PROPERTIES AND PERFORMANCE OF THE FLORISTIC QUALITY INDEX IN GREAT LAKES COASTAL WETLANDS

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ABSTRACT

The Floristic Quality Index (*FQI*) has been proposed as a tool that can be used to: identify areas that have high conservation value, monitor sites over time, and measure the ecological condition of an area. *FQI* is based on the Coefficient of Conservatism (*C*), a numerical score assigned to each plant species in a local flora that reflects the likelihood that a species is found in natural habitats. *FQI* is computed by multiplying the mean Coefficient of Conservatism ($\overline{C}$) by the square root of species richness for an observational unit. The properties and performance of various species richness, Coefficient of Conservatism, and Floristic Quality indices were assessed in Great Lakes coastal wetlands. Both *FQI* and species richness increased with sampling area according to a power function. $\overline{C}$ however, was not sensitive to sampling area except in fen communities. *FQI* should therefore be interpreted with knowledge of the underlying species-area relationship. Michigan and Wisconsin assign different values of *C* to the same species, highlighting the subjective nature of *C*. This could affect index scores and confound regional interpretations. *FQI* was correlated with diversity indices that are sensitive to changes in species richness. $\overline{C}$ was also correlated with these same diversity indices. Both Coefficient of Conservatism and Floristic Quality indices were better than species richness indices at discriminating differences between sites, independent of a surrogate condition gradient, but neither one type outperformed the other. Both types of indices were also found to be acceptable ecological indicators of condition, although Floristic Quality indices consistently outperformed Coefficient of Conservatism indices in this capacity. Regardless of the subjectivity involved with the assignment of *C*-values and that 'floristic quality' is a human concept and not a true ecosystem property, both Coefficient of Conservatism and Floristic Quality indices appear to be effective indicators of condition in Great Lakes coastal wetlands.

INTRODUCTION

Humans have affected many of the ecosystems of this planet. Terms such as ecosystem health and ecosystem integrity are often used to describe the condition of ecosystems that are exposed to anthropogenic stress. Though these terms are vague and depend as much on human perception as ecosystem condition (Regier 1993), there is no question that the actions of humans have led to the loss and alteration of some ecosystems. Thus, there is a need for well-defined tools that can assess these anthropogenic effects.

The Floristic Quality Index (*FQI*) has been proposed as such a tool. *FQI* is a biotic, or content based index (Washington 1984, Rooney and Rogers 2002), that is based on a numerical score called the Coefficient of Conservatism (*C*). *C*-values range from 0-10 and are assigned, subjectively, to each plant species within a local flora by a panel of experienced botanists. The theory behind *C* is that plant species differ in their tolerance to the type, frequency, and amplitude of disturbance and that plants exhibit a varying degree of fidelity to remnant natural habitats (Wilhelm 1977, Swink and Wilhelm 1994, Taft et al. 1997). *C*-values therefore reflect the likelihood that a species can be found in remnant natural habitats. Consider the following examples that illustrate assignment of *C*-values for the state of Wisconsin (Bernthal et al. 2001). *Acer negundo* L. (box elder) is a common tree in floodplain forests and riverbanks but is also found in urban waste areas (Chadde 1998). *Acer negundo* shows little fidelity to remnant natural habitats and is given a *C*-value of 0. Conversely, *Sarracenia purpurea* L. (pitcher plant) is restricted to relatively undisturbed peatlands (Chadde 1998), and is given a *C*-value of 10. Early versions of *FQI* emphasized rareness and native status by assigning *C*-values greater than 10 to rare or endangered species and assigning *C*-values less than 0 to introduced species (Wilhelm 1977). Current versions of *FQI* have de-emphasized rare and introduced taxa by assigning *C*-values between 0-10 (Swink and Wilhelm 1994).

FQI is calculated from the following formula

$$FQI = \overline{C}\sqrt{S} \tag{1}$$

where $\overline{C}$ is the mean *C* and *S* is the number of species, or species richness, of the area censused.

FQI was originally developed to identify areas that have high conservation value in the state of Illinois (Wilhelm 1977). Since then, *FQI* has been proposed as tool that can: compare different sites regardless of plant community type, monitor sites over time, assess the anthropogenic impacts on an area, and ultimately, measure the ecological condition of an area (Swink and Wilhelm 1994, Lopez and Fennessy 2002). *FQI* has been used to analyze historical data to determine long-term changes in plant communities in Wisconsin lakes (Nichols 2001), monitor a tall grass prairie restoration in Ohio (Poling et al. 2003), and set wetland mitigation standards in Illinois (Matthews 2003). The U.S. Environmental Protection Agency recommends the use of *FQI* for bioassessments of wetland condition (Fennessy et al. 2001). The Ohio Environmental Protection Agency currently uses *FQI* as a component in a multimetric Vegetation Index of Biotic Integrity (VIBI) for wetlands (Mack 2001). Seven states and one Canadian Province have assigned *C*-values to their floras, facilitating the use of *FQI*: Illinois (Wilhelm 1977, Swink and Wilhelm 1994, Taft et al. 1997), Missouri (Ladd 1993), Ohio (Andreas 1995), Southern Ontario (Oldham et al. 1995), Michigan (Herman et al. 2001), North and South Dakota (NGPFQAP 2001), and Wisconsin (Bernthal et al. 2001).

Critical evaluation of *FQI* has come only recently, even though it has been in use since 1977 and is gaining popularity among regulating agencies. A common criticism of *FQI* is the subjective assignment of *C*-values (Mushet et al. 2002). As previously stated, *C*-values are assigned by a panel of botanists, using their collective opinion of the fidelity of each species to natural habitats. Mushet et al. (2002) evaluated the effect of subjectivity by generating *C*-values for species from 204 prairie pothole wetlands in North Dakota, which were distributed along a known gradient of ecological condition. $\overline{C}$ and *FQI* scores were computed in four independent wetlands using both the panel generated and data generated *C*-values. Index scores were consistently greater when data generated *C*-values were used; however, the authors concluded that panel derived *C*-values were adequate because conclusions on wetland condition were the same based on both methods.

FQI has also been found to be positively correlated with sampling area due to the species richness parameter in the index (Francis et al. 2000, Rooney and Rogers 2002,

Matthews 2003). Species richness increases with area according to the well-known power function.

$$S = cA^z \tag{2}$$

where S is the expected number of species occurring in an area A, c is the expected number of species in a unit area (when $A = 1$), and z is the rate of species richness increase per incremental increase in area (Arrhenius 1921). Because *FQI* is essentially a weighted species richness index it should therefore depend on sampling area as does species richness.

In addition, *FQI* has been found to vary between different community types, by the time of the year sampled due to phenology, and by the ability of observers to identify plants (Rooney and Rogers 2002, Matthews 2003). *FQI* has been correlated with surrogate measures of ecological condition and therefore considered a useful ecological indicator (Mack 2001, Lopez and Fennessy 2002). *FQI* in depressional wetlands in Ohio was negatively correlated with plant biomass, which may be an indicator of wetland eutrophication (Fennessy et al. 2001, Lopez and Fennessy 2002).

While these results provide valuable information about *FQI*, further questions remain about its properties and performance. Studies that have examined the effect of sampling area on index scores have fit equation (2) to data, but did not use nested plots (Rooney and Rogers 2002, Matthews 2003). This leads to an overestimation of the slope (z) (Rosenzweig 1995), and could generate misleading results. Also, individual users have computed index scores in systematically different ways. Swink and Wilhelm (1994) advocate that introduced taxa should be excluded from index calculations, because introduced taxa were not involved with the evolution of native plant communities. Other authors include introduced taxa in index calculations ($C = 0$) (Rooney and Rogers 2002, Lopez and Fennessy 2002). Poling et al. (2003) have also presented index calculations that include measures of abundance. Users need to be further informed of the properties and performance of *FQI*, and the various ways to compute it, to be able to effectively and consistently apply it in an ecological monitoring and assessment program.

The objective of this study is to investigate several properties of *FQI*, as well as test its performance, in Great Lakes coastal wetlands. Specifically, this study seeks to: 1) determine the relationship between sampling area, species richness, and *FQI*; 2) compare

Michigan and Wisconsin *C*-values; 3) explore possible *FQI*-diversity index relationships; 4) compare the relative discriminant ability of various *FQI* indices, independent of a condition gradient; 5) test *FQI* as an indicator of ecological condition; and 6) compare the performance of various indices that include, or exclude, introduced species and/or indices that are weighted, or not weighted, by abundance.

Study Area

Great Lakes coastal wetlands occur in areas where the geomorphology of the shoreline of the Great Lakes allows for some protection against wave action, thereby allowing wetland plants to grow (Maynard and Wilcox 1997). This includes wetlands that are not directly fringing into the Great Lakes but have a hydrologic connection. These wetlands form a transition between the Great Lakes and the surrounding land area and are therefore influenced by abiotic factors and anthropogenic stressors from both (Minc and Albert 1998, Maynard and Wilcox 1997).

Coastal wetlands can be classified based on hydrogeomorphological characteristics. This study follows the general classifications by Keough et al. (1999), which include open-coast, river-influenced (or riverine), and protected wetlands. Open-coast wetlands occur where wetland vegetation fringes directly into the lake along a gently sloping water depth gradient. These wetlands are connected with lake waters and are therefore directly influenced by wave action, currents, and lake level. Soils of this wetland type are typically inorganic, with only a minimal surface organic layer. Open-coast wetlands can occupy long stretches of shoreline, such as in Saginaw Bay, MI, but more often are restricted to smaller embayments. Riverine wetlands are characterized by wetland vegetation bordering a river that empties into the Great Lakes. Riverine wetlands are influenced by both river and lake hydrology. Soil types are highly variable in riverine wetlands, ranging from sandy to clayey inorganic soils to thick organic soils, due to the dynamic nature of river channels and sediment deposition (Johnston et al. 2001). Protected wetlands form behind natural structures, such as sandbars formed by longshore currents. Protected wetlands may or may not be directly connected to lake surface waters by narrow channels through lakeward sandbars. Thick organic soils typically form in protected wetlands, because they are isolated from direct lake

disturbances. In the northern portion of the upper Great Lakes, floating peatmats are often found in protected wetlands.

Human activities have affected all of the coastal wetlands within the Great Lakes basin to some degree. Effects range from subtle basin-wide differences in precipitation and temperature due to global climate change (Mortsch and Quinn 1996, Mortsch 1998) to the complete loss of wetlands due to draining and filling. In Saginaw Bay, MI, there was an estimated loss of 19,620 acres of coastal wetlands between the time the area was settled by European descendents and the 1970s (Herdendorf et al. 1981). The loss was attributed to draining and filling for agricultural and residential purposes, as well as coastal erosion. Other anthropogenic stressors, human caused external factors that in turn cause change in ecosystems (Rapport et al. 1985, Detenbeck et al. 1999), have led to the alteration of ecological condition in coastal wetlands.

Humans have altered the patterns of natural water level fluctuation in many coastal wetlands. Lake levels have been under partial control in Lake Superior for most of the 20th century and in Lake Ontario since the opening of the St. Lawrence Seaway (Maynard and Wilcox 1997). In this case, the stress to coastal wetlands is the reduction of lake level variation. The local hydrology of many coastal wetlands has also been altered by man-made structures such as dikes, ditches, and roadbeds. The natural fluctuation in water level is likely a major factor in the maintenance of plant diversity in coastal wetlands (Keddy and Reznicek 1986). The diking of coastal marshes in Lake Erie, with the purpose of preserving them, has often resulted in a decrease of diversity and increased abundance of invasive plant species (Sanzone and McElroy 1998). The dampening of water levels in the St. Lawrence River has also promoted monocultures of invasive plants (Hudon 1997). The disruption of natural water level fluctuations has also been implicated as causing a decline in condition in other types of wetlands (Wilcox et al. 1985, Wilcox and Meeker 1991, Wilcox 1995, Findlay and Bourdages 2000).

Coastal wetlands are also stressed by point and non-point pollution in the form of increased nutrient and sediment loading, as well as chemical contamination. Excess nutrient loading leads to wetland eutrophication, where aggressive and/or exotic invasive species are released from nutrient limitation and out compete other species for light, thereby decreasing diversity (Kadlec and Bevis 1990, Otte 2001, Gustafson and Wang

2002). Bedford et al. (1999) reported that species richness appears to decline at some threshold as nutrient availability increases in wetlands across North America. Man made chemicals may also directly or indirectly stress wetland plants. A chemical, such as an herbicide, that is toxic to plants is an example of a direct stress. An example of an indirect stress is the contamination of a wetland by oil, where the oil may not be toxic but disrupts gas exchange and photosynthesis by coating the surface of the plant (Otte 2001). Increased sediment loading introduces multiple stressors into wetlands. Sediment can physically disturb wetland plant communities by scouring or burying plants. In turn, litter decomposition dynamics may be altered and seeds may be buried too deep to germinate (Jurik et al. 1994, Wang et al. 1994, Vargo et al. 1998). Nutrients, particularly P, and other chemicals may be adsorbed to sediments and are therefore also deposited in wetlands as sediments are deposited (Johnston 1991).

Exotic invasive species are biological stressors in Great Lakes coastal wetlands (Maynard and Wilcox 1997). Exotic invasive species have the ability to out compete native species for resources and increase their abundance, at the expense of native species. Increase of exotic invasive species in wetlands is often linked to other stressors. Localized physical disturbances of wetland plant communities increase the likelihood of initial invasion (Galatowitsch et al. 1999, Rachich and Reader 1999). Man-made structures that disrupt hydrology in wetlands, such as roads and dikes, can act as corridors and introduction vectors of exotic invasive species into wetlands (Parendes and Jones 2000). Increased nutrient loading and dampening of natural water level fluctuations have been directly implicated as causing the dominance of exotic invasives in several wetlands (Wilcox et al. 1985, Kadlec and Bevis 1990, Hudon 1997). Thus, exotic invasive species can be an ecosystem response to stress, as well as a cause. The following species are considered as exotic invasive species in this study: *Lythrum salicaria* L., *Typha angustifolia* L., *Typha* x *glauca* Godr. (pro sp.), *Butomus umbellatus* L., *Frangula alnus* P. Mill., *Myriophyllum spicatum* L., *Potamogeton crispus* L., *Hydrocharis morsus-ranae* L., and two cryptic invasives (species that have native and aggressive introduced genotypes) *Phalaris arundinacea* L. and *Phragmites australis* (Cav.) Trin. ex Steud. (Detenbeck et al. 1999, Galatowitsch et al. 1999, Morrison and Molofsky 1999, Saltonstall 2002).

Localized anthropogenic disturbances, not associated with structures that alter wetland hydrology, can also affect coastal wetland plant communities. Mowing for hay in wetlands was once a common farming practice in the northern U.S. (Steinberg 2002). Prescribed burning and mowing are common management practices to enhance waterfowl habitats in wetlands (de Szalay and Resh 1997). The effects of these two disturbances differ. Both burning and mowing remove above ground vegetation, but burning also removes litter, which remobilizes nutrients stored in litter and creates gaps for seedling germination (de Szalay and Resh 1997).

Land-use within the Great Lakes basin ranges from nature preserves to intensive row-crop agriculture and urban/industrial centers (Minc and Albert 1998, Detenbeck et al. 1999). Measurements of the surrounding upland area of wetlands, such as land use, are not in of themselves stressors to wetland plant communities, though they might be the cause of specific stressors. For example, agriculture within a watershed can cause increased wetland sediment and nutrient loading, which in turn affects wetland plant communities. Often these landscape measurements are used to estimate stressors. Watershed land use has been found to be a significant predictor of total N and P in South Carolina streams (Tufford et al. 1998). Wetland vegetation diversity has been found to be negatively correlated with increased agriculture and urban land use in the adjacent upland area (Findlay and Houlahan 1997, Mensing et al. 1998, Smith and Haukos 2002). Diversity has also been negatively correlated with watershed road density (Findlay and Bourdages 2000).

METHODS

Vegetation Sampling

This research was part of an ongoing study to develop and test ecological indicators in Great Lakes coastal waters and adjacent uplands. Fifty-five coastal wetlands were selected as study sites (Fig. 1) in the Laurentian Mixed Forest Ecoprovince (Keys et al. 1995). Coastal wetlands were selected to span multiple stressor gradients to facilitate indicator development and testing (Danz et al. *in press*).

Prior to vegetation sampling, randomly located transects were mapped at each selected site using a GIS. First, the sampling domain of a site, defined as the emergent

vegetation area of the wetland, was delineated. This was done by using a combination of National Wetlands Inventory (NWI) and Wisconsin Wetlands Inventory (WWI) data, as well as aerial photography interpretation. A first approximation of the sampling domain was made by selecting EM, or emergent, NWI or WWI polygons. This area was then edited to correspond to the interpreted emergent wetland area from the aerial photography. A program called Sample (http://www.quantdec.com/sample) was then used to generate random potential transect location points within the sampling domain. At a randomly selected point, a potential transect was generated that followed the perceived water depth gradient and extended from the shrub dominated zone, or the upland boundary, of the wetland to open water. A transect was accepted and mapped if it was at least 40 m long, not within 100 m of another transect, and was accessible. The total transect length and target number of quadrats to sample within each site was determined in proportion to the size of the sampling domain (20 quadrats/60 ha, with a minimum of 10-15 and 20 m of transect length/quadrat). Following the mapping of an appropriate total transect length, transect endpoints were generated and a text file giving the location of the endpoints in decimal degrees was created which could then be uploaded into a GPS unit.

Field teams from the University of Minnesota Duluth and the University of Wisconsin Madison, consisting of two-three people each, sampled the vegetation of the selected wetland study sites from 2001 to 2003. Sampling took place during July and August to ensure that most of the vegetation could be identified and peak annual growth was observed. Before each field season, field teams trained together to ensure sampling consistency between teams. Once at a site, field teams located transect endpoints with a handheld GPS unit. Transects were sampled in the field according to the order that they were mapped. One meter square quadrats were established along transects by dividing a transect into 20 m segments and randomly locating a quadrat in each 20 m segment, using a random number table. Quadrats were sampled along a transect until the entire acceptable transect length, extending from the continuous shrub zone or upland boundary of the wetland to 1 m depth of standing water, had been sampled or the target number of quadrats for the site had been reached. Nonrandomly placed quadrats were also located

at the ‘wet end’ of transects because the wetland/deep-water interface often had distinctively different vegetation.

Within each quadrat all vascular plant species were identified to the lowest taxonomic division possible. Percent cover was estimated visually for each taxon according to the following cover classes: <1%, 1-<5%, 5-<25%, 25-<50%, 50-<75%, 75-100%. Summed individual taxa percent cover in a quadrat could exceed 100% because canopies of different species overlap one another. Large, identifiable non-vascular plants, such as *Chara vulgaris* L. and *Sphagnum* spp., if present, were also given cover estimations. Cover estimates were also recorded for the following features if present: total plant cover, open water, herbaceous litter, woody litter, driftwood, bare soil, standing water, and brown moss. If a plant species could not be identified in the field, it was collected, pressed, and identified in the lab. If an unknown species was suspected to be rare and/or protected, a digital photograph of the plant was taken in place of collection. Plant nomenclature conventions in this study follow the Interagency Taxonomic Information System (ITIS, http://www.itis.usda.gov), which is a national standardized plant nomenclature source. Prior to data analyses, cover classes were converted to the midpoint percent cover of each class.

Of the selected sites in this study 18 open-coast, 21 riverine, and 16 protected wetlands were sampled. Data from one of the open-coast wetlands (Hog Island, in the Duluth/Superior harbor) were excluded from data analysis due to being sampled too early in the field season.

Index Calculations

Because the study area spans portions of both Michigan and Wisconsin, *C*-values were created specifically for the study area that combine the *C*-values from both states (Table 1). This was done to reduce possible confounding factors due to differences in *C*-values of the same species from the two states. Study area specific *C*-values were generated by taking the mean of the two individual state *C*-values for an observed species.

The indices listed in Table 2 were calculated for every quadrat, using the study area *C*-values, and the average quadrat score was used as the site score. Weighted Coefficient of Conservatism indices were calculated from the general formula

$$wC = \sum_{j=1}^{S} p_j C_j \tag{3}$$

where *wC* is the weighted Coefficient of Conservatism index which is equal to the product of the proportional abundance (*p*; expressed as percent cover) and the *C*-value of the *j*th species, summed for all species (*S*). Weighted Floristic Quality indices were computed by multiplying weighted Coefficient of Conservatism indices by the square root of *S*. Plants that were observed but could not be identified to species were excluded from all of the various index calculations, because assigning *C*-values to higher taxonomic levels was thought to be inappropriate.

Sampling Area-Index Relationships

Additional field sampling was performed during the 2002 field season to determine sampling area-index relationships at multiple scales. Data were collected from nine nested quadrat plots, in a protected wetland in Allouez Bay (Superior, WI), to develop species-area curves and test the sensitivity of *FQI* to the species-area relationship. Three nested quadrat plots were sampled in each of the following community types: wet meadow, marsh, and fen. Potential plot location points were mapped randomly using a GIS and were located in the field using a handheld GPS. Potential plot location points were evaluated in numbered order on site and selected as a plot location until three plots were selected in each of the three community types. Within each plot, all plant species were recorded in nested quadrats of 0.25, 1, 2, 4, 8, and 16 m^2. In addition, field teams recorded additional plant species occurring within 1 m of each side of sampling transects at 12 study sites (Fig. 1). These data were collected to compare index scores computed from quadrat data (smaller sampling area) and transect data (larger sampling area).

S, $\overline{C}$, and *FQI*, were calculated for each of the nested quadrats and averaged within community type over each quadrat area. Wisconsin *C*-values were used for these

calculations. Simple linear regression was then used to determine sampling area-index relationships from the nested plot data.

S, $\overline{C}$, and *FQI* were also calculated from the data collected in the 12 sampling area comparison sites by the cumulative site species lists generated from both quadrat and transect data. Because all 12 sites are within the state of Michigan (Fig. 1), Michigan *C*-values were used to compute these index scores.

Michigan and Wisconsin C-*value Comparison*

In addition to computing $\overline{C}$ and *FQI* by the study area specific *C*-values, these indices were computed using both Michigan and Wisconsin *C*-values independently for each site regardless of which state the site actually occurred in. This was done to examine the effect of the differences between the two sets of *C*-values on index scores.

