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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 of high conservation value, monitor sites over time, assess the anthropogenic impacts affecting an area, and measure the ecological condition of an area. *FQI* is based on the Coefficient of Conservatism (*C*), which is a numerical score assigned to each plant species in a local flora, primarily from best professional judgment, 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. Great Lakes coastal wetlands were used to assess the properties and performance of various species richness, Coefficient of Conservatism, and Floristic Quality indices, as well as compare *C*-value assignments from two U.S. states (Wisconsin and Michigan). *FQI* and species richness increased with sampling area according to a power function, but $\overline{C}$ more or less remained constant. Sampling schemes should therefore focus on controlling sampling area and minimally sampling each community type at a site. In some cases, Wisconsin and Michigan assigned different values of *C* to the same species, highlighting possible effects due to the somewhat subjective nature of *C*-value assignment. Coefficient of Conservatism and Floristic Quality indices were better at discriminating differences between sites, independent of a condition gradient, than species richness alone, but neither index 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 seem to be effective indicators of condition in Great Lakes coastal wetlands.

Key Words: Floristic Quality Index (*FQI*), Coefficient of Conservatism, species richness, biological indicator, wetland condition, condition gradient, species-area relationship, discriminant ability, Great Lakes coastal wetland

INTRODUCTION

Biological indicators have been increasingly used to assess and monitor ecosystems (Fennessy et al. 2001, Mack 2004, Niemi and McDonald 2004). This is due to concern over human-caused ecosystem alteration and the resulting loss of biological diversity, as well as research suggesting that biological responses to anthropogenic stressors can integrate past and ongoing disturbances and therefore be effective indicators. Environmental laws, such as the Clean Water Act in the U.S., also have provisions to protect and promote biological diversity. Thus, there is a need for well-defined tools that can assess the biological condition of ecosystems.

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 to each plant species within a local flora by a panel of experienced botanists, primarily based on their best professional judgment. The theory behind *C* is that plant species differ in their tolerance to the type, frequency, and amplitude of disturbance and that plants have 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. *FQI* is calculated from the following general 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 Chicago region of Illinois (Wilhelm 1977). Since then, *FQI* has been proposed as a tool that can compare different sites regardless of plant community type, monitor sites over time, assess the anthropogenic impacts affecting an area, and ultimately measure the ecological condition of an area (Swink and Wilhelm 1994, Lopez and Fennessy 2002, Cohen et al. 2004). *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 2004). Eight 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 and Lichvar 1995), Southern Ontario (Oldham et al. 1995), Michigan (Herman et al. 2001), North and South Dakota (NGPFQAP 2001), Wisconsin (Bernthal 2003), and Indiana (Rothrock 2004).

Critical evaluation of *FQI* has come only recently, even though it has been in use since 1977 and is gaining popularity among regulating agencies. Mushet et al. (2002) evaluated the effect of panel subjectivity by generating *C*-values with data from 204 prairie pothole wetlands in North Dakota and concluded that panel derived *C*-values were adequate because wetland condition assessments 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 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 depend on sampling area, as does species richness. In addition, *FQI* has been found to vary systematically between different community types, by the time of the year sampled due to phenology, and by the ability of observers to identify plants correctly (Rooney and Rogers 2002, Matthews 2003). *FQI* has been correlated with surrogate measures of ecological condition and therefore considered a useful ecological indicator (Lopez and Fennessy 2002, Cohen et al. 2004 Mack 2004).

Although 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 present during the evolution of native plant communities. Other authors include introduced taxa in index calculations ($C = 0$) (Lopez and Fennessy 2002, Rooney and Rogers 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 apply it effectively and consistently towards ecological monitoring and assessment.

The objective of this study was to investigate several properties of *FQI* and test its performance as a potential ecological indicator in Great Lakes coastal wetlands. Specifically, this study sought to 1) determine the relationship between sampling area, species richness, $\overline{C}$, and *FQI*; 2) compare Michigan and Wisconsin *C*-values; 3) compare the relative discriminant ability of various *FQI* indices, independent of a condition gradient; 4) test *FQI* as an indicator of ecological condition; and 5) 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 but that have a hydrologic connection with the Great Lakes. 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. Riverine wetlands are characterized by wetland vegetation bordering a river that empties into the Great Lakes. Protected wetlands form behind natural structures, such as sandbars formed by longshore currents.

