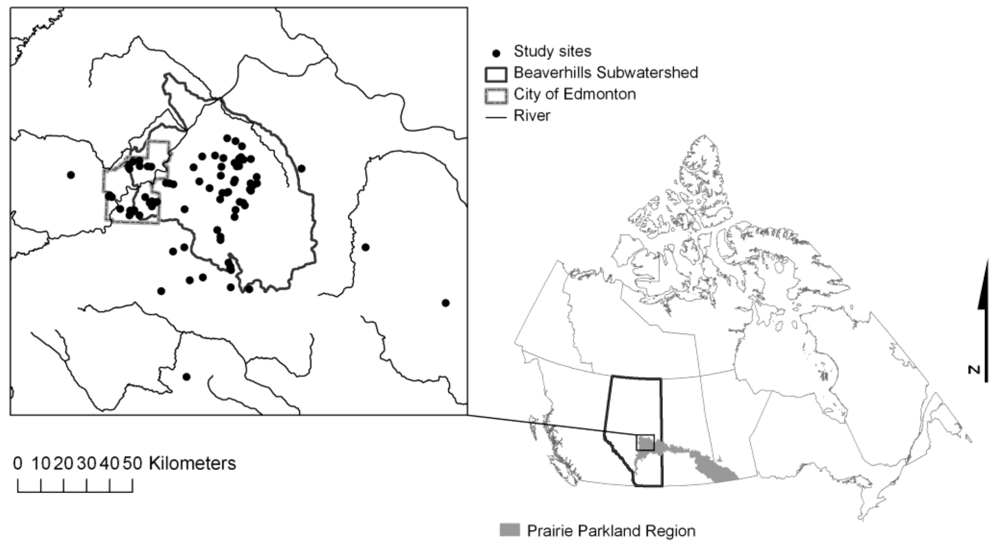


Figure 2.1. Location of Aspen Parkland Ecoregion in Canada and inset of study sites in the Beaverhills Subwatershed of the Central Parkland Subregion of Alberta.

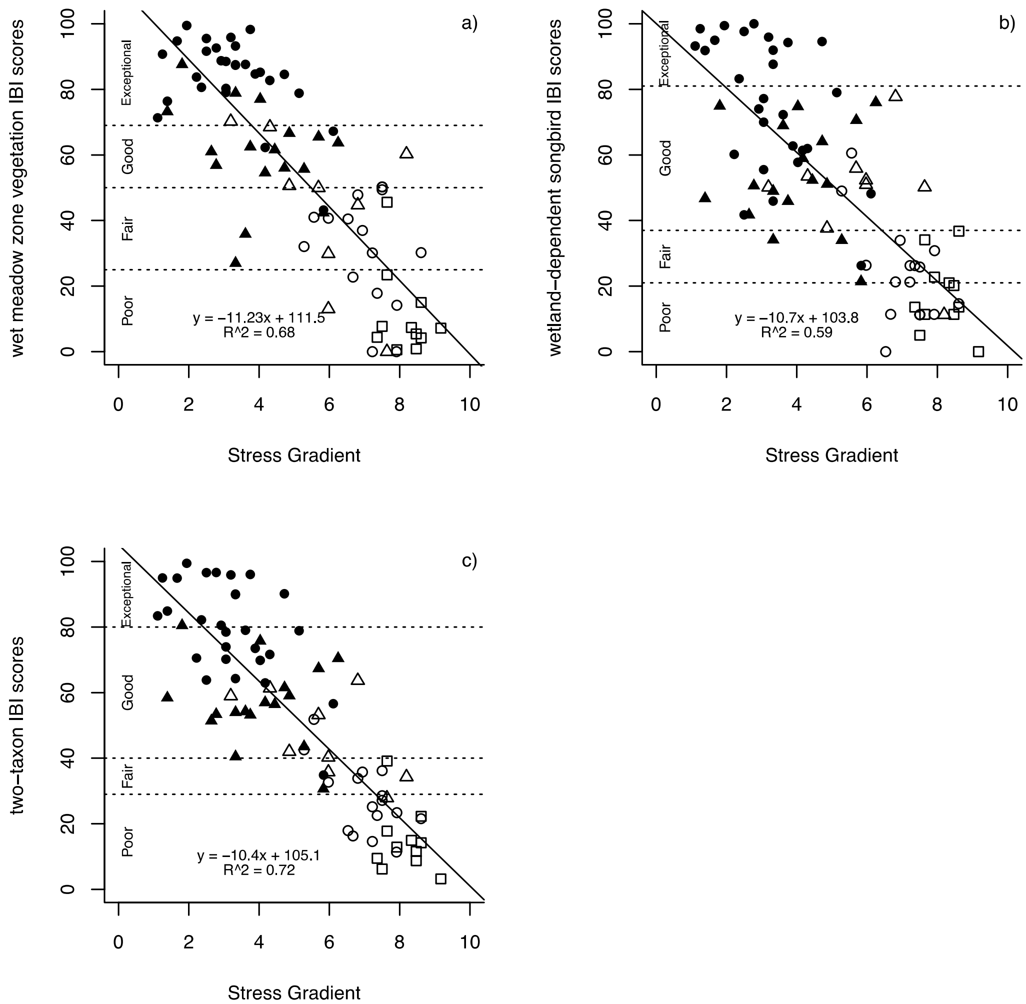


Figure 2.2. Site IBI scores regressed against stress scores for IBIs developed from a) wet meadow vegetation; b) wetland-dependent songbirds, and c) two-taxons (wet meadow vegetation + wetland-dependent songbirds). Thresholds for health categories are shown as defined by classification and regression tree analysis (CART). Legend: ● = reference; ▲= agricultural, △ = restored, ○= naturalized constructed ponds and □ = stormwater ponds

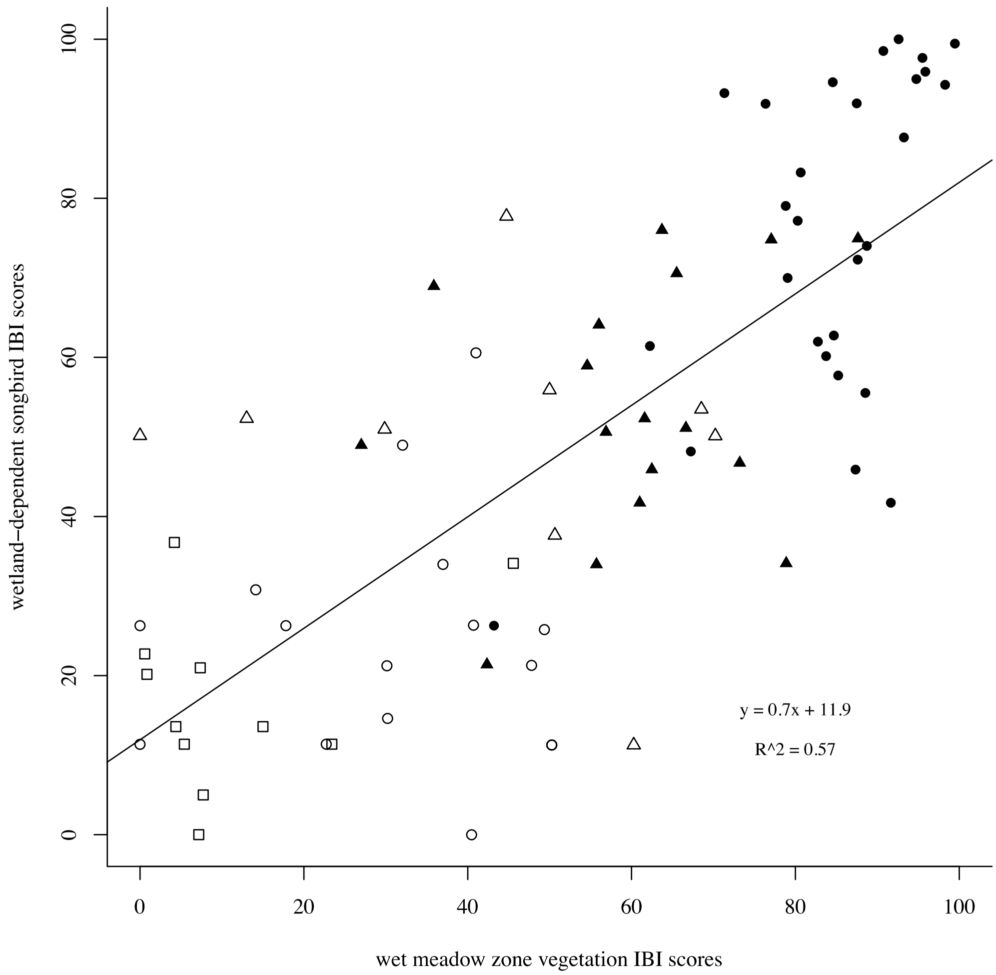


Figure 2.3. Linear relationship between songbird IBI scores and the wet meadow vegetation IBI scores. Legend: ● = reference; ▲= agricultural, △= restored, ○= naturalized constructed and □ = stormwater ponds

3. *Reproducibility of a wet meadow vegetation IBI to assess wetlands in wet and dry conditions*

3.1 Introduction

Wetlands provide significant functions and values in terms of water storage and flood attenuation, ground water recharge, element cycling and wildlife habitat (Keddy, 2000; Wray and Bayley, 2006). Wetland plant communities play a major role in water quality improvement (Kirby et al., 2002) and support a rich biodiversity of organisms (Gibbs, 1995). Wildlife such as songbirds and waterfowl rely on wetland plant communities for foraging, nesting and breeding habitat. Agricultural pressures, however, have led to extensive loss and deterioration of wetlands. Approximately 70% of wetlands have been converted for agricultural reclamation in the prairie region of Canada (Kennedy and Mayer, 2002). In Alberta, compensatory some constructed stormwater management ponds are accepted as compensation for the loss or alteration of natural wetlands. To ensure the proper management of healthy ecosystems and provisioning of wetland functions and values, biological monitoring tools are needed that can accurately quantify ecosystem health to assess whether constructed wetlands are adequately replacing natural wetland functions and values.

Initiatives to monitor ecological health stemmed from the growing concern for biodiversity in aquatic systems (Karr, 1993). Karr (1981) introduced an approach to measure ecological health, called the Index of Biotic Integrity (IBI), to assess fish communities in streams and rivers in Midwestern US. IBIs integrate several biological metrics, defined as measurable attributes that are sensitive to changes in environmental disturbance (Karr and Chu, 1999), into a multi-metric IBI to indicate a site's biological integrity. IBIs use a comprehensive suite of metrics that together can discriminate the "signal," or biological condition, from "noise" induced by natural variation (Karr and Chu, 1999). Multi-metric indices are based on the premise that an IBI's precision improves by adding metrics together (Karr, 1993).

IBIs have been widely adapted to wetlands in North America using plants as indicators of ecosystem health. Plant-based IBIs have been reported as effective tools to assess various types of wetlands in diverse regions, including coastal wetlands of the Great Lakes (Simon et al., 2001), wetlands across the state of Ohio (Mack, 2007), headwater wetlands in Pennsylvania (Miller et al., 2006) and prairie pothole wetlands (DeKeyser et al., 2003; Hargiss et al., 2008). While plant-based IBIs have been broadly adopted by several jurisdictions in the United States (e.g. Ohio, Minnesota, Michigan, North Dakota), some wetland scientists have criticized them for being sensitive to inter-annual variation in water levels.

Vegetation in wetlands is closely connected with water levels (Hutchinson, 1975; Keddy, 2000). Differing functional groups (e.g. emergent species, drawdown species) occupy certain elevation ranges based on moisture gradients, making discernable zonations (Hutchinson, 1975; Van der Valk, 2005). For example, in a normal emergent phase submersed and floating aquatic vegetation inhabit the open water zone, robust emergent plants occupy deep marsh areas and fine-textured grasses, rushes and sedges occupy the wet meadow zone (Stewart and Kantrud, 1971).

Wetlands are routinely subject to wet-dry cycles, whereby plant communities change in response to dynamic inter-annual variation in water level and hydroperiod (Euliss et al., 2004). Wet-dry cycles are known to be an important part of wetland functioning. For instance, droughts provide conditions for germination and regeneration of emergent plants from seed banks (Van der Valk and Davis, 1978) and enhance energy and nutrient flow in wetlands by altering plant community composition and production (Laubhan and Roelle, 2001). The benefits of these processes are well known to wetland scientists; purposefully draining and flooding wetlands is a common management tactic to enhance waterfowl populations (Kadlec, 1962).