FQI-*Diversity Relationships*

Margalef diversity, Shannon diversity, Shannon evenness, and Simpson dominance indices were computed to explore possible *FQI*-diversity relationships (Table 3). All of these diversity indices combine both species richness and the abundance distribution, or evenness, of the community (Magurran 1988). The Margalef diversity index is sensitive to changes in species richness and is often referred to as a species richness index. The Shannon diversity index originates from information theory and is a measure of the entropy of the distribution of biomass, or in this case cover, amongst species. Dividing Shannon diversity by the possible maximum entropy, ln *S* or when all of the species have equal relative abundance, of the sampled community produces the Shannon evenness index. The Shannon evenness index increases as the abundance distribution of the sampled community becomes more equally distributed. The Simpson dominance index gives the probability that any two units of abundance, drawn at random from the sampled community, are the same species. This index is most influenced by abundant species and is therefore referred to as a measure of dominance.

Diversity indices were calculated using the cumulative site values of *S*, *N*, and p_i from quadrat data for a site. Correlations between the diversity indices and the number of plots sampled in a site were performed to determine if sampling area alone had an effect

on diversity index scores. The Margalef diversity index was found to be positively correlated with the number of plots at a site ($\rho = 0.42$, $P < 0.01$) and was recalculated on a per quadrat basis and averaged over a site. Shannon diversity and evenness, and Simpson dominance indices were uncorrelated with the number of plots. Correlations between the diversity indices and S, $\overline{C}$, and FQI were performed in each wetland type to explore possible FQI-diversity relationships.

Relative Discriminant Ability

To be useful an index must be capable of detecting differences between sites or within a site over time (Magurran 1988). F-ratios can be used to assess the between site discriminant ability of a particular index. In this case, the F-ratio is a measure of the differences in index scores between sites divided by within-site index score differences, or the between-site variance divided by the within-site variance (Kempton 1979, Magurran 1988). An index that maximizes F compared to other indices for a given dataset is thus the best index at discriminating differences between sites. This approach to assessing discriminant ability is independent of a stressor or condition gradient. Thus, if a particular index consistently outperforms other indices in multiple trials, it may be reasonable to conclude that that index has inherently greater discriminant ability (Magurran 1988).

F-ratios were computed for all ten species richness, Coefficient of Conservatism, and Floristic Quality indices (Table 2) in each of the three wetland types using ANOVA. Because F-ratios can be computed only once per index per wetland type, a data resampling (i.e., bootstrapping) procedure was performed to allow comparisons between indices. The bootstrapping procedure randomly sampled, with replacement, all of the plots within each of the sites 500 times. F-ratios were computed for each of the 500 iterations to produce a bootstrap mean F-ratio and a 95% confidence interval for each index. Prior to bootstrapping, F-ratios were computed for all indices, the residuals were plotted, and Box-Cox transformations were performed to check statistical assumptions of normality and constant variance. Statistical assumptions were met for all of the indices, except S and S_i, in all three wetland types. For these two indices, square root transformations were found to be adequate.

Indicator Testing

Testing of ecological indicators presents unique challenges to ecologists. The most important property of a proposed indicator is that it discriminates sites along a known condition gradient (Jackson et al. 2000, Fore 2003). The human concept of condition is thus introduced into scientific arguments. To reduce subjectivity, indicators are often tested against estimates of anthropogenic stress, which are assumed to act as surrogates of condition (Kantrud and Newton 1996, Jackson et al. 2000, Lopez and Fennessy 2002, Wilcox et al. 2002, Fore 2003). This is appropriate because anthropogenic stressors are the causal agents of ecosystem degradation (Rapport et al. 1985).

This study uses estimates of anthropogenic stressors as a surrogate of condition to test the various species richness, Coefficient of Conservatism, and Florist Quality indices as indicators of condition. The primary anthropogenic stressors to coastal wetlands include: alteration of lake levels and local hydrologic disturbance; excess nutrient, sediment, and chemical loading; exotic invasive species; and localized disturbance. Because this is an observational study and because multiple stressors may act simultaneously in addition to the variation caused by natural abiotic factors (Keough et al. 1999), cause-effect relationships may only be suggested. An indicator testing strategy that used an *a priori* set of stressor measurements to act as a surrogate condition gradient, as opposed to a model building strategy, was therefore decided to be appropriate.

Three measurements were chosen as predictors, to act as an overall condition gradient, in multiple regressions to test indices as indicators. The goal was to choose measurements that summarized as many of the primary stressors as possible. The first two predictors, Agriculture pc1 and Population Density pc1, were principal components that summarized multiple landscape scale variables and were measured at a scale called the shoreline segment-shed (Danz et al. *in press*). Shoreline segment-sheds are the watersheds that drain a length of shoreline with endpoints at the half-way distance between second order or higher streams. Agriculture pc1 has been interpreted as the overall amount of agriculture, and Population Density pc1 has been interpreted as the overall population density, in a segment-shed. Both of these measurements can be thought of as being integrated stressor measurements, because they likely summarize

multiple stressors. An estimate of local hydrologic modification was also developed and used as the third predictor. The length (m) of features that likely disrupt the natural flow and fluctuation of water (e.g., road beds, dikes, ditches) in wetlands was measured in all sampled sites using a GIS. The summed length of these features was then divided by the sampling domain area (ha) of the wetland to produce an index of local hydrologic modification that was unbiased in terms of wetland area.

A key feature of any ecological indicator program is setting specific, *a priori*, performance criteria (Jackson et al. 2000). The performance criteria adopted in this study proceed as follows: an index is an acceptable indicator if 1) there is a significant negative relationship between the index and the condition gradient, and 2) the relationship is strong enough that the index can detect a twenty percent difference in the condition gradient. Power, the probability of rejecting the null hypothesis when the null hypothesis is indeed false, was used to examine if an index met the second criterion. Power is traditionally used to determine the number of samples needed to test hypotheses or add evidence to correctly reject null hypotheses. In this study, power was used as a benchmark to determine if a particular index was able to detect changes, when reasonable model simulations were performed, along the condition gradient.

Multiple linear regressions of all ten species richness, Coefficient of Conservatism, and Floristic Quality indices (Table 2) were performed against the three predictors in each wetland type. Residual plots were examined and Box-Cox transformations were performed to check statistical assumptions of normality, constant variance, and linearity. All of the regressions adequately met statistical assumptions. Regressions were also performed using normal standardized predictor values to facilitate comparisons between the regression coefficients of the three predictors. The non-standardized regression models were used to predict ‘reference’ and percent predictor change index values. Reference values were computed by first finding the tenth percentile of the distribution of each of the predictors and then plugging those values into regression models. Predictor change values were computed by adding 1-50%, in one percent increments, of the remaining predictor distributions to tenth percentile predictor values and plugging those values into regression models. Power was then computed by performing a hypothetical *t*-test between simulated ‘sample means’ of reference predicted

values and predictor change predicted values, where each of the two 'sample means' were model predicted values, regression mean squared errors served as 'sample variances', and an arbitrary number (20) was chosen for *n*. An index met the second indicator performance criterion if it had a 0.80 level of power by a twenty percent increase in the predictors, when all three predictors were changed simultaneously. Power was also calculated from differences of predicted values when only one of the predictors was increased twenty percent, while holding the other two predictors constant.

RESULTS

Sampling Area-Index Relationships

Species richness increased with sampling area according to the ln-ln transformation of equation (2) in all three (marsh, wet meadow, and fen) community types (Fig. 2A). $\overline{C}$ remained constant with an increase in sampling area in marsh and wet meadow communities, but decreased with sampling area in the fen community type (Fig. 2B). *FQI* however, also increased with sampling area according to the ln-ln transformation of equation (2) in all three community types (Fig. 2C), though the increase was somewhat dampened compared to the increase in species richness, because the square root of species richness is an index parameter.

The increase in species richness per incremental increase in sampling area (*z*) was greatest in the wet meadow community (0.24), but was similar in both marsh and fen communities, 0.16 and 0.18 respectively. Unit area species richness (*c*), however, was greatest in the fen community (11.36) and was similar in the marsh and wet meadow communities (9.12 and 9.21). In general, species-area curves from the marsh community had low *c* and *z*, those from the wet meadow community had low *c* and high *z*, and those from the fen community had high *c* and low *z*. These patterns were also somewhat reflected in the *c* and *z* values derived from *FQI*-area relationships, where the wet meadow had the greatest *z* (0.10) compared to the marsh and fen communities (0.08 and 0.06), and *c* was greatest in the fen community (22.87) compared to the marsh and wet meadow communities (17.29 and 13.87).

There was a similar pattern of *S*, $\overline{C}$, and *FQI* behavior at a larger scale. *S* and *FQI* were both greater when computed from data collected along transects (larger

sampling area) compared to index scores from quadrat data (Fig. 3A, 3C) in twelve sites in Michigan (Fig. 1). $\overline{C}$ scores, on the other hand, were very similar when computed from both data sets (Fig. 3B).

Michigan and Wisconsin C-*value comparison*

On average, *C*-values, assigned to the 403 species observed in this study, were greater in Wisconsin than in Michigan (Table 1). Michigan *C*-values, however, had greater variance. There were 220 *C*-value differences between the two states, with the largest difference being 6, for *Symphyotrichum pilosum* var. *pilosum* (Willd.) Nesom. There were nine other species with a *C*-value difference greater than or equal to 4 (*Carex leptalea* Wahlenb., *Carex lurida* Wahlenb., *Elymus trachycaulus* (Link) Gould ex Shinners, *Myriophyllum sibiricum* Komarov, *Salix bebbiana* Sarg., *Salix petiolaris* Sm., *Solidago uliginosa* Nutt., *Thelypteris palustris* Schott, and *Thuja occidentalis* L.).

There were also differences between the two states in classifying species as introduced or native. Seven species were observed which are considered introduced in Wisconsin but native in Michigan (*Beckmannia syzigachne* (Steud.) Fern., *Galium palustre* L., *Hibiscus moscheutos* L., *Juncus articulatus* L., *Peltandra virginica* (L.) Schott, *Phalaris arundinacea* L., and *Polygonum hydropiper* L.). There is also one species that is considered native in Wisconsin but introduced in Michigan (*Xanthium strumarium* L.). The USDA PLANTS Database (http://plants.usda.gov) was used to make a final determination of native status for these species in this study.

To examine the effect of the differences between the two sets of *C*-values, $\overline{C}$ and *FQI* were computed using both Michigan and Wisconsin *C*-values at all sites regardless of their actual location. Of the 54 sites, 51 sites had greater $\overline{C}$ and 53 sites had greater *FQI* when calculated with Wisconsin *C*-values compared to Michigan *C*-values. The greatest differences of $\overline{C}$ and *FQI* for an individual site were 1.21 and 3.75 respectively.

FQI-*Diversity Relationships*

FQI was positively correlated with *S* and D_{Mg} in all three (open-coast, riverine, and protected) wetland types (Table 4). $\overline{C}$ was also positively correlated with *S* in all

three wetland types but was correlated with D_{Mg} only in open-coast wetlands. *FQI* and $\overline{C}$ were not correlated with *H'*, *relH'*, or 1-D_S in either open-coast or protected wetlands. However, in riverine wetlands *FQI* was positively correlated with all three, and $\overline{C}$ was correlated with *relH'* and 1-D_S. *S* was positively correlated with D_{Mg} in all three wetland types but was not correlated with any other index, except for *H'* in riverine wetlands, in any of the wetland types.

Relative Discriminant Ability

Bootstrap mean *F*-ratios for the Coefficient of Conservatism indices were significantly greater than the two measures of species richness in both open-coast and protected wetlands (Fig. 4). In riverine wetlands however, only the bootstrap mean *F*-ratios for the Coefficient of Conservatism indices that include introduced species were significantly greater than S_i. All of the bootstrap mean *F*-ratios for the Floristic Quality indices were significantly greater than species richness indices in all three wetland types, except *FQI* in riverine wetlands. There were no significant differences between Coefficient of Conservatism and Floristic Quality indices in open-coast and riverine wetlands. In protected wetlands however, Coefficient of Conservatism indices had significantly greater bootstrap mean *F*-ratios than Floristic Quality indices. There were no significant differences between indices that included or excluded introduced species in any of the three wetland types. There were also no significant differences between indices that were or were not weighted by abundance in any of the three wetland types.

Indicator Testing

Index-condition multiple regression results are listed in Table 5. All of the species richness, Coefficient of Conservatism, and Floristic Quality indices declined significantly ($P < 0.05$) with declines in condition in open-coast and protected wetlands. Only $\overline{C}_i$, wC_i, *FQI*, FQI_i, and $wFQI_i$-condition relationships were significant in riverine wetlands.

As a group, Floristic Quality indices had stronger condition relationships than Coefficient of Conservatism indices in open-coast wetlands (Table 5). In turn, Coefficient of Conservatism indices had stronger condition relationships than species

richness indices in this wetland type. This general pattern was also found in the riverine and protected wetland types, but there were several exceptions. $\overline{C}_i$ and wC_i both had slightly stronger condition relationships than their Floristic Quality index counterparts, FQI_i and $wFQI_i$, in riverine wetlands. Also, wC and wC_i -condition relationships were weaker than either S or S_i in protected wetlands.

The power probabilities showed the same general pattern of index performance (Table 5, Fig. 5). In each of the three wetland types, the power to detect a significant difference between predicted reference index values and predicted index values when all three predictors increased was greatest for Floristic Quality indices, followed by Coefficient of Conservatism, and then species richness indices. In open-coast and protected wetlands, all four Floristic Quality indices had greater than 0.99 power by a 20% change in the predictors. All four Coefficient of Conservatism indices also met the 0.80 power criterion in both open-coast and protected wetlands. Neither of the two species richness indices met the power criterion in any of the three wetland types. Riverine wetlands, where index-condition relationships were not as strong, had a slightly different pattern. $\overline{C}_i$ and wC_i had greater power to detect a change in condition than their counterpart Floristic Quality indices. Only one index, $\overline{C}_i$, met the power criterion in this wetland type. None of the indices, in any of the three wetland types, reached the power criterion under model simulations in which only one of the predictors was increased and the other two held constant.

There were consistent predictor contribution patterns within wetland types but few patterns between types (Table 6). Standardized regression coefficients for Agriculture pc1 and Population Density pc1 were very similar between indices in open-coast wetlands. Agriculture pc1 coefficients were consistently less than (i.e., contribute more to an overall negative slope) Population Density pc1 coefficients in riverine wetlands, and consistently greater than in protected wetlands. Hydrologic Modification coefficients were consistently greater than (i.e., contribute less to an overall negative slope) either of the other predictors in each of the three wetland types.

DISCUSSION

Sampling Area-Index Relationships

The nested plot and the quadrat vs. transect data comparison results (Fig. 2, Fig. 3) clearly show that *FQI* is sensitive is to sampling area due to the species richness parameter in the index. Interpretation of *FQI* therefore depends on knowledge of the underlying species-area relationships. This agrees with results from other studies where *FQI* increased with sampling area (Francis et al. 2000, Rooney and Rogers 2002, Matthews 2003).

Procedures that take sampling area into account need to be used to be able to make unbiased comparisons with *FQI*. This can either be done by standardizing the site sampling area, or by sampling sites with multiple randomly placed plots and then using the mean plot index scores as the site index value, as was done in this study. The first sampling procedure has the disadvantage of not being flexible with respect to wetland size, and therefore may not accurately characterize plant communities. The second procedure can be flexible with respect to wetland size (i.e., larger wetlands have more plots). Also, this method has the advantage of a site being characterized by a distribution of index scores, as opposed to one value. Mean plot index scores are often normally distributed, which allows comparisons of different sites using parametric statistics (Magurran 1988). This property is particularly useful for *FQI*, which is generally computed by cumulative site species data to produce one value.

The nested plot results also suggest that quadrat sizes larger than 1 m^2 are needed to characterize wetland plant communities in terms of species richness. However, one single plot size may not be sufficient to characterize all three community types because the species area-curves of the different communities cross. The species-area curve in the marsh community flattens out by 16 m^2, but the wet meadow and fen species-area curves appear to be increasing past the maximum quadrat size (Fig. 2A). More specific research is needed to determine appropriate plot sizes for Great Lakes coastal wetlands.

Michigan and Wisconsin C-*value comparison*

Comparisons of index scores and overall statistics of Michigan and Wisconsin *C*-values indicate that the assignment of *C*-values by state may be somewhat flawed. The

behavior of plant species does vary geographically. *C*-values should therefore reflect that variation. However, in terms of plant communities, it does not seem likely that Michigan and Wisconsin are very different. Each state has, or once had, large areas of northern deciduous and coniferous forest, as well as limited tall-grass prairie communities. The difference of the overall mean *C*-values between the two states may not be great (0.59), but it does appear that $\overline{C}$ scores calculated using Wisconsin *C*-values are consistently greater than $\overline{C}$ scores calculated using Michigan *C*-values. These results highlight the subjective nature of *C*-value assignment and raise the question: Are plants in Wisconsin naturally more 'conservative' or is this an artifact of *C*-value assignment methodology? Assigning *C*-values by an ecologically meaningful unit, such as Ecological Province or Ecological Section (Keys et al. 1995), may improve accuracy and reduce subjective effects, because these units are based on broad climate and vegetation patterns. Thus, it is reasonable to assume that plant species will behave consistently within these types of units making *C*-value assignment more accurate and consistent. However, because natural resource and heritage programs are usually organized at the state level, assigning *C*-values in this manner would require a degree of coordination between states, which may be very difficult.

FQI-*Diversity Relationships*

Correlations between *FQI* parameters and diversity indices were done to explore possible relationships between these parameters and the two aspects of diversity: species richness and evenness. Significant positive *FQI-S* and *FQI-*D_{Mg} correlations were expected, because *FQI* is essentially a weighted species richness index. Significant correlations between $\overline{C}$ and *S*, in all three wetland types, and significant $\overline{C}$*-relH'* and $\overline{C}$-1-D_S correlations in riverine wetlands, however, were not expected. These results suggest two possible relationships. Positive correlations between $\overline{C}$ and *S* indicate that at more species-rich sites there is a greater proportion of species with higher *C*-values. Positive correlations between $\overline{C}$ and *relH'* and 1-D_S indicate that at sites that are dominated by a few species, those species have low *C*-values. The first relationship appears to be inconsistent with results from the nested quadrat sampling. *S* increases

with sampling area, but $\overline{C}$ remains constant, or decreases (Fig. 2). This may lead one to predict that *S* and $\overline{C}$ would be uncorrelated. However, the two correlations are different. The first is a within site correlation, and the second is a between site correlation.

The significant correlations between $\overline{C}$ and *S* may be explained by the theory behind the *C*-value. In theory, *C* reflects the fidelity of a species to natural habitats. Species that have high *C*-values are therefore thought of as being intolerant to anthropogenic stress. Rare species are often assigned high *C*-values. Intolerant species are likely the first to be lost in stressed communities, and rare species are usually associated with species rich communities (Bedford et al. 1999). Thus, if *C* is truly a measure of a species' intolerance to anthropogenic stress, we would expect $\overline{C}$ and *S* to be correlated in sites that were selected across multiple anthropogenic stress gradients, such as in this study. The significant correlations between $\overline{C}$ and *relH'* and 1-D_S in riverine wetlands may also be explained by this. The most stressed sites are dominated by a few taxa, which are tolerant to stress and therefore have low *C*-values. These results may be an indirect validation of *C*.

The strengths of the correlations between $\overline{C}$ and *relH'* and 1-D_S differed by wetland type (Table 4). Correlations were significantly positive in riverine wetlands, positive but not significant in open-coast wetlands, and near zero in protected wetlands. A possible explanation for this result is the degree to which the abundance distribution of each wetland type relates to exotic invasive species. While invasive species do not directly affect $\overline{C}$ scores, because introduced species were excluded from calculations, they can indirectly affect scores by competitively reducing the abundance of native species, particularly rare species that often have high *C*-values. Correlations between the proportional abundance of exotic invasive species and 1-D_S in open-coast (ρ = -0.50) and riverine wetlands (ρ = -0.66) were both significant ($P < 0.05$). These correlations suggest that in sites that are more dominated by a few species, those species are exotic invasives, which indirectly depress $\overline{C}$ scores, leading to positive correlations between $\overline{C}$ and 1-D_S. The strength of the invasive species-1-D_S correlations in these two wetland types may explain why the correlation between $\overline{C}$ and 1-D_S was significant in riverine wetlands and not significant in open-coast wetlands. Riverine wetlands may be subjected to more

pressure from invasive species, because streams act as corridors for introduced plant species by transporting propagules and streams can also create local disturbances that allow exotic species to colonize a wetland (Parendes and Jones 2000). The proportional abundance of exotic invasive species was uncorrelated with 1-D_S in protected wetlands (ρ = -0.11), which may explain the $\overline{C}$-1-D_S correlation in this wetland type. However, a more likely explanation for the $\overline{C}$-1-D_S correlation result is that some of the protected wetlands were peatlands, and that most species that inhabit bogs and fens have high *C*-values. Peatlands may be dominated by a few vascular species, but those species have high *C*-values.

Relative Discriminant Ability

The ability of an index to discriminate between sites that are not inherently different (i.e., sites of the same ecosystem or community type) is an important property (Kempton 1979, Magurran 1988). Indices that have low discriminant ability will not be particularly useful in applied situations. The goal of the relative discriminant ability testing was to find performance consistencies, independent of stressor or condition gradients, across wetland types. In general, Coefficient of Conservatism and Floristic Quality indices are better at discriminating sites than species richness indices (Fig. 4). In open-coast and riverine wetlands, Coefficient of Conservatism and Floristic Quality indices perform at the same level. In protected wetlands however, Coefficient of Conservatism indices significantly outperform Floristic Quality indices. This is again likely due to some of the protected wetland sites being peatlands. As previously stated, *C*-values of species that inhabit fen and bog communities are often greater than species that typically inhabit wet meadow and marsh communities. This leads to greater differences in Coefficient of Conservatism index scores between sites because of the two different community types alone. Rooney and Rogers (2002) have found that different plant community types have different $\overline{C}$ in Wisconsin. This appears to be producing the high *F*-ratios of Coefficient of Conservatism indices in protected wetlands and therefore may not be indicative of the performance of these indices.

Indicator Testing

Indicator testing results produced a somewhat different pattern of index performance. In general, Floristic Quality indices performed better than Coefficient of Conservatism indices, which, in turn, performed better than species richness indices (Table 5, Fig. 5). A possible explanation for the performance of Floristic Quality indices can be made by looking at the performance of the individual index parameters. Both species richness and Coefficient of Conservatism indices decrease along the condition gradient. Floristic Quality indices are the product of the two, thereby strengthening the response. Also, $\overline{C}$ was found to be positively correlated with *S* (Table 4). This indicates a further inflation of Coefficient of Conservatism indices at least-stressed sites, which could lead to a greater response in these indices and therefore contribute to the greater response of Floristic Quality indices.