Human activities have affected all of the coastal wetlands within the Great Lakes basin to some degree. Effects range from basin-wide impacts due to global climate change (Mortsch 1998) to the complete loss of wetlands due to draining and filling activities (Herdendorf et al. 1981). Other anthropogenic stressors (human-caused external factors that in turn cause change in ecosystems, acting singly or in combination; Rapport et al. 1985, Detenbeck et al. 1999) have led to the alteration of ecological condition in coastal wetlands. The primary anthropogenic stressors to wetlands in the Great Lakes region include alteration of wetland hydrology due to partial control of lake levels and local disturbances (Keddy and Reznicek 1986, Hudon 1997, Maynard and Wilcox 1997, Sanzone and McElroy 1998); excess sediment, nutrient, and chemical loading (Kadlec and Bevis 1990, Jurik et al. 1994, Bedford et al. 1999, Detenbeck et al. 1999, Otte 2001); and exotic invasive species (Maynard and Wilcox 1997, Galatowitsch et al. 1999).

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 (Figure 1) in the Laurentian Mixed Forest Ecoprovince (Keys et al. 1995), of which 18 were classified as open-coast, 21 riverine, and 16 protected. The wetlands were selected to span multiple stressor gradients to facilitate indicator development and testing (Danz et al. 2005).

Field teams from the University of Minnesota Duluth and the University of Wisconsin Madison, consisting of two to three people each, sampled the vegetation of the wetland study sites during July and August from 2001 to 2003. Before each field season, field teams trained together to ensure consistency between teams. The primary vegetation sampling units were $1 \times 1\ m^2$ quadrats that were established in each wetland site along randomly located transects. Transects were generated for each selected site using GIS, and a target number of quadrats was determined based on wetland size (20 quadrats/60 ha of emergent class wetland, with a minimum of 10–15 quadrats) prior to vegetation sampling. Transects were evaluated in the field according to the order that they were mapped. Quadrats were established along transects by dividing a transect into 20-m segments and randomly locating a quadrat in each 20-m segment. Quadrats were established and 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.

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 modified Braun-Blanquet cover class ranges (ASTM 1997) : <1%, 1 to <5%, 5 to <25%, 25 to <50%, 50 to <75%, 75 to 100%. Summed individual taxa percent cover in a quadrat could exceed 100% because canopies of different