A major challenge to wetland bioassessments is to ensure IBIs are not sensitive to plant community changes driven by inter-annual climatic variation. If wetland bioassessments are confounded by natural climatic variability, it makes it difficult to tell if a change in IBI score is due to human disturbance or natural variation in plant community structure. Inaccurate IBI scores could have serious implications for the management and conservation of wetlands. Wilcox et (2002) concluded that fluctuating IBI scores in coastal wetlands of the Great Lakes were due to confounding variation caused by changing lake levels. Likewise, Euliss and Mushet (2011) found that an Index of Plant Community Integrity (IPCI) developed to assess temporary to semi-permanent prairie pothole wetlands was also sensitive to climatic variability, with sites improving as water levels increased.

The ability to produce IBIs that are robust to inter-annual variation could vary by region and wetland type. Regional difference in climate could permit IBIs to be developed more easily in the Aspen Parkland Ecoregion than elsewhere in the Prairie Pothole Region. For example, The Aspen Parkland has less extreme moisture deficits (0-15mm; Hogg, 1994) than the grassland prairies, where evapotranspiration exceeds precipitation by 30mm (Winter, 1989). IBI scores are probably only robust to natural variation within a limited range, which potentially could make IBIs more useful for assessing semi-permanent and permanent wetlands than temporary and seasonal wetlands. Semi-permanent and permanent wetlands do not dry out as often as temporary and seasonal wetlands, and can have more stable water levels (Euliss and Mushet, 1996) if they receive water inputs from the groundwater (Euliss and Mushet, 1996; Smerdon et al., 2005). In addition, temporary and seasonal wetlands exhibit water loss to both groundwater recharge and evapotranspiration (Euliss and Mushet, 1996) whereas more permanent wetlands are usually areas of groundwater discharge (LaBaugh et al., 1996).

Monitoring wetland vegetation by sampling along natural moisture gradients could also negatively influence an IBI's ability to sufficiently distinguish human disturbance from natural variation. Uzarski et al. (2004) recommended the calculation of richness metrics by plant zone to control for the number of inundated zones present at a site. Two IBIs were developed for reclamation oil sands marshes in the boreal region of Alberta by monitoring the wet meadow zone plant community alone (Raab and Bayley, 2012) and open-water zone plant community alone (Rooney and Bayley, 2011). I hypothesize that an IBI that calculates metrics from a single zone could factor out some natural variation induced by sampling along the entire moisture gradient.

The goal of this study is to determine whether the wet meadow vegetation IBI developed to assess the health of semi-permanent and permanent wetlands in dry conditions could be used to assess wetland health in wet conditions. The detailed objectives of this study are to 1) examine whether the wet meadow vegetation IBI is sensitive to plant community changes in the wet meadow zone between dry and wet years; 2) to test the reproducibility of individual component metric scores and IBI scores in dry to wet conditions; and 3) identify problematic metrics that have high variability and need modification or replacement. I hypothesize that if IBI scores are relatively insensitive to changes in the plant community structure related to wet-dry cycles, then they should accurately reflect biological integrity. In contrast, if IBI scores are sensitive to wet-dry cycles, then a change in IBI scores cannot be taken as evidence of a change in biological integrity.

3.2 Methods

Study sites and climatic variability

Land use in the Central Parkland Subregion is dominated by agriculture interspersed with urban and industrial development as well as parks and protected areas. Relatively undisturbed or least-impacted wetlands, which I will refer to as reference wetlands, were surrounded by >50% forest cover within a 500 m buffer. Reference sites were mostly restricted to protected areas of aspen and mixedwood forests. Agricultural sites were surrounded by >50% agriculture within a 500 m buffer. Constructed ponds were all constructed stormwater management ponds surrounding predominantly by urban development. Half of the constructed sites were "naturalized" to mimic the vegetation structure of natural wetlands while the other half were classic stormwater management facilities. Sites were between 1 and 10 ha, at least 1km apart and within 60 km of Edmonton. All sites were semi-permanent/permanent wetlands containing standing water throughout the entire sampling period. Mean age of constructed wetlands was 20.5 years and only 3 sites were < 7 years old.

The Aspen Parkland Ecoregion has intermediate climate and vegetation characteristics between that of drier grasslands to the south and moister boreal forests to the north (GOA, 2011). Mean temperature between May and September was approximately 14° in all 4 years. Annual precipitation (September-August) was low in 2008 and 2009 (250-350 mm) and near-average in 2010 and 2011 (400-500 mm) (AgroClimatic Information Service). Annual precipitation increased over the 4 years (Figure 3.1a). The highest amount of precipitation fell in the summer in all 4 years (Figure 3.1b).

Sampling methods

IBIs were initially developed from several taxonomic groups using data sampled on a suite of test and reference sites in 2008-09. Although I found that the IBI developed from plants in the wet meadow zone had the strongest linear relationship to environmental stress, I was concerned that the wet meadow vegetation IBI might be sensitive to plant community changes related to increasing precipitation. During the summer of 2010, which had higher precipitation than the previous two summers (Figure 3.1b), I resampled 32 sites (8 reference, 8 agricultural and 16 constructed sites) that were initially sampled in 2008. Then, a heavy and sustained rainfall in July of 2011 (Figure 3.1b) prompted us to increase sampling effort to capture a larger range of climatic variability at natural sites, where I suspected changes in plant community structure would be most evident. Hence, in 2011 I resampled a new subset of wetlands (8 reference and 8 agricultural sites) initially sampled in 2009 during the driest conditions. Sites were chosen based on accessibility and approval from landowners to revisit the site. A Mann-Whitney U test was performed to ensure that IBI scores in 2008 and 2009 were similar (p-value = 0.69) and that IBI scores in 2010 and 2011 were similar (p-value = 0.50). A Mann-Whitney U test had to be used because consecutive years did not have paired samples whereas alternate years did. Hereafter, I will call the initial sampling years of 2008 and 2009 *dry* years and subsequent resampling years of 2010 and 2011 *wet* years. Thus, a total of 48 wetlands were resampled in *dry* and *wet* years, and were distributed evenly among wetland types: (16 reference, 16 agricultural and 16 constructed). One agricultural site was removed because it dried up during the sampling period and thus was probably of an intermediate class between a seasonal and semi-permanent wetland. Two other constructed sites were also removed from analysis due to the lack of a wet meadow zone in wet years.

To sample wet meadow vegetation, 3 evenly spaced radial transects were chosen. At each transect, two 1x1 m quadrats were sampled in the middle of the wet meadow zone, with quadrat pairs spaced 5 m apart. This totaled 6 quadrats per wetland, which power analysis showed was sufficient to detect among-site variance (see methods in chapter 2). All herbaceous plants present in the quadrat were identified to species following Moss and Packer (1983) and species percent cover was estimated to the nearest 5%. Width of the wet meadow zone was also measured at each transect and averaged per site.

Calculating IBI scores and wetland health categories

I calculated wet meadow vegetation IBI scores as described in Chapter 2. Four non-redundant metrics that exhibited the strongest sensitivity to an environmental stress gradient were selected for use in the wet meadow vegetation IBI (see Chapter 1). These metrics were:

1) Width of the wet meadow zone
2) Floristic Quality Assessment Index (FQAI)
3) % of total cover of *Carex* sp.
4) % of total cover of native sp.

Individual metrics were scored on a continuous scale using the reference range approach explained in Chapter 2. The individual metric scores were summed and rescaled between 0 and 100. Sites were also placed into health categories (see Table 2.4 of Chapter 2) by objectively defining thresholds using classification and regression tree analysis (CART) (see methods section of Chapter 2). Sites fell into 1 of 4 health categories based on their IBI scores (health category ranges are in parenthesis): “Exceptional” (> 70), “Good” (51-69), “Fair” (26-51), and “Poor” (< 25).

Sensitivity of IBI scores to inter-annual variability in plant community

Multivariate analysis was used to depict whether IBI scores were sensitive to plant community changes in the wet meadow zone as a result of dry and wet conditions. Nonmetric multidimensional scaling (NMS) was performed on community data using the software package PC-ORD version 6 (McCune and Mefford, 2011). Species abundance was relativized by its maximum abundance and all rare species found in <5% of the total number of quadrats were eliminated as recommended by McCune and Grace (2002). This latter step pared the number of species in the ordination from over 300 species down to 45 species. The number of dimensions in the final solution was determined by running the NMS ordination 50 times with real data and 50 times with randomized data for 1 to 4-dimensional configurations using Bray-Curtis distances. NMS ordinations position sites in species space with distances between them as a linear function of their dissimilarity in species composition. Hence, sites consisting of similar community composition are plotted close together while sites with dissimilar composition are plotted further apart. Species whose abundances were reasonably correlated ($R^2 > 0.2$) were identified with community gradients summarized by ordination axes. A rigid rotation was then performed to align the IBI with the ordination's first axis. This procedure aided interpretation but did not alter the relative positions of sites in species space (McCune and Grace, 2002). Identical sites were connected by vectors to display each site's dissimilarity in community composition in dry to wet conditions. Changes in community composition within sites were determined by calculating the absolute change in coordinate position along each axis. Site changes were summed and expressed as a proportion of total change in inter-annual variation represented by each axis. To aid the visual interpretation of the NMS ordination and vector diagrams, I also calculated Clarke's *R* for each site, which is a non-parametric statistic able to test for a significant change in community composition over time (Clarke, 1993). To calculate Clarke's *R*-values, inter-annual variance in community composition (same site, different years) was compared to variance within replicates (same site, same year). I tested the significance of Clarke's *R*-values using Monte Carlo testing with 999 randomized permutations.

Reproducibility of the wet meadow vegetation IBI

Initial and subsequent IBI scores in dry versus wet years were compared to assess the reproducibility of the wet meadow vegetation IBI and its constituent metrics. Specifically, Pearson's correlation tests were used to evaluate if IBI scores in wet years were correlated with IBI scores in dry years. Wilcoxon's test was used to evaluate if IBI scores ranked sites in a consistent order in dry and wet years. A signal to noise test was performed to compare the among-site variance, indicating variation in biological integrity among sites, with the within-site variance, which included inter-annual variation as well as other sources of error. High signal to noise ratios indicate higher precision whereas low signal to noise ratios indicate high within-site variability. A signal to noise ratio >3 was considered to be adequate based on criteria defined by Whittier et al. (2007) for IBIs in streams and rivers in the western US. Each of these three tests (i.e.

Pearson's correlation, Wilcoxon's test, signal to noise test) was also performed on individual metrics in the IBI to identify problematic metrics.

Natural and constructed sites may respond differently to climatic variability since constructed stormwater ponds do not exhibit natural hydroperiods. Analysis of variance (ANOVA) was used to test whether the reproducibility of IBI scores was similar among site types. Levene's test was performed on individual metric and IBI scores to ensure homogeneity of variance. For each ANOVA, the absolute difference in metric or IBI score between dry and wet years was used as the response variable. Tukey's *post-hoc* test determined which site types differed.

3.3 Results

Sensitivity of IBI scores to inter-annual variability in plant community

I tested whether IBI scores were sensitive to plant community changes resulting from dry and wet conditions. NMS ordinations were used to depict compositional changes in the wet meadow vegetation community among sites and to identify species abundances that were correlated with community gradients (see Appendix E). The best solution for the NMS ordination had 3 dimensions. The final 3-dimensional solution had a stress of 18.16, an instability of < 0.00001 and ran for 62 iterations. Stress is a measure of goodness of fit, whereby lower stress indicates a better solution. Low instability indicates a higher likelihood that the ordination found a global minimum rather than local minima. The final solution represented 69% of the cumulative variance, of which 34% was represented by axis 1, 20% by axis 2 and 14% by axis 3. The wet meadow vegetation IBI was strongly correlated with the ordination's 1^{st} axis (Pearson's R = 0.87). The abundance of *Carex atherodes* and *Calamagrostis stricta* were positively correlated with the ordination's 1^{st} axis while the abundance of *Hordeum jubatum* was negatively correlated with the 1^{st} axis. The NMS's 2^{nd} axis was positively correlated with the abundance of *Carex aquatilis* and negatively correlated with the abundance *Carex atherodes*, and the abundance of *Calamagrostis stricta* was positively correlated with the NMS's 3^{rd} axis.