Index-condition relationships of each of the indices were weaker in riverine wetlands than in the other two wetland types (Table 5). None of the indices passed both of the performance criteria, though there were some significant relationships. The methods used to gather data for both Agriculture pc1 and Human Population pc1 may explain this inconsistency. Landscape scale data were used to compute these predictors. Data were collected, and measurements were computed, by segment-shed area. Segment-shed areas can be quite large, because the segment-shed encompasses the watershed of a second order or higher stream. This may particularly be the case for riverine wetlands on higher order streams. For example, the segment-shed area for a coastal wetland sampled at the mouth of the Peshtigo River in Wisconsin, which drains into Lake Michigan, is 303,029 ha. This scale may be too coarse to accurately estimate local stressors, which may have the largest effects on wetland condition (Mensing et al. 1998, Tufford et al. 1998). A two-tailed *t*-test indicated that there was no significant difference in the log segment-shed area (ha) between open-coast and protected wetlands ($P = 0.56$). One-tailed *t*-tests however, indicated that the log segment-shed area of riverine wetlands was greater than both open-coast and protected wetlands ($P < 0.05$). Local stressors may not be characterized as well in riverine wetlands as the other two types, because of the larger average segment-shed area, which could have led to the weaker index-condition relationships. It is important to note that all of the study sites, regardless of wetland type,

were characterized locally only in terms of hydrologic modification, not by the other two predictors. Local stressor measurements may more accurately portray wetland condition and therefore be a 'truer' test of ecological indicators.

The goal of the indicator testing portion of this study was to test the various indices as indicators of ecological condition, using stressor measurements as a surrogate condition gradient, and compare performance. It is not a goal of this study to turn around the research question and use these indices to test which of the various stressors effect condition. While this is an important research question, there is a subtle difference between the two and therefore they must be addressed independently. Thus the inconsistent predictor contribution results (Table 6) between wetland types does not necessarily mean that particular stressors are either more or less important in a general sense, or in specific wetland types. The three predictors were specifically chosen in advance to summarize multiple stressors that would likely affect wetland condition based on knowledge gleaned from previous research. For example, Agriculture pc1 was interpreted as the amount of agriculture in a segment-shed. While the amount of agriculture surrounding a wetland is not a stressor in of itself, this measurement likely characterizes excess nutrient, chemical, and sediment loading due to agriculture. In this case, it is not possible to tease out the effect of individual stressors, because principal component analysis has summarized individual measurements. To test for the effects of individual stressors to communities or ecosystems, a model building approach should be used to add individual stressors to models based on statistical criteria. To test ecological indicators of condition, the exact opposite approach should be used, where predictors are selected based on *a priori* knowledge, as was done here. It is important to note that the two principal components used as predictors were computed to facilitate site selection for the larger indicator study and were not designed to be used for indicator testing. While these two pc's summarize multiple stressors indirectly, they do not summarize all of the primary stressors that can affect the extent and composition of coastal wetlands. For example, local anthropogenic disturbances, such as mowing, can alter plant community composition (de Szalay and Resh 1997). Specific site disturbances, either historical or current, were not accounted for in this study. Further study is therefore needed to test for

the effect of individual stressors, which includes a more complete account of primary stressors.

Previous work on the development of ecological indicators has highlighted that to be useful as indicators of condition, measurements should posses the following attributes: 1) be conceptually relevant to assessment questions and to ecological function; 2) implementation is straightforward and cost effective; 3) have the ability to discriminate sites along a known condition gradient; 4) show low, and/or understandable variability; and 5) have clear linkages to management action (NRC 2000, Jackson et al. 2000, Dale and Beyeler 2001, Fore 2003). Results from this study show that both Coefficient of Conservatism and Floristic Quality indices, in Great Lakes coastal wetlands, have most of these attributes. *C*-values are conceptually simple to understand and have apparent ecological meaning, therefore predications of index behavior are easy to make. The implementation of these indices requires plant identification skills, as well as a background in plant ecology and data analysis. It is likely that most natural resource agencies have existing personnel with these skills; therefore there is little need for agencies to hire personnel with unique skills and abilities to implement these indices. Sampling requirements are also likely within the normal range of what regulating agencies already perform, thereby making implementation likely cost effective. The ability of an indicator to discriminate sites along a known condition gradient is the most important property of an indicator. Both of these types of indices perform well in that capacity. The ability of an indicator to detect changes in condition within the context of natural variability is also an extremely important property; however specific variability components were not analyzed in this study. Jackson et al. (2000) has identified the following error components in indicator-condition relationships: measurement error, within year variability, between year variability, and spatial variability. In this study, within year variability was controlled for by sampling only in July and August. An attempt to control measurement error, between field teams and across years, was made by annually training together before sampling. Spatial variability was somewhat controlled for by classifying wetlands by geomorphic type and by bounding the study area within the Laurentian Mixed Forest Ecological Province. However, the effect of some of the protected wetlands being peatlands on index scores was not anticipated, and future

studies should stratify coastal wetlands by this factor. Between year variability was not addressed in this study. The effect of these variance components on the performance of these indices remains a subject for future research. Linking indicators to management action requires specification of management goals, making it difficult to assess how well these indices would address this indicator attribute. Results presented in this paper show that these indices would likely perform well indicating levels of condition, if different levels of condition were defined. Users could then apply these indices to management goals, as is found to be appropriate.

Conclusions

As the use of both Coefficient of Conservatism and Floristic Quality indices increases, users should apply these indices consistently to allow comparisons between studies and over time. This then begs the question: What is the best index to use? Conceptual criteria contribute to answering this question, as well as performance criteria.

Performance results of indices that include or exclude introduced species (Table 5, Fig. 4) are indistinguishable. Conceptually however, it seems appropriate to include introduced species, because introduced species are simultaneously a source of and a response to anthropogenic stress. Index scores should be directly affected by introduced species and thus included in index computation.

Performance results between indices that are weighted or not weighted by abundance are also indistinguishable (Table 5, Fig. 4). In this case it appears that weighting indices by abundance does not increase performance and should not be done, because of the increased data and computational requirements needed to compute these types of indices. However, there could be applications where weighted indices outperform non-weighted indices. Poling et al. (2003) has identified a situation where abundance weighting improved index performance in a tallgrass prairie restoration in Ohio. Successional shifts from colonizing grasses (lower *C*-values) to later successional grasses (higher *C*-values) could be detected only by using weighted indices, because the abundance distribution of the community was changing over time but species composition was not. Results presented here and results presented by Poling et al. (2003) suggest that weighting indices by abundance does not increase performance to detect

differences between sites, but does increase performance tracking changes within a particular site over time. More research is needed to clarify the utility of weighting these indices by abundance.

The decision to use a Coefficient of Conservatism or a Floristic Quality index appears to also depend on the application, as well as the sampling design. Results presented in this study (Fig. 2, Fig. 3) clearly show that Floristic Quality indices are affected by sampling area due to the species richness parameter. Another criticism of Floristic Quality indices is that they can give misleading results because they are the product of two parameters. Rooney and Rogers (2002) found that *FQI* of an abandoned forest road, inhabited by species with low *C*-values, was actually higher than the surrounding forest, simply because the road had almost twice as many species. The justification of including species richness in Floristic Quality indices is to increase discriminant ability (Taft et al. 1997). The relative discriminant ability, independent of a condition gradient, though, of both types of indices is more or less the same (Fig. 4). All of these lines of evidence point to discarding Floristic Quality indices and only using Coefficient of Conservatism indices, which are also conceptually easier to understand. However, Floristic Quality indices consistently outperform Coefficient of Conservatism indices as indicators of ecological condition (Table 5, Fig. 5) and thus should be used in that capacity. Therefore, users need to take into account the application the index is being asked to perform and how the data were collected, and then decide which index is more appropriate.

Before either of these types of indices can be fully implemented as assessment tools in Great Lakes coastal wetlands, further research is needed to calibrate index scores. This could be in the form of defining different levels of condition such as ‘good’, ‘fair’, and ‘poor’ or an impairment threshold which would signal managers to perform some action. Calibrating index scores to condition levels or thresholds is beyond the scope of this study. One factor that may hinder index score calibration, or the application of any indicator in Great Lakes coastal wetlands, is the effect that lake levels have on the extent and composition of coastal wetlands (Wilcox et al. 2002). Index scores could change due to this natural variation, independent of the effect of anthropogenic stressors, and perhaps

lead users to incorrect conclusions concerning wetland condition. Studies that measure the effect of lake levels over time may be necessary to properly calibrate index scores.

Regardless of the subjectivity involved with the assignment of *C*-values and that 'floristic quality' is a human concept and not a true ecosystem property, both Coefficient of Conservatism and Floristic Quality indices appear to be effective indicators of condition in Great Lakes coastal wetlands, and therefore likely have bright futures as ecological assessment tools.

LITERATURE CITED

Andreas, B. K. and R. W. Lichvar. 1995. Floristic index for establishing assessment standards: a case study for northern Ohio. U.S. Army Corps of Engineers, Waterways Experiment Station, Vicksburg, MS, USA. Technical Report WRP-DE-8.

Arrhenius, O. 1921. Species and area. Journal of Ecology 9:95-99.

Bedford, B. L., M. R. Walbridge, and A. Aldous. 1999. Patterns in nutrient availability and plant diversity of temperate North American wetlands. Ecology 80:2151-2169.

Bernthal, T., T. Cochrane, G. Fewless, R. Freckmann, R. Henderson, E. Judziewicz, and L. Leitner. 2001. Wisconsin floristic quality assessment (WFQA). Wisconsin State Herbarium Home page, http://wiscinfo.doit.wisc.edu/herbarium/WFQA.html, Madison, WI, USA.

Chadde, S. W. 1998. A Great Lakes Wetland Flora. Pocketflora Press, Calumet, MI, USA.

Dale, V. H. and S. C. Beyeler. 2001. Challenges in the development and use of ecological indicators. Ecological Indicators 1:3-10.

Danz, N. P., R. R. Regal, G. J. Niemi, V. Brady, T. Hollenhorst, L. B. Johnson, G. E. Host, J. M. Hanowski, C. Johnston, T. Brown, J. Kingston, and J. R. Kelly. In Review. Environmentally stratified sampling design for the development of Great Lakes ecological indicators.

de Szalay, F. C. and V. H. Resh. 1997. Responses of wetland invertebrates and plants important in waterfowl diets to burning and mowing of emergent vegetation. Wetlands 17:149-156.

Detenbeck, N.E., S.M. Galatowitsch, J. Atkinson, and H. Ball. 1999. Evaluating perturbations and developing restoration strategies for inland wetlands in the Great Lakes basin. Wetlands 19:789-820.

Fennessy, S., M. Gernes, J. Mack, and D. H. Wardrop. 2001. Methods for evaluating wetland condition: using vegetation to assess environmental conditions in wetlands. U.S. Environmental Protection Agency, Office of Water; Washington, DC. EPA 822-R-01-007j.

Findlay, C. S. and J. Bourdages. 2000. Response time of wetland biodiversity to road construction of adjacent lands. Conservation Biology 14:86-94.

Findlay, C. S. and J. Houlahan. 1997. Anthropogenic correlates of species richness in southeastern Ontario wetlands. Conservation Biology 11:1000-1009.

Fore, L. S. 2003. Developing biological indicators: lessons learned from mid-Atlantic streams. U.S. Environmental Protection Agency, Office of Environmental Information and Mid-Atlantic Integrated Assessment Program, Region 3, Ft. Meade, MD, USA. EPA 903/R-003/003.

Francis, C. M., M. J. W. Austen, J. M. Bowles, and W. B. Draper. 2000. Assessing floristic quality in southern Ontario woodlands. Natural Areas Journal 20:66-77.

Galatowitsch, S. M., N. O. Anderson, and P. D. Ascher. 1999. Invasiveness in wetland plants in temperate North America. Wetlands 19:733-755.

Gustafson, S. and D. Wang. 2002. Effects of agricultural runoff on vegetation composition of a priority conservation wetland, Vermont, USA. Journal of Environmental Quality 31:350-357.

Herdendorf, C. E., S. M. Hartley, and M. D. Barnes (eds.). 1981. Fish and wildlife resources of the Great Lakes coastal wetlands within the United States, Vol. 1: overview. U.S. Fish and Wildlife Service, Washington, DC, USA. FQS/OBS-81/02-v1.

Herman, K. D., L. A. Masters, M. P. Penskar, A. A. Reznicek, G. S. Wilhelm, W. W. Brodovich, and K. P. Gardiner. 2001. Floristic quality assessment with wetland categories and examples of computer applications for the state of Michigan, second edition. Michigan Department of Natural Resources, Wildlife Division, Natural Heritage Program, In partnership with U.S. Department of Agriculture Natural Resources Conservation Service, Rose Lake Plant Materials Center, MI, USA.

Hudon, C. 1997. Impact of water level fluctuations on St. Lawrence River aquatic vegetation. Canadian Journal of Fisheries and Aquatic Sciences 54:2853-2865.

Jackson, L. E., J. C. Kurtz, and W. S. Fisher (eds.) 2000. Evaluation guidelines for ecological indicators. U.S. Environmental Protection Agency, Office of Research and Development, Research Triangle Park, NC, USA. EPA/620/R-99/005.

Johnston, C. A. 1991. Sediment and nutrient retention by freshwater wetlands: effects on surface water quality. Critical Reviews in Environmental Control 21:491-565.

Johnston, C. A., S. D. Bridgham, and J. P. Schubauer-Berigan. 2001. Nutrient dynamics in relation to geomorphology of riverine wetlands. Soil Science Society of America Journal 65:557-577.

Jurik, T. W., S. C. Wang, and A. G. van der Valk. 1994. Effects of sediment load on seedling emergence from wetland seed banks. Wetlands 14:159-165.

Kadlec, R. H. and F. B. Bevis. 1990. Wetlands and wastewater: Kinross, Michigan. Wetlands 10:77-92.

Kantrud, H. A. and W. E. Newton. 1996. A test of vegetation-related indicators of wetland quality in the prairie pothole region. Journal of Aquatic Ecosystem Health 5:177-191.

Keddy, P. A. and A. A. Reznicek. 1986. Great Lakes vegetation dynamics: the role of fluctuating water levels and buried seeds. Journal of Great Lakes Research 12:25-36.

Kempton, R. A. 1979. The structure of species abundance and measurement of diversity. Biometrics 35:307-321.

Keough, J. R., T. A. Thompson, G. R. Guntenspergen, and D. A. Wilcox. 1999. Hydrogeomorphic factors and ecosystem responses in coastal wetlands of the Great Lakes. Wetlands 19:821-834.

Keys, J. E., Jr., C. A. Carpenter, S. L. Hooks, F. G. Koeneg, W. H. McNab, W. Russell, and M. L. Smith. 1995. Ecological units of the eastern United States: first approximation. Map (scale 1:3,500,000). U.S. Dept. of Agriculture, Forest Service, Atlanta, GA, USA. Technical Publication R8-TP 21.

Ladd, D. 1993 The Missouri floristic quality assessment system. Nature Conservancy. The Nature Conservancy, St. Louis, MO, USA.

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

Mack, J. J. 2001. Vegetation index of biotic integrity (VIBI) for wetlands: ecoregional, hydrogeomorphic, and plant community comparisons with preliminary wetland aquatic life use designations. Wetland Ecology Group, division of Surface Water, Ohio EPA, Columbus, OH, USA. Final report to U.S. EPA Grant No. CD985875-01.

Magurran, A. E. 1988. Ecological Diversity and its Measurement. Princeton University Press, Princeton, NJ, USA.

Matthews, J. W. 2003. Assessment of the floristic quality index for use in Illinois, USA, wetlands. Natural Areas Journal 23:53-60.

Maynard, L. and D. Wilcox. 1997. State of the lakes ecosystem conference 1996 background paper: Coastal wetlands. U.S. Environmental Protection Agency, Great Lakes National Program Office, Chicago, IL, USA. EPA 905-R97-015b.

Mensing, D. M., S. M. Galatowitsch, and J. R. Tester. 1998. Anthropogenic effects on the biodiversity of riparian wetlands of a northern temperate landscape. Journal of Environmental Management 53:349-377.

Minc, L. D. and D. A. Albert. 1998. Great Lakes coastal wetlands: abiotic and floristic characterization. Michigan Natural Features Inventory, Lansing, MI, USA.

Morrison, S. L. and J. Molofsky. 1999. Environmental and genetic effects on the early survival and growth of the invasive grass *Phalaris arundinacea*. Canadian Journal of Botany 77:1447-1453.

Mortsch, L.D. 1998. Assessing the impact of climate change on the Great Lakes shoreline wetlands. Climatic Change 40:391-416.

Mortsch, L. D. and F. H. Quinn. 1996. Climate change scenarios for Great Lakes basin ecosystem studies. Limnology and Oceanography 41:903-911.

Mushet, D. M., N. H. Euliss, Jr., and T. H. Shaffer. 2002. Floristic quality assessment of one natural and three restored wetland complexes in North Dakota, USA. Wetlands 22:126-138.

National Research Council (NRC). 2000. Ecological Indicators for the Nation. National Academy Press, Washington, DC, USA.

Nichols, S. A. 2001. Long-term change in Wisconsin lake plant communities. Journal of Freshwater Ecology 16:1-13.

Northern Great Plains Floristic Quality Assessment Panel (NGPFQAP). 2001. Floristic quality assessment for plant communities of North Dakota, South Dakota (excluding the Black Hills), and adjacent grasslands. Northern Prairie Wildlife Research Center Home Page. http://www.npwrc.usgs.gov/resource/2001/fqa/fqa.htm (Version 26Jan2001). Jamestown, ND, USA.

Oldham, M. J., W. D. Bakowsky, and D. A. Sutherland. 1995. Floristic quality assessment system for southern Ontario. Ontario Ministry of Natural Resources, Natural Heritage Information Centre, Peterborough, Ontario, Canada.

Otte, M. L. 2001. What is stress to a wetland plant? Environmental and Experimental Botany 46:195-202.

Parendes, L. A. and J. A. Jones. 2000. Role of light availability and dispersal in exotic plant invasion along roads and streams in the H.J. Andrews Experimental Forest, Oregon. Conservation Biology 14:64-75.

Poling, T. C., M. G. Banker, and L. M. Jablonski. 2003. Quadrat-level floristic quality index reflects shifts in composition of a restored tallgrass prairie (Ohio). Ecological Restoration 21:144-145.

Rachich, J. and R. J. Reader. 1999. An experimental study of wetland invisibility by purple loosestrife (*Lythrum salicaria*). Canadian Journal of Botany 77:1499-1503.

Rapport, D. J., H. A. Regier, and T. C. Hutchinson. 1985. Ecosystem behavior under stress. American Naturalist 125:617-640.

Regier, H. A., 1993. The notion of natural and cultural integrity. In: Woodley, S., J. Kay, G. Francis. Ecological integrity and the management of ecosystems. St. Lucie Press, Delray Beach, FL, USA.

Rooney, T. P., and D. A. Rogers. 2002. The modified floristic quality index. Natural Areas Journal 22:340-344.

Rosenzweig, M. L. 1995. Species Diversity in Space and Time. Cambridge University Press. Cambridge UK.

Saltonstall, K. 2002. Cryptic invasion by a non-native genotype of the common reed, *Phragmites australis*, into North America. Proceedings of the National Academy of Sciences, U.S.A. 99:2445-2449.

Sanzone, S., and A. McElroy (eds.). 1998. Ecological impacts and evaluation criteria for the use of structures in marsh management. Marsh Management Subcommittee, Ecological Processes and Effects Committee, U.S. Environmental Protection Agency Science Advisory Board, Washington, DC, USA. EPA-SAB-EPEC-98-003.

Smith, L. M. and D. A. Haukos. 2002. Floral diversity in relation to playa wetland area and watershed disturbance. Conservation Biology 16:964-974.

Steinberg, T. 2002. Down to Earth: Nature's Role in American History. Oxford University Press, New York, NY, USA.

Swink, F. A. and G. S. Wilhelm. 1994. Plants of the Chicago Region (4th ed.). Morton Arboretum. Lisle, IL, USA.

Taft, J. B., G. S. Wilhelm, D. M. Ladd, and L. A. Masters. 1997. Floristic quality assessment for vegetation in Illinois: a method for assessing vegetation integrity. Erigenia 15:3-95.

Tufford, D. L., H. N. McKellar, Jr., and J. R. Hussey. 1998. In-stream nonpoint source nutrient prediction with land-use proximity and seasonality. Journal of Environmental Quality 27:100-111.

Vargo, S. M., R. K. Neely, and S. M. Kirkwood. 1998. Emergent plant decomposition and sedimentation: response to sediments varying in texture, phosphorus content and frequency of deposition. Environmental and Experimental Botany 40:43-58.

Wang, S. C., T. W. Jurik, and A. G. van der Valk. 1994. Effects of sediment load on various stages in the life and death of cattail (*Typha* X *glauca*). Wetlands 14:166-173.

Washington, H. G. 1984. Diversity, biotic, and similarity indices: a review with special relevance to aquatic ecosystems. Water Resources 18:653-694.

Wilcox, D. A. 1995. Wetland and aquatic macrophytes as indicators of anthropogenic hydrologic disturbance. Natural Areas Journal 15:240-248.

Wilcox, D. A., S. I. Apfelbaum, and R. D. Hiebert. 1985. Cattail invasion of sedge meadows following hydrologic disturbance in the Cowles Bog wetland complex, Indiana Dunes National Lakeshore. Wetlands 4:115-128.

Wilcox, D. A. and J. E. Meeker. 1991. Disturbance effects on aquatic vegetation in regulated and unregulated lakes in northern Minnesota. Canadian Journal of Botany 69:1542-1551.

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

Wilhelm, G. S. 1977. Ecological assessment of open land areas in Kane County, Illinois. Kane County Urban Development. Geneva, IL, USA.

Table 1. Plant species observed with Michigan (Herman et al. 2001), Wisconsin (Bernthal et al. 2001), and study area specific *C*-values. Missing values indicate that the species does not occur in that state, and * indicates that the species is considered introduced.

