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# **Ecological Indicators**



# Use of single versus multiple biotic communities as indicators of biological integrity in northern prairie wetlands

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# ABSTRACT

As much as 70% of prairie wetlands in Canada have been lost. Although further degradation of natural wetlands is considered to be somewhat offset by wetland construction and restoration, Canada lacks bioassessment tools that can track ecosystem health in prairie wetlands. Indices of biological integrity (IBIs) use one or more biotic communities to compare the biological condition of a particular site to conditions found in least-impacted reference sites. Using the IBI approach, we evaluated the potential of 5 biotic communities to assess wetland health in northern prairie wetlands in Canada. Vegetation in the wet meadow, emergent and open-water zones as well as wetland-dependent songbirds and waterbirds were sampled at 81 semi-permanent/permanent natural and compensation wetlands spanning an environmental stress gradient. Metrics with strong linear relationships to the stress gradient ( $R^2 > 0.2$ ) were combined into an IBI for each biotic community and were subsequently validated at a suite of test sites. After validation, the entire data set was combined and each IBI was evaluated based on its linear relationship to environmental stress. Wet meadow zone vegetation was a strong indicator of environmental stress ( $R^2 = 0.68$ , p < 0.001), as was the wetland-dependent songbird community ( $R^2 = 0.59$ , p < 0.001). The emergent zone vegetation community was a relatively weak and inconsistent indicator of environmental stress, while the open-water zone vegetation and waterbird communities were poor indicators. To evaluate whether monitoring more than one biotic community provided additional information about a site's biological and environmental condition, we produced a two-taxon IBI that combined wet meadow zone vegetation and wetland-dependent songbird metrics. The two-taxon IBI had a marginally stronger linear relationship to the stress gradient ( $R^2 = 0.72$ , p < 0.001) than any single biotic community alone, although we argue that this added information would not warrant the extra cost, effort, and logistical barriers of sampling both plants and birds. The wet meadow zone vegetation and wetland-dependent songbird IBIs were strong surrogates of one another ( $R^2 = 0.57$ ), suggesting that wet meadow zone vegetation can be used to predict the health of wetland-dependent songbirds, and visa versa. Our results suggest that habitat for healthy wet meadow zone vegetation and wetland-dependent songbird communities is being degraded as compensation sites are replacing their natural analogs.

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# 1. Introduction

The prairie provinces in Canada have experienced persistent loss of wetlands, significantly altering the landscape and compromising the functioning of natural ecosystems (Dahl and Watmough, 2007). As much as 70% of prairie wetlands in Canada have been lost since European settlement, primarily due to agricultural reclamation (Kennedy and Mayer, 2002; Mitsch and Gosselink, 2007). Today, urban and suburban development is continuing to degrade the remaining wetlands on the landscape. The deterioration of aquatic ecosystems has presently shifted the focus to conserving wetland functions and biodiversity (Kennedy and Mayer, 2002). Wetlands provide critical breeding habitat for waterfowl and wetland-dependent songbirds and support a high diversity of organisms specifically adapted to living in the ecotone between aquatic and terrestrial habitats.

Although further degradation of natural wetlands is considered to be somewhat offset by mitigation projects, wetland management and conservation in this region lacks bioassessment tools that can compare and monitor ecosystem health of compensation sites relative to their natural counterparts. 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 in Canada is lacking (Rubec and Hanson, 2009).

The index of biological integrity (IBI) is a bioassessment tool that uses indicators to assess the condition of a particular site relative

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to conditions found at reference sites that are least-impacted by human influence. In the IBI approach, a taxonomically related biotic community whose biological condition reliably predicts human disturbance is typically monitored. A biotic community is a broad term for a group of organisms found in a particular place or habitat (Morin, 1999). Northern prairie marshes typically have physically defined vegetation communities making up 3–4 zonations along the moisture gradient, each of which can be used to indicate ecosystem health (Adamus, 1996).

Although 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 organisms including plant and bird communities. In the IBI approach, scientists search for biological attributes (metrics) that can predict underlying environmental stress. Environmental stress can be quantified by measuring 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 the stress gradient are then selected, standardized and combined into an IBI. Rigorous testing needs to be done to insure that the predictive relationship between biological integrity and environmental stress remains consistent across space and time (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.