Inter-annual variation in vegetation community structure occurred at many sites between dry and wet conditions (Table 3.1, Figure 3.2). Clarke's *R*-values (Table 3.1) revealed that community structure significantly changed at 57% of the sites between dry and wet years. Changes in community structure were predominantly correlated with the ordination's 2^{nd} and 3^{rd} axis (Figure 3.2). IBI scores, which were correlated with the 1^{st} axis, seemed to be fairly insensitive to these shifts in plant community structure within sites (Figure 3.2). Out of the total within-site change in plant community structure from dry to wet years, 80% was represented along the ordination's 2^{nd} and 3^{rd} axes representing change in abundance of *Carex* species and *Calamagrostis stricta*. In contrast, only 20% of the plant community changes were represented by the 1^{st} axis (which most represented IBI scores).

Reproducibility of the wet meadow vegetation IBI

Average IBI scores were similar under dry and wet conditions, increasing across all sites by 4% from dry to wet conditions (Table 3.2). IBI scores at reference sites decreased by 2%, whereas IBI scores at agricultural and constructed sites both improved by 7% (Table 3.2). The inter-annual difference in IBI score between dry and wet conditions, however, was variable, ranging between 0 and 41% (Table 3.2, see Appendix F for individual site scores). Sites were placed into health categories (Exceptional, Good, Fair, Poor) determined by CART analysis. In total, 67% of sites stayed in the same health category in dry and wet years. Of the 33% of sites that changed health categories, 4% were reference sites, 13% were agricultural sites, and 16% were constructed sites. Sites never moved more than one health category. Ten of thirteen sites that changed health categories improved with wetter conditions.

Overall, IBI scores strongly corresponded between dry and wet conditions (Pearson's R = 0.84, Figure 3.2). Looking at constituent metrics individually, width of the wet meadow zone was most correlated (R = 0.84) and % of total cover of native perennials was least correlated (R = 0.56) between dry and wet conditions (Figure 3.3). Wilcoxon's test demonstrated that IBI scores calculated in dry and wet conditions ranked sites in a similar order, as did each of the IBI's constituent metrics (Table 3.2). The wet meadow vegetation IBI had a signal to noise ratio of 6.2 (Figure 3.4), indicating a fairly strong signal or effect among sites compared to the effect of year (dry vs. wet). Individual constituent metrics had signal to noise ratios between 2.2 and 5.9 (Figure 3.4).

I tested whether differences in the reproducibility of IBI scores existed among site types. Mean difference in IBI scores was similar among site types ($F_{2,41}$ = 3.05, p-value = 0.06). As for individual metrics, width of the wet meadow zone significantly differed between reference and agricultural sites ($F_{2, 41}$ = 3.2, p-value = 0.05) and % of total native perennials differed between reference and constructed sites ($F_{2, 41}$ = 4.4, p-value = 0.02).

3.4 Discussion

Plants are perhaps the most widely used indicator to assess wetland health. Although plant-based IBIs have been reported to produce precise estimates of biological integrity (DeKeyser et al., 2003; Mack, 2007; Miller et al., 2006; Simon et al., 2001), few studies have assessed inter-annual variability in IBI score. A few studies, however, have criticized plant-based IBIs for not being able to adequately partition anthropogenic disturbance from natural variation induced by variable weather conditions. Wilcox et al. (2002) first presented this issue for Great Lake's coastal wetlands and surmised that IBI scores would not be reproducible over time because variability in water levels would change plant communities and invalidate metric scoring ranges. More recently, Euliss and Mushet (2011) evaluated inter-annual variability in IPCI scores for temporary to semi-permanent prairie pothole wetlands (DeKeyser et al., 2003; Hargiss et al., 2008; Kirby and DeKeyser, 2003), concluding that the IBI was sensitive to changes in plant communities related to water levels. Wilcox et al. (2002) and Euliss and Mushet (2011) both argued that these plant-based IBIs did not meet two biological monitoring criteria defined by Herricks and Shaefer (1985) in that they failed to give reproducible estimates of biological integrity and the variability in IBI scores was too high. Comparing IBI scores for the same sites measured in both initial dry years and subsequent wet years enabled us to evaluate whether the wet meadow vegetation IBI's met these two criteria.

Sensitivity of IBI scores inter-annual variability in plant community

I found that plant community composition changed as a result of inter-annual variation in precipitation. Inter-annual variation in climate conditions are known to cause dynamic shifts in water level and hydroperiod (especially in prairie pothole wetlands), resulting in changes in plant community structure (Euliss et al., 2004; Van der Valk, 2005; Van der Valk and Davis, 1978; Wilcox, 2004). If the metrics used in plant-based IBIs are sensitive to changes in community structure caused by climatic variability, then it is difficult to discern whether changes in IBI scores are due to changes in biological condition or natural variation. In other words, IBI scores would be unable to partition anthropogenic stress from inter-annual variation in the plant community.
NMS ordinations revealed that the wet meadow vegetation community did change from dry to wet conditions (more than 50% of the sites had significant community shifts). IBI scores, however, were relatively insensitive to these inter-annual changes in community structure. In fact, IBI scores seemed to be able to differentiate plant community changes caused by anthropogenic disturbance from plant community changes related to natural climatic variability. For instance, the 1st NMS axis (which most represents IBI scores) was negatively correlated with abundance of *Hordeum jubatum* and positively correlated with abundance of *Carex atherodes*. Since *H. jubatum* is adapted to a wide range of environmental conditions (Best et al., 1978) and *C. atherodes* is an obligate wetland species, variance in the 1st NMS axis likely reflects plant community changes in response to a gradient of anthropogenic disturbance. This plant community gradient could

also be related to differences in life history traits that reflect changes from dry to wet conditions. For example, *Hordeum jubatum* is a perennial that propogates by seed whereas *Carex atherodes* is a clonal perennial. Differences in life history traits will determine how a species will respond to changes in moisture and how quickly it will move up and down the moisture gradient. Dry conditions could favour *Hordeum jubatum* while wet conditions could favor *Carex atherodes*. *Hodeum jubatum*, however, was rarely present at reference sites, which suggests that this plant community gradient is primarily reflecting anthropogenic disturbance. In contrast, changes in the plant community along NMS axes 2 and 3 are highly related to inter-annual variation in plant communities (Figure 3.2). For example, changes along the 2^{nd} axis were correlated with inter-annual variation in the abundance of *Carex aquatilis* and *Carex atherodes*. Although inter-annual shifts in the abundance of these two species of sedges occurred, functional guild metrics like % of total cover of *Carex* sp. and % of total cover of native perennials were likely relatively insensitive to these changes in species composition. Hence, the constituent metrics in the IBI measure a suite of characteristics that are broad enough to be relatively robust to changes in species composition.

The type and class of wetlands being assessed may influence the effectiveness of plant-based IBIs at assessing wetland biological integrity. For example, I developed the wet meadow vegetation IBI to assess semi-permanent and permanent wetlands, whereas the IBI in North Dakota was developed to assess temporary, seasonal and semi-permanent wetlands (Euliss and Mushet, 2011). Semi-permanent and permanent wetlands do not dry out as often as temporary and seasonal wetlands because they often receive continued water from groundwater sources (Euliss and Mushet, 1996). Moreover, temporary and seasonal wetlands tend to be recharge areas that lose water to the groundwater table as well as to the atmosphere via evapotranspiration (Euliss and Mushet, 1996). In addition, the loss of water due to evapotranspiration is lower in the Aspen Parkland Ecoregion than in the rest of the Prairie Pothole Region (Hogg, 1994; LaBaugh et al., 1996), which may moderate seasonal hydroperiods and may be another reason why the wet meadow zone vegetation IBI developed in this region was less sensitive to changes in plant communities responding to variable in weather conditions than other studies. Clearly, more dramatic changes in weather, such as sustained droughts or flooding have the potential to invalidate the IBI scores and make it difficult to distinguish between anthropogenic stress and weather cycles.

Although this 4-year study suggests that IBI scores are relatively insensitive to variability in dry versus wet conditions, it did not address lag times in plant response to environmental conditions that could affect the IBI's precision over long-term data. Unlike rivers and lakes, many wetlands are dominated by clonal perennials that often exhibit a lag time between environmental change and plant response (Galatowitsch et al., 1999). Weather conditions in preceding years had been dry over the past decade, and a lag time could influence the response of the plant community to changing conditions resulting from immediate increases in summer rainfall, such as were seen in 2010 and 2011. Long-term data sets need to be collected to confirm that the IBI remains relatively insensitive over long term wet-dry cycles.

Differences in sampling methods could have also enabled the wet meadow zone vegetation IBI to factor out some natural variation that may have influenced the reproducibility of IBIs developed for the Great Lakes (Wilcox et al., 2002) and prairie potholes (Euliss and Mushet, 2011). The wet meadow zone vegetation IBI calculated metrics based on the plant community within a single zone; that is, if the wet meadow zone moved between sampling years, my sampling moved accordingly. In contrast, other IBIs that were tested over time (i.e. Great Lakes and prairie potholes) calculated metrics from the entire wetland plant community along a moisture gradient. Inter-annual climatic variation is known to influence zonation structure and species composition (Van der Valk and Davis, 1978). In coastal wetlands in Great Lakes' where Wilcox (2002) reported the failure of a plant-based IBI, Uzarski et al. (2004) demonstrated that an IBI developed from macroinvertebrates could effectively assess biological condition over time when metrics were calculated by individual zone. I believe that sampling within a single vegetation zone could factor out some of the natural variation in plant community captured by sampling along the entire moisture gradient, at least in semi-permanent and permanent wetlands. The wet meadow zone is a suitable zone to monitor at semi-permanent/permanent sites because it is more likely to exhibit fluctuations in the relative abundance of species than changes in species composition (Van der Valk, 2005).

Reproducibility of the wet meadow vegetation IBI

Reproducibility and variability of IBI scores in dry versus wet years was tested using Pearson's correlation, Wilcoxon's test and a signal to noise test. IBI scores in dry and wet years remained strongly correlated and ranked sites in a similar order. The wet meadow vegetation IBI had a signal to noise ratio of 6.2, which was stronger than any of the IBI's constituent individual metrics (Figure 3.4) and adequately higher than the minimum requirement of 3 recommended in IBIs for streams and rivers (Whittier et al., 2007). These three tests suggest that IBI scores are fairly consistent over relatively dry to wet conditions.

Whereas Euliss and Mushet (2011) found that improvements in IBI scores corresponded with increasing water levels, IBI scores in this study did not exhibit obvious directional changes to precipitation across the entire range of sites. The average IBI score of reference sites only decreased by 2% from dry to wet years, whereas both agricultural and constructed sites had higher biological integrity in wet years (IBI scores improved by 7%). Connectivity to groundwater (Smerdon et al., 2005), large basins (Larson, 1995) and gentle slopes (Forrest, 2010) could have all tempered the drying and reflooding effects in reference wetlands, resulting in more consistent IBI scores.

Agricultural and constructed sites tended to have higher biological integrity in wetter conditions, potentially because of changes in soil or germination from seed banks that changed plant community composition between years. Lower biological condition of agricultural sites in dry years could be due to agricultural disturbances (i.e. tilling, grazing) that are coupled with changes in climate conditions. For example, dry conditions expose more area to support agriculture and can result in encroachment by cultivation and grazing that can suppress perennials and increase annuals and invasives (Kantrud et al., 1989). IBI scores could also be sensitive to the larger fluctuations in water levels that occur in agricultural sites that have tilled catchment basins. Tilled agricultural landscapes increase water fluctuations in wetlands by reducing a catchment's capacity to mitigate surface flow into wetland basins (Euliss and Mushet, 1996). Although IBI scores at agricultural sites tended to improve in wet conditions, I could not conclude this relationship was due to rainfall variability or whether it reflected actual changes in biological integrity as a result of agricultural disturbances.