Scientific Name	MI	WI	Study Area
Acer negundo L.	0	0	0
Acer rubrum L.	1	3	2
Acorus calamus L.	6	7	6.5
Agalinis paupercula (Gray) Britt.		7	7
Agalinis tenuifolia (Vahl) Raf.	5	6	5.5
Agrostis hyemalis (Walt.) B.S.P.	4	4	4
Agrostis stolonifera L.	*	*	*
Alisma subcordatum Raf.	1	3	2
Alisma triviale Pursh	1	4	2.5
Alnus incana ssp. *rugosa* (Du Roi) Clausen	5	4	4.5
Alopecurus aequalis Sobol.	4	4	4
Ambrosia artemisiifolia L.	0	0	0
Ammophila breviligulata Fern.	10	10	10
Andromeda polifolia var. *glaucophylla* (Link) DC.	10	10	10
Anemone canadensis L.	4	4	4
Apios americana Medik.	3	5	4
Arethusa bulbosa L.	10	10	10
Argentina anserina (L.) Rydb.	5	4	4.5
Arrhenatherum elatius (L.) Beauv. Ex J.& K. Presl	*	*	*
Artemisia vulgaris L.	*	*	*
Asclepias incarnata L.	6	5	5.5
Asclepias syriaca L.	1	1	1
Asclepias tuberosa L.	5	6	5.5
Azolla caroliniana Willd.	10	10	10
Barbarea vulgaris Ait. f.	*	*	*
Beckmannia syzigachne (Steud.) Fern.	4	*	4
Betula papyrifera Marsh.	2	3	2.5
Betula pumila L.	8	7	7.5
Bidens cernua L.	3	4	3.5
Bidens connata Muhl. ex Willd.	5	6	5.5
Bidens frondosa L.	1	1	1
Bidens vulgata Greene	0	1	0.5
Boehmeria cylindrica (L.) Sw.	5	6	5.5
Brasenia schreberi J.F. Gmel.	6	6	6
Brassica nigra (L.) W.D.J. Koch	*	*	*
Butomus umbellatus L.	*	*	*
Calamagrostis canadensis (Michx.) Beauv.	3	5	4
Calamagrostis stricta (Timm) Koel.	10	7	8.5
Calla palustris L.	10	9	9.5
Callitriche palustris L.	6	8	7
Caltha palustris L.	6	6	6

Table 1. Continued.

Scientific Name	MI	WI	Study Area
Calystegia sepium (L.) R.Br.	2	2	2
Campanula aparinoides Pursh	7	7	7
Cardamine pensylvanica Muhl. ex Willd.	1	3	2
Carex aquatilis Wahlenb.	7	7	7
Carex aurea Nutt.	3	5	4
Carex bebbii Olney ex Fern.	4	4	4
Carex brunnescens (Pers.) Poir.	5	7	6
Carex buxbaumii Wahlenb.	10	8	9
Carex chordorrhiza Ehrh. ex L. f.	10	10	10
Carex comosa Boott	5	5	5
Carex cristatella Britt.	3	4	3.5
Carex cryptolepis Mackenzie	10	8	9
Carex diandra Schrank	8	9	8.5
Carex echinata Murr.	6	8	7
Carex exilis Dewey	10	9	9.5
Carex flava L.	4	6	5
Carex granularis Muhl. ex Willd.	2	3	2.5
Carex gynocrates Wormsk. ex Drej.	10	10	10
Carex hystericina Muhl. ex Willd.	2	3	2.5
Carex lacustris Willd.	6	6	6
Carex lasiocarpa var. *americana* Fern.	8	9	8.5
Carex leptalea Wahlenb.	5	9	7
Carex limosa L.	10	10	10
Carex livida (Wahlenb.) Willd.	10	10	10
Carex lurida Wahlenb.	3	8	5.5
Carex michauxiana Boeckl.	10	10	10
Carex normalis Mackenzie	5	5	5
Carex pellita Muhl ex Willd.	2	4	3
Carex prairea Dewey ex Wood	10	10	10
Carex pseudocyperus L.	5	8	6.5
Carex retrorsa Schwein.	3	6	4.5
Carex rostrata Stokes	10	10	10
Carex sartwellii Dewey	5	7	6
Carex scoparia Schkuhr ex Willd.	4	4	4
Carex stipata Muhl. ex Willd.	1	2	1.5
Carex stricta Lam.	4	7	5.5
Carex sychnocephala Carey	6	9	7.5
Carex trisperma Dewey	9	9	9
Carex utriculata Boott	5	7	6
Carex vesicaria L.	7	7	7
Carex viridula Michx.	4	6	5
Carex vulpinoidea Michx.	1	2	1.5
Cephalanthus occidentalis L.	7	9	8
Ceratophyllum demersum L.	1	3	2
Ceratophyllum echinatum Gray	10	10	10
Chamaedaphne calyculata (L.) Moench	8	9	8.5
Chelone glabra L.	7	7	7
Cicuta bulbifera L.	5	7	6

Table 1. Continued.

Scientific Name	MI	WI	Study Area
Cirsium arvense (L.) Scop.	*	*	*
Cirsium muticum Michx.	6	8	7
Cirsium palustre (L.) Scop.	*	*	*
Cirsium vulgare (Savi) Ten.	*	*	*
Cladium mariscoides (Muhl.) Torr.	10	10	10
Clematis virginiana L.	4	4	4
Clinopodium arkansanum (Nutt.) House	10	10	10
Comarum palustre L.	7	8	7.5
Cornus amomum P. Mill.	2	4	3
Cornus drummondii C.A. Mey.	6		6
Cornus racemosa Lam.	1	2	1.5
Cornus sericea ssp. *sericea* L.	2	3	2.5
Crataegus douglasii Lindl.	7		7
Crepis tectorum L.	*	*	*
Cuscuta gronovii Willd. ex J.A. Schultes	3	4	3.5
Cyperus strigosus L.	3	1	2
Dasiphora floribunda (Pursh) Kartesz, comb. nov. ined.	10	9	9.5
Daucus carota L.	*	*	*
Decodon verticillatus (L.) Ell.	7	7	7
Deschampsia caespitosa (L.) Beauv.	9	10	9.5
Dichanthelium acuminatum var. *fasciculatum* (Torr.) Freckmann	3	2	2.5
Dichanthelium acuminatum var. *lindheimeri* (Nash) Gould & C.A.Clark	8	8	8
Doellingeria umbellata var. *umbellata* (P. Mill.) Nees	5	6	5.5
Drosera rotundifolia L.	6	7	6.5
Dryopteris carthusiana (Vill.) H. P. Fuchs	5	7	6
Dulichium arundinaceum (L.) Britt.	8	9	8.5
Echinochloa muricata (Beauv.) Fern.	1	1	1
Echinochloa walteri (Pursh) Heller	7	8	7.5
Eleocharis acicularis (L.) Roemer & J.A. Schultes	7	5	6
Eleocharis elliptica Kunth	6	7	6.5
Eleocharis erythropoda Steud.	4	3	3.5
Eleocharis palustris (L.) Roemer & J.A. Schultes	5	6	5.5
Eleocharis quinqueflora (F.X. Hartmann) Schwarz	10	8	9
Eleocharis robbinsii Oakes	8	10	9
Elodea canadensis Michx.	1	3	2
Elodea nuttallii (Planch.) St. John	5	7	6
Elymus repens (L.) Gould	*	*	*
Elymus trachycaulus (Link) Gould ex Shinners	8	4	6
Elymus virginicus L.	4	6	5
Epilobium ciliatum Raf.	3	3	3
Epilobium coloratum Biehler	3	3	3
Epilobium hirsutum L.	*	*	*
Epilobium leptophyllum Raf.	6	8	7
Epilobium strictum Muhl. ex Spreng.	8	10	9
Equisetum arvense L.	0	1	0.5
Equisetum fluviatile L.	7	7	7
Equisetum palustre L.	10	8	9

Table 1. Continued.

Scientific Name	MI	WI	Study Area
Equisetum sylvaticum L.	5	7	6
Erechtites hieracifolia (L.) Raf. ex DC.	2	2	2
Erigeron annuus (L.) Pers.	0	0	0
Erigeron strigosus Muhl. ex Willd.	4	2	3
Eriocaulon aquaticum (Hill) Druce	9	9	9
Eriophorum angustifolium Honckeny	10	9	9.5
Eriophorum gracile W.D.J. Koch	10	10	10
Eriophorum vaginatum var. spissum (Fern.) Boivin	10	10	10
Erucastrum gallicum (Willd.) O.E. Schulz	*	*	*
Eupatorium maculatum L.	4	4	4
Eupatorium perfoliatum L.	4	6	5
Euthamia graminifolia (L.) Nutt.	3	4	3.5
Fragaria virginiana Duchesne	2	1	1.5
Frangula alnus P. Mill.	*	*	*
Fraxinus pennsylvanica Marsh.	2	2	2
Galeopsis tetrahit L.	*	*	*
Galium asprellum Michx.	5	7	6
Galium labradoricum (Wieg.) Wieg.	8	10	9
Galium obtusum ssp. *obtusum* Bigelow	5	6	5.5
Galium palustre L.	3	*	3
Galium tinctorium L.	5	5	5
Galium trifidum L.	6	6	6
Glyceria borealis (Nash) Batchelder	6	8	7
Glyceria grandis S. Wats.	6	6	6
Glyceria striata (Lam.) A.S. Hitchc.	4	4	4
Gratiola neglecta Torr.	5	5	5
Hackelia virginiana (L.) I.M. Johnston	1	3	2
Helianthus grosseserratus Martens	2	2	2
Hibiscus moscheutos L.	7	*	7
Hippuris vulgaris L.	10	10	10
Hordeum jubatum L.	*	*	*
Hydrocharis morsus-ranae L.	*		*
Hypericum boreale (Britt.) Bickn.	5	6	5.5
Hypericum ellipticum Hook.	9	9	9
Hypericum kalmianum L.	10	9	9.5
Hypericum majus (Gray) Britt.	4	5	4.5
Ilex verticillata (L.) Gray	5	7	6
Impatiens capensis Meerb.	2	2	2
Iris pseudacorus L.	*	*	*
Iris versicolor L.	5	5	5
Iris virginica L.	5	5	5
Juncus alpinoarticulatus Chaix in Villars	5	6	5.5
Juncus articulatus L.	3	*	3
Juncus balticus var. *littoralis* Engelm.	4	5	4.5
Juncus brevicaudatus (Engelm.) Fern.	8	6	7
Juncus dudleyi Wieg.	1	4	2.5
Juncus effusus L.	3	4	3.5
Juncus militaris Bigelow	10		10

Table 1. Continued.

Scientific Name	MI	WI	Study Area
Juncus nodosus L.	5	6	5.5
Juncus pelocarpus E. Mey.	8	8	8
Juncus tenuis Willd.	1	1	1
Justicia americana (L.) Vahl	9		9
Kalmia polifolia Wangenh.	10	10	10
Laportea canadensis (L.) Weddell	4	4	4
Larix laricina (DuRooi) K. Koch	5	8	6.5
Lathyrus palustris L.	7	5	6
Ledum groenlandicum Oeder	8	8	8
Leersia oryzoides (L.) Sw.	3	3	3
Lemna minor L.	5	4	4.5
Lemna trisulca L.	6	6	6
Leucanthemum vulgare Lam.	*	*	*
Lilium michiganense Farw.	5	6	5.5
Linaria vulgaris P. Mill.	*	*	*
Lobelia kalmii L.	10	9	9.5
Lonicera oblongifolia (Goldie) Hook.	8	9	8.5
Ludwigia palustris (L.) Ell.	4	4	4
Lycopus americanus Muhl. ex W. Bart	2	4	3
Lycopus europaeus L.	*	*	*
Lycopus uniflorus Michx.	2	4	3
Lycopus virginicus L.	8	8	8
Lysimachia nummularia L.	*	*	*
Lysimachia quadriflora Sims	10	9	9.5
Lysimachia terrestris (L.) B.S.P.	6	7	6.5
Lysimachia thyrsiflora L.	6	7	6.5
Lythrum salicaria L.	*	*	*
Maianthemum trifolium (L.) Sloboda	10	10	10
Megalodonta beckii (Torr. ex Spreng.) Greene	10	8	9
Mentha arvensis L.	3	3	3
Menyanthes trifoliata L.	8	10	9
Mimulus alatus Ait.	9		9
Mimulus glabratus Kunth	10	9	9.5
Mimulus ringens L.	5	6	5.5
Muhlenbergia glomerata (Willd.) Trin.	10	9	9.5
Myosotis laxa Lehm.	6	9	7.5
Myosotis scorpioides L.	*	*	*
Myrica gale L.	6	9	7.5
Myriophyllum farwellii Morong	10	8	9
Myriophyllum humile (Raf.) Morong		10	10
Myriophyllum sibiricum Komarov	10	6	8
Myriophyllum spicatum L.	*	*	*
Myriophyllum verticillatum L.	6	8	7
Najas flexilis (Willd.) Rostk. & Schmidt	5	6	5.5
Najas guadalupensis (Spreng.) Magnus	7	8	7.5
Najas marina L.	*	*	*
Nelumbo lutea Willd.	8	7	7.5
Nemopanthus mucronatus (L.) Loes.	7	8	7.5

Table 1. Continued.

Scientific Name	MI	WI	Study Area
Nuphar lutea ssp. *pumila* (Timm) E.O. Beal	10	9	9.5
Nuphar lutea ssp. *variegata* (Dur.) E.O. Beal	7	6	6.5
Nymphaea odorata Ait.	6	6	6
Oligoneuron houghtonii (Torr. & Gray ex Gray) Nesom	10		10
Oligoneuron ohioense (Frank ex Riddell) G.N. Jones	8	9	8.5
Onoclea sensibilis L.	2	5	3.5
Osmunda cinnamomea L.	5	7	6
Osmunda regalis L.	5	7	6
Parnassia glauca Raf.	8	8	8
Parthenocissus quinquefolia (L.) Planch	5	5	5
Peltandra virginica (L.) Schott	6	*	6
Penthorum sedoides L.	3	3	3
Phalaris arundinacea L.	0	*	0
Phleum pratense L.	*	*	*
Phlox maculata L.	10		10
Phragmites australis (Cav.) Trin. ex Steud.	0	1	0.5
Pilea fontana (Lunell) Rydb.	5	7	6
Pilea pumila (L.) Gray	5	3	4
Pinus strobus L.	3	5	4
Plantago major L.	*	*	*
Platanthera psycodes (L.) Lindl.	7	7	7
Poa compressa L.	*	*	*
Poa palustris L.	3	5	4
Poa pratensis L.	*	*	*
Poa trivialis L.	*	*	*
Pogonia ophioglossoides (L.) Ker-Gawl.	10	9	9.5
Polygonum amphibium L.	6	5	5.5
Polygonum amphibium var. *stipulaceum* Coleman	6	6	6
Polygonum convolvulus L.	*	*	*
Polygonum hydropiper L.	1	*	*
Polygonum hydropiperoides Michx.	5	6	5.5
Polygonum lapathifolium L.	0	2	1
Polygonum pensylvanicum L.	0	1	0.5
Polygonum punctatum Ell.	5	5	5
Polygonum sagittatum L.	5	6	5.5
Pontederia cordata L.	8	8	8
Populus balsamifera L.	2	4	3
Populus deltoides Bartr. ex Marsh.	1	2	1.5
Populus tremuloides Michx.	1	2	1.5
Potamogeton crispus L.	*	*	*
Potamogeton diversifolius Raf.	10	8	9
Potamogeton epihydrus Raf.	8	8	8
Potamogeton foliosus Raf.	4	6	5
Potamogeton friesii Rupr.	6	8	7
Potamogeton gramineus L.	5	7	6
Potamogeton illinoensis Morong	5	6	5.5
Potamogeton natans L.	5	5	5
Potamogeton nodosus Poir.	6	7	6.5

Table 1. Continued.

Scientific Name	MI	WI	Study Area
Potamogeton richardsonii (Benn.) Rydb.	5	5	5
Potamogeton zosteriformis Fern.	5	6	5.5
Potentilla norvegica L.	0	0	0
Primula mistassinica Michx.	10	10	10
Proserpinaca palustris L.	6	8	7
Prunella vulgaris L.	0	1	0.5
Prunella vulgaris ssp. *lanceolata* (Bart.) Hulten	0	1	0.5
Pycnanthemum virginianum (L.) T. Dur. & B.D. Jackson ex B.L. Robins. & Fern.	5	6	5.5
Pyrola asarifolia Michx.	8	8	8
Ranunculus hispidus Michx.	5	6	5.5
Ranunculus sceleratus L.	1	3	2
Rhynchospora alba (L.) Vahl	6	9	7.5
Rhynchospora capillacea Torr.	10	10	10
Rhynchospora capitellata (Michx.) Vahl.	6	8	7
Rhynchospora fusca (L.) Ait. f.	7	10	8.5
Ribes americanum P. Mill.	6	4	5
Rorippa palustris (L.) Bess.	1	3	2
Rorippa palustris ssp. *hispida* (Desv.) Jonsell	1	3	2
Rosa palustris Marsh.	5	7	6
Rubus allegheniensis Porter	1	2	1.5
Rubus idaeus ssp. *strigosus* (Michx.) Focke	2	3	2.5
Rubus pubescens Raf.	4	7	5.5
Rudbeckia hirta L.	1	4	2.5
Rumex orbiculatus Gray	9	8	8.5
Rumex verticillatus L.	7	6	6.5
Sagittaria brevirostra Mackenzie & Bush	10	9	9.5
Sagittaria cuneata Sheldon	6	7	6.5
Sagittaria graminea Michx.	10	9	9.5
Sagittaria latifolia Willd.	1	3	2
Sagittaria rigida Pursh	6	8	7
Salix amygdaloides Anderss.	3	4	3.5
Salix bebbiana Sarg.	1	7	4
Salix candida Flugge ex Willd.	9	10	9.5
Salix discolor Muhl.	1	2	1.5
Salix interior Rowlee	1	2	1.5
Salix lucida Muhl.	3	5	4
Salix nigra Marsh.	5	4	4.5
Salix pedicellaris Pursh.	8	8	8
Salix petiolaris Sm.	1	6	3.5
Salix serissima (Bailey) Fern.	8	8	8
Sambucus nigra ssp. *canadensis* (L.) R. Bolli	3	3	3
Sarracenia purpurea L.	10	10	10
Sassafras albidum (Nutt.) Nees	5	5	5
Scheuchzeria palustris L.	10	10	10
Schizachyrium scoparium var. *scoparium* (Michx.) Nash	5	4	4.5
Schoenoplectus acutus var. *acutus* (Muhl. ex Bigelow) A.& D. Love	5	6	5.5

Table 1. Continued.

Scientific Name	MI	WI	Study Area
Schoenoplectus americanus (Pers.) Volk. ex Schinz & R. Keller	10		10
Schoenoplectus fluviatilis (Torr.) M.T. Strong	6	6	6
Schoenoplectus pungens var. *pungens* (Vahl) Palla	5	5	5
Schoenoplectus subterminalis (Torr.) Sojak	8	9	8.5
Schoenoplectus tabernaemontani (K.C. Gmel.) Palla	4	4	4
Scirpus atrovirens Willd.	3	3	3
Scirpus cyperinus (L.) Kunth	5	4	4.5
Scirpus microcarpus J.& K. Presl.	5	6	5.5
Scutellaria galericulata L.	5	5	5
Scutellaria lateriflora L.	5	5	5
Sium suave Walt.	5	5	5
Solanum dulcamara L.	*	*	*
Solidago canadensis L.	1	1	1
Solidago canadensis var. *scabra* Torr. & Gray	1	1	1
Solidago gigantea Ait.	3	3	3
Solidago juncea Ait.	3	4	3.5
Solidago rugosa P. Mill.	3		3
Solidago uliginosa Nutt.	4	8	6
Sonchus arvensis L.	*	*	*
Sparganium androcladum (Engelm.) Morong	6	8	7
Sparganium erectum ssp. *stoloniferum* (Graebn.) Hara	6	8	7
Sparganium eurycarpum Engelm. ex Gray	5	5	5
Sparganium fluctuans (Morong) B.L. Robbins.	10	10	10
Spartina pectinata Bosc ex Link	5	5	5
Sphenopholis intermedia (Rydb.) Rydb.	4	7	5.5
Spiraea alba Du Roi	4	4	4
Spiranthes cernua (L.) L.C. Rich.	4	5	4.5
Spirodela polyrrhiza (L.) Schleiden	6	5	5.5
Sporobolus cryptandrus (Torr.) Gray	3	3	3
Stachys palustris L.	5	5	5
Stachys pilosa Nutt.	5	5	5
Stachys tenuifolia Willd.	5	6	5.5
Stuckenia filiformis (Pers) Boerner	7	8	7.5
Stuckenia pectinatus (L.) Boerner	3	3	3
Symphyotrichum boreale (Torr. & Gray) A.& D. Love	9	10	9.5
Symphyotrichum lanceolatum var. *lanceolatum* (Willd.) Nesom	2	4	3
Symphyotrichum lateriflorum var. *lateriflorum* (L.) A.& D. Love	2	3	2.5
Symphyotrichum pilosum var. *pilosum* (Willd.) Nesom	1	7	4
Symphyotrichum puniceum var. *puniceum* (L.) A.& D. Love	5	5	5
Tanacetum vulgare L.	*	*	*
Taraxacum officinale G.H. Weber ex Wiggers	*	*	*
Teucrium canadense L.	4	4	4
Thalictrum dasycarpum Fisch. & Ave-Lall.	3	4	3.5
Thelypteris palustris Schott	2	7	4.5
Thuja occidentalis L.	4	9	6.5
Tofieldia glutinosa (Michx.) Pers.	10	10	10
Torreyochloa pallida var. *pallida* (Torr.) Church	7	9	8
Toxicodendron radicans (L.) Kuntze	2	4	3

Table 1. Continued.

Scientific Name	MI	WI	Study Area
Triadenum fraseri (Spach) Gleason	6	8	7
Triadenum virginicum (L.) Raf.	10	9	9.5
Trichophorum alpinum (L.) Pers.	10	10	10
Trichophorum caespitosum (L.) Hartman	10	10	10
Trientalis borealis Raf.	5	7	6
Trifolium dubium Sibthorp	*	*	*
Trifolium hybridum L.	*	*	*
Triglochin maritimum L.	8	10	9
Typha angustifolia L.	*	*	*
Typha latifolia L.	1	1	1
Typha X *glauca* Godr. (pro sp.)	*	*	*
Urtica dioica L.	1	1	1
Utricularia cornuta Michx.	10	10	10
Utricularia intermedia Hayne	10	9	9.5
Utricularia macrorhiza Le Conte	6	7	6.5
Utricularia minor L.	10	10	10
Vaccinium angustifolium Ait.	4	4	4
Vaccinium macrocarpon Ait.	8	9	8.5
Vaccinium oxycoccos L.	8	9	8.5
Vallisneria americana Michx.	7	6	6.5
Verbena hastata L.	4	3	3.5
Veronica anagallis-aquatica L.	4	4	4
Veronicastrum virginicum (L.) Farw.	8	6	7
Viola pubescens Ait.	4	5	4.5
Vitis riparia Michx.	3	2	2.5
Wolffia columbiana Karst.	5	5	5
Xanthium strumarium L.	*	1	1
Xyris torta Sm.	10	10	10
Zizania aquatica L.	9	8	8.5
mean	5.42	6.01	5.73
standard deviation	2.95	2.68	2.74

Table 2. Species richness, Coefficient of Conservatism, and Floristic Quality indices.