The sensitivity of a particular taxon to human influence varies by region and wetland type depending on the kinds of stressors being assessed and the scale at which they are measured (Brazner et al., 2007). Hence, it is important to evaluate several biotic communities to determine suitable bioindicators that are able to predict ecosystem health (i.e. Cvetkovic and Chow-Fraser, 2011; Seilheimer et al., 2009). Plant communities are perhaps the most widely used bioindicators for wetland monitoring. Advantages of using plants as bioindicators include their presence in almost all wetlands, low cost of sampling, and known sensitivity to specific stressors (Adamus and Brandt, 1990; Nevel et al., 2004). The disadvantages of using plants as bioindicators include their limited social value, laborious sampling methods, and lagged response to some stressors, which may allow plants to survive in poor conditions for many years (Adamus, 1996; Nevel et al., 2004). In addition, plant community composition in prairie wetlands changes in response to fluctuations in water level and hydroperiod (Stewart and Kantrud, 1971; Van der Valk and Davis, 1978). Some scientists have contended that this inter-annual variation in species composition leads to inconsistent IBI scores over time (Euliss and Mushet, 2011; Wilcox et al., 2002).

Birds also hold potential as bioindicators because of their high social value and their relatively easy 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). However, their migratory behavior could introduce uncertainties as to whether changes in species populations are indeed due to local factors or are a product of events occurring elsewhere (Nevel et al., 2004). Other disadvantages of birds as bioindicators include variability associated with differing detection probabilities among species and among sites (Hutto et al., 1986); variability in observer detection; and uncontrolled variables such as noise, time of day, and weather. Furthermore, it is difficult to confirm breeding pairs of waterbirds from individuals that are using the wetland as a staging area.

Although IBIs usually use a single, taxonomically related community as a surrogate proxy of ecosystem health, there is concern whether one community adequately reflects the biological integrity of other biota. Other studies have found that biotic communities 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 community congruence that is stressand scale-dependent (Brazner et al., 2007). A multi-community approach could be necessary for a comprehensive monitoring program (Cairns et al., 1993). However, the trade-off between increased cost and sampling effort and the amount of information gained may not be worthwhile, especially if different groups of organisms have similar responses to stress and are thus redundant (Karr et al., 1986; O'Connor et al., 2000). Furthermore, it may not be practical to monitor multiple communities if the goal of monitoring is to reduce complex ecological information into a simple and applicable tool for policy and management objectives. Hence, unless monitoring multiple biotic communities results in a substantially better evaluation of wetland health, a single community could suffice for most bioassessment goals.

The research for this study involved sampling vegetation communities from the wet meadow, emergent and open-water zones as well as wetland-dependent songbird and waterbird communities at 81 wetlands in the northern prairies of Canada spanning a range of environmental stress. Our first goal was to use the IBI approach to evaluate the potential of each biotic community to assess biological integrity. Our second goal was to evaluate whether a single community represented the health of other biota. We specifically hypothesized that (1) the 5 biotic communities would exhibit varying sensitivities to the underlying gradient of environmental stress; (2) the two-taxon IBI combining IBIs developed from plant and bird communities would represent more variance in environmental stress than any single biotic community alone; and (3) biotic communities might be good predictors of each other if their sensitivity to environmental stress.

# 2. Methods

#### 2.1. Study area

Our study area was located in the northern prairies in Alberta, Canada, within the Aspen Parkland Ecoregion (Fig. 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 depressional wetlands in remnant natural areas. The landscape is dominated by agriculture as well as some urban and suburban areas. 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 15 °C in 2008 and 14°C in 2009 (AgroClimatic Information Service). Accumulated precipitation during the growing season between May and September was 180 mm in 2008 and 173 mm in 2009, which is considerably lower than recent averages of 286 mm between 1971 and 2000 (AgroClimatic Information Service). Temperature and precipitation data was acquired by taking averages from 5 weather stations distributed throughout the study area.

# 2.2. Sampling design

In 2008 and 2009, we sampled 81 natural and compensation wetlands, of which 27 were least-impacted reference sites, 19 were agricultural sites, 9 were restored sites, 16 were naturalized stormwater management ponds, and 11 were classic stormwater management ponds. Compensation sites were compared to their natural analogs, semi-permanent to permanent prairie wetlands (Stewart and Kantrud, 1971), which contain water for most or all of the year. Natural wetlands in this region typically have 3–4 zonations characterized by the following vegetation communities:



Fig. 1. Location of Aspen Parkland Ecoregion in Canada and inset of study sites in the northern prairies of Alberta.