Biological integrity at constructed sites also tended to improve from dry to wet conditions. Wetland vegetation around the fringes of constructed sites may experience more severe effects of drought because they have significantly steeper shoreline slopes than natural wetlands (Forrest, 2010) and impermeable clay substrates (determined by taking soil cores as described in chapter 2). Furthermore, constructed stormwater ponds have heavy hydraulic loadings as a result of stormwater run-off (Wong and Geiger, 1997). Episodic flooding events cause pronounced fluctuations in water levels in stormwater ponds, which can increase the likelihood of flooding vegetation in the wetland fringe (Mitsch and Wilson, 1996). If inter-annual variation in biological condition of constructed sites is due to hydraulic loading, this has major implications for wetland biota as some constructed stormwater ponds are currently considered acceptable replacements for the loss of natural wetlands in this region.

Based on thresholds defined by CART analysis (see chapter 2), wetlands were placed into 1 of the following 4 health categories: “Exceptional,” “Good,” “Fair,” and “Poor.” A total of 67% of wetlands remained in the same health category. Of the 33% of sites that changed health categories, the majority were constructed and agricultural sites. This result corresponds with the higher variability in IBI scores of agricultural and constructed sites. I contend, however, that wetland health categories are not a good measure by which to evaluate the reproducibility of IBI scores, as the probability of a site switching health categories is dependent on its position relative to the health category threshold. That is, sites with IBI scores near the threshold between two health categories have a higher probability of switching health categories than sites squarely in the middle of a health category.

Evaluation of IBI’s individual metrics

Euliss and Mushet (2011) criticized the constituent metrics of the IPCI because several metrics favored plant species present in wet conditions over those in dry conditions. For instance, FQAI-based metrics in their study were problematic because annual species that proliferate during drawdown phases have lower coefficients of conservatism than species found during the normal emergent phase, consequently lowering IBI scores at sites in dry conditions. Clearly if metrics favor plants associated with different hydroperiods, the assessment technique is limited because wetland drying and reflooding is fundamental to wetlands (Van der Valk, 1981; Van der Valk and Davis, 1978). Wetland managers have intentionally drained and reflooded impoundments to maintain waterfowl productivity as dynamic hydroperiods increase seed abundance, enhance establishment of emergent cover and improve soil and aquatic plant production upon reflooding (Kadlec, 1962).

My evaluation of individual metrics revealed that there were weaknesses in all 4 metrics that were used in the IBI calculations. The % of total cover of native perennials had more variable scores in constructed sites than in reference sites, and width of the wet meadow zone scores were more variable in agricultural sites than in reference sites. The % of total cover of *Carex* sp. and % of total cover of native perennials also had signal to noise ratios that were lower than minimum acceptable level (Figure 3.4). I suspect that modifications of individual metrics in the IBI could mitigate some of these weaknesses. The % of total cover of *Carex* species and % of total cover of native perennials are relativized by the total cover of vegetation in the quadrat, which in some instances gave unrealistic overestimations of vegetation abundance, particularly at highly disturbed sites with sparse vegetation cover. Because the amount of vegetation cover was not equal among site types, I recommend that these 2 metrics be converted to their percent cover per quadrat (i.e. de-relativized). In contrast to the dense vegetation present at natural wetlands, constructed sites had very patchy distributions of vegetation due to their proximity to roads, pathways and residences. Increasing patchiness means that variance among quadrats is higher, which in turn lowers the power to detect differences at the wetland level. I suggest that sampling effort should be increased at constructed sites to account for wetland patchiness and hopefully provide a more precise estimation of biological condition. The main weakness of the FQAI is that it is weighted by richness of native species. Species richness can increase with disturbance due to the introduction of pioneering and edge species. Miller and Wardrop (2006) recommended using an adjusted FQAI that is weighted by relative number of native species to total species instead of richness of native species.

Combining individual metrics into the wet meadow vegetation IBI produced a stronger signal than any of the individual metrics alone (Figure 3.4). This is likely due to the ability of multi-metric indices to reduce natural variation. A metric is a measureable component of a biological system that is sensitive to anthropogenic stress (Karr, 1991). As such, it is subject to natural variation as well as measurement and sampling error, or noise that can mask the signal of the metric's response to human disturbance. Providing that errors among multiple metrics are uncorrelated, combining multiple metrics mitigates the effects of sampling error as the uncorrelated errors cancel each other out (Figure 3.4). This increases the robustness of the multi-metric index and can amplify the signal of human disturbance relative to the error, yielding an index that is more sensitive than any of its constituent metrics (Figure 3.4). If, however, the errors associated with metric values are correlated, then incorporating multiple metrics may compound the noise, thereby reducing its signal to anthropogenic stress (Figure 3.4). It is standard practice in IBI development (Whittier et al., 2007) to reduce redundancy among metrics by discarding one of the metrics that have Pearson correlation coefficients > 0.7 (see chapter 2). The idea behind the elimination of highly correlated metrics is that by reducing redundancy among metrics, correlated errors can be avoided. In the prairie pothole wetlands, 6 of 9 metrics incorporated in the IBI (DeKeyser et al., 2003; Euliss and Mushet, 2011) were variants of richness metrics, several which likely had correlated errors and influenced the reproducibility of the IBI. In contrast, I used redundancy analysis in developing the wet meadow IBI to pare down the number of metrics from 23 to only 4. I believe this practice is overly conservative, and that the wet meadow IBI could be improved if more metrics were retained; additional metrics can increase the resolution of the IBI (Karr, 1991) and may also improve its robustness to inter-annual variation. Metric values *should* be strongly correlated because their sensitivity to human disturbance covaries. Since correlated error is the issue of concern, then metrics should be discarded if they posses strongly correlated *errors*, not if they possess strongly correlated *values*. To fully make the wet meadow vegetation IBI appropriate for the typical hydrologic conditions across the entire Aspen Parkland Ecoregion, I recommend revisiting the original 23 metrics to assess the degree to which their errors are correlated, and reintegrating metrics providing they have uncorrelated errors.

Conclusion

Although wetland scientists have questioned the ability of plant-based IBI to produce consistent scores due to the dynamic nature of wetlands, the wet meadow vegetation IBI developed for wetlands in the Aspen Parkland Ecoegion appears able to distinguish environmental human disturbance from natural climatic variability. IBI scores had a strong signal to anthropogenic stress despite changes in climatic variability. Most interestingly, my findings suggest that although plant community composition changed in dry versus wet years, changes in IBI scores were relatively insensitive to these shifts in community composition. I suspect that several factors enabled the wet meadow vegetation IBI to adequately separate anthropogenic stress from natural variation: First, the wet meadow vegetation IBI was developed to assess more permanent wetlands that generally do not have extreme droughts and flooding; second, sampling the vegetation community within a single zone (i.e. wet meadow) factors out some variability induced by sampling along a natural moisture gradient; third, the IBI and its constituent metrics measure broad attributes such as functional guilds and habitat structure, ensuring that IBI scores are relatively robust to changes in community composition; and fourth, the wet meadow vegetation IBI integrates a comprehensive set of metrics that act synergistically when they are combined to improve IBI signal. I argue that plant-based IBIs are useful to assess wetland health of semi-permanent and permanent wetlands as I found that plant-based IBIs are indeed able to estimate a site's biological integrity by partitioning anthropogenic stress from natural variation.

3.5 References

AgroClimatic Information Service, retrieved from http://www.agric.gov.ab.ca, November 2011.

Best, K.F., Banting, J.D., Bowes, G.G., 1978. The Biology of Canadian Weeds: *Hordeum jubatum* L. Canadian Journal of Plant Science. 58, 699-708.

Clarke, K.R., 1993. Non-parametric multivariate analyses of changes in community structure. Aust. J. Ecol. 18, 117-143.

DeKeyser, E.S., Kirby, D.R., Ell, M.J., 2003. An index of plant community integrity: Development of methodology for assessing prairie wetland plant communities. Ecol. Indic. 3, 119-133.

Euliss, N., Mushet, D., 2011. A multi-year comparison of IPCI scores for prairie pothole wetlands: Implications of temporal and spatial variation. Wetlands. 31, 713-723.

Euliss, N.H., Labaugh, J.W., Fredrickson, L.H., Mushet, D.M., Laubhan, M.R.K., Swanson, G.A., Winter, T.C., Rosenberry, D.O., Nelson, R.D., 2004. The wetland continuum: A conceptual framework for interpreting biological studies. Wetlands. 24, 448-458.

Euliss, N.H., Mushet, D.M., 1996. Water-level fluctuation in wetlands as a function of landscape condition in the prairie pothole region. Wetlands. 16, 587-593.

Forrest, A.S., 2010. Created stormwater wetlands as wetland compensation and a floristic quality approach to wetland condition assesssment in central Alberta. Biological Sciences, University of Alberta, M.Sc., 117 pp.

Galatowitsch, S.M., Whited, D.C., Tester, J.R., 1999. Development of community metrics to evaluate recovery of Minnesota wetlands. J. Aquat. Ecosyst. Stress Recovery. 6, 217-234.

Gibbs, J.P., 1995. Hydrologic needs of wetland animals, Academic Press, New York, 267-276 pp.

GOA, 2011, retrieved from http://www.tpr.alberta.ca, November 2011.

Hargiss, C.L.M., DeKeyser, E.S., Kirby, D.R., Ell, M.J., 2008. Regional assessment of wetland plant communities using the index of plant community integrity. Ecol. Indic. 8, 303-307.

Herricks, E.E., Schaeffer, D.J., 1985. Can we optimize biomonitoring? Environ. Manage. 9, 487-492.

Hogg, E.H., 1994. Climate and the southern limit of the Western Canadian Boreal forest. Canadian Journal of Forest Research. 24, 1835-1845.

Hutchinson, G.E., 1975. A Treatise on Limnology, Volume 3: Limnological Botany, John Wiley, New York, NY, 660 pp.

Kadlec, J.A., 1962. Effects of a drawdown on a waterfowl impoundment. Ecology. 43, 267-281.

Kantrud, H.A., Krapu, G.L., Swanson, G.A., 1989. Prairie Basin Wetlands of the Dakotas: a community profile. Biological Report 85(7.28). U.S. Department of the Interior, Fish and Wildlife Service, Washington, DC.

Karr, J.R., 1981. Assessment of biotic integrity using fish communities. Fisheries. 6, 21-27.

Karr, J.R., 1991. Biological Integrity - a long-neglected aspect of water-resource management. Ecol. Appl. 1, 66-84.

Karr, J.R., 1993. Defining and assessing ecological integrity: beyond water quality. Environ. Toxicol. Chem. 12, 1521-1531.

Karr, J.R., Chu, E.W., 1999. Restoring Life in Running Waters: Better biological monitoring, Island Press, Washington, DC, 220 pp.

Keddy, P.A., 2000. Wetland Ecology: Principles and conservation, Cambridge University Press, New York, NY, 614 pp.

Kennedy, G., Mayer, T., 2002. Natural and constructed wetlands in Canada: An overview. Water Qual. Res. J. Can. 37, 295-325.

Kirby, D.R., DeKeyser, E.S., 2003. Index of wetland biological integrity development and assessment of semi-permanent wetlands in the Missouri Coteau Region of North Dakota. Section 104[b], Final Report for North Dakota Department of Health, (3) Wetland Grant Funds,.

Kirby, D.R., Krabbenhoft, K.D., Sedivec, K.K., DeKeyser, E.S., 2002. Wetlands in northern plains prairies: Offer societal values too. Rangelands. 24, 26-29.

LaBaugh, J.W., Winter, T.C., Swanson, G.A., Rosenberry, D.O., Nelson, R.D., Euliss, N.H., Jr., 1996. Changes in atmospheric circulation patterns affect midcontinent wetlands sensitive to climate. Limnol. Oceanogr. 41, 864-870.

Larson, D.L., 1995. Effects of climate on numbers of northern prairie wetlands. Clim. Change. 30, 169-180.

Laubhan, M.K., Roelle, J.E.,2001. Managing Wetlands for Waterbirds. In Rader, R.B., Batzer, D.P., Wissinger, S.A. (Eds.), Bioassessment and Management of North American Freshwater Wetlands. John Wiley & Sons, New York, pp. 387-411.

Mack, J.J., 2007. Developing a wetland IBI with statewide application after multiple testing iterations. Ecol. Indic. 7, 864-881.