Category	Index	Notation
Species Richness	Native Species Richness	S
	Total Species Richness	S_i
Coefficient of Conservatism	Native Species Mean Coefficient of Conservatism	$\overline{C}$
	Total Species Mean Coefficient of Conservatism	$\overline{C}_i$
	Native Weighted Coefficient of Conservatism	wC
	Total Weighted Coefficient of Conservatism	wC_i
Floristic Quality	Native Floristic Quality Index	FQI
	Total Floristic Quality Index	FQI_i
	Native Weighted Floristic Quality Index	$wFQI$
	Total Weighted Floristic Quality Index	$wFQI_i$

Table 3. Diversity indices where *S* is species richness, *N* is abundance (summed percent cover), and p_i is the relative abundance of the *i*th species. All formulas from Magurran (1988).

Index	Formula
Margalef diversity	$D_{Mg} = (S - 1) / \ln N$
Shannon diversity	$H' = -\sum_{i=1}^{S} p_i \ln p_i$
Shannon evenness	$relH' = H' / \ln S$
Simpson dominance	$D_s = \sum_{i=1}^{S} p_i^2$

Table 4. Pearson correlation coefficients of *FQI* variables and diversity indices. Diversity indices are arranged along the top axis of the table that range from measurements of species richness to measurements that reflect changes in the abundance distribution of the community. * = $P < 0.05$, ** = $P < 0.01$.

Open Coast Wetlands

Index	S	D_{Mg}	H'	$relH'$	$1\text{-}D_S$
S	1	0.83**	0.40	0.27	0.38
$\overline{C}$	0.70**	0.68**	0.41	0.23	0.33
FQI	0.90**	0.77**	0.42	0.25	0.38

Riverine Wetlands

Index	S	D_{Mg}	H'	$relH'$	$1\text{-}D_S$
S	1	0.72**	0.45**	0.33	0.49*
$\overline{C}$	0.57**	0.31	0.32	0.55**	0.52*
FQI	0.87**	0.53**	0.42*	0.50*	0.55**

Protected Wetlands

Index	S	D_{Mg}	H'	$relH'$	$1\text{-}D_S$
S	1	0.76**	0.43	0.11	0.14
$\overline{C}$	0.62*	0.42	-0.05	-0.02	-0.05
FQI	0.85**	0.63**	0.16	0.03	0.03

Table 5. Multiple regression results in each of the wetland types. Values listed under Agriculture pc1, Population Density pc1, and Hydrologic Modification are regression coefficients for the three predictors (* = $P < 0.05$, ** = $P < 0.01$). Values listed under Power are power probabilities from model simulations when all three predictors are increased 20% simultaneously.

Open Coast Wetlands

Index	F	P	r^2	MSE	Intercept	Agriculture pc1	Pop. Dens. pc1	Hydro. Mod.	Power
S	8.460	0.002	0.661	2.355	8.070**	-0.920	-0.996	-0.003	0.746
S_i	6.345	0.007	0.594	2.399	8.486**	-0.737	-0.938	-0.003	0.663
$\overline{C}$	17.528	<0.001	0.802	0.298	5.538**	-0.541*	-0.432	-0.002	0.976
$\overline{C}_i$	13.008	<0.001	0.750	0.566	5.254**	-0.721*	-0.440	-0.002	0.888
wC	13.220	<0.001	0.753	0.599	5.677**	-0.635	-0.585	-0.002	0.911
wC_i	14.987	<0.001	0.776	0.802	5.466**	-0.947*	-0.544	-0.002	0.892
FQI	31.401	<0.001	0.879	2.844	15.124**	-2.014*	-2.001*	-0.008*	0.999
FQI_i	26.946	<0.001	0.861	3.556	14.743**	-2.177*	-1.985*	-0.008	0.998
$wFQI$	22.903	<0.001	0.841	5.103	15.611**	-2.242*	-2.416*	-0.007	0.990
$wFQI_i$	24.806	<0.001	0.851	5.611	15.483**	-2.803*	-2.252*	-0.009	0.995

Riverine Wetlands

Index	F	P	r^2	MSE	Intercept	Agriculture pc1	Pop. Dens. pc1	Hydro. Mod.	Power
S	1.754	0.194	0.236	3.473	6.842**	-1.174	0.211	-0.002	0.328
S_i	1.269	0.317	0.183	3.323	7.026**	-1.080	0.429	-0.001	0.179
$\overline{C}$	2.757	0.074	0.327	0.836	5.330**	-0.483	-0.326	-0.001	0.453
$\overline{C}_i$	5.165	0.010	0.477	0.994	5.115**	-0.827*	-0.336	-0.002	0.806
wC	2.158	0.131	0.276	1.569	5.376**	-0.382	-0.697	-0.002	0.500
wC_i	4.396	0.018	0.437	1.883	5.054**	-1.013	-0.488	-0.002	0.666
FQI	3.188	0.050	0.360	11.462	13.625**	-2.322	-0.636	-0.004	0.506
FQI_i	3.933	0.027	0.410	12.262	13.380**	-2.716	-0.674	-0.005	0.620
$wFQI$	2.592	0.086	0.314	17.571	13.828**	-2.019	-1.680	-0.006	0.515
$wFQI_i$	3.551	0.037	0.385	19.516	13.345**	-2.995	-1.275	-0.006	0.581

Table 5. Continued.

Protected Wetlands

Index	F	P	r^2	MSE	Intercept	Agriculture pc1	Pop. Dens. pc1	Hydro. Mod.	Power
S	8.408	0.003	0.678	2.193	8.126**	-0.960	-1.507*	0.000	0.645
S_i	7.903	0.004	0.664	2.092	8.175**	-0.906	-1.403*	0.002	0.581
$\overline{C}$	8.410	0.003	0.678	1.432	7.415**	-0.938	-1.196*	-0.015	0.880
$\overline{C}_i$	9.731	0.002	0.709	1.378	7.399**	-0.984	-1.278*	-0.018	0.935
wC	7.204	0.005	0.643	1.951	7.637**	-1.028	-1.278	-0.016	0.825
wC_i	7.768	0.004	0.660	2.001	7.609**	-1.015	-1.412*	-0.017	0.853
FQI	25.011	<0.001	0.862	7.733	20.918**	-2.911*	-5.521**	-0.047	0.999
FQI_i	25.705	<0.001	0.865	7.698	20.899**	-2.926*	-5.621**	-0.051	0.999
$wFQI$	18.898	<0.001	0.825	11.519	21.658**	-3.273*	-5.699**	-0.052	0.994
$wFQI_i$	18.724	<0.001	0.824	11.955	21.615**	-3.109	-5.962**	-0.051	0.992

Table 6. Normal standardized multiple regression coefficients. * = $P < 0.05$, ** = $P < 0.01$.

Open-Coast Wetlands

Index	Intercept	Agriculture pc1	Pop. Dens. pc1	Hydro. Mod.
S	6.602**	-0.963	-1.006	-0.411
$\overline{C}$	4.765**	-0.566*	-0.437	-0.234
FQI	11.948**	-2.106*	-2.021*	-1.012*

Riverine Wetlands

Index	Intercept	Agriculture pc1	Pop. Dens. pc1	Hydro. Mod.
S	6.842**	-1.174	0.211	-0.002
$\overline{C}$	5.330**	-0.483	-0.326	-0.001
FQI	13.625**	-2.322	-0.636	-0.004

Protected Wetlands

Index	Intercept	Agriculture pc1	Pop. Dens. pc1	Hydro. Mod.
S	7.376**	-0.807	-1.282*	-0.013
$\overline{C}$	6.515**	-0.789	-1.017*	-0.404
FQI	17.464**	-2.448*	-4.697**	-1.293

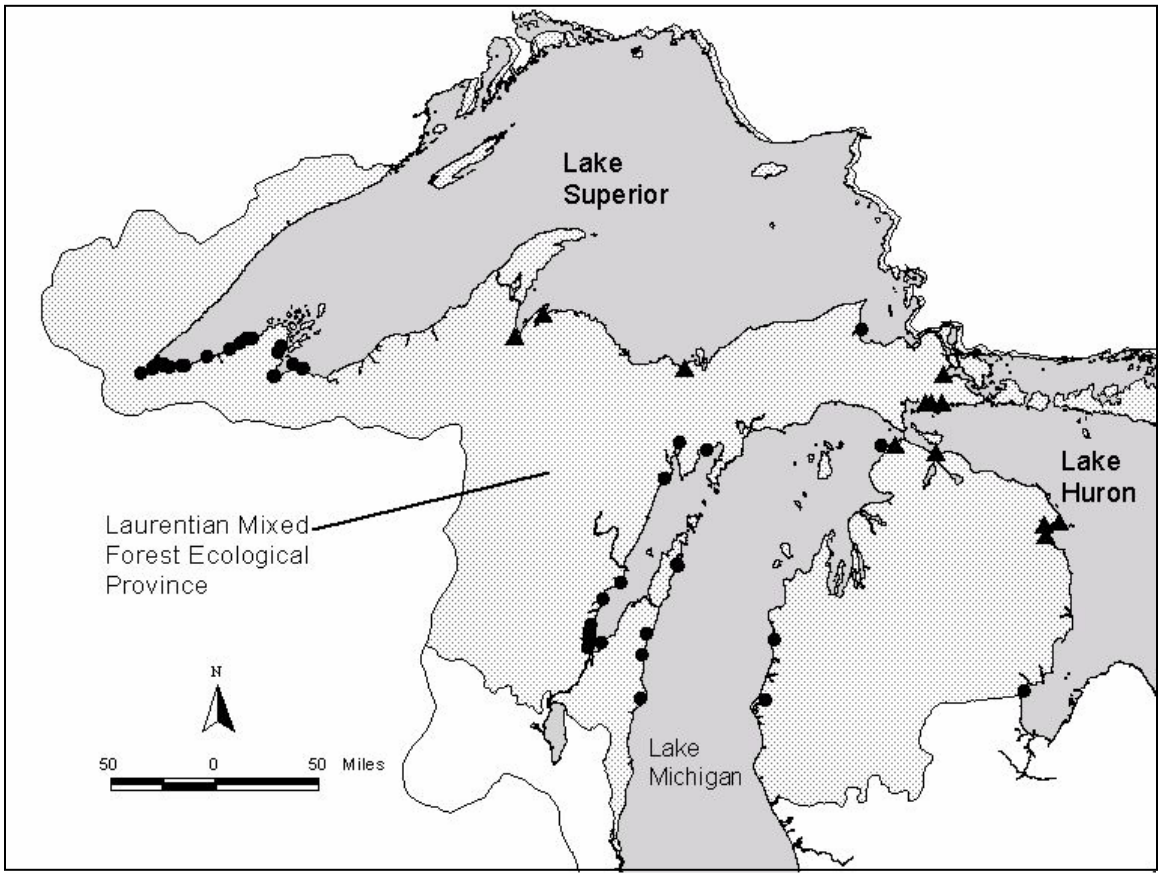


Figure 1. Coastal wetland study sites. Triangles indicate sites that were additionally sampled along transects.

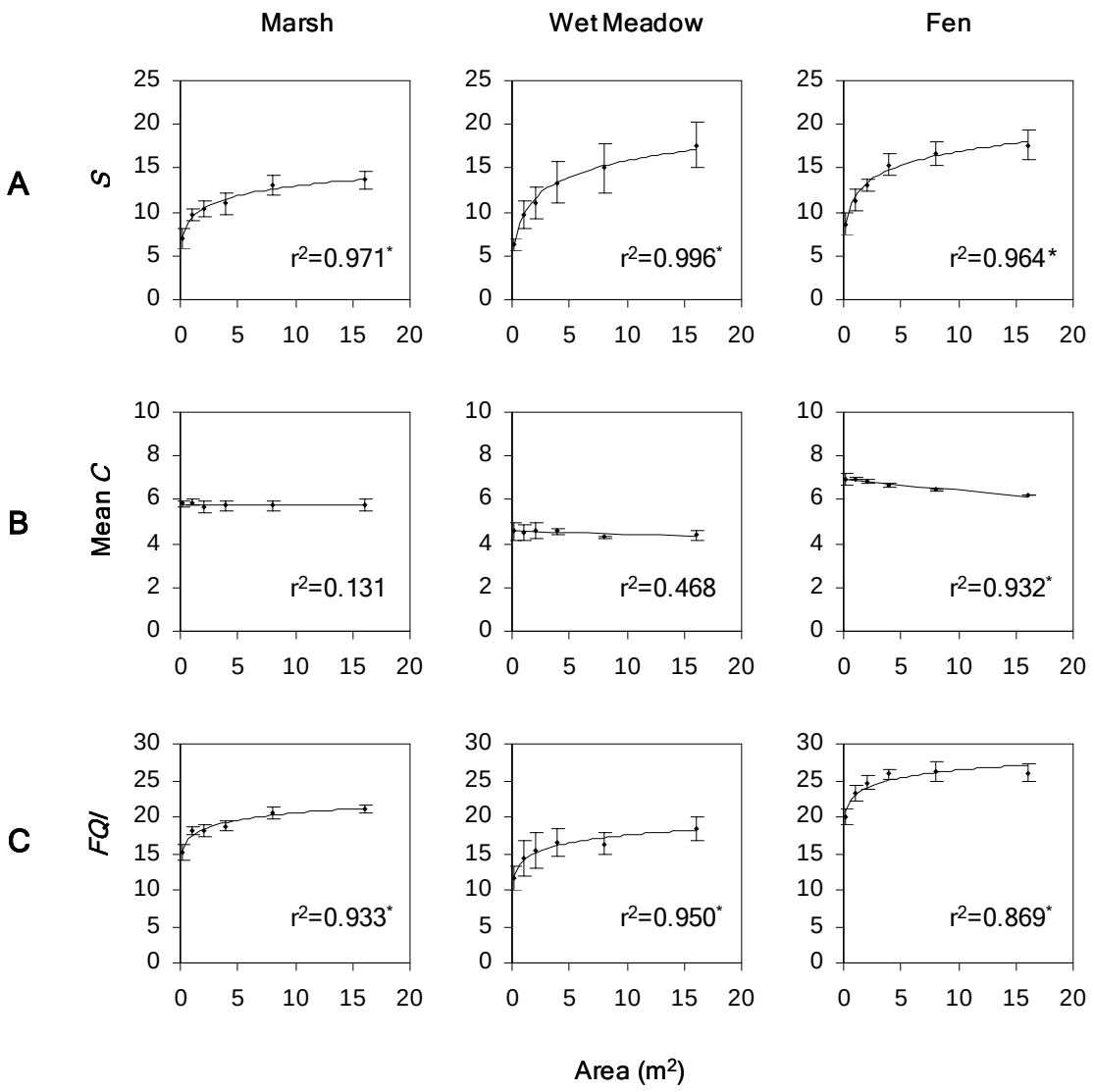


Figure 2. Species-area (A), mean Coefficient of Conservatism-area (B), and *FQI*-area (C) curves for three wetland community types in Allouez Bay, WI. * = Significant at $P < 0.01$, error bars represent one standard error.

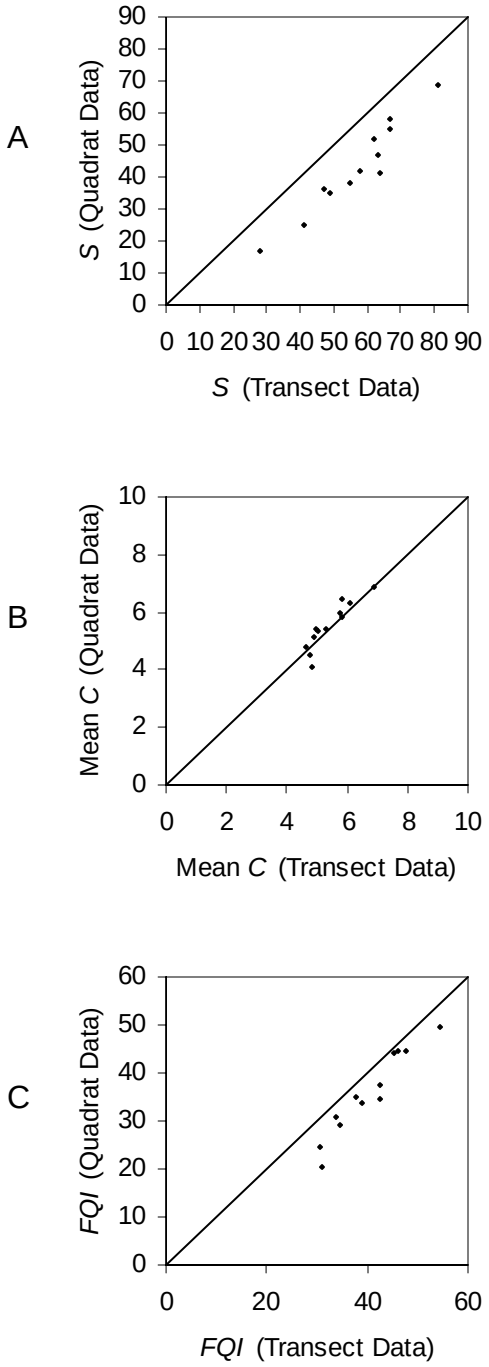


Figure 3. Scatter diagrams comparing (A) species richness, (B) mean Coefficient of Conservatism, and (C) *FQI* derived from transect and quadrat sampling from twelve sites. Each diamond represents one site. A one-one line has been added as a reference of equality.

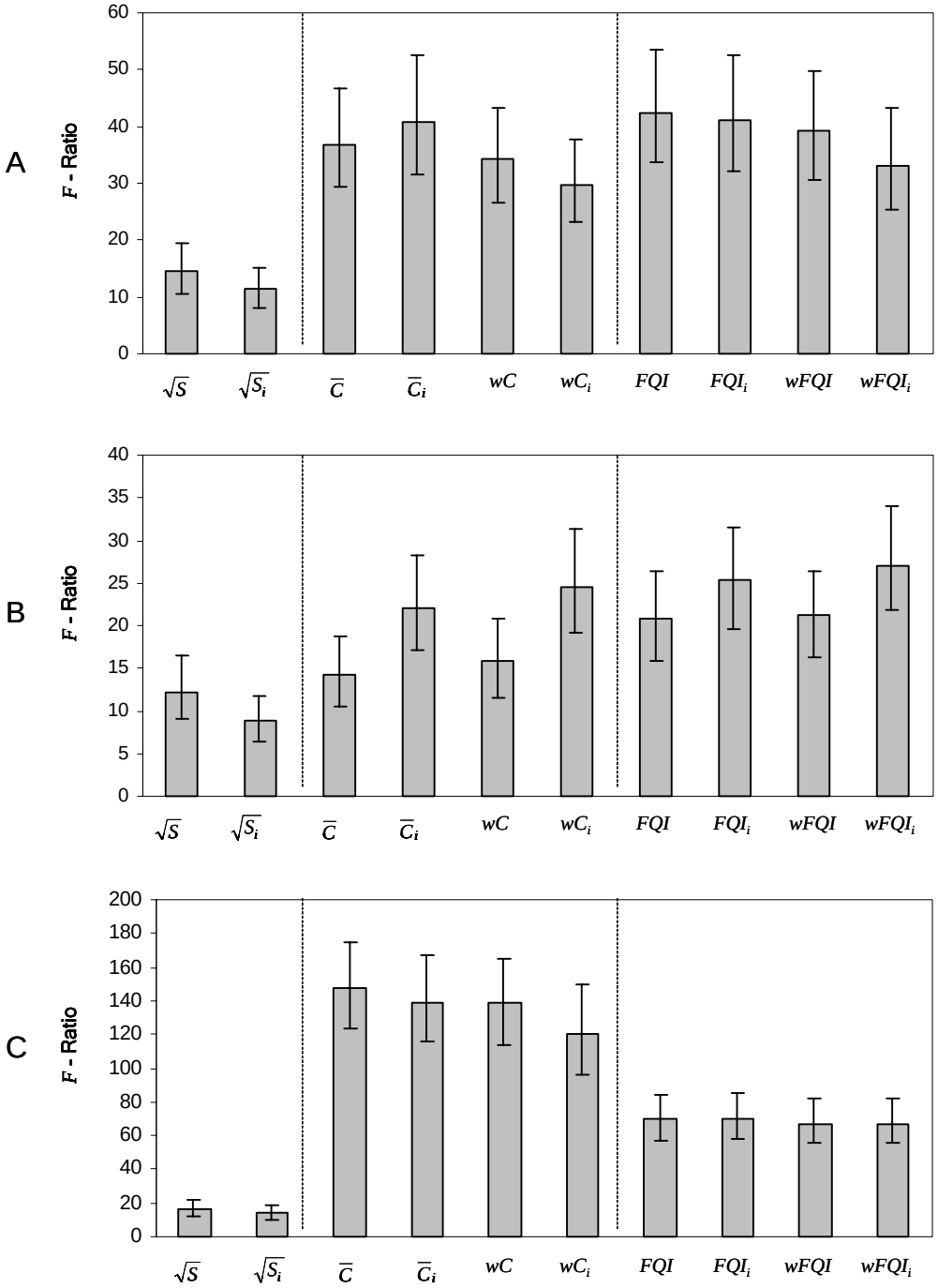
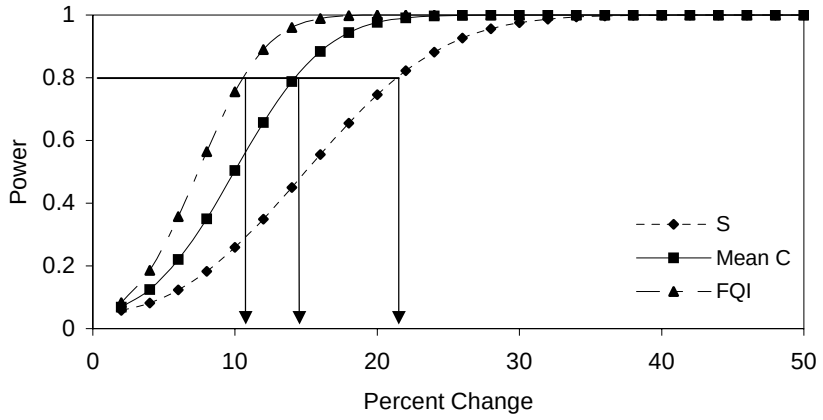
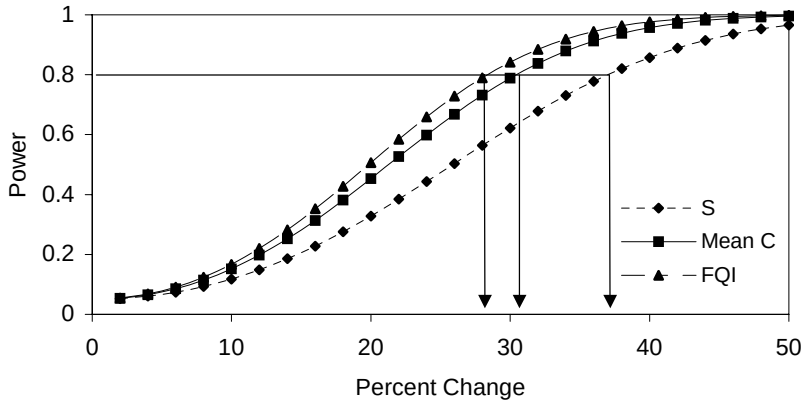


Figure 4. Bootstrap mean *F*-ratios, with 95% confidence intervals, for all species richness, Coefficient of Conservatism, and Floristic Quality indices in (A) open-coast, (B) riverine, and (C) protected wetlands. Dashed lines delineate index categories.