submersed and floating aquatic plants in the shallow-open water zone; annuals and young emergent species in the drawdown zone; robust emergents (i.e. Typha latifolia) in the emergent zone; and a mixture of grasses, sedges and other herbaceous plants in the wet meadow zone. Sampling from the drawdown zone was not included in the analysis. Sites were between 1 and 13 ha, including the open-water and emergent vegetation zones. The wet meadow zone was not included in total wetland area measurements because it was too difficult to distinguish from the surrounding upland using aerial imagery. Reference sites were surrounded by >50% undisturbed forest within a 500 m radius, representing the region's least-disturbed range of natural variability. Agricultural sites were surrounded by >50% cultivated or grazing land within 500 m. Restored sites were generally surrounded by pasture and had been restored >3 years prior to sampling. Naturalized stormwater management ponds differ from classic stormwater management ponds in that they are designed to mimic some appearances and functions of natural wetlands. All constructed sites were >3 years old (mean age = 17). Permission to access all sites on private land was obtained.

#### 2.2.1. Biological sampling

Vegetation in the emergent and wet meadow zones was sampled between late July and August, when peak biomass is expected. Due to an extensive drought over the past several years, which eliminated the established emergent vegetation at some sites, the sample size in the emergent zone was reduced to 57 sites rather than 81. Likewise, 79 sites were included in wet meadow vegetation sampling because the wet meadow zone was missing at 2 sites.

To sample macrophytes in the emergent and wet meadow zones, sites were randomly divided into thirds by three radial transects. At each transect, two 1 m × 1 m quadrats were sampled in the middle of every zone present with quadrat pairs spaced 5 m apart, totaling 6 quadrats per zone (Raab and Bayley, 2012). Sometimes additional quadrats were sampled in very wide zones at natural sites, but exactly 6 quadrats per zone were randomly selected for use in analysis. An a priori power analysis found that a sample size of 4 quadrats in the wet meadow zone could 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 (or genus) following Moss and Packer (1983) and their percent cover was estimated to the nearest 5%. Width of each zone was measured at each transect. In addition, the Robel pole

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 vegetation height and a stem count of *T. latifolia* was conducted instead. All plant species names were updated according to the International Code of Botanical Nomenclature (see Appendix A).

Submersed and floating aquatic vegetation (SAV) in the openwater zone was sampled using the rake technique described by Rooney and Bayley (2011b). The sampler navigated by kayak to 10 stratified-random locations in the open water zone (>50 cm deep) and made a 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 had to be sampled at depths < 50 cm because of low water levels. All plant species names were updated according to the International Code of Botanical Nomenclature (see Appendix B).

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

#### 2.2.2. Physicochemical sampling

Water samples were collected at each wetland in late July and analyzed for ammonia ( $NH_4^+$ ), nitrate ( $NO_2NO_3$ ), total nitrogen (TN), total dissolved nitrogen (TDN), soluble reactive phosphorous (SRP), total phosphorous (TP), total dissolved phosphorous (TDP), chloride, sulfate ( $SO_4^{2-}$ ), sodium (Na), potassium (K), calcium (Ca), magnesium (Mg), iron (Fe), aluminum (Al), dissolved organic carbon (DOC), silicone dioxide ( $SiO_2$ ), non-filterable residue, and chlorophyll a, as described by Bayley and Prather (2003). All water analyses were done in the Limnological Laboratory, University of Alberta. In addition, pH and conductivity were measured in situ with a Hach meter.

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. Samples from sediment cores were analyzed for percent nitrogen (% N) and carbon (% C) in the Limnological Laboratory, University of Alberta. Percent phosphorous (% P) was also 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  $60 \circ C$  for 48 h) of a  $50 \pm 1 g$  sample (wet mass) was weighed to calculate the percent water content in the sediment. We used a Mettler Toledo AE240 balance  $(\pm 0.0001 \text{ mg})$  to weigh a 0.5 g sample that was placed in a muffle furnace at 550 °C for 4 h and re-weighed to determine the loss on ignition (Rooney and Bayley, 2010).

Shoreline slope was determined at each vegetation transect by measuring the height of a laser beam 10 m away from the edge of the open water and calculating the rise over run. In July, water clarity was estimated with a secchi disk at 10 locations in the open-water zone. Wetland area was estimated by digitizing aerial imagery in ESRI's ArcGIS 9.0 (2008).