McCune, B., Grace, J.B., 2002. Analysis of ecological communities, MjM Software Design, Gleneden Beach, Oregon,

McCune, B., Mefford, M.J., 2011. PC-ORD: Multivariate Analysis of Ecological Data version 6. MjM Software, Gleneden Beach, Oregon.

Miller, S.J., Wardrop, D.H., 2006. Adapting the floristic quality assessment index to indicate anthropogenic disturbance in central Pennsylvania wetlands. Ecol. Indic. 6, 313-326.

Miller, S.J., Wardrop, D.H., Mahaney, W.M., Brooks, R.P., 2006. A plant-based index of biological integrity (IBI) for headwater wetlands in central Pennsylvania. Ecol. Indic. 6, 290-312.

Mitsch, W.J., Wilson, R.F., 1996. Improving the success of wetland creation and restoration with know-how, time, and self-design. Ecol. Appl. 6, 77-83.

Moss, E.H., Packer, J.G., 1983. Flora of Alberta, University of Toronto Press, Toronto, 687 pp.

Raab, D.J., Bayley, S.E., 2012. Reclamation of wetland habitat in the Alberta oil sands: Generating assessment targets using boreal marsh vegetation communities. Ecol. Indic. 15, 43-51.

Rooney, R.C., Bayley, S.E., 2011. 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. Environ. Monit. Assess. 184, 749-761.

Simon, T.P., Stewart, P.M., Rothrock, P.E., 2001. Development of multimetric indices of biotic integrity for riverine and palustrine wetland plant communities along Southern Lake Michigan. Aquat. Ecosyst. Health Manage. 4, 293-309.

Smerdon, B.D., Devito, K.J., Mendoza, C.A., 2005. Interaction of groundwater and shallow lakes on outwash sediments in the sub-humid Boreal Plains of Canada. Journal of Hydrology. 314, 246-262.

Stewart, R.E., Kantrud, H.A., 1971. Classification of natural ponds and lakes in the glaciated prairie region, U.S. Bureau of Sport Fisheries and Wildlife, Washington, DC, 57 pp.

Uzarski, D.G., Burton, T.M., Genet, J.A., 2004. Validation and performance of an invertebrate index of biotic integrity for Lakes Huron and Michigan fringing wetlands during a period of lake level decline. Aquat. Ecosyst. Health Manage. 7, 269-288.

Van der Valk, A.G., 1981. Succession in wetlands: A Gleasonian approach. Ecology. 62, 688-696.

Van der Valk, A.G., 2005. Water-level fluctuations in North American prairie wetlands. Hydrobiologia. 539, 171-188.

Van der Valk, A.G., Davis, C.A., 1978. The role of seed banks in the vegetation dynamics of prairie glacial marshes. Ecology. 59, 322-335.

Whittier, T.R., Hughes, R.M., Stoddard, J.L., Lomnicky, G.A., Peck, D.V., Herlihy, A.T., 2007. A structured approach for developing indices of biotic integrity: Three examples from streams and rivers in the western USA. Trans. Am. Fish. Soc. 136, 718-735.

Wilcox, D.A., 2004. Implications of hydrologic variability on the succession of plants in the Great Lakes wetlands. Aquat. Ecosyst. Health Manage. 7, 223-231.

Wilcox, D.A., Meeker, J.E., Hudson, P.L., Armitage, B.J., Black, M.G., Uzarski, D.G., 2002. Hydrologic variability and the application of index of biotic integrity metrics to wetlands: A Great Lakes evaluation. Wetlands. 22, 588-615.

Winter, T.C.,1989. Hydrologic studies of wetlands in the northern prairie. In Van der Valk, A.G. (Ed.) Northern Prairie Wetlands. Iowa State University Press, pp. 16-54.

Wong, T.H.F., Geiger, W.F., 1997. Adaptation of wastewater surface flow wetland formulae for application in constructed stormwater wetlands. Ecol. Eng. 9, 187-202.

Wray, H.E., Bayley, S.E., 2006. A review of indicators of wetland health and function in alberta's prairie, aspen parkland and boreal dry mixedwood regions. The Water Research Users Group, A.E.,

Table 3.1. Clarke's *R*-values representing the amount of variance in plant community structure at identical sites in dry versus wet conditions. Values range between 0 and 1, where higher values signify increasing inter-annual dissimilarity in community composition.

Site type	Clarke's *R*
Reference	0.36
Agricultural	0.44
Constructed	0.34
All types	0.38

Table 3.2. Summary statistics of differences in IBI scores and health categories in dry versus wet conditions

Site type	Mean IBI score dry years	Mean IBI score wet years	Mean difference in IBI score	Range of difference	% of sites that changed health categories
Reference	85	83	2	0-22	4
Agricultural	58	65	7	3-28	13
Constructed	29	37	8	0-41	16
All	59	63	4	0-41	33

Table 3.3. Pearson correlations (R-values) of IBI scores between dry and wet conditions and Wilcoxon (p-values) testing whether sites were ranked in a similar order in dry and subsequent wet conditions. WM = wet meadow.

Metric	Pearson's R-value	Wilcoxon's p-value
Width of WM zone	0.84	0.07
FQAI	0.70	0.93
% of total cover of *Carex* sp.	0.63	0.72
% of total cover of native perennials	0.56	0.07
WM vegetation IBI	0.85	0.10

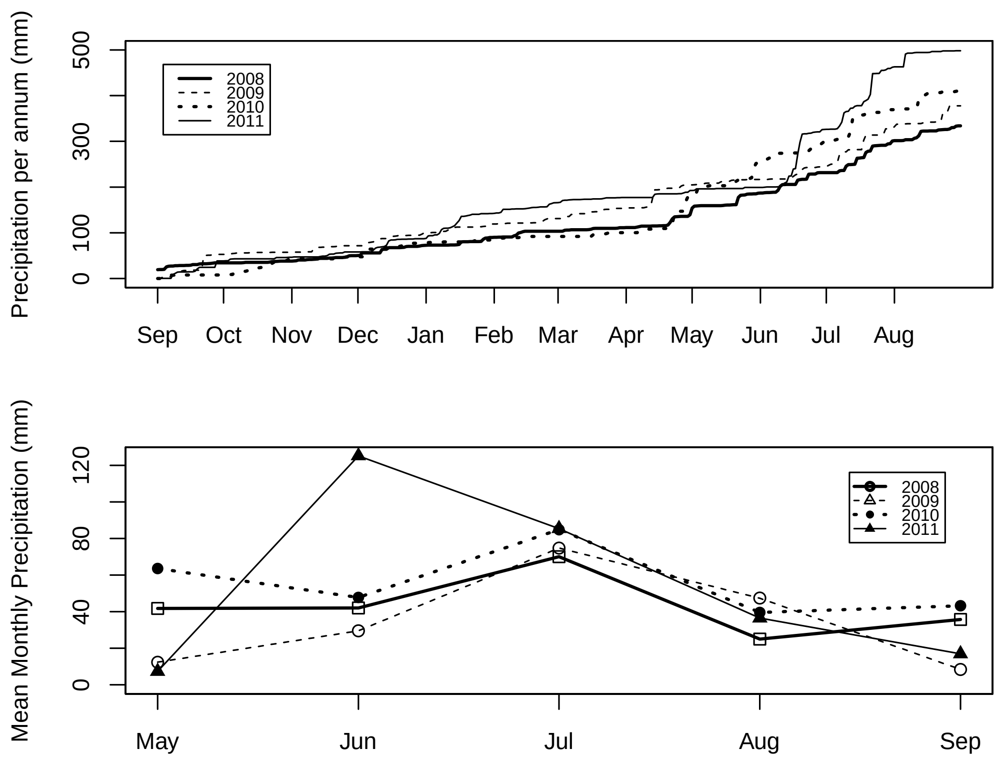


Figure 3.1. Plots of accumulated precipitation per annum (above) and mean monthly precipitation during the growing season (below) in the Central Parkland.

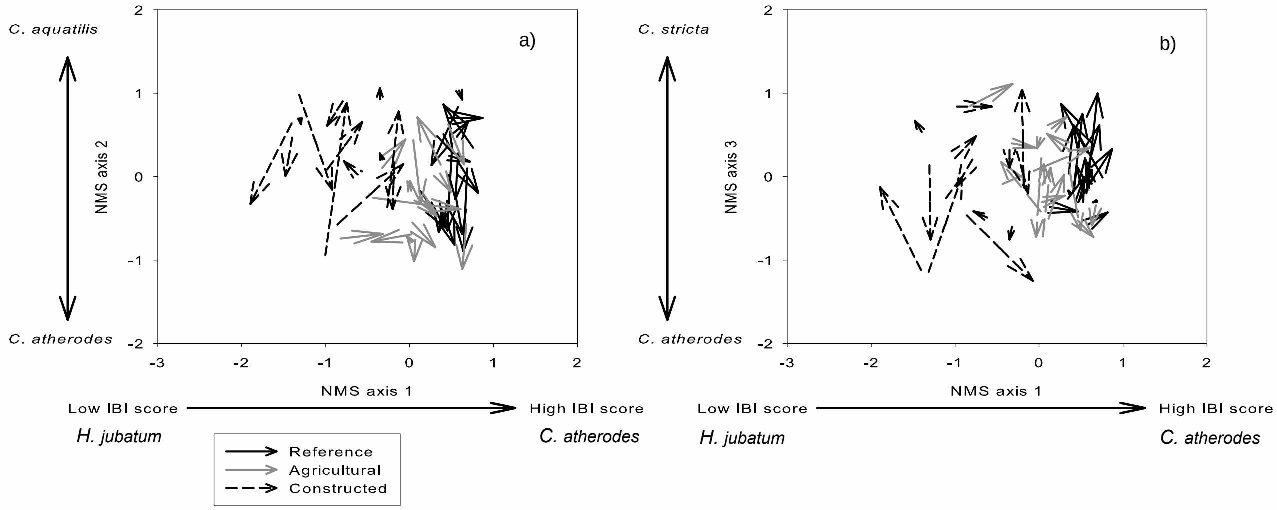


Figure 3.2. Non-metric multi-dimensional scaling (NMS) plots displaying inter-annual change in plant community in the wet meadow zone. Vector arrows represent the change of an identical site in species space from dry to wet conditions.

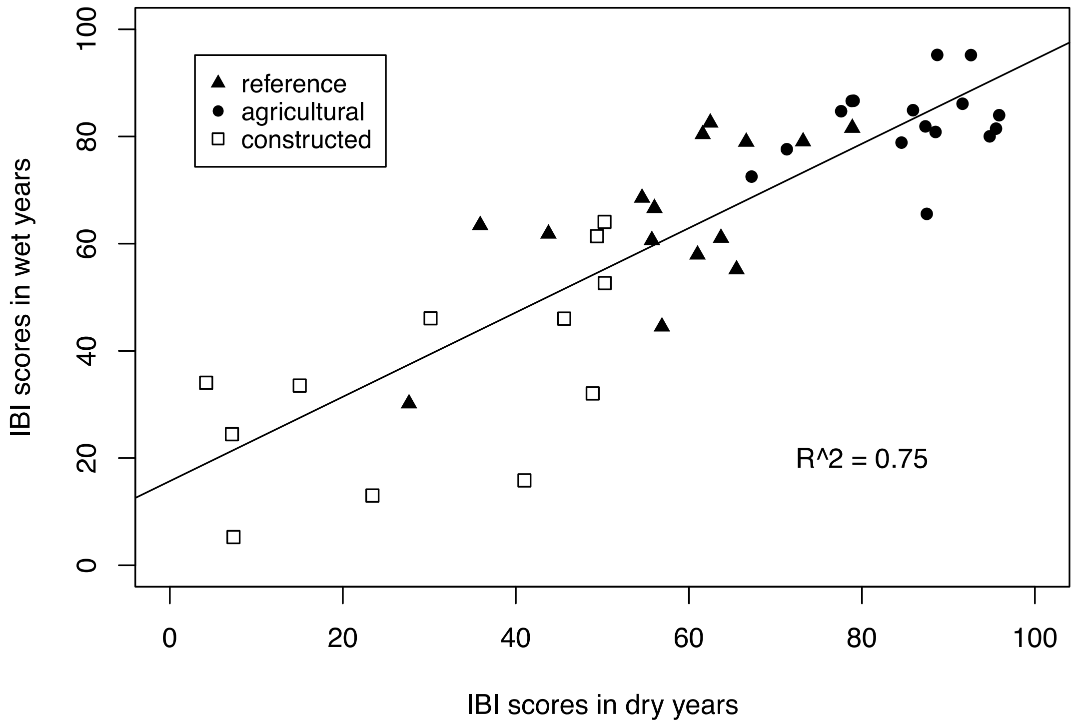


Figure 3.3. Correlation between IBI scores calculated in dry and wet conditions at 43 sites ranging from least-disturbed reference sites to highly disturbed constructed stormwater ponds.