Open-Coast Wetlands



Riverine Wetlands



Protected Wetlands

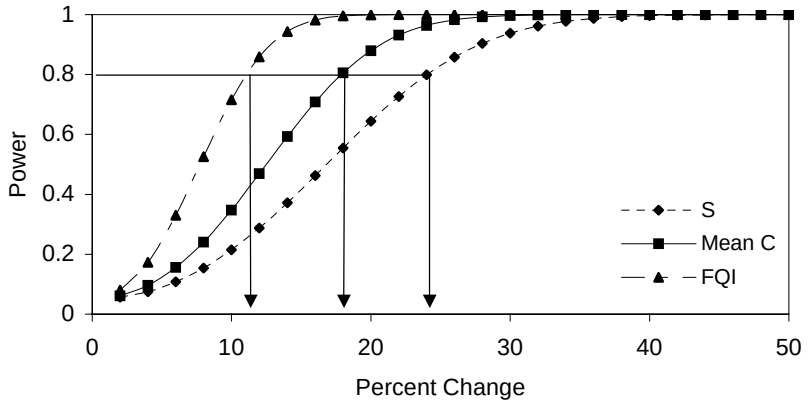


Figure 5. Power curves for detecting differences between predicted index reference values and percent predictor change values for *S*, $\overline{C}$, and *FQI* in all three wetland types.

Appendix 1. Metadata.

The purpose of this appendix is to describe data included in appendices 2-6. Data field descriptions are listed for each appendix below.

All data used in this manuscript were collected for the purposes of the Great Lakes Environmental Indicators (GLEI) project. Raw GLEI project data are available to the public and are archived by the Environmental Protection Agency (EPA) and can be accessed through EPA's Environmental Information Management System (EMIS, http://oaspub.epa.gov/eims/query.page).

Field descriptions:

Appendix 2. Study site information.

Field Name	Description
Site Name	Name assigned to site
seg_hash	A unique site identification number that includes the shoreline-segment number and a sub-segment location
Lake	Great Lake site was located on
Geomorphic Type	Site geomorphic classification: Cw = open-coast wetland, Rw = riverine wetland, Pw = protected wetland
State	State site was located in
dd_east	Centroid easting site location coordinate in decimal degrees
dd_north	Centroid northing site location coordinate in decimal degrees
Year Sampled	Year site was sampled

Appendix 3. Nested quadrat data.

Field Name	Description
Plot	Entire nested quadrat plot identifier
Sub-Plot	Specific within-plot quadrat identifier
Type	Wetland plant community type of plot: marsh, wet meadow, or fen
Area	Quadrat size: 0.25, 1, 2, 4, 8, 16 m^2
sci_name	Latin plant name

Appendix 4. Site species richness, Coefficient of Conservatism, and Floristic Quality index scores.

Field Name	Description
seg_hash	Site identifier
S	Native species richness
Si	Total species richness
C	Native species mean Coefficient of Conservatism
Ci	Total species mean Coefficient of Conservatism
wC	Native weighted Coefficient of Conservatism
wCi	Total weighted Coefficient of Conservatism
FQI	Native Floristic Quality Index
FQIi	Total Floristic Quality Index
wFQI	Native Weighted Floristic Quality Index
wFQIi	Total Weighted Floristic Quality Index

Appendix 5. Site diversity index scores.

Field Name	Description
seg_hash	Site identifier
H	Shannon diversity index
Ds	Simpson dominance index
Hrel	Shannon evenness index
m_plot_Dmg	Mean quadrat Margalef diversity index

Appendix 6. Site stressor data.

Field Name	Description
seg_hash	Site identifier
ac1	Agriculture principal component 1
pc1	Population Density principal component 1
hydr_mod	Local hydrologic modification

Appendix 2. Study site information.

Site Name	seg_hash	Lake	Geomorphic Type	State	dd_east	dd_north	Year Sampled
St. Louis River	56001	Superior	Rw	WI	-92.2737	46.6499	2002
Dwights Point	59004	Superior	Rw	WI	-92.1456	46.6757	2002
Kimball's Bay	59005	Superior	Rw	WI	-92.1456	46.6935	2001
Hog Island	61004	Superior	Cw	WI	-92.0368	46.7029	2001
Nemadji River	61005	Superior	Rw	WI	-92.0332	46.6944	2001
Allouez Bay	63004	Superior	Pw	WI	-91.99	46.6817	2001
Amnicon River	66003	Superior	Rw	WI	-91.8615	46.6884	2001
Middle River	68006	Superior	Rw	WI	-91.8271	46.688	2001
Brule River	78003	Superior	Rw	WI	-91.6095	46.7438	2001
Flag River	84004	Superior	Pw	WI	-91.3786	46.7895	2001
Cranberry River	85002	Superior	Rw	WI	-91.267	46.8277	2001
Bark Bay	87004	Superior	Pw	WI	-91.1895	46.8498	2001
Lost Creek	88001	Superior	Pw	WI	-91.1346	46.8555	2001
Pike Creek	96004	Superior	Rw	WI	-90.8617	46.7888	2001
Sioux River	98002	Superior	Rw	WI	-90.8831	46.7335	2001
Sioux River	98006	Superior	Pw	WI	-90.8825	46.7458	2001
Three Highway	103004	Superior	Cw	WI	-90.9433	46.5893	2001
Fish Creek East	104004	Superior	Rw	WI	-90.942	46.5862	2001
Fish Creek West	105006	Superior	Pw	WI	-90.9335	46.583	2001
Sand Cut Slough	109003	Superior	Cw	WI	-90.7428	46.6643	2003
Bad River	110004	Superior	Pw	WI	-90.6262	46.6201	2003
Baraga	192006	Superior	Rw	MI	-88.4907	46.7521	2002
Peterson Creek	200001	Superior	Pw	MI	-88.1938	46.8963	2002
Au Train	215001	Superior	Rw	MI	-86.8268	46.43	2002
Tahquamenon River	229001	Superior	Pw	MI	-85.0441	46.5572	2002
Parker Creek	242001	Huron	Cw	MI	-84.262	46.1945	2002
McKay Creek	247001	Huron	Cw	MI	-84.3167	45.9948	2002
McKay Creek	247002	Huron	Cw	MI	-84.4127	46.0028	2002
Pine River 1	248001	Huron	Cw	MI	-84.4665	46.009	2002
Sturgeon River	274001	Michigan	Pw	MI	-86.6897	45.8459	2002
Rapid River	278004	Michigan	Cw	MI	-86.961	45.9176	2003
Ford River	281001	Michigan	Rw	MI	-87.1442	45.6743	2002
Peshtigo River	291001	Michigan	Rw	WI	-87.6525	44.9834	2001
Oconto River	294001	Michigan	Cw	WI	-87.8416	44.8844	2002

Little Suamico River	299001	Michigan	Cw	WI	-87.9929	44.7055	2002
Little Tail Point	300001	Michigan	Cw	WI	-88.0012	44.6724	2002
Saumico River	302001	Michigan	Cw	WI	-88.0123	44.6276	2001
Saumico River	302011	Michigan	Cw	WI	-88.0142	44.6377	2002
Dead Horse Bay	303002	Michigan	Cw	WI	-88.0239	44.5917	2001
Howard	304001	Michigan	Cw	WI	-88.0393	44.5644	2002
Point Sable Wetland	306004	Michigan	Pw	WI	-87.903	44.58	2001
Toft Point	313003	Michigan	Pw	WI	-87.083	45.0655	2001
Three Springs Creek	313011	Michigan	Pw	WI	-87.0944	45.0809	2001
Ahnappe River	319001	Michigan	Rw	WI	-87.4506	44.6227	2001
Kewaunee River	321002	Michigan	Rw	WI	-87.5101	44.4733	2001
West Twin River	325003	Michigan	Rw	WI	-87.5792	44.1749	2002
Big Sable	389001	Michigan	Rw	MI	-86.3665	44.0849	2002
Arcadia Lake	393005	Michigan	Pw	MI	-86.223	44.4868	2002
Crossville	407001	Michigan	Cw	MI	-84.9846	45.7527	2002
Cecil Bay	408001	Michigan	Pw	MI	-84.8202	45.7484	2002
Cheboygan River	411001	Huron	Pw	MI	-84.4255	45.6632	2002
Long Luke Creek	421001	Huron	Cw	MI	-83.3136	45.071	2002
Thunder Bay River	422001	Huron	Rw	MI	-83.4576	45.0687	2002
Devils Lake	423001	Huron	Pw	MI	-83.4641	44.9975	2002
Great Lakes Marina	442003	Huron	Cw	MI	-83.8838	43.9416	2003

Appendix 3. Nested quadrat data.

Plot	Sub-Plot	Type	Area	sci_name
1	A	Marsh	0.25	Carex lasiocarpa var. americana Fern.
1	A	Marsh	0.25	Carex rostrata Stokes
1	A	Marsh	0.25	Sagittaria latifolia Willd.
1	A	Marsh	0.25	Schoenoplectus fluviatilis (Torr.) M.T. Strong
1	A	Marsh	0.25	Sparganium eurycarpum Engelm.
1	B	Marsh	1	Carex lasiocarpa var. americana Fern.
1	B	Marsh	1	Carex rostrata Stokes
1	B	Marsh	1	Ceratophyllum demersum L.
1	B	Marsh	1	Eleocharis palustris L.
1	B	Marsh	1	Glyceria borealis (Nash) Batch.
1	B	Marsh	1	Sagittaria latifolia Willd.
1	B	Marsh	1	Schoenoplectus fluviatilis (Torr.) M.T. Strong
1	B	Marsh	1	Sparganium eurycarpum Engelm.
1	B	Marsh	1	Utricularia macrorhiza Le Conte
1	C	Marsh	2	Carex lasiocarpa var. americana Fern.
1	C	Marsh	2	Carex rostrata Stokes
1	C	Marsh	2	Ceratophyllum demersum L.
1	C	Marsh	2	Eleocharis palustris L.
1	C	Marsh	2	Glyceria borealis (Nash) Batch.
1	C	Marsh	2	Sagittaria latifolia Willd.
1	C	Marsh	2	Schoenoplectus fluviatilis (Torr.) M.T. Strong
1	C	Marsh	2	Sparganium eurycarpum Engelm.
1	C	Marsh	2	Utricularia macrorhiza Le Conte
1	D	Marsh	4	Carex lasiocarpa var. americana Fern.
1	D	Marsh	4	Carex rostrata Stokes
1	D	Marsh	4	Ceratophyllum demersum L.
1	D	Marsh	4	Eleocharis palustris L.
1	D	Marsh	4	Glyceria borealis (Nash) Batch.
1	D	Marsh	4	Sagittaria latifolia Willd.
1	D	Marsh	4	Schoenoplectus fluviatilis (Torr.) M.T. Strong
1	D	Marsh	4	Sparganium eurycarpum Engelm.
1	D	Marsh	4	Utricularia macrorhiza Le Conte
1	E	Marsh	8	Carex lasiocarpa var. americana Fern.
1	E	Marsh	8	Carex rostrata Stokes
1	E	Marsh	8	Ceratophyllum demersum L.
1	E	Marsh	8	Comarum palustre L.
1	E	Marsh	8	Eleocharis palustris L.
1	E	Marsh	8	Glyceria borealis (Nash) Batch.
1	E	Marsh	8	Sagittaria latifolia Willd.
1	E	Marsh	8	Schoenoplectus fluviatilis (Torr.) M.T. Strong
1	E	Marsh	8	Sium suave Walt.
1	E	Marsh	8	Sparganium eurycarpum Engelm.
1	E	Marsh	8	Utricularia macrorhiza Le Conte
1	F	Marsh	16	Carex lasiocarpa var. americana Fern.
1	F	Marsh	16	Carex rostrata Stokes
1	F	Marsh	16	Ceratophyllum demersum L.
1	F	Marsh	16	Comarum palustre L.
1	F	Marsh	16	Eleocharis palustris L.
1	F	Marsh	16	Equisetum fluviatile L.
1	F	Marsh	16	Glyceria borealis (Nash) Batch.
1	F	Marsh	16	Sagittaria latifolia Willd.
1	F	Marsh	16	Schoenoplectus fluviatilis (Torr.) M.T. Strong
1	F	Marsh	16	Sium suave Walt.
1	F	Marsh	16	Sparganium eurycarpum Engelm.
1	F	Marsh	16	Utricularia macrorhiza Le Conte
2	A	Marsh	0.25	Carex rostrata Stokes
2	A	Marsh	0.25	Potamogeton strictifolius Benn.

2	A	Marsh	0.25	Sagittaria latifolia Willd.
2	A	Marsh	0.25	Schoenoplectus subterminalis (Torr.) Sojak
2	A	Marsh	0.25	Sium suave Walt.
2	A	Marsh	0.25	Sparganium eurycarpum Engelm.
2	A	Marsh	0.25	Typha latifolia L.
2	A	Marsh	0.25	Utricularia macrorhiza Le Conte
2	A	Marsh	0.25	Zizania aquatica L.
2	B	Marsh	1	Campanula aparinoides Pursh
2	B	Marsh	1	Carex rostrata Stokes
2	B	Marsh	1	Cicuta bulbifera L.
2	B	Marsh	1	Potamogeton strictifolius Benn.
2	B	Marsh	1	Sagittaria latifolia Willd.
2	B	Marsh	1	Schoenoplectus subterminalis (Torr.) Sojak
2	B	Marsh	1	Sium suave Walt.
2	B	Marsh	1	Sparganium eurycarpum Engelm.
2	B	Marsh	1	Typha latifolia L.
2	B	Marsh	1	Utricularia macrorhiza Le Conte
2	B	Marsh	1	Zizania aquatica L.
2	C	Marsh	2	Campanula aparinoides Pursh
2	C	Marsh	2	Carex lacustris Willd.
2	C	Marsh	2	Carex rostrata Stokes
2	C	Marsh	2	Cicuta bulbifera L.
2	C	Marsh	2	Potamogeton strictifolius Benn.
2	C	Marsh	2	Sagittaria latifolia Willd.
2	C	Marsh	2	Schoenoplectus subterminalis (Torr.) Sojak
2	C	Marsh	2	Sium suave Walt.
2	C	Marsh	2	Sparganium eurycarpum Engelm.
2	C	Marsh	2	Typha latifolia L.
2	C	Marsh	2	Utricularia macrorhiza Le Conte
2	C	Marsh	2	Zizania aquatica L.
2	D	Marsh	4	Campanula aparinoides Pursh
2	D	Marsh	4	Carex lacustris Willd.
2	D	Marsh	4	Carex lasiocarpa var. americana Fern.
2	D	Marsh	4	Carex rostrata Stokes
2	D	Marsh	4	Cicuta bulbifera L.
2	D	Marsh	4	Potamogeton strictifolius Benn.
2	D	Marsh	4	Sagittaria latifolia Willd.
2	D	Marsh	4	Schoenoplectus subterminalis (Torr.) Sojak
2	D	Marsh	4	Sium suave Walt.
2	D	Marsh	4	Sparganium eurycarpum Engelm.
2	D	Marsh	4	Typha latifolia L.
2	D	Marsh	4	Utricularia macrorhiza Le Conte
2	D	Marsh	4	Zizania aquatica L.
2	E	Marsh	8	Campanula aparinoides Pursh
2	E	Marsh	8	Carex lacustris Willd.
2	E	Marsh	8	Carex lasiocarpa var. americana Fern.
2	E	Marsh	8	Carex rostrata Stokes
2	E	Marsh	8	Ceratophyllum demersum L.
2	E	Marsh	8	Cicuta bulbifera L.
2	E	Marsh	8	Eleocharis palustris L.
2	E	Marsh	8	Equisetum fluviatile L.
2	E	Marsh	8	Potamogeton strictifolius Benn.
2	E	Marsh	8	Sagittaria latifolia Willd.
2	E	Marsh	8	Schoenoplectus subterminalis (Torr.) Sojak
2	E	Marsh	8	Sium suave Walt.
2	E	Marsh	8	Sparganium eurycarpum Engelm.
2	E	Marsh	8	Typha latifolia L.
2	E	Marsh	8	Utricularia macrorhiza Le Conte
2	E	Marsh	8	Zizania aquatica L.
2	F	Marsh	16	Campanula aparinoides Pursh
2	F	Marsh	16	Carex lacustris Willd.

2	F	Marsh	16	Carex lasiocarpa var. americana Fern.
2	F	Marsh	16	Carex rostrata Stokes
2	F	Marsh	16	Ceratophyllum demersum L.
2	F	Marsh	16	Cicuta bulbifera L.
2	F	Marsh	16	Eleocharis palustris L.
2	F	Marsh	16	Equisetum fluviatile L.
2	F	Marsh	16	Potamogeton strictifolius Benn.
2	F	Marsh	16	Sagittaria latifolia Willd.
2	F	Marsh	16	Schoenoplectus subterminalis (Torr.) Sojak
2	F	Marsh	16	Sium suave Walt.
2	F	Marsh	16	Sparganium eurycarpum Engelm.
2	F	Marsh	16	Typha latifolia L.
2	F	Marsh	16	Utricularia macrorhiza Le Conte
2	F	Marsh	16	Zizania aquatica L.
3	A	Marsh	0.25	Acorus calamus L.
3	A	Marsh	0.25	Campanula aparinoides Pursh
3	A	Marsh	0.25	Carex lasiocarpa var. americana Fern.
3	A	Marsh	0.25	Carex utriculata F. Boott
3	A	Marsh	0.25	Comarum palustre L.
3	A	Marsh	0.25	Equisetum fluviatile L.
3	A	Marsh	0.25	Impatiens capensis Meerb.
3	A	Marsh	0.25	Iris versicolor L.
3	A	Marsh	0.25	Lysimachia thyrsiflora L.
3	A	Marsh	0.25	Typha X glauca (T. angustifolia X T. latifolia)
3	B	Marsh	1	Acorus calamus L.
3	B	Marsh	1	Campanula aparinoides Pursh
3	B	Marsh	1	Carex lasiocarpa var. americana Fern.
3	B	Marsh	1	Carex utriculata F. Boott
3	B	Marsh	1	Comarum palustre L.
3	B	Marsh	1	Equisetum fluviatile L.
3	B	Marsh	1	Galium trifidum L.
3	B	Marsh	1	Impatiens capensis Meerb.
3	B	Marsh	1	Iris versicolor L.
3	B	Marsh	1	Lysimachia thyrsiflora L.
3	B	Marsh	1	Sparganium eurycarpum Engelm.
3	B	Marsh	1	Typha X glauca (T. angustifolia X T. latifolia)
3	C	Marsh	2	Acorus calamus L.
3	C	Marsh	2	Campanula aparinoides Pursh
3	C	Marsh	2	Carex lasiocarpa var. americana Fern.
3	C	Marsh	2	Carex utriculata F. Boott
3	C	Marsh	2	Comarum palustre L.
3	C	Marsh	2	Equisetum fluviatile L.
3	C	Marsh	2	Galium trifidum L.
3	C	Marsh	2	Impatiens capensis Meerb.
3	C	Marsh	2	Iris versicolor L.
3	C	Marsh	2	Lysimachia thyrsiflora L.
3	C	Marsh	2	Lythrum salicaria L.
3	C	Marsh	2	Sparganium eurycarpum Engelm.
3	C	Marsh	2	Typha X glauca (T. angustifolia X T. latifolia)
3	D	Marsh	4	Acorus calamus L.
3	D	Marsh	4	Campanula aparinoides Pursh
3	D	Marsh	4	Carex lasiocarpa var. americana Fern.
3	D	Marsh	4	Carex utriculata F. Boott
3	D	Marsh	4	Comarum palustre L.
3	D	Marsh	4	Equisetum fluviatile L.
3	D	Marsh	4	Galium trifidum L.
3	D	Marsh	4	Impatiens capensis Meerb.
3	D	Marsh	4	Iris versicolor L.
3	D	Marsh	4	Lysimachia thyrsiflora L.
3	D	Marsh	4	Lythrum salicaria L.
3	D	Marsh	4	Rumex orbiculatus Gray