### 2.3. IBI approach

We evaluated the suitability of five biotic communities as indicators of biological integrity based on their sensitivity to underlying environmental stress. First, individual metrics were tested and selected based on their sensitivity to a gradient in environmental stress at a stratified random subset of sample sites (N = 54). Metrics were standardized and combined into an IBI, which was subsequently validated against stress scores at a suite of independent test sites (N = 27). A strong linear relationship between IBI and stress scores at test sites insured that IBIs represented changes in biological integrity reflecting underlying environmental stress. Once validated, we used the entire data set to evaluate each IBI based on the amount of variance explained by the linear regression. Pearson's correlation was also performed to test for correspondence between IBIs developed from differing biotic communities.

#### 2.3.1. Stress gradient development

A stress gradient based on underlying physical and chemical variables was developed to establish a representative gradient of environmental conditions at sites in our study area. We quantified the gradient in environmental stress using methodologies described in detail by Rooney and Bayley (2010). In brief, forty-one initial variables that we expected would influence plant community health were measured at each of the 81 study sites. These variables were grouped into three abiotic categories (physical variables, water chemistry, and sediment chemistry). We performed a Principal Components Analysis (PCA) to reveal metrics that were most strongly correlated with the PCA's orthogonal axes. The following 8 metrics were selected into the stress index: shoreline slope, proportion secchi depth; NO<sub>2</sub>NO<sub>3</sub>, TN, and conductivity in the open water zone; and % N, % P, and % water content in the sediment. To standardize variables, we used a percentile binning approach recommended by Rooney and Bayley (2010). First, the direction of correlation was corrected so that the values of all the variables increased in correspondence with increasing doses of stress. We used 20th percentile increments to convert each variable into a score between 1 and 5 (i.e. 1st to 20th percentile = 1, ..., 81st-100th percentile = 5). Variable scores within each abiotic category were averaged so that physical variables, water chemistry, and sediment chemistry were weighted equally. Category scores were then

#### Table 1

Types of metrics that were tested for a linear relationship to the environmental stress gradient. Candidate metrics ( $R^2 > 0.2$ ) from biotic communities are underlined. WMZ=wet meadow zone vegetation, EMZ=emergent zone vegetation, SAV= submersed/floating aquatic vegetation, SB=wetland-dependent songbirds, WB= waterbirds.

Type of metric	Biotic community
Biological productivity	
Robel height	WMZ
T. latifolia stem count	EMZ
Community richness and composition	
Diversity indices	WMZ, <u>EMZ</u> , <u>SB</u> , WB, SAV
Total species richness	WMZ, EMZ, SB, WB, SAV
Total species abundance	WMZ, <u>EMZ</u> , SB, WB, SAV
Native species	<u>WMZ</u> , <u>EMZ</u> , SAV
Invasive species	WMZ, EMZ
Species of concern	SB, WB
Taxonomic/structural guilds	
Group (e.g. monocots)	WMZ, EMZ, <u>SAV</u>
Life cycle (e.g. perennials)	WMZ, EMZ, SAV
Growth habit (e.g. graminoids)	WMZ, <u>EMZ</u> , SAV
Dietary need (e.g. carnivore)	<u>SB</u> , WB
Foraging mode (e.g. aerial)	<u>SB</u> , WB
Nesting location (e.g. ground)	<u>SB</u> , WB
Taxonomic groups	<u>WMZ</u> , EMZ, <u>SB</u> , WB
Habitat	
Wetland-associated species	<u>WMZ</u> , <u>EMZ</u> , SB, WB, SAV
Width of vegetation zone	<u>WMZ</u> , EMZ
Floristic quality assessment index (FQAI)	<u>WMZ</u> , <u>EMZ</u>
Sensitive/tolerant species	WMZ, EMZ, SAV

combined and rescaled to produce a stress score ranging between 0 and 10.

#### 2.3.2. Metric testing, selection, and standardization

To determine whether communities had indicators that were sensitive to underlying environmental stress, we used linear regression to test for a relationship between individual metric scores and stress scores. Potential metrics measured a variety of attributes ranging from richness and composition to guild structure to habitat quality (Table 1). An additional metric we tested was the floristic quality assessment index (FQAI), which was calculated in each vegetation community. In a previous study by Forrest (2010), every marsh plant species in the region was assigned a coefficient of conservatism (*C*-value) between 0 and 10 based on its fidelity to specific habitat types and tolerance to disturbance (Lopez and Fennessy, 2002; Miller and Wardrop, 2006). We calculated FQAI scores by multiplying the mean *C*-value at each site by the square root of its native species richness.