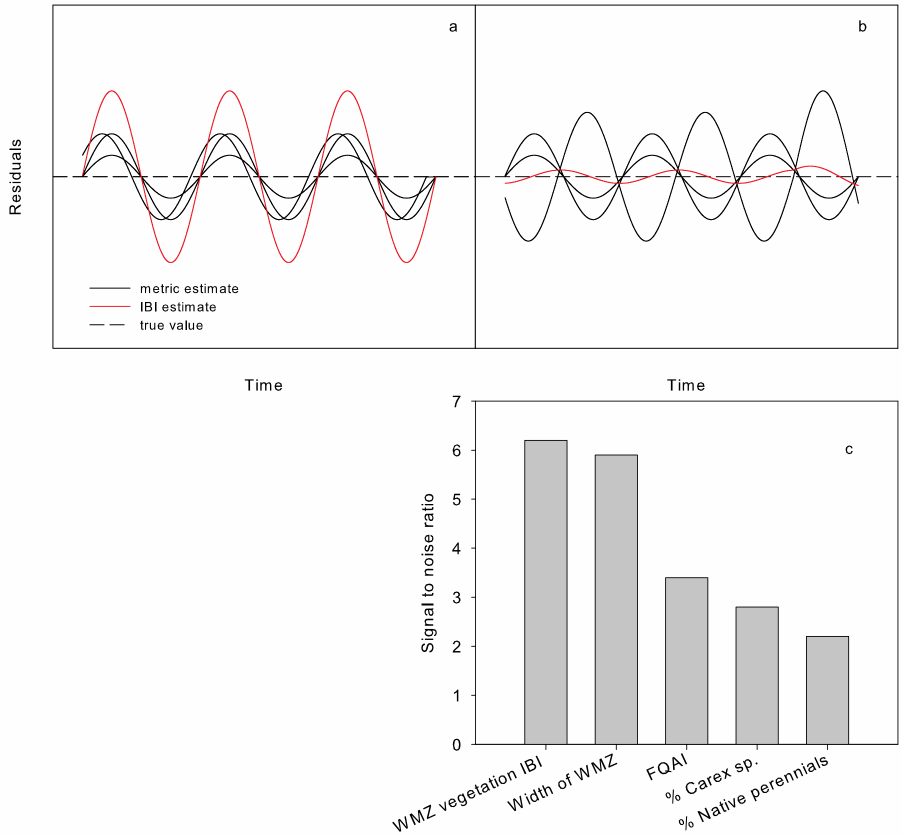


Figure 3.4. Hypothetical data depicting how IBIs that incorporate constituent metrics with correlated residuals amplify noise contributed by inter-annual variation, masking the biological signal (panel a), whereas residual error is canceled when metrics have uncorrelated residuals (panel b), thereby improving the IBI's signal. Metrics with uncorrelated errors results in a stronger IBI signals, as shown with actual data in panel c. Abbreviations: WMZ = wet meadow zone.

4. General discussion and conclusion

Despite the present concerns for the health of our ecosystems, the new driving forces of urbanization are altering the remaining wetlands in the Parkland Ecoregion. Wetland management in Canada is trailing the sophisticated programs developed in numerous jurisdictions in the United States that have established statewide wetland monitoring programs (i.e. Ohio, North Dakota, Oregon, Michigan). Although the province of Alberta already accepts some constructed stormwater management ponds as compensation for the loss of natural wetlands, no wetland management program exists that can assess whether these sites are actually replacing the ecosystem functions of natural wetlands. Research into the performance of stormwater management ponds is still in its infancy in terms of water quality objectives (Wong and Geiger, 1997) and faces major challenges because of the episodic nature of hydrologic loading from stormwater runoff (Carleton et al., 2001; Wong and Geiger, 1997). Forrest (2010) revealed that steeper shoreline slopes of stormwater management ponds were a major factor impairing their ability to provide habitat to support wetland-dependent songbirds. He also developed a Floristic Quality Assessment Index (FQAI) for the Parkland Ecoregion of Alberta based on previous floristic quality assessment methods devised in the United States (Lopez and Fennessy, 2002; Miller et al., 2006; Swink and Wilhelm, 1994), finding that floristic quality was lower at constructed sites than at natural wetlands. However, the FQAI may not capture the broad suite of ecological functions that wetland managers may be interested in, as it is based on a single metric. Furthermore, other single metric indices such as indices of diversity are known to be inconsistent and ambiguous (Karr and Chu, 1999).

In contrast, multi-metric approaches such as Indices of Biological Integrity (IBIs) are broader in scope and incorporate several biological attributes that are predictive of human disturbance (Karr and Chu, 1999). That is, the purpose of my research was to develop and evaluate tools that could effectively assess ecosystem condition of semi-permanent and permanent natural and constructed wetlands.

The 1st chapter of this thesis provided a background overview of bioassessment practices and literature concerning the use of indicators to assess wetlands and their reliability over variable climate conditions over time. The Aspen Parkland Ecoregion is in need of wetland bioassessment tools that can aid management to maintain the integrity of our waters and wetlands.

In the 2nd chapter, building upon Forrest's (2010) research and recommendations as well as the urgent need for an assessment method that quantifies human disturbance and provides an estimate of biological condition, I developed IBIs from several taxonomic groups to assess biological condition across the full range of disturbances. I built the IBIs by comparing test sites to least-impacted reference sites that spanned a gradient of underlying environmental conditions. This research determined that vegetation in the wet meadow zone and wetland-dependent songbirds had the highest potential for use as indicators of wetland health whereas other indicators such as emergent vegetation, submersed aquatic vegetation and waterfowl are not good indicators to assess local disturbance and environmental condition in this region. Furthermore, I reinforced Forrest's (2010) findings by revealing that constructed stormwater ponds indeed had significantly lower biological integrity than natural wetlands. This suggests that constructed stormwater ponds are not in fact replacing the natural ecosystem functions of lost wetlands.

These results have a number of implications for wetland management and research on biological indicators for wetland assessment. My study revealed that differing biological communities within the same wetlands have varying sensitivities to the same environmental stress gradient. Although wet meadow vegetation was the most sensitive taxonomic group to environmental stress, it may be of particular interest to managers that songbirds were also fairly predictive indicators of stress. Birds are of particularly high value to society, and bird-based IBIs could facilitate communication of monitoring programs to the public (Bryce et al., 2002). Although there is strong interest in monitoring biological condition of waterfowl as well, this study did not find any waterfowl-based metrics that were sensitive to environmental stress. Forrest (2010) found similar results in that waterfowl richness and abundance was similar in constructed sites and reference sites. Notwithstanding, waterfowl may be sensitive to other types and scales of disturbance that may not be measured in the IBI approach. Another bioassessment tool could potentially be produced that monitors waterfowl to predict changes in land-use at larger scales.

The wet meadow vegetation and wetland-dependent songbird IBIs were correlated, indicating that they are fairly good surrogates of one another and both could be considered as suitable indicators to assess wetland health. This further suggests that the IBI scores of wet meadow vegetation could be used a proxy measurement of the biological condition of songbirds, and vise versa. I suggest that they be used independently to address different management objectives. Either of these IBIs can be used for a number of purposes: to report on the biological condition of constructed sites (and potentially restored sites after further testing with increased sample sizes), to evaluate the health of a natural site that will be degraded or lost as a result of development, and to convey information to the public.

In the 3nd chapter of this thesis, I evaluated the reproducibility of the wet meadow vegetation IBI to determine whether plant-based IBIs are in fact useful tools for more permanent wetlands in the Aspen Parkland Ecoregion. There has been notable criticism that natural climatic variability in wetlands influences the effectiveness of an IBIs estimation of biological condition because metrics are sensitive to plant community changes responding to dynamic water levels and hydroperiods. These results demonstrated that the wet meadow vegetation IBI was largely insensitive to inter-annual shifts in plant community and was reproducible in relatively variable climate conditions. This has major implications for wetland research on biological indicators and wetland assessment tools. Only a few studies have examined the reproducibility of plant-based IBIs to assess wetlands over time, of which they have been found to be sensitive to climatic variability. A number of factors likely permit IBIs to successfully estimate biological condition in some cases and not in others. IBIs might be more appropriate for assessment of more permanent water bodies with stable water levels, as IBI scores are likely only robust to a limited range of natural variation. Clearly these results would have differed in the case of very severe drought or flooding. IBI sensitivity to natural variation in hydroperiod could also be reduced by sampling the plant community within a single vegetation zone, which would factor out some natural variation caused by sampling the wetland plant community along moisture gradients. This implies that future development of IBIs should be cautious of calculating metrics along natural gradients since the goal of a bioassessment tool is to measure human disturbance and not natural variation (Karr and Chu, 1999). The last factor that contributes to IBI robustness is ensuring the IBI is composed of a comprehensive set of component metrics that act synergistically to reduce the noise of IBI scores. The next step for future research is to revisit some of the candidate metrics to make minor modifications to the IBI's existing constituent metrics and to add more metrics to increase IBI robustness.

Future research should focus on validating the wet meadow vegetation and songbird IBI across the entire Aspen Parkland Region by testing the IBIs on new wetlands distributed across the region as well as evaluating whether the IBIs could be expanded to other wetland classes. Other research could focus on identifying potential indicators such as waterfowl that may respond to larger scales of land use disturbance. This is the first research to develop a broadly applicable wetland assessment tool for wetland management programs in Alberta. The research pertaining to this thesis is broad in scope and contributes to research on biological indicators and wetland bioassessment tools and will hopefully be of use to future wetland management programs in the Aspen Parkland.

References

Bryce, S.A., Hughes, R.M., Kaufmann, P.R., 2002. Development of a bird integrity index: Using bird assemblages as indicators of riparian condition. Environ. Manage. 30, 294-310.

Carleton, J.N., Grizzard, T.J., Godrej, A.N., Post, H.E., 2001. Factors affecting the performance of stormwater treatment wetlands. Water Res. 35, 1552-1562.

Forrest, A.S., 2010. Created stormwater wetlands as wetland compensation and a floristic quality approach to wetland condition assesssment in central Alberta. Biological Sciences, University of Alberta, M.Sc., 117 pp.

Karr, J.R., Chu, E.W., 1999. Restoring Life in Running Waters: Better biological monitoring, Island Press, Washington, DC, 220 pp.

Lopez, R.D., Fennessy, M.S., 2002. Testing the floristic quality assessment index as an indicator of wetland condition. Ecol. Appl. 12, 487-497.

Miller, S.J., Wardrop, D.H., Mahaney, W.M., Brooks, R.P., 2006. A plant-based index of biological integrity (IBI) for headwater wetlands in central Pennsylvania. Ecol. Indic. 6, 290-312.

Swink, F.A., Wilhelm, G.S., 1994. Plants of the Chicago Region, fourth edition, Morton Arboretum, Lisle, IL, USA, 900 pp pp.

Wong, T.H.F., Geiger, W.F., 1997. Adaptation of wastewater surface flow wetland formulae for application in constructed stormwater wetlands. Ecol. Eng. 9, 187-202.

Appendix A. Functional guilds of wet meadow and emergent macrophyte species. Lifecycle: P = perennial, A = annual; Group: M = monocot, D = dicot; Growth habit: F = Forb, G = graminoid, V = vines, S = shrub, T = tree; Native status: N = native, I = introduced; Wetland indicator status: OBL = obligate, FACW = facultative wetland; FAC = facultative; FACU = facultative upland; UPL = upland.