3	D	Marsh	4	Sagittaria latifolia Willd.
3	D	Marsh	4	Sparganium eurycarpum Engelm.
3	D	Marsh	4	Typha X glauca (T. angustifolia X T. latifolia)
3	E	Marsh	8	Acorus calamus L.
3	E	Marsh	8	Campanula aparinoides Pursh
3	E	Marsh	8	Carex lasiocarpa var. americana Fern.
3	E	Marsh	8	Carex utriculata F. Boott
3	E	Marsh	8	Cicuta bulbifera L.
3	E	Marsh	8	Comarum palustre L.
3	E	Marsh	8	Equisetum fluviatile L.
3	E	Marsh	8	Galium trifidum L.
3	E	Marsh	8	Impatiens capensis Meerb.
3	E	Marsh	8	Iris versicolor L.
3	E	Marsh	8	Lysimachia thyrsiflora L.
3	E	Marsh	8	Lythrum salicaria L.
3	E	Marsh	8	Rumex orbiculatus Gray
3	E	Marsh	8	Sagittaria latifolia Willd.
3	E	Marsh	8	Sparganium eurycarpum Engelm.
3	E	Marsh	8	Typha X glauca (T. angustifolia X T. latifolia)
3	F	Marsh	16	Acorus calamus L.
3	F	Marsh	16	Campanula aparinoides Pursh
3	F	Marsh	16	Carex lasiocarpa var. americana Fern.
3	F	Marsh	16	Carex utriculata F. Boott
3	F	Marsh	16	Cicuta bulbifera L.
3	F	Marsh	16	Comarum palustre L.
3	F	Marsh	16	Equisetum fluviatile L.
3	F	Marsh	16	Galium trifidum L.
3	F	Marsh	16	Impatiens capensis Meerb.
3	F	Marsh	16	Iris versicolor L.
3	F	Marsh	16	Lysimachia thyrsiflora L.
3	F	Marsh	16	Lythrum salicaria L.
3	F	Marsh	16	Rumex orbiculatus Gray
3	F	Marsh	16	Sagittaria latifolia Willd.
3	F	Marsh	16	Scirpus cyperinus (L.) Kunth
3	F	Marsh	16	Sparganium eurycarpum Engelm.
3	F	Marsh	16	Typha X glauca (T. angustifolia X T. latifolia)
4	A	Wet meadow	0.25	Calamagrostis canadensis (Michx.) Nutt.
4	A	Wet meadow	0.25	Campanula aparinoides Pursh
4	A	Wet meadow	0.25	Carex lacustris Willd.
4	A	Wet meadow	0.25	Carex stricta Lam.
4	A	Wet meadow	0.25	Cicuta bulbifera L.
4	A	Wet meadow	0.25	Impatiens capensis Meerb.
4	A	Wet meadow	0.25	Lycopus uniflorus Michx.
4	A	Wet meadow	0.25	Triadenum fraseri (Spach) Gl.
4	B	Wet meadow	1	Calamagrostis canadensis (Michx.) Nutt.
4	B	Wet meadow	1	Campanula aparinoides Pursh
4	B	Wet meadow	1	Carex lacustris Willd.
4	B	Wet meadow	1	Carex stricta Lam.
4	B	Wet meadow	1	Cicuta bulbifera L.
4	B	Wet meadow	1	Eupatorium maculatum L.
4	B	Wet meadow	1	Impatiens capensis Meerb.
4	B	Wet meadow	1	Lycopus uniflorus Michx.
4	B	Wet meadow	1	Lysimachia thyrsiflora L.
4	B	Wet meadow	1	Poa palustris L.
4	B	Wet meadow	1	Rumex orbiculatus Gray
4	B	Wet meadow	1	Scutellaria galericulata L.
4	B	Wet meadow	1	Spiraea alba Du Roi
4	B	Wet meadow	1	Triadenum fraseri (Spach) Gl.
4	C	Wet meadow	2	Alnus incana ssp. rugosa (Du Roi) Clausen
4	C	Wet meadow	2	Calamagrostis canadensis (Michx.) Nutt.
4	C	Wet meadow	2	Campanula aparinoides Pursh

4	C	Wet meadow	2	Carex lacustris Willd.
4	C	Wet meadow	2	Carex stricta Lam.
4	C	Wet meadow	2	Cicuta bulbifera L.
4	C	Wet meadow	2	Eupatorium maculatum L.
4	C	Wet meadow	2	Galium trifidum L.
4	C	Wet meadow	2	Impatiens capensis Meerb.
4	C	Wet meadow	2	Lycopus uniflorus Michx.
4	C	Wet meadow	2	Lysimachia thyrsiflora L.
4	C	Wet meadow	2	Poa palustris L.
4	C	Wet meadow	2	Rumex orbiculatus Gray
4	C	Wet meadow	2	Scutellaria galericulata L.
4	C	Wet meadow	2	Spiraea alba Du Roi
4	C	Wet meadow	2	Triadenum fraseri (Spach) Gl.
4	D	Wet meadow	4	Alnus incana ssp. rugosa (Du Roi) Clausen
4	D	Wet meadow	4	Calamagrostis canadensis (Michx.) Nutt.
4	D	Wet meadow	4	Campanula aparinoides Pursh
4	D	Wet meadow	4	Carex lacustris Willd.
4	D	Wet meadow	4	Carex stricta Lam.
4	D	Wet meadow	4	Cicuta bulbifera L.
4	D	Wet meadow	4	Eupatorium maculatum L.
4	D	Wet meadow	4	Galium trifidum L.
4	D	Wet meadow	4	Impatiens capensis Meerb.
4	D	Wet meadow	4	Iris versicolor L.
4	D	Wet meadow	4	Lycopus uniflorus Michx.
4	D	Wet meadow	4	Lysimachia thyrsiflora L.
4	D	Wet meadow	4	Lythrum salicaria L.
4	D	Wet meadow	4	Mentha arvensis L.
4	D	Wet meadow	4	Poa palustris L.
4	D	Wet meadow	4	Rumex orbiculatus Gray
4	D	Wet meadow	4	Scutellaria galericulata L.
4	D	Wet meadow	4	Spiraea alba Du Roi
4	D	Wet meadow	4	Triadenum fraseri (Spach) Gl.
4	D	Wet meadow	4	Typha latifolia L.
4	E	Wet meadow	8	Alnus incana ssp. rugosa (Du Roi) Clausen
4	E	Wet meadow	8	Calamagrostis canadensis (Michx.) Nutt.
4	E	Wet meadow	8	Campanula aparinoides Pursh
4	E	Wet meadow	8	Carex lacustris Willd.
4	E	Wet meadow	8	Carex stricta Lam.
4	E	Wet meadow	8	Cicuta bulbifera L.
4	E	Wet meadow	8	Cirsium arvense (L.) Scop.
4	E	Wet meadow	8	Eupatorium maculatum L.
4	E	Wet meadow	8	Galium trifidum L.
4	E	Wet meadow	8	Impatiens capensis Meerb.
4	E	Wet meadow	8	Iris versicolor L.
4	E	Wet meadow	8	Lycopus uniflorus Michx.
4	E	Wet meadow	8	Lysimachia thyrsiflora L.
4	E	Wet meadow	8	Lythrum salicaria L.
4	E	Wet meadow	8	Mentha arvensis L.
4	E	Wet meadow	8	Poa palustris L.
4	E	Wet meadow	8	Potentilla norvegica L.
4	E	Wet meadow	8	Rubus idaeus ssp. strigosus (Michx.) Focke
4	E	Wet meadow	8	Rumex orbiculatus Gray
4	E	Wet meadow	8	Scutellaria galericulata L.
4	E	Wet meadow	8	Spiraea alba Du Roi
4	E	Wet meadow	8	Triadenum fraseri (Spach) Gl.
4	E	Wet meadow	8	Typha latifolia L.
4	F	Wet meadow	16	Alnus incana ssp. rugosa (Du Roi) Clausen
4	F	Wet meadow	16	Calamagrostis canadensis (Michx.) Nutt.
4	F	Wet meadow	16	Calla palustris L.
4	F	Wet meadow	16	Campanula aparinoides Pursh
4	F	Wet meadow	16	Carex lacustris Willd.

4	F	Wet meadow	16	Carex stricta Lam.
4	F	Wet meadow	16	Cicuta bulbifera L.
4	F	Wet meadow	16	Cirsium arvense (L.) Scop.
4	F	Wet meadow	16	Eupatorium maculatum L.
4	F	Wet meadow	16	Galium trifidum L.
4	F	Wet meadow	16	Geum aleppicum Jacq.
4	F	Wet meadow	16	Impatiens capensis Meerb.
4	F	Wet meadow	16	Iris versicolor L.
4	F	Wet meadow	16	Lycopus uniflorus Michx.
4	F	Wet meadow	16	Lysimachia thyrsiflora L.
4	F	Wet meadow	16	Lythrum salicaria L.
4	F	Wet meadow	16	Mentha arvensis L.
4	F	Wet meadow	16	Poa palustris L.
4	F	Wet meadow	16	Potentilla norvegica L.
4	F	Wet meadow	16	Rubus idaeus ssp. strigosus (Michx.) Focke
4	F	Wet meadow	16	Rumex orbiculatus Gray
4	F	Wet meadow	16	Scutellaria galericulata L.
4	F	Wet meadow	16	Spiraea alba Du Roi
4	F	Wet meadow	16	Thalictrum dasycarpum Fisch. & Lall.
4	F	Wet meadow	16	Triadenum fraseri (Spach) Gl.
4	F	Wet meadow	16	Typha latifolia L.
5	A	Wet meadow	0.25	Calamagrostis canadensis (Michx.) Nutt.
5	A	Wet meadow	0.25	Carex lacustris Willd.
5	A	Wet meadow	0.25	Carex stricta Lam.
5	A	Wet meadow	0.25	Cirsium arvense (L.) Scop.
5	A	Wet meadow	0.25	Lysimachia thyrsiflora L.
5	A	Wet meadow	0.25	Typha latifolia L.
5	B	Wet meadow	1	Calamagrostis canadensis (Michx.) Nutt.
5	B	Wet meadow	1	Carex lacustris Willd.
5	B	Wet meadow	1	Carex stricta Lam.
5	B	Wet meadow	1	Cirsium arvense (L.) Scop.
5	B	Wet meadow	1	Cornus sericea ssp. sericea L.
5	B	Wet meadow	1	Lysimachia thyrsiflora L.
5	B	Wet meadow	1	Typha latifolia L.
5	C	Wet meadow	2	Calamagrostis canadensis (Michx.) Nutt.
5	C	Wet meadow	2	Carex lacustris Willd.
5	C	Wet meadow	2	Carex stricta Lam.
5	C	Wet meadow	2	Cirsium arvense (L.) Scop.
5	C	Wet meadow	2	Cornus sericea ssp. sericea L.
5	C	Wet meadow	2	Lysimachia thyrsiflora L.
5	C	Wet meadow	2	Typha latifolia L.
5	D	Wet meadow	4	Calamagrostis canadensis (Michx.) Nutt.
5	D	Wet meadow	4	Carex lacustris Willd.
5	D	Wet meadow	4	Carex stricta Lam.
5	D	Wet meadow	4	Cirsium arvense (L.) Scop.
5	D	Wet meadow	4	Cornus sericea ssp. sericea L.
5	D	Wet meadow	4	Lysimachia thyrsiflora L.
5	D	Wet meadow	4	Onoclea sensibilis L.
5	D	Wet meadow	4	Rumex orbiculatus Gray
5	D	Wet meadow	4	Scutellaria galericulata L.
5	D	Wet meadow	4	Typha latifolia L.
5	E	Wet meadow	8	Calamagrostis canadensis (Michx.) Nutt.
5	E	Wet meadow	8	Carex lacustris Willd.
5	E	Wet meadow	8	Carex stricta Lam.
5	E	Wet meadow	8	Cirsium arvense (L.) Scop.
5	E	Wet meadow	8	Cornus sericea ssp. sericea L.
5	E	Wet meadow	8	Lysimachia thyrsiflora L.
5	E	Wet meadow	8	Lythrum salicaria L.
5	E	Wet meadow	8	Onoclea sensibilis L.
5	E	Wet meadow	8	Rumex orbiculatus Gray
5	E	Wet meadow	8	Scutellaria galericulata L.

5	E	Wet meadow	8	Typha latifolia L.
5	F	Wet meadow	16	Alnus incana ssp. rugosa (Du Roi) Clausen
5	F	Wet meadow	16	Calamagrostis canadensis (Michx.) Nutt.
5	F	Wet meadow	16	Carex lacustris Willd.
5	F	Wet meadow	16	Carex stricta Lam.
5	F	Wet meadow	16	Cirsium arvense (L.) Scop.
5	F	Wet meadow	16	Cornus sericea ssp. sericea L.
5	F	Wet meadow	16	Lysimachia thyrsiflora L.
5	F	Wet meadow	16	Lythrum salicaria L.
5	F	Wet meadow	16	Onoclea sensibilis L.
5	F	Wet meadow	16	Ribes sp.
5	F	Wet meadow	16	Rumex orbiculatus Gray
5	F	Wet meadow	16	Scutellaria galericulata L.
5	F	Wet meadow	16	Typha latifolia L.
5	F	Wet meadow	16	Urtica dioica L.
6	A	Wet meadow	0.25	Calamagrostis canadensis (Michx.) Nutt.
6	A	Wet meadow	0.25	Carex stricta Lam.
6	A	Wet meadow	0.25	Cirsium arvense (L.) Scop.
6	A	Wet meadow	0.25	Rubus idaeus ssp. strigosus (Michx.) Focke
6	A	Wet meadow	0.25	Scutellaria galericulata L.
6	B	Wet meadow	1	Calamagrostis canadensis (Michx.) Nutt.
6	B	Wet meadow	1	Carex stricta Lam.
6	B	Wet meadow	1	Cirsium arvense (L.) Scop.
6	B	Wet meadow	1	Dryopteris cristata (L.) Gray
6	B	Wet meadow	1	Impatiens capensis Meerb.
6	B	Wet meadow	1	Polygonum hydropiperoides Michx.
6	B	Wet meadow	1	Rubus idaeus ssp. strigosus (Michx.) Focke
6	B	Wet meadow	1	Scutellaria galericulata L.
6	B	Wet meadow	1	Spiraea alba Du Roi
6	C	Wet meadow	2	Calamagrostis canadensis (Michx.) Nutt.
6	C	Wet meadow	2	Carex lacustris Willd.
6	C	Wet meadow	2	Carex stricta Lam.
6	C	Wet meadow	2	Cirsium arvense (L.) Scop.
6	C	Wet meadow	2	Dryopteris cristata (L.) Gray
6	C	Wet meadow	2	Impatiens capensis Meerb.
6	C	Wet meadow	2	Polygonum hydropiperoides Michx.
6	C	Wet meadow	2	Rubus idaeus ssp. strigosus (Michx.) Focke
6	C	Wet meadow	2	Scutellaria galericulata L.
6	C	Wet meadow	2	Spiraea alba Du Roi
6	D	Wet meadow	4	Calamagrostis canadensis (Michx.) Nutt.
6	D	Wet meadow	4	Carex lacustris Willd.
6	D	Wet meadow	4	Carex stricta Lam.
6	D	Wet meadow	4	Cirsium arvense (L.) Scop.
6	D	Wet meadow	4	Dryopteris cristata (L.) Gray
6	D	Wet meadow	4	Impatiens capensis Meerb.
6	D	Wet meadow	4	Polygonum hydropiperoides Michx.
6	D	Wet meadow	4	Rubus idaeus ssp. strigosus (Michx.) Focke
6	D	Wet meadow	4	Scutellaria galericulata L.
6	D	Wet meadow	4	Spiraea alba Du Roi
6	E	Wet meadow	8	Calamagrostis canadensis (Michx.) Nutt.
6	E	Wet meadow	8	Carex lacustris Willd.
6	E	Wet meadow	8	Carex stricta Lam.
6	E	Wet meadow	8	Cirsium arvense (L.) Scop.
6	E	Wet meadow	8	Dryopteris cristata (L.) Gray
6	E	Wet meadow	8	Impatiens capensis Meerb.
6	E	Wet meadow	8	Lysimachia thyrsiflora L.
6	E	Wet meadow	8	Polygonum hydropiperoides Michx.
6	E	Wet meadow	8	Rubus idaeus ssp. strigosus (Michx.) Focke
6	E	Wet meadow	8	Scutellaria galericulata L.
6	E	Wet meadow	8	Spiraea alba Du Roi
6	F	Wet meadow	16	Calamagrostis canadensis (Michx.) Nutt.

6	F	Wet meadow	16	Campanula aparinoides Pursh
6	F	Wet meadow	16	Carex lacustris Willd.
6	F	Wet meadow	16	Carex stricta Lam.
6	F	Wet meadow	16	Cirsium arvense (L.) Scop.
6	F	Wet meadow	16	Dryopteris cristata (L.) Gray
6	F	Wet meadow	16	Impatiens capensis Meerb.
6	F	Wet meadow	16	Lycopus uniflorus Michx.
6	F	Wet meadow	16	Lysimachia thyrsiflora L.
6	F	Wet meadow	16	Poa palustris L.
6	F	Wet meadow	16	Polygonum hydropiperoides Michx.
6	F	Wet meadow	16	Ribes hirtellum Michx.
6	F	Wet meadow	16	Rubus idaeus ssp. strigosus (Michx.) Focke
6	F	Wet meadow	16	Rumex orbiculatus Gray
6	F	Wet meadow	16	Scutellaria galericulata L.
6	F	Wet meadow	16	Spiraea alba Du Roi
7	A	Fen	0.25	Campanula aparinoides Pursh
7	A	Fen	0.25	Carex lasiocarpa var. americana Fern.
7	A	Fen	0.25	Cicuta bulbifera L.
7	A	Fen	0.25	Equisetum fluviatile L.
7	A	Fen	0.25	Lysimachia thyrsiflora L.
7	A	Fen	0.25	Menyanthes trifoliata L.
7	B	Fen	1	Campanula aparinoides Pursh
7	B	Fen	1	Carex lasiocarpa var. americana Fern.
7	B	Fen	1	Cicuta bulbifera L.
7	B	Fen	1	Equisetum fluviatile L.
7	B	Fen	1	Galium trifidum L.
7	B	Fen	1	Lysimachia thyrsiflora L.
7	B	Fen	1	Menyanthes trifoliata L.
7	B	Fen	1	Rumex orbiculatus Gray
7	B	Fen	1	Sium suave Walt.
7	C	Fen	2	Campanula aparinoides Pursh
7	C	Fen	2	Carex lasiocarpa var. americana Fern.
7	C	Fen	2	Cicuta bulbifera L.
7	C	Fen	2	Equisetum fluviatile L.
7	C	Fen	2	Eriophorum gracile Koch
7	C	Fen	2	Galium trifidum L.
7	C	Fen	2	Iris versicolor L.
7	C	Fen	2	Lysimachia thyrsiflora L.
7	C	Fen	2	Menyanthes trifoliata L.
7	C	Fen	2	Rumex orbiculatus Gray
7	C	Fen	2	Sium suave Walt.
7	C	Fen	2	Typha latifolia L.
7	D	Fen	4	Campanula aparinoides Pursh
7	D	Fen	4	Carex lasiocarpa var. americana Fern.
7	D	Fen	4	Carex limosa L.
7	D	Fen	4	Cicuta bulbifera L.
7	D	Fen	4	Equisetum fluviatile L.
7	D	Fen	4	Eriophorum gracile Koch
7	D	Fen	4	Galium trifidum L.
7	D	Fen	4	Iris versicolor L.
7	D	Fen	4	Lysimachia thyrsiflora L.
7	D	Fen	4	Menyanthes trifoliata L.
7	D	Fen	4	Rumex orbiculatus Gray
7	D	Fen	4	Sium suave Walt.
7	D	Fen	4	Typha latifolia L.
7	E	Fen	8	Campanula aparinoides Pursh
7	E	Fen	8	Carex lasiocarpa var. americana Fern.
7	E	Fen	8	Carex limosa L.
7	E	Fen	8	Cicuta bulbifera L.
7	E	Fen	8	Equisetum fluviatile L.
7	E	Fen	8	Eriophorum gracile Koch

7	E	Fen	8	Galium trifidum L.
7	E	Fen	8	Iris versicolor L.
7	E	Fen	8	Lysimachia thyrsiflora L.
7	E	Fen	8	Lythrum salicaria L.
7	E	Fen	8	Menyanthes trifoliata L.
7	E	Fen	8	Rumex orbiculatus Gray
7	E	Fen	8	Sium suave Walt.
7	E	Fen	8	Typha latifolia L.
7	F	Fen	16	Campanula aparinoides Pursh
7	F	Fen	16	Carex lasiocarpa var. americana Fern.
7	F	Fen	16	Carex limosa L.
7	F	Fen	16	Cicuta bulbifera L.
7	F	Fen	16	Equisetum fluviatile L.
7	F	Fen	16	Eriophorum gracile Koch
7	F	Fen	16	Galium trifidum L.
7	F	Fen	16	Iris versicolor L.
7	F	Fen	16	Lysimachia thyrsiflora L.
7	F	Fen	16	Lythrum salicaria L.
7	F	Fen	16	Menyanthes trifoliata L.
7	F	Fen	16	Rumex orbiculatus Gray
7	F	Fen	16	Sium suave Walt.
7	F	Fen	16	Typha latifolia L.
8	A	Fen	0.25	Calla palustris L.
8	A	Fen	0.25	Campanula aparinoides Pursh
8	A	Fen	0.25	Carex lacustris Willd.
8	A	Fen	0.25	Carex lasiocarpa var. americana Fern.
8	A	Fen	0.25	Cicuta bulbifera L.
8	A	Fen	0.25	Eleocharis palustris L.
8	A	Fen	0.25	Galium trifidum L.
8	A	Fen	0.25	Lysimachia thyrsiflora L.
8	A	Fen	0.25	Sagittaria latifolia Willd.
8	A	Fen	0.25	Sium suave Walt.
8	A	Fen	0.25	Sparganium eurycarpum Engelm.
8	A	Fen	0.25	Utricularia intermedia Hayne
8	B	Fen	1	Calla palustris L.
8	B	Fen	1	Campanula aparinoides Pursh
8	B	Fen	1	Carex lacustris Willd.
8	B	Fen	1	Carex lasiocarpa var. americana Fern.
8	B	Fen	1	Cicuta bulbifera L.
8	B	Fen	1	Comarum palustre L.
8	B	Fen	1	Eleocharis palustris L.
8	B	Fen	1	Equisetum fluviatile L.
8	B	Fen	1	Eriophorum gracile Koch
8	B	Fen	1	Galium trifidum L.
8	B	Fen	1	Lysimachia thyrsiflora L.
8	B	Fen	1	Sagittaria latifolia Willd.
8	B	Fen	1	Sium suave Walt.
8	B	Fen	1	Sparganium eurycarpum Engelm.
8	B	Fen	1	Utricularia intermedia Hayne
8	C	Fen	2	Calla palustris L.
8	C	Fen	2	Campanula aparinoides Pursh
8	C	Fen	2	Carex lacustris Willd.
8	C	Fen	2	Carex lasiocarpa var. americana Fern.
8	C	Fen	2	Cicuta bulbifera L.
8	C	Fen	2	Comarum palustre L.
8	C	Fen	2	Eleocharis palustris L.
8	C	Fen	2	Equisetum fluviatile L.
8	C	Fen	2	Eriophorum gracile Koch
8	C	Fen	2	Galium trifidum L.
8	C	Fen	2	Lysimachia thyrsiflora L.
8	C	Fen	2	Sagittaria latifolia Willd.