Approximately 65 metrics were tested from each of the 5 biotic communities (Table 1). Metrics with  $R^2$ -values > 0.2 were considered candidate metrics (Mack, 2007). Arcsine transformations were performed 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. The final selection of metrics was determined by performing a redundancy analysis of candidate metrics. In cases where two metrics had a Pearson's correlation coefficient > 0.7 (Rooney and Bayley, 2011a), only the metric with the largest  $R^2$ -value was retained for use in the IBI.

A continuous reference range approach was used to score metrics, which 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 we set as the difference between the 75th percentile of reference sites (upper bound) and the 25th percentile of constructed sites (lower bound) (Table 2). Metric scores were summed and rescaled to produce an IBI score between 0 and 100. In addition, a two-taxon IBI was produced by combining IBIs from the wet meadow zone vegetation and songbird communities.

#### 2.4. Wetland health categories

We used classification and regression tree analysis (CART) to objectively group sites into health categories, as described by Wardrop et al. (2007). CART builds a regression tree by recursively partitioning data into two mutually exclusive groups. It chooses the predictor variable that best describes the response variable and provides 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 community. Each model was pruned to yield 4 health categories: Exceptional, Good, Fair, and Poor.

# 3. Results

#### 3.1. Metrics selected for use in IBIs

A subset of sites was designated to test and select several non-redundant metrics exhibiting a strong relationship to the environmental stress gradient. After redundancy analysis, four non-redundant vegetation metrics were selected for use in the wet meadow zone vegetation IBI (Table 2): average width of the wet meadow zone ( $R^2 = 0.65$ , p < 0.001); % of total cover of *Carex* spp. ( $R^2 = 0.44$ , p < 0.001); % of total cover of native perennials ( $R^2 = 0.35$ , p < 0.001); and the FQAI ( $R^2 = 0.39$ , p < 0.001). In the emergent zone, the FQAI ( $R^2 = 0.43$ , p < 0.001) and % cover of native species ( $R^2 = 0.42$ , p < 0.001) were the only sensitive metrics after redundancy analysis (Table 2). Only 1 non-redundant metric from the submersed and floating vegetation community, % cover of dicots, had an acceptable relationship to the stress gradient ( $R^2 = 0.22$ , p < 0.001).

Five non-redundant metrics were selected for use in the wetland-dependent songbird IBI (Table 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),

#### Table 2

Linear relationship of individual metrics to the environmental stress gradient. Upper and lower bounds represent metric values used to standardize metrics into scores. Upper bounds represent the 75th percentile of reference sites and lower bounds represent the 25th percentile of all constructed stormwater management ponds. WMZ = wet meadow zone vegetation, EMZ = emergent zone vegetation, SB = wetland-dependent songbirds.

Metric	Biotic community	<i>R</i> <sup>2</sup>	Upper bound	Lower bound
Width of wet meadow zone (meters)	WMZ	0.65***	31.2	1.8
% of total species of insecti- vores/granivores	SB	0.47***	85.7	33.3
% of total cover of <i>Carex</i> spp.	WMZ	0.44***	41.5	0.0
% of total species of ground nesters	SB	0.44***	88.8	37.3
FQAI	EMZ	0.43***	11.3	2.4
% cover of native species	EMZ	0.42***	60.4	3.3
FQAI	WMZ	0.39***	13.3	4.0
No. of temperate migratory spp.	SB	0.37***	2	0
% of total cover of native perennials	WMZ	0.35***	90.7	17.6
Abundance of canopy foraging spp.	SB	0.35***	20.9	0.0
No. of passerines	SB	0.31***	7	3

\*\*\* p < 0.001.

#### Table 3

Linear relationship ( $R^2$  values) between IBI and stress gradient scores. Two-taxon IBI = wet meadow zone vegetation + wetland-dependent songbirds.

Biotic community	Development	Test	All sites
Wet meadow zone vegetation	0.66***	0.73***	0.68***
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***	0.72***
* = <0.05			

p < 0.05.

\*\*\* *p* < 0.001.

relative abundance of canopy foraging species ( $R^2 = 0.35$ , p < 0.001) and number of passerine species ( $R^2 = 0.31$ , p < 0.001). The waterbird community did not have any metrics that were sensitive to the environmental stress gradient.