Latin name	Lifecycle	Group	Growth habit	Native status	Wetland indicator status	Coefficient of conservatism
Achillea millefolium	P	D	F	I	FACU	0
Achillea sibirica	P	D	F	N	UPL	5
Agrostis scabra	P	M	G	N	FAC	2
Agrostis stolonifera	P	M	G	I	FAC	0
Alisma trivial	P	M	F	N	OBL	4
Alopecurus aequilis	P	M	G	N	OBL	4
Amelanchier alnifolia	P	D	T	N	FACU	3
Anemone canadensis	P	D	F	N	FACW	6
Argentina anserina	P	D	F	N	OBL	3
Artemisia biennis	A	D	F	I	FAC	2
Beckmannia syzigachne	A	M	G	N	OBL	2
Bidens cernua	A	D	F	N	OBL	4
Bromus ciliatus	P	M	G	N	FAC	5
Calamagrostis canadensis	P	M	G	N	FACW	2
Calamagrostis stricta	P	M	G	N	FACW	4
Calla palustris	P	M	F	N	OBL	8
Caltha palustris	P	D	F	N	OBL	6
Carex aquatilis	P	M	G	N	OBL	3
Carex atherodes	P	M	G	N	OBL	5

Appendix A continued

Latin name	Lifecycle	Group	Growth habit	Native status	Wetland indicator status	Coefficient of conservatism
Carex bebbii	P	M	G	N	OBL	4
Carex diandra	P	M	G	N	OBL	6
Carex lasiocarpa	P	M	G	N	OBL	6
Carex sartwellii	P	M	G	N	FACW	5
Carex sychnocephala	P	M	G	N	FACW	5
Carex utriculata	P	M	G	N	OBL	5
Carex spp.	P	M	G	—	—	—
Chamerion angustifolium	P	D	F	N	FAC	1
Chenopodium album	A	D	F	I	FAC	0
Chenopodium capitatum	A	D	F	N	UPL	2
Cicuta bulbifera	P	D	F	N	OBL	6
Cicuta maculata	P	D	F	N	OBL	5
Cicuta spp	P	D	F	—	—	—
Cirsium arvense	P	D	F	I	FACU	0
Conyza canadensis	P	D	F	N	FACU	—
Cornus sericea	P	D	S	N	UPL	3
Unknown mustard	A	D	F	I	FACU	0
Eleocharis acicularis	P	M	G	N	OBL	4
Eleocharis palustris	P	M	G	N	OBL	5
Elymus repens	P	M	G	I	FAC	0
Epilobium ciliatum	P	D	F	N	FACW	2
Epilobium palustre	P	D	F	N	OBL	3
Equisetum arvense/pratense	P	H	F	N	FAC	1
Equisetum fluviatile	P	H	F	N	OBL	5
Equisetum hyemale	P	H	F	N	FACW	4
Equisetum sylvaticum	P	H	F	N	FACW	6
Erigeron acris	P	D	F	N	FAC	3
Erigeron philadelphicus	P	D	F	N	FACW	3

Appendix A continued

Latin name	Lifecycle	Group	Growth habit	Native status	Wetland indicator status	Coefficient of conservatism
Fragaria vesca	P	D	F	N	UPL	4
Fragaria virginiana	P	D	F	N	FACU	2
Galeopsis tetrahit	A	D	F	I	NO	0
Galium trifidum	P	D	F	N	OBL	5
Galium triflorum	P	D	F	N	FACU	5
Geum rivale	P	D	F	N	FACW	6
Geum aleppicum	P	D	F	N	FAC	3
Glaux maritima	P	D	F	N	OBL	6
Glyceria grandis	P	M	G	N	FACW	5
Gnaphalium uliginosum	A	D	F	I	FAC	0
Hieracium umbellatum	P	D	F	N	UPL	3
Hippuris vulgaris	P	D	F	N	OBL	5
Hordeum jubatum	P	M	G	N	FACW	1
Impatiens capensis	A	D	F	N	FACW	4
Juncus alpinus	P	M	G	N	OBL	4
Juncus arcticus	P	M	G	N	OBL	3
Juncus bufonius	A	M	G	N	OBL	2
Juncus longistylis	P	M	G	N	FACW	5
Juncus nodosus	P	M	G	N	OBL	4
Lathyrus venosus	P	D	V	N	FACW	—
Linaria vulgaris	P	D	F	I	UPL	0
Lotus corniculatus	P	D	F	I	FACU	0
Lycopus asper	P	D	F	N	OBL	4
Lycopus spp	P	D	F	—	—	—
Lycopus uniflorus	P	D	F	N	OBL	3
Lysimachia ciliata	P	D	F	N	FACW	—
Lysimachia thyrsiflora	P	D	F	N	OBL	6
Maianthemum canadense	P	M	F	N	FACU	—

Latin name	Lifecycle	Group	Growth habit	Native status	Wetland indicator status	Coefficient of conservatism
Matricaria discoidea	A	D	F	I	FAC	0
Melilotus alba	A	D	F	I	FACU	0
Melilotus officinalis	A	D	F	I	FACU	0
Melilotus spp	A	D	F	—	—	0
Mentha arvensis	P	D	F	N	FACW	4
Myosotis laxa	P	D	F	N	OBL	—
Petasites frigidus	P	D	F	N	FAC	4
Phacelia franklinii	A	D	F	N	UPL	—
Phalaris arundinacea	P	M	G	N	FACW	2
Phleum pratense	P	M	G	I	FACU	0
Plantago major	P	D	F	I	FAC	0
Poa palustris	P	M	G	N	FACW	3
Poa pratensis	P	M	G	UNK	FACU	0
Polygonum amphibium	P	D	F	N	OBL	4
Polygonum aviculare	A	D	F	I	FACU	—
Polygonum lapathifolium	A	D	F	I	OBL	2
Polygonum spp.	A/P	D	F	—	—	—
Populus spp.	P	D	T	—	—	—
Potentilla gracilis	P	D	F	N	FAC	5
Potentilla norvegica	A	D	F	N	FAC	2
Potentilla palustris	P	D	F	N	OBL	7
Puccinellia nuttaliana	P	M	G	N	OBL	5
Ranunculus cymbalaria	P	D	F	N	FACW	4
Ranunculus macounii	P	D	F	N	OBL	5
Ranunculus sceleratus	A	D	F	UNK	OBL	3
Ranunculus spp.	P	D	F	—	—	—
Ribes spp.	P	D	S	—	—	—
Rorippa palustris	A	D	F	N	OBL	4

Appendix A continued

Latin name	Lifecycle	Group	Growth habit	Native status	Wetland indicator status	Coefficient of conservatism
Rosa spp	P	D	S	N	FACU	—
Rubus arcticus	P	D	S	N	UPL	5
Rubus idaeus	P	D	S	N	FACU	1
Rubus pubescens	P	D	F	N	FACW	5
Rumex britannica	P	D	F	N	OBL	4
Rumex crispus/occidentalis	P	D	F	UNK	FACW	—
Rumex maritimus	A	D	F	N	FACW	2
Saggitaria cuneata	P	M	F	N	OBL	5
Salix spp.	P	D	T	—	—	—
Schoenoplectus maritimus	P	M	G	N	OBL	—
Schoenoplectus tabernaemontani	P	M	G	N	OBL	4
Scirpus microcarpus	P	M	G	N	OBL	3
Scolochloa festucacea	P	M	G	N	OBL	4
Scutellaria galericulata	P	D	F	N	OBL	5
Senecio congestus	A	D	F	N	FACW	3
Senecio eremophilus	P	D	F	N	FAC	5
Sium suave	P	D	F	N	OBL	5
Solidago canadensis	P	D	F	N	FACU	3
Sonchus arvensis	P	D	F	I	FAC	0
Sonchus asper	A	D	F	I	FACW	0
Sparganium spp.	P	M	F	N	OBL	5
Stachys pilosa	P	D	F	N	OBL	4
Stellaria calycantha	P	D	F	N	FACW	4
Stellaria longipes	P	D	F	N	OBL	3
Stellaria spp.	P	D	F	—	—	—
Symphyotrichum ciliatum	A	D	F	N	FACW	4
Symphyotrichum lanceolatum	P	D	F	N	OBL	6
Symphyotrichum puniceum	P	D	F	N	OBL	5

Appendix A continued

Latin name	Lifecycle	Group	Growth habit	Native status	Wetland indicator status	Coefficient of conservatism
Symphoricarpos spp.	P	D	S	—	—	—
Taraxacum officinale	P	D	F	UNK	FACU	0
Thlaspi arvense	A	D	F	I	—	0
Trifolium hybridum	P	D	F	I	FACU	0
Triglochin maritima	P	M	G	N	OBL	5
Typha latifolia	P	M	F	N	OBL	2
Urtica dioica	P	D	F	N	FACW	3
Vicia americana	P	D	V	N	—	3

Appendix B. Functional guilds of submersed aquatic vegetation species. Group: M = monocot, D = dicot; Growth habit: F = Forb, G = graminoid, V = vines, S = shrub, T = tree; Native status: N = native, I = introduced; Wetland indicator status: OBL = obligate, FACW = facultative wetland; FAC = facultative; FACU = facultative upland; UPL = upland.

Latin name	Group	Growth habit	Native status	Wetland Indicator Status	Coefficient of conservatism	Nutrient Regime
Aquatic moss	—	—	—	OBL	—	—
Callitriche palustris	D	F	N	OBL	7	M
Ceratophyllum demersum	D	F	N	OBL	4	P
Chara spp.	—	A	N	OBL	—	—
Elodea canadensis	D	F	N	OBL	5	S
Lemna minor	M	F	N	OBL	4	VR
Lemna trisulca	M	F	N	OBL	4	VR
Myriophyllum verticillatum	D	F	N	OBL	4	VR
Najas flexilis	M	F	N	OBL	8	S
Nuphar lutea	D	F	N	OBL	5	NL
Potamogeton foliosus	M	F	N	OBL	4	M
Potamogeton natans	M	F	N	OBL	5	M
Potamogeton pusillus	M	F	N	OBL	4	NL
P. richardsonii	M	F	N	OBL	4	P
P. zosteriformis	M	F	N	OBL	5	P
Ranunculus aquatilis	D	F	N	OBL	5	R
Sagittaria cuneata	M	F	N	OBL	5	R
Stuckenia filiformis	M	F	N	OBL	5	NL
Stuckenia pectinata	M	F	N	OBL	3	VR
Stuckenia vaginatus	M	F	N	OBL	5	NL
Utricularia vulgaris	D	F	N	OBL	4	R
Zannichellia palustris	M	F	N	OBL	5	R

Appendix C. Functional guilds of songbird and other point count bird species. Wetland indicator status: OBL = obligate species, FACW = facultative wetland species, FAC = facultative species, UPL = upland species; Dietary: O = insectivore/granivore, C = carnivore, G = granivore, V = insectivore; Foraging: A = aerial, G = ground; C – canopy; P = predatory; Nesting: G = ground, C = canopy, P = parasite nester; Migratory: NM = neotropical migrant, T = temperate migrant; Species concern: SC = secure, SN = sensitive; Prevalence: C = common, U = uncommon, RU = relatively uncommon.