8	C	Fen	2	Sium suave Walt.
8	C	Fen	2	Sparganium eurycarpum Engelm.
8	C	Fen	2	Utricularia intermedia Hayne
8	D	Fen	4	Calla palustris L.
8	D	Fen	4	Campanula aparinoides Pursh
8	D	Fen	4	Carex diandra Schrank
8	D	Fen	4	Carex lacustris Willd.
8	D	Fen	4	Carex lasiocarpa var. americana Fern.
8	D	Fen	4	Cicuta bulbifera L.
8	D	Fen	4	Comarum palustre L.
8	D	Fen	4	Eleocharis palustris L.
8	D	Fen	4	Equisetum fluviatile L.
8	D	Fen	4	Eriophorum gracile Koch
8	D	Fen	4	Galium trifidum L.
8	D	Fen	4	Lysimachia thyrsiflora L.
8	D	Fen	4	Sagittaria latifolia Willd.
8	D	Fen	4	Sium suave Walt.
8	D	Fen	4	Sparganium eurycarpum Engelm.
8	D	Fen	4	Typha latifolia L.
8	D	Fen	4	Utricularia intermedia Hayne
8	E	Fen	8	Calla palustris L.
8	E	Fen	8	Campanula aparinoides Pursh
8	E	Fen	8	Carex diandra Schrank
8	E	Fen	8	Carex lacustris Willd.
8	E	Fen	8	Carex lasiocarpa var. americana Fern.
8	E	Fen	8	Cicuta bulbifera L.
8	E	Fen	8	Comarum palustre L.
8	E	Fen	8	Eleocharis palustris L.
8	E	Fen	8	Equisetum fluviatile L.
8	E	Fen	8	Eriophorum gracile Koch
8	E	Fen	8	Galium trifidum L.
8	E	Fen	8	Lysimachia thyrsiflora L.
8	E	Fen	8	Rumex orbiculatus Gray
8	E	Fen	8	Sagittaria latifolia Willd.
8	E	Fen	8	Sium suave Walt.
8	E	Fen	8	Sparganium eurycarpum Engelm.
8	E	Fen	8	Typha latifolia L.
8	E	Fen	8	Utricularia intermedia Hayne
8	F	Fen	16	Calla palustris L.
8	F	Fen	16	Campanula aparinoides Pursh
8	F	Fen	16	Carex diandra Schrank
8	F	Fen	16	Carex lacustris Willd.
8	F	Fen	16	Carex lasiocarpa var. americana Fern.
8	F	Fen	16	Cicuta bulbifera L.
8	F	Fen	16	Comarum palustre L.
8	F	Fen	16	Eleocharis palustris L.
8	F	Fen	16	Equisetum fluviatile L.
8	F	Fen	16	Eriophorum gracile Koch
8	F	Fen	16	Galium trifidum L.
8	F	Fen	16	Iris versicolor L.
8	F	Fen	16	Lysimachia thyrsiflora L.
8	F	Fen	16	Lythrum salicaria L.
8	F	Fen	16	Rumex orbiculatus Gray
8	F	Fen	16	Sagittaria latifolia Willd.
8	F	Fen	16	Sium suave Walt.
8	F	Fen	16	Sparganium eurycarpum Engelm.
8	F	Fen	16	Typha latifolia L.
8	F	Fen	16	Utricularia intermedia Hayne
9	A	Fen	0.25	Carex lasiocarpa var. americana Fern.
9	A	Fen	0.25	Carex utriculata F. Boott
9	A	Fen	0.25	Comarum palustre L.

9	A	Fen	0.25	Eriophorum gracile Koch
9	A	Fen	0.25	Galium trifidum L.
9	A	Fen	0.25	Lysimachia thyrsiflora L.
9	A	Fen	0.25	Menyanthes trifoliata L.
9	A	Fen	0.25	Sagittaria latifolia Willd.
9	A	Fen	0.25	Salix pedicellaris Pursh.
9	A	Fen	0.25	Typha latifolia L.
9	A	Fen	0.25	Utricularia intermedia Hayne
9	B	Fen	1	Carex lasiocarpa var. americana Fern.
9	B	Fen	1	Carex utriculata F. Boott
9	B	Fen	1	Comarum palustre L.
9	B	Fen	1	Equisetum fluviatile L.
9	B	Fen	1	Eriophorum gracile Koch
9	B	Fen	1	Galium trifidum L.
9	B	Fen	1	Lysimachia thyrsiflora L.
9	B	Fen	1	Menyanthes trifoliata L.
9	B	Fen	1	Rumex orbiculatus Gray
9	B	Fen	1	Sagittaria latifolia Willd.
9	B	Fen	1	Salix pedicellaris Pursh.
9	B	Fen	1	Typha latifolia L.
9	B	Fen	1	Utricularia intermedia Hayne
9	C	Fen	2	Campanula aparinoides Pursh
9	C	Fen	2	Carex lasiocarpa var. americana Fern.
9	C	Fen	2	Carex utriculata F. Boott
9	C	Fen	2	Comarum palustre L.
9	C	Fen	2	Equisetum fluviatile L.
9	C	Fen	2	Eriophorum gracile Koch
9	C	Fen	2	Galium trifidum L.
9	C	Fen	2	Lysimachia thyrsiflora L.
9	C	Fen	2	Menyanthes trifoliata L.
9	C	Fen	2	Myrica gale L.
9	C	Fen	2	Rumex orbiculatus Gray
9	C	Fen	2	Sagittaria latifolia Willd.
9	C	Fen	2	Salix pedicellaris Pursh.
9	C	Fen	2	Typha latifolia L.
9	C	Fen	2	Utricularia intermedia Hayne
9	D	Fen	4	Campanula aparinoides Pursh
9	D	Fen	4	Carex lasiocarpa var. americana Fern.
9	D	Fen	4	Carex utriculata F. Boott
9	D	Fen	4	Cicuta bulbifera L.
9	D	Fen	4	Comarum palustre L.
9	D	Fen	4	Equisetum fluviatile L.
9	D	Fen	4	Eriophorum gracile Koch
9	D	Fen	4	Galium trifidum L.
9	D	Fen	4	Iris versicolor L.
9	D	Fen	4	Lysimachia thyrsiflora L.
9	D	Fen	4	Lythrum salicaria L.
9	D	Fen	4	Menyanthes trifoliata L.
9	D	Fen	4	Myrica gale L.
9	D	Fen	4	Rumex orbiculatus Gray
9	D	Fen	4	Sagittaria latifolia Willd.
9	D	Fen	4	Salix pedicellaris Pursh.
9	D	Fen	4	Sparganium eurycarpum Engelm.
9	D	Fen	4	Typha latifolia L.
9	D	Fen	4	Utricularia intermedia Hayne
9	E	Fen	8	Campanula aparinoides Pursh
9	E	Fen	8	Carex lasiocarpa var. americana Fern.
9	E	Fen	8	Carex utriculata F. Boott
9	E	Fen	8	Cicuta bulbifera L.
9	E	Fen	8	Comarum palustre L.
9	E	Fen	8	Equisetum fluviatile L.

9	E	Fen	8	Eriophorum gracile Koch
9	E	Fen	8	Galium trifidum L.
9	E	Fen	8	Iris versicolor L.
9	E	Fen	8	Lycopus uniflorus Michx.
9	E	Fen	8	Lysimachia thyrsiflora L.
9	E	Fen	8	Lythrum salicaria L.
9	E	Fen	8	Menyanthes trifoliata L.
9	E	Fen	8	Myrica gale L.
9	E	Fen	8	Rumex orbiculatus Gray
9	E	Fen	8	Sagittaria latifolia Willd.
9	E	Fen	8	Salix pedicellaris Pursh.
9	E	Fen	8	Sparganium eurycarpum Engelm.
9	E	Fen	8	Triadenum fraseri (Spach) Gl.
9	E	Fen	8	Typha latifolia L.
9	E	Fen	8	Utricularia intermedia Hayne
9	F	Fen	16	Campanula aparinoides Pursh
9	F	Fen	16	Carex lasiocarpa var. americana Fern.
9	F	Fen	16	Carex utriculata F. Boott
9	F	Fen	16	Cicuta bulbifera L.
9	F	Fen	16	Comarum palustre L.
9	F	Fen	16	Equisetum fluviatile L.
9	F	Fen	16	Eriophorum gracile Koch
9	F	Fen	16	Galium trifidum L.
9	F	Fen	16	Impatiens capensis Meerb.
9	F	Fen	16	Iris versicolor L.
9	F	Fen	16	Lycopus uniflorus Michx.
9	F	Fen	16	Lysimachia thyrsiflora L.
9	F	Fen	16	Lythrum salicaria L.
9	F	Fen	16	Menyanthes trifoliata L.
9	F	Fen	16	Myrica gale L.
9	F	Fen	16	Rumex orbiculatus Gray
9	F	Fen	16	Sagittaria latifolia Willd.
9	F	Fen	16	Salix pedicellaris Pursh.
9	F	Fen	16	Sparganium eurycarpum Engelm.
9	F	Fen	16	Triadenum fraseri (Spach) Gl.
9	F	Fen	16	Typha latifolia L.
9	F	Fen	16	Utricularia intermedia Hayne

Appendix 4. Site species richness, Coefficient of Conservatism, and Floristic Quality index scores. All values are mean 1 m^2 quadrat index scores.

seg_hash	S	Si	C	Ci	wC	wCi	FQI	FQIi	wFQI	wFQIi
56001	6.7857	6.7857	5.9948	5.9948	5.8376	5.8376	15.4480	15.4480	14.8867	14.8867
59004	5.9048	5.9524	5.8760	5.8610	6.5005	6.4683	13.9147	13.8883	15.2468	15.1529
59005	6.5833	7.2500	4.4703	4.0458	4.3289	3.7225	10.5170	10.0728	10.2564	9.4717
61005	6.4000	6.8500	5.9727	5.5955	5.9282	5.4290	14.7962	14.3343	14.7028	13.9255
63004	9.2083	9.5000	5.3658	5.2519	5.5786	5.3725	16.0210	15.8208	16.8896	16.4003
66003	7.0000	7.0500	6.3266	6.3009	6.6405	6.6393	15.6338	15.5964	16.6062	16.6349
68006	7.6000	7.6000	5.5740	5.5740	5.8797	5.8797	14.3007	14.3007	15.0029	15.0029
78003	8.2143	8.2857	6.2853	6.1853	6.2846	6.2826	17.9022	17.7966	17.8532	17.9545
84004	8.8462	8.8462	6.5603	6.5603	6.9870	6.9870	18.9875	18.9875	20.3355	20.3355
85002	4.0588	4.2941	2.9371	2.6940	2.4996	2.3629	6.4681	6.2455	5.5612	5.5214
87004	7.1250	7.1250	8.6604	8.6604	8.6262	8.6262	22.5206	22.5206	22.3638	22.3638
88001	10.5556	10.5926	7.4890	7.4647	7.9247	7.9234	24.1422	24.1048	25.6451	25.6850
96004	7.2500	7.8000	4.5942	4.2501	4.0023	3.6353	12.9006	12.4557	11.8533	11.0841
98002	10.4545	10.4545	5.7542	5.7542	6.4644	6.4644	18.0372	18.0372	20.4007	20.4007
98006	9.7500	9.8750	7.2316	7.1066	7.4482	7.4450	22.3928	22.2338	23.0990	23.2368
103004	5.7500	6.3750	4.6225	4.3277	4.9335	4.5513	10.4224	10.0081	11.3041	10.7110
104004	5.4545	5.9091	4.7372	4.3634	4.5513	4.0832	11.0415	10.6148	10.5275	9.8839
105006	5.2308	5.4615	5.4731	5.2603	5.4407	5.0429	12.2864	12.0611	12.4891	11.7242
109003	7.5682	7.7727	6.7605	6.5950	7.0987	6.8942	17.8572	17.6751	18.7917	18.5187
110004	9.3846	9.3846	8.2486	8.2486	8.6296	8.6296	24.8402	24.8402	25.9804	25.9804
192006	6.8750	6.8750	4.1914	4.1914	3.0476	3.0476	10.7887	10.7887	7.7551	7.7551
200001	9.9429	9.9429	8.5903	8.5903	8.9544	8.9544	26.3616	26.3616	27.5599	27.5599
215001	10.1538	10.1538	6.6837	6.6837	6.8720	6.8720	21.0089	21.0089	21.7073	21.7073
229001	3.5833	3.5833	5.6583	5.6583	5.4625	5.4625	10.4944	10.4944	10.1380	10.1380
242001	8.0519	8.1266	5.6991	5.2831	6.0670	5.3677	15.8407	15.1910	17.0994	15.7877
247001	8.9000	9.3000	5.7167	5.5521	5.3107	5.2637	16.6740	16.3786	15.3606	15.4652
247002	10.8750	11.6250	5.1681	4.9232	5.3592	5.2900	16.2003	15.7643	17.0162	17.2531
248001	9.5385	9.9231	5.8360	5.5634	6.3984	6.3694	17.1289	16.7631	19.0610	19.4118
274001	10.0667	10.0667	8.4025	8.4025	8.6562	8.6562	26.2328	26.2328	27.0404	27.0404
278004	9.2400	9.6400	5.0392	4.8450	5.0081	4.9613	14.8811	14.5920	14.6576	14.7557
281001	7.7500	7.7500	5.2728	5.2728	5.8088	5.8088	14.3297	14.3297	15.7998	15.7998
291001	6.5593	7.2203	4.6337	4.1834	4.6008	4.1197	11.4056	10.8924	11.3589	10.8614
294001	4.5952	6.0000	4.4363	3.3394	4.3378	3.4707	9.0969	7.9494	8.9465	8.2800
299001	7.0000	7.7500	4.0528	3.5312	3.9044	2.7749	10.2515	9.6816	9.8200	8.0629
300001	4.6429	5.5714	3.5964	3.0083	2.3004	1.9313	7.5630	6.9298	4.9304	4.6069
302001	5.0938	5.5625	4.6123	4.1739	4.6854	3.7222	10.0646	9.6459	10.1607	8.7019
302011	5.0667	6.3125	3.1691	2.3478	2.9802	1.8536	7.2126	5.9525	6.9329	4.6624
303002	3.7759	4.7759	3.1065	2.4288	3.1819	2.6716	5.9983	5.3670	6.1381	5.8666
304001	2.6061	3.7879	2.9959	2.0187	2.6460	1.5175	4.8375	4.0410	4.2739	3.0148
306004	2.5667	3.2333	2.5394	2.1200	2.5774	2.0682	4.0844	3.7663	4.0862	3.6183
313003	5.5000	5.5000	3.7902	3.7902	3.6404	3.6404	8.7599	8.7599	8.2644	8.2644
313011	6.2143	6.3571	4.9334	4.8764	4.7638	4.7535	11.2912	11.2014	11.1046	11.1604
319001	6.1250	6.6875	3.6432	3.3154	3.2426	3.0264	8.4757	8.1065	7.5834	7.4291
321002	3.0556	3.7500	3.9029	2.2983	3.6901	1.4152	6.7377	4.7632	6.2675	2.9136
325003	4.1200	4.6154	3.5885	2.9212	3.1003	2.0611	7.0612	6.2950	6.0416	4.4382
389001	4.6471	5.0556	5.4223	4.1720	5.8230	3.1382	11.0918	9.7520	11.7860	7.6346
393005	6.4706	6.5490	4.4326	4.3667	3.7484	3.7391	10.9314	10.8724	9.2016	9.2503
407001	4.6667	4.7333	5.6055	5.5578	5.5678	5.5541	11.6431	11.5898	11.5771	11.6380
408001	5.0000	5.0000	8.2289	8.2289	8.6374	8.6374	17.2323	17.2323	18.2602	18.2602
411001	6.0667	6.2000	6.8697	6.7558	7.0420	7.0280	16.5784	16.4530	17.2657	17.3573
421001	9.1333	9.4000	5.8738	5.7328	6.5449	6.5269	17.4685	17.2578	19.6267	19.7663
422001	8.6364	8.8182	5.4095	5.3258	5.7116	5.5427	15.5869	15.4498	16.5836	16.0872

423001	6.0909	6.5455	7.4233	6.6885	7.9608	7.5539	16.7674	16.0619	18.1363	18.0308
442003	5.7368	6.0526	4.7116	4.4962	4.6264	4.4954	9.9827	9.7403	9.6374	9.5348

Appendix 5. Site diversity index scores.

seg_hash	H	Ds	Hrel	m_plot_Dmg
56001	2.7395	0.0769	0.8620	1.2631
59004	2.6505	0.0997	0.7455	1.0603
59005	2.9337	0.0764	0.7389	1.3060
61005	2.8530	0.0829	0.7843	1.3049
63004	2.9999	0.0761	0.7630	1.5994
66003	3.0614	0.0624	0.7951	1.2909
68006	2.5721	0.1122	0.6882	1.5409
78003	2.3974	0.1371	0.6743	1.5741
84004	2.9427	0.0937	0.7130	1.6323
85002	2.4040	0.2071	0.6432	0.7203
87004	2.2243	0.1743	0.5990	1.4450
88001	2.6427	0.1212	0.6429	2.1058
96004	2.5529	0.1296	0.6631	1.3653
98002	2.6067	0.0976	0.7332	1.9417
98006	2.6963	0.1084	0.7467	1.7752
103004	3.0152	0.0653	0.7789	1.1755
104004	2.3471	0.1211	0.7709	0.9555
105006	2.2479	0.1521	0.6984	0.8998
109003	2.6156	0.1308	0.6266	1.8402
110004	2.7810	0.0804	0.7223	2.2001
192006	1.7424	0.2770	0.5413	1.2141
200001	2.7413	0.1069	0.6810	2.0696
215001	3.1650	0.0610	0.7730	1.9205
229001	2.0223	0.1867	0.7138	0.7308
242001	3.1311	0.0687	0.7252	1.5356
247001	2.4467	0.1463	0.6679	1.9185
247002	3.0407	0.0717	0.7258	2.2514
248001	2.4483	0.1273	0.6509	1.9610
274001	2.0855	0.2508	0.5580	2.3295
278004	3.1198	0.0788	0.7319	2.3968
281001	2.5692	0.1169	0.6637	2.0016
291001	3.1998	0.0690	0.7010	2.3033
294001	2.5574	0.1553	0.6173	1.2302
299001	2.7701	0.0909	0.7509	1.7550
300001	1.7538	0.2729	0.5263	1.0928
302001	2.6648	0.1242	0.6620	1.3751
302011	2.6841	0.0941	0.7228	1.2537
303002	2.3974	0.1598	0.5662	1.3035
304001	2.3099	0.1427	0.6860	0.6475
306004	2.4911	0.1130	0.6951	0.8652
313003	2.3225	0.1583	0.5850	1.9281
313011	2.6325	0.1113	0.6915	1.8519
319001	2.8804	0.1071	0.7094	1.7735
321002	2.2562	0.1931	0.5710	0.9871
325003	2.4437	0.1301	0.6873	0.9619
389001	2.3916	0.1463	0.6247	1.5131
393005	2.2189	0.2429	0.5465	1.3529
407001	3.1633	0.0605	0.7930	1.7473
408001	1.8640	0.1939	0.6579	0.8729
411001	2.6821	0.0975	0.7176	1.6764
421001	2.7379	0.1009	0.6999	2.1933
422001	2.8682	0.0913	0.7295	1.7582
423001	2.5594	0.1195	0.6416	1.7112
442003	2.3533	0.2149	0.6016	2.1152

Appendix 6. Site stressor data.

seg_hash	ac1	pd1	hydr_mod
56001	-0.3381	0.1304	0.0000
59004	-0.2871	0.3685	0.8900
59005	-0.2871	0.3685	41.9694
61005	-0.0718	0.0328	48.0625
63004	-0.2914	0.9082	10.0610
66003	-0.1862	0.0777	0.0000
68006	-0.1821	0.2887	0.0000
78003	-0.1475	0.0287	0.0000
84004	-0.2841	-0.2539	47.7893
85002	-0.2839	-0.3386	31.1036
87004	-0.2840	-0.4513	0.0000
88001	-0.2853	-0.0747	0.0000
96004	-0.2845	0.0216	153.1043
98002	-0.2839	-0.2810	77.2815
98006	-0.2839	-0.2810	80.9060
103004	-0.2861	0.4727	540.2970
104004	-0.2199	-0.1078	904.4780
105006	-0.2869	1.4688	0.0000
109003	-0.3591	-1.5181	0.0000
110004	-0.0721	-0.4567	0.0000
192006	-0.7882	-0.0078	177.9627
200001	-0.8060	-0.5015	5.5028
215001	-0.5637	-0.1857	6.2785
229001	-0.4637	-0.7497	6.1047
242001	-0.3014	-0.2738	27.7709
247001	-0.5492	0.0526	0.0000
247002	-0.5492	0.0526	25.4243
248001	-0.5247	-0.6637	44.3904
274001	-0.3434	-1.1456	45.1820
278004	0.2964	-0.6771	21.0966
281001	0.3637	-0.6473	15.4306
291001	0.4310	0.6359	57.9686
294001	0.9762	0.5201	0.0000
299001	1.7779	1.3887	98.0931
300001	1.8024	1.4909	109.9361
302001	1.9501	1.2400	25.5770
302011	1.9501	1.2400	86.0093
303002	1.8804	2.2341	22.7065
304001	1.9825	1.7004	128.6690
306004	1.9468	1.4983	0.0000
313003	1.7204	1.3414	0.0000
313011	1.7204	1.3414	0.0000
319001	1.8523	1.0212	115.9605
321002	1.9511	1.0696	31.0991
325003	2.3340	1.2509	4.3778
389001	1.0010	-0.0642	0.0000
393005	0.1582	0.3201	50.3778
407001	0.2404	1.4278	0.0000
408001	0.2247	1.2500	0.0000
411001	0.1344	0.2893	0.0000
421001	0.5846	-0.0226	19.9014
422001	0.2727	-0.1132	6.0471
423001	0.5730	0.3551	55.7222
442003	1.5939	0.8455	92.9616