#### 3.2. Evaluation of IBIs developed from a single biotic community

The wet meadow zone vegetation, emergent zone vegetation, and wetland-dependent songbird communities all had multiple sensitive metrics to the environmental stress gradient. These metrics were standardized and summed into an IBI score representing a site's biological integrity and its underlying environmental conditions. Once validated, we used all sites in the data set to evaluate IBIs based on their relationship to the stress gradient. The wet meadow zone vegetation community was the best predictor of environmental stress ( $R^2 = 0.68$ , p < 0.001, Table 3 and Fig. 2a), followed by the wetland-dependent songbird community ( $R^2 = 0.59$ , p < 0.001, Table 3 and Fig. 2b), and the emergent zone vegetation community ( $R^2 = 0.41$ , p < 0.001, Table 3). There was also a strong correlation between the wetland-dependent songbird IBI and the wet meadow zone vegetation IBI ( $R^2 = 0.57$ , p < 0.001, Fig. 3), indicating that these 2 biotic communities are strong surrogates of each other.

IBI scores were tallied at a subset of independent test sites reserved to insure that the correlation between biological integrity (IBI score) and environmental condition (stress score) was real. The wet meadow zone vegetation IBI had a very consistent linear relationship to the stress gradient at test sites ( $R^2 = 0.73$ , p < 0.001, Table 3), as did wetland-dependent songbirds ( $R^2 = 0.50$ , p < 0.001, Table 3). The emergent zone vegetation IBI had a comparatively weaker but still significant relationship to the stress gradient at test sites ( $R^2 = 0.30$ , p < 0.05, Table 3).

#### 3.3. Evaluation of the two-taxon IBI

We combined IBIs from out two best indicator communities, wet meadow zone vegetation and wetland-dependent songbirds, into a two-taxon IBI to evaluate whether integrating multiple biotic communities strengthened the correlation between biological integrity and environmental stress. The two-taxon IBI had a slightly stronger relationship to the stress gradient ( $R^2 = 0.72$ , p < 0.001, Table 3 and Fig. 2c) than either the wet meadow zone vegetation IBI ( $R^2 = 0.68$ ) or the wetland-dependent songbird IBI ( $R^2 = 0.59$ ) alone.

# 3.4. Wetland health categories

CART provided thresholds that delineated wetland health categories (Table 4). The wet meadow zone vegetation IBI had the most conservative threshold for sites in "Good" health (Fig. 2a and Table 4), whereas the wetland-dependent songbird IBI had the most conservative threshold for sites in "Exceptional" health (Fig. 2b and Table 4). The wet meadow zone vegetation IBI ranked nearly all reference sites as "Exceptional" while the wetland-dependent songbird IBI ranked some reference sites as "Exceptional" and others as "Good" (Table 5). Both the wet meadow zone vegetation and wetland-dependent songbird IBIs had comparable thresholds for



Fig. 2. Site IBI scores regressed against stress scores using the (a) wet meadow zone vegetation IBI; (b) wetland-dependent songbird IBI, and (c) two-taxon IBI (wet meadow zone vegetation + wetland-dependent songbirds). Thresholds for health categories are shown as defined by classification and regression tree analysis (CART). (●) Reference; (▲) agricultural, (△) restored, (○) naturalized constructed stormwater management ponds and (□) classic stormwater management ponds.



**Fig. 3.** Linear relationship between wetland-dependent songbird and wet meadow zone vegetation IBI scores. (•) Reference; (•) agricultural, ( $\triangle$ ) restored, ( $\bigcirc$ ) naturalized constructed stormwater management ponds and ( $\Box$ ) classic stormwater management ponds.

"Poor" and "Fair" categories (Table 4). The emergent vegetation IBI had the least conservative thresholds for all health categories (Table 4). Thresholds produced by the two-taxon IBI were very similar to those of the songbird IBI (Fig. 2c and Table 4).

# 4. Discussion

We chose to investigate plant and bird communities as bioindicators for several reasons: first, field sampling of birds and plants are staggered, with breeding bird sampling taking place in May-June and vegetation sampling taking place in July-August. Second, sampling and identification of vegetation and bird communities is straightforward and less costly in comparison to other taxonomic groups (i.e. macroinvertebrates, microbes, algae). Third, both plants and birds are known to be sensitive to human influence. Plants have been used widely as wetland indicators in other studies, while birds, although not as established as wetland indicators, would be of particular interest to managers because of their value to society. However, we made no presuppositions as to which biotic communities held the highest potential as bioindicators, as no equivalent bioassessment study has been conducted in this region before. Thus, we assumed that a comprehensive study evaluating the potential of several biotic communities was required to identify suitable indicators of biological integrity and environmental stress.