Common Name	Latin Name	Wetland indicator status	Dietary	Foraging	Nesting	Migratory	Species concern	Prevalence
Alder flycatcher	*Empidonax alnorum*	FACW	O	A	G	NM	SC	C
American bittern	*Botaurus lentiginosus*	OBL	C/V	G	G	NM	SN	U
American goldfinch	*Carduelis tristis*	UPL	G	C	C	NM	SC	C
Baltimore oriole	*Icterus galbula*	UPL	O	C	C	NM	SC	C
Barn swallow	*Hirundo rustica*	FACW	V	A	C	NM	SN	C
Brown-headed cowbird	*Molothrus ater*	UPL	O	G	P	NM	SC	C
Black-necked stilt	*Himantopus mexicanus*	FACW	V	G	G	NM	SN	RU
Black tern	*Chlidonias niger*	OBL	C/V	A	G	NM	SN	C
Brewer's blackbird	*Euphagus cyanocephalus*	UPL	G	G	C	NM	SC	C
Common grackle	*Quiscalus quiscula*	FACW	O	G	C	TM	SC	C
Common tern	*Sterna hirundo*	FACW	C	A	G	NM	SC	C
Common yellowthroat	*Geothlypis trichas*	OBL	O	C	G	NM	SN	C
Eastern kingbird	*Tyrannus tyrannus*	FACW	V	A	C	NM	SC	UC
Eastern phoebe	*Sayornis phoebe*	FAC	O	A	C	NM	SN	C
Great blue heron	*Ardea herodias*	FACW	C/V	G	C	NM	SN	C
Killdeer	*Charadrius vociferus*	FAC	V	G	G	NM	SC	VC

Appendix C continued

Common Name	Latin Name	Wetland indicator status	Dietary	Foraging	Nesting	Migratory	Species concern	Prevalence
Le Conte's sparrow	*Ammodramus leconteii*	FACW	O	G	G	TM	SC	UC
Lesser yellowlegs	*Tringa flavipes*	OBL	V	G	G	NM	SC	C
Lincoln's sparrow	*Melospiza lincolnii*	FACW	O	G	G	NM	SC	UC
Marsh wren	*Cistothorus platensis*	OBL	V	G	G	NM	SC	UC
Northern harrier	*Circus cyaneus*	FACW	C/V	P	G	NM	SN	C
Purple martin	*Progne subis*	FAC	I	A	C	NM	SC	C
Red-winged blackbird	*Agelaius phoeniceus*	OBL	O	G	G	NM	SC	VC
Sora	*Porzana carolina*	OBL	O	G	G	NM	SN	UC
Solitary sandpiper	*Tringa solitaria*	FACW	V	G	C	NM	SC	U
Song sparrow	*Melospiza melodia*	FAC	O	G	G	TM	SC	VC
Spotted sandpiper	*Actitis macularius*	FAC	C/V	G	G	NM	SC	C
Sharp-tailed sparrow	*Ammodramus nelsoni*	OBL	O	G	G	TM	SC	UC
Swamp sparrow	*Melospiza georgiana*	OBL	O	G	G	NM	SC	U
Tree swallow	*Tachycineta bicolor*	FACW	V	A	C	NM	SC	C
Willet	*Tringa semipalmata*	FACW	C/V	G	G	NM	SC	UC
Wilson's pharalope	*Phalaropus tricolor*	FACW	V	G	G	NM	SC	UC
Wilson's snipe	*Gallinago delicata*	OBL	O	G	G	NM	N/A	C
Yellow-headed blackbird	*Xanthocephalus xanthocephalus*	OBL	O	G	C	NM	SC	C

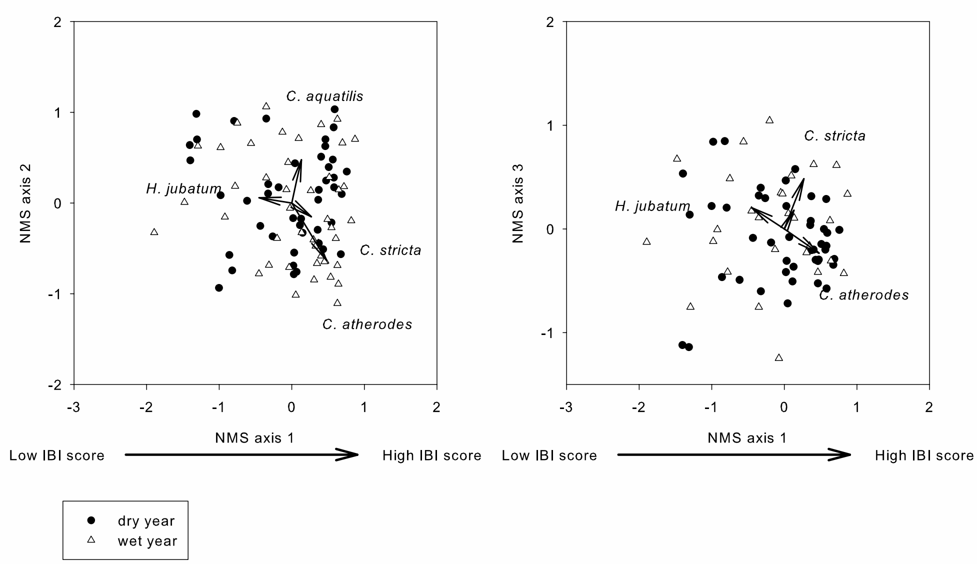
Appendix D. Functional guilds of waterfowl and other waterbird species. Wetland indicator status: OBL = obligate species, FACW = facultative wetland species, FAC = facultative species, UPL = upland species; Dietary: O = omnivore, C = carnivore, G = granivore, V = insectivore; Foraging: A = aerial, G = ground; C – canopy; P = predatory; Nesting: G = ground, C = canopy, P = parasite nester; Migratory: NM = neotropical migrant, T = temperate migrant; Species concern: SC = secure, SN = sensitive; Prevalence: C = common, U = uncommon, RU = relatively uncommon.

Common Name	Latin Name	Dietary	Foraging	Nesting	Migratory	Species concern	Prevelance
American Avocet	*Recurvirostra americana*	O	G	G	NM	SC	U
American Bittern	*Botaurus lentiginosus*	C/V	G	G	NM	SN	U
American Coot	*Fulica americana*	O	DB/DV	F	NM	SC	VC
American Wigeon	*Anas americana*	H	DB	G	NM	SC	C
Black-crowned Night Heron	*Nycticorax nycticorax*	C/V	G	C	NM	SN	U
Black-necked Stilt	*Himantopus mexicanus*	V	G	G	NM	SN	RU
Black Tern	*Chlidonias niger*	C/V	A	G	NM	SN	C
Bufflehead	*Bucephala albeola*	C/V	DV	C	TM	SC	UC
Blue-winged Teal	*Anas discors*	O	DB	G	NM	SC	C
Canada Goose	*Branta canadensis*	H	DB	G	TM	SC	C
Canvasback	*Aythya valisineria*	O	DV	F	NM	SC	U
Cinnamon Teal	*Anas cyanoptera*	O	DB	G	NM	SC	U
Common Goldeneye	*Bucephala clangula*	C/V	DV	C	TM	SC	C
Common Loon	*Gavia immer*	C/V	DV	G	TM	SC	UC
Common Merganser	*Mergus merganser*	O	DV	C	TM	SC	UC
Double-crested Cormorant	*Phalacrocorax auritus*	C	P	G	NM	SC	U
Eared Grebe	*Podiceps migricollis*	C/V	DV	F	NM	SC	UC
Franklin's Gull	*Larus pipixcan*	O	G	G	NM	SC	C
Gadwall	*Anas strepera*	H	DB	G	NM	SC	C
Great Blue Heron	*Ardea herodias*	C/V	G	C	NM	SN	C
Green-winged Teal	*Anas crecca*	H	DB	G	NM	SN	UC
Horned Grebe	*Podiceps auritus*	C/V	DV	F	TM	SN	UC
Killdeer	*Charadrius vociferus*	V	G	G	NM	SC	VC
Lesser Scaup	*Aythya affinis*	O	DV	G	NM	SN	C

Appendix D continued

Common Name	Latin Name	Dietary	Foraging	Nesting	Migratory	Species concern	Prevelance
Lesser Yellowlegs	*Tringa flavipes*	V	G	G	NM	SC	C
Marbled Godwit	*Limosa fedoa*	C/V	G	G	NM	SC	C
Mallard	*Anas platyrhynchos*	O	DB	G	NM	SC	VC
Northern Harrier	*Circus cyaneus*	C/V	P	G	NM	SN	C
Northern Pintail	*Anas acuta*	O	DB	G	NM	SN	C
Northern Shoveler	*Anas clypeata*	O	DB	G	NM	SC	C
Pied-billed Grebe	*Podilymbus podiceps*	C/V	DV	F	NM	SN	UC
Redhead	*Aythya americana*	H	DV	F	NM	SC	U
Ring-necked Duck	*Aythya collaris*	H	DV	G	NM	SC	U
Red-necked Grebe	*Podiceps grisegena*	C/V	DV	F	TM	SC	UC
Ruddy Duck	*Oxyura jamaicensis*	O	DV	G	NM	SC	C
Sandhill Crane	*Grus canadensis*	O	G	G	TM	SN	U
Short-billed Dowitcher	*Limnodromus griseus*	V	G	G	NM	N/A	UC
Sora	*Porzana carolina*	O	G	G	NM	SN	UC
Solitary Sandpiper	*Tringa solitaria*	V	G	C	NM	SC	U
Spotted Sandpiper	*Actitis macularius*	C/V	G	G	NM	SC	C
Trumpeter Swan	*Cygnus buccinator*	H	DB	G	TM	SN	U
Willet	*Tringa semipalmata*	C/V	G	G	NM	SC	UC
Wilson's Pharalope	*Phalaropus tricolor*	V	G	G	NM	SC	UC
Wilson's Snipe	*Gallinago delicata*	O	G	G	NM	N/A	C

Appendix E. Non-metric multidimensional scaling ordinations depicting wedge-shaped community composition in dry and wet conditions. NMS used wet meadow vegetation community data of 44 sites sampled in dry conditions and subsequent wet conditions. Identical sites were joined to depict within-site changes in plant community structure as shown in Figure 3.2.



Appendix F. Wet meadow vegetation IBI scores and health categories in dry conditions (2008-09) and resultant scores in wet conditions (2010-11). Health category thresholds were as follows: “Poor” = 0-25, “Fair” = 26=51, “Good” = 52-69, “Exceptional” = 70-100.

Site type	Site name	Years sampled	IBI score dry years	IBI score wet years	Health category dry years	Health category wet years
Reference	cl03	2008, 2010	87	82	Exceptional	Exceptional
	cl07	2008, 2010	89	95	Exceptional	Exceptional
	ei00	2008, 2010	96	84	Exceptional	Exceptional
	ei12	2008, 2010	93	95	Exceptional	Exceptional
	ei45	2008, 2010	92	86	Exceptional	Exceptional
	ei46	2008, 2010	79	87	Exceptional	Exceptional
	ei47	2008, 2010	86	85	Exceptional	Exceptional
	ml10	2008, 2010	67	73	Good	Exceptional
	cl50	2009, 2011	87	66	Exceptional	Good
	cl52	2009, 2011	89	81	Exceptional	Exceptional
	clcen	2009, 2011	85	79	Exceptional	Exceptional
	curr	2009, 2011	71	78	Exceptional	Exceptional
	ei02	2009, 2011	79	87	Exceptional	Exceptional
	einw	2009, 2011	78	85	Exceptional	Exceptional
	eisand	2009, 2011	95	80	Exceptional	Exceptional
	mnse	2009, 2011	96	81	Exceptional	Exceptional
Agricultural	ag20	2008, 2010	64	61	Good	Good
	ag27	2008, 2010	44	62	Fair	Good
	ag28	2008, 2010	55	69	Good	Good
	ag37	2008, 2010	36	63	Fair	Good
	ag41	2008, 2010	56	61	Good	Good
	ag43	2008, 2010	57	45	Good	Fair
	ag67	2008, 2010	62	80	Good	Exceptional
	ag18	2009, 2011	62	83	Good	Exceptional
	ag23	2009, 2011	61	58	Good	Good

Appendix F continued

Agricultural	ag34	2009, 2011	73	79	Exceptional	Exceptional
	ag45	2009, 2011	79	82	Exceptional	Exceptional
	ag66	2009, 2011	56	67	Good	Good
	ag78	2009, 2011	67	79	Good	Exceptional
	ag80	2009, 2011	66	55	Good	Good
	ag81	2009, 2011	28	30	Fair	Fair
Constructed	bear	2008, 2010	46	46	Fair	Fair
	brin	2008, 2010	7	24	Poor	Poor
	call	2008, 2010	41	16	Fair	Poor
	cano	2008, 2010	50	53	Fair	Good
	chir	2008, 2010	23	13	Poor	Poor
	clark	2008, 2010	49	61	Fair	Good
	holken	2008, 2010	15	34	Poor	Fair
	huds	2008, 2010	5	46	Poor	Fair
	mead	2008, 2010	4	34	Poor	Fair
	ruth	2008, 2010	30	46	Fair	Fair
	silv	2008, 2010	50	64	Fair	Good
	terw	2008, 2010	49	32	Fair	Fair
	twin	2008, 2010	7	5	Poor	Poor
	vale	2008, 2010	1	0	Poor	Poor