#### Table 4

Wetland health category ranges on a scale between 0 and 100. Thresholds were determined by classification and regression tree analysis (CART). WMZ=wet meadow zone vegetation, EMZ=emergent zone vegetation, SB=wetland-dependent songbirds, two-taxon=WM+SB.

1					
	Health category	WMZ-IBI	EMZ-IBI	SB-IBI	Two-taxon IBI
	Exceptional Good Fair Poor	>69 >51 and <70 >25 and <52 <26	>62 >21 and <63 >11 and <22 <12	>81 >37 and <82 >21 and <38 <22	>80 >40 and <81 >29 and <41 <30

#### 4.1. Metrics selected for use in IBIs

The width of the wet meadow zone, which decreased with environmental stress as expected, has been used in previous IBIs developed in Alberta (Raab and Bayley, 2012). It is a simple and straightforward metric to measure in the field and can be scaled up by measuring it by remote sensing (Creed, pers. comm.). Wider wet meadow zones likely correspond to greater habitat availability for species in other trophic levels. FQAI scores also had a negative relationship to environmental stress as expected. FOAIs have been used as indicators in several other studies (DeKeyser et al., 2003; Mack, 2007; Raab and Bayley, 2012), and although calculating this metric requires more intensive sampling effort and identification skills, it is useful because it estimates habitat quality (Miller and Wardrop, 2006). Metrics related to % cover of Carex and native species were also negatively correlated with the stress gradient as expected, as these metrics are known to be sensitive to both agricultural and urban impacts (Galatowitsch et al., 1998) and have been used as indicators in other regions as well (DeKeyser et al., 2003; Simon et al., 2001). Carex plays a vital role in wetland functioning via nutrient uptake, cycling and primary production (Bernard et al., 1988) and are a dominant genus in northern prairie wetlands.

The richness and composition of insectivores/granivores, ground nesting species, temperate migratory species, canopy foraging species and passerines all had negative linear relationships to environmental stress as expected. These metrics are likely correlated to the stress gradient because both environmental conditions and songbird integrity likely covary with surrounding habitat and land use alterations. Other bird community indices have been found to be strongly correlated with habitat quality indices (Canterbury et al., 2000).

#### 4.2. Evaluation of IBIs developed from a single biotic community

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 Canada (Raab and Bayley, 2012; Rooney and Bayley, 2011a). The wet meadow zone vegetation IBI had the strongest and most reliable relationship to the environmental stress gradient out of all 5 communities that were examined.

#### Table 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 stormwater management ponds made up 20% of total sites, and classic stormwater management ponds made up 14% of total sites. WMZ = wet meadow zone vegetation, EMZ = emergent zone vegetation, SB = songbirds, Two-taxon = WM + SB.

Health category	WMZ-IBI	EMZ-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

Since the wet meadow zone vegetation community was the only plant community found to be consistently sensitive to environmental stress, we argue that monitoring vegetation in the wet meadow zone alone is better than calculating plant metrics based on the entire wetland. Our results showed that different plant communities have varying sensitivities to environmental stress, and when assessed together, the individual signal of the wet meadow zone vegetation community to stressors would likely be masked. Raab and Bayley (2012) also developed a successful IBI for natural and oil sands reclamation boreal marshes using the wet meadow zone vegetation community. Out of the 4 metrics we selected in our wet meadow zone vegetation IBI, 3 of them were either similar or identical to those used by Raab and Bayley (2012): width of the wet meadow zone, FQAI and % of total cover of Carex spp. The similarity in component metrics between studies suggests that these metrics are reliably sensitive to a variety of stressors in differing wetland types and regions.

Wetlands can be parceled into health categories that are more comprehensible to the public and more practical for managers to use than an index score. Wetland health categories can be used by managers to monitor wetland compliance and compensation success and to set benchmarks or thresholds for certification and approval. Typically, studies have recommended preliminary wetland health categories that are often based on best professional judgment in order to satisfy management requirements; however, we used an objective approach (i.e. CART) to delineate health categories based on a study by Wardrop et al. (2007). The wet meadow zone vegetation IBI produced fairly conservative health categories and grouped nearly all our reference sites as "Exceptional," which verified that our choice of reference sites did in fact represent the reference condition.

Vegetation in the emergent zone could only be sampled at 60 of the 81 sites because the vegetation in this zone had died off at some sites due to prolonged dry conditions. Low sample size likely reduced the power of the emergent zone vegetation IBI. Furthermore, the emergent zone vegetation IBI had only 2 metrics after redundancy analysis, which is likely not sufficient to produce a robust IBI that is reliable over years with varying water levels. The logistical issues we encountered in the emergent zone support Wilcox et al.'s (2002) argument that hydrologic fluctuation would make plant metrics in some zones inconsistent from year to year.

An IBI could not be produced with the submersed and floating vegetation community even though it was successfully developed for northern natural and reclamation marshes in Alberta (Rooney and Bayley, 2011a). In northern reclamation marshes, submersed and floating species were likely sensitive to hydrocarbon and salt-related toxicity (Rooney and Bayley, 2011a), suggesting that differences in types of stressors likely led to differing results between studies. Low species richness (mean total richness = 4) likely influenced the poor outcome of submersed and floating aquatic vegetation as an indicator of biological integrity in this region.

The IBI developed from wetland-dependent songbirds had the second strongest relationship to the stress gradient after the wet meadow zone vegetation IBI. The wetland-dependent songbird IBI had the most conservative threshold for sites in "Exceptional" health, suggesting that managers could use songbirds as bioindicators to identify wetlands that support healthy songbird communities. Perhaps most interesting, however, is that the wet meadow zone vegetation IBI scores could predict wetland-dependent songbird IBI scores and visa versa, which supports evidence found by Canterbury et al. (2000) that habitat indices are good surrogates of bird integrity. Several metrics in the wet meadow zone, are proxies of habitat quality. Rooney and Bayley (2012) found in a related study that community congruence between wetland-dependent songbird and wet meadow zone vegetation communities was relatively low, despite both covarying with similar environmental variables. As a result, they concluded that bioassessment tools using a single biotic community might not adequately represent biological health of other biota. Because we found that IBI scores produced from wet meadow zone vegetation and wetland-dependent songbird communities are correlated, we argue that IBIs *can* reflect the biological integrity of other organisms even if they do not have congruent community compositions. The constituent metrics in the wet meadow zone vegetation IBI (i.e. width of the wet meadow vegetation, FQAI) are likely good indicators of biological integrity because they are broad, represent habitat quality, and subsequently reflect other biotic communities.

In contrast to the songbird community, 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 particularly interested in managing waterfowl populations.

#### 4.3. Evaluation of the two-taxon IBI

The two-taxon IBI had a marginally stronger relationship to the stress gradient than any single biotic community alone. This result probably occurred because the wet meadow zone vegetation and wetland-dependent songbird communities are strong surrogates of one another and contain redundant metrics. Monitoring both plants and birds would substantially raise the cost and effort of a wetland bioassessment program and could reduce the utility of the IBI. Hence, we argue that combining multiple biotic communities into a single index of health might not be worthwhile in some situations where the added information does not offset the increase in cost and effort of sampling multiple communities. Different sampling methods and field training skills would be required to monitor both breeding birds and marsh vegetation, and differing times of year are required for sampling each of these organisms. The wet meadow zone vegetation and wetland-dependent songbird IBIs both covaried with the stress gradient and were correlated with each other. Hence, we believe that they could be utilized separately during different times of the year. In central Alberta, for example, wetland-dependent songbirds could be used as a bioindicator during the breeding season in May-June, whereas wet meadow zone vegetation could be used in late July-August.

# 4.4. Conclusions

We found that the suitability of different biotic communities varied widely, and that not all communities are sensitive to a gradient of environmental stress. We determined that the wet meadow zone vegetation and wetland-dependent songbird communities are good indicators of environmental stress while emergent zone vegetation, open-water zone vegetation and waterbirds are relatively poor to poor indicators of environmental stress. The wet meadow zone vegetation and wetland-dependent songbird IBIs were strong surrogates of each other, indicating that sampling a single biotic community can reflect the health of other organisms of differing trophic levels. Contrary to our 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 zone vegetation IBI alone. Both the wet meadow zone vegetation and wetland-dependent songbird communities revealed that constructed sites are in poorer biological health than natural sites and this decrease in biological condition reflects unhealthy underlying

environmental conditions. This finding suggests that wetland ecosystem health is deteriorating as constructed sites are replacing natural wetlands on the landscape.

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#### Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.ecolind.2012.02.009.

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