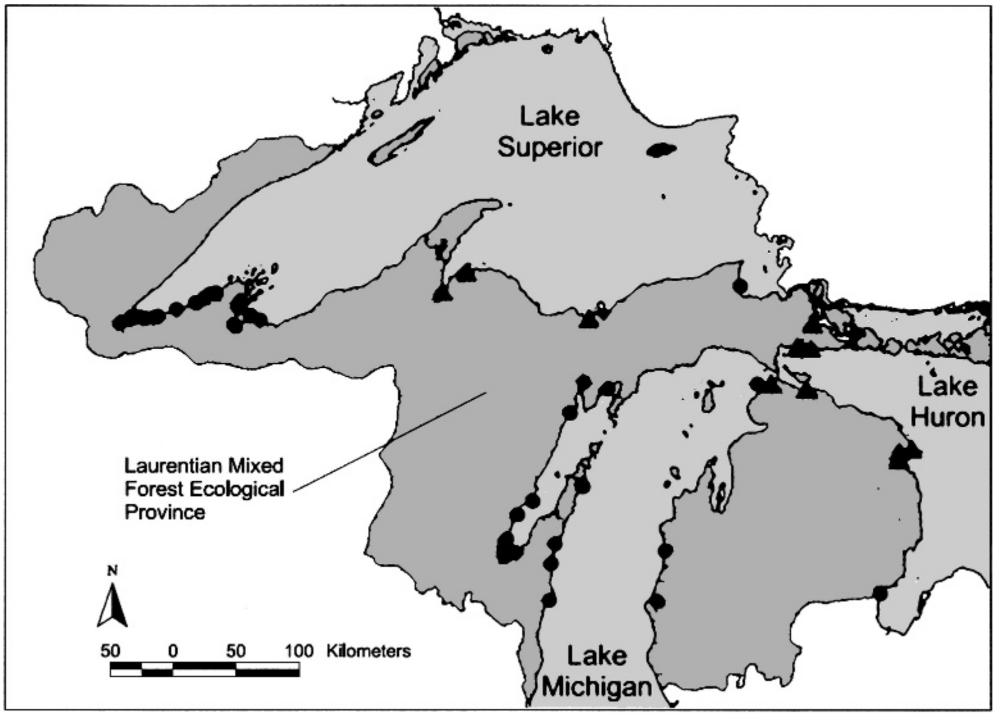

Figure 1. Coastal wetland study sites. Triangles indicate sites that were additionally sampled along transects.

species overlap one another. If a plant species could not be identified in the field, it was collected, pressed, and identified in the lab. Plant nomenclature conventions 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 mid-point percent cover of each class. Data from one of the open-coast wetlands were excluded from data analysis due to being sampled too early in the field season. Wetland mapping and vegetation sampling methodological details are fully described in Bourdaghs (2004).

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. 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 1 were calculated for every quadrat using the study area *C*-values, and average quadrat scores were used as site index scores. Weighted Coefficient of Conservatism indices were calculated from the general formula

$$wC = \sum_{j=1}^{S} p_j C_j \qquad (3)$$

Table 1. 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	$wFQI_i$

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 jth 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.

Individual C-value differences for the same species occurring in both Michigan and Wisconsin were also tallied. In addition, both $\overline{C}$ and FQI were computed using each set of 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.

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. Within each plot, all plant species were recorded in nested quadrats of 0.25, 1, 2, 4, 8, and 16 m^2. Field teams also recorded additional plant species occurring within a 2-m-wide belt centered on sampling transects at 12 of the study sites (Figure 1). These data were collected to compare index scores computed from quadrat data (smaller sampling area) versus a continuous belt transect (larger sampling area).

The indices S, $\overline{C}$, and FQI (Table 1) were calculated for each of the nested quadrats and averaged within community type over each quadrat area. Simple linear regression was then used to determine sampling area-index relationships from the nested plot data. Values for S, $\overline{C}$, and FQI were also calculated from the cumulative site species lists generated from quadrat and transect data collected in the 12 sampling area comparison sites.

Relative Discriminant Ability

Relative discriminant ability was tested by computing F-ratios for all ten species richness, Coefficient of Conservatism, and Floristic Quality indices (Table 1) in each of the three wetland types using ANOVA. An index that maximizes F compared to other indices for a given dataset is best at discriminating differences between sites, independent of a stressor or condition gradient (Kempton 1979, Magurran 1988). Because F-ratios can be computed only once per index per wetland type, a 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

Three measurements were chosen as predictors to act as an overall condition gradient in multiple regressions to test the various 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 within shoreline segment-sheds (watersheds that drain to a length of shoreline; Danz et al. 2005). The third predictor, local hydrologic modification, was computed for sampled wetlands by summing the length (m) of features that likely disrupt water flow and fluctuation (e.g., road beds, dikes, ditches) occurring in the wetland and dividing that sum by the wetland area (ha). All of the hydrologic modification measurements were made using digital orthophotoquads and wetland inventory maps displayed in GIS.

A key feature of any ecological indicator program is setting specific, *a priori*, performance criteria (Jackson et al. 2000). Indices were accepted as indicators where 1) there was a significant negative relationship between the index and the condition gradient and 2) the relationship was strong enough to detect a potentially important degradation in condition. The second criterion was judged based on the power (i.e., the probability of rejecting the null hypothesis when the null hypothesis is indeed false) to detect changes in condition as reflected by increases in stress in model simulations. More specifically, we required 80% power to detect a degradation of ecological condition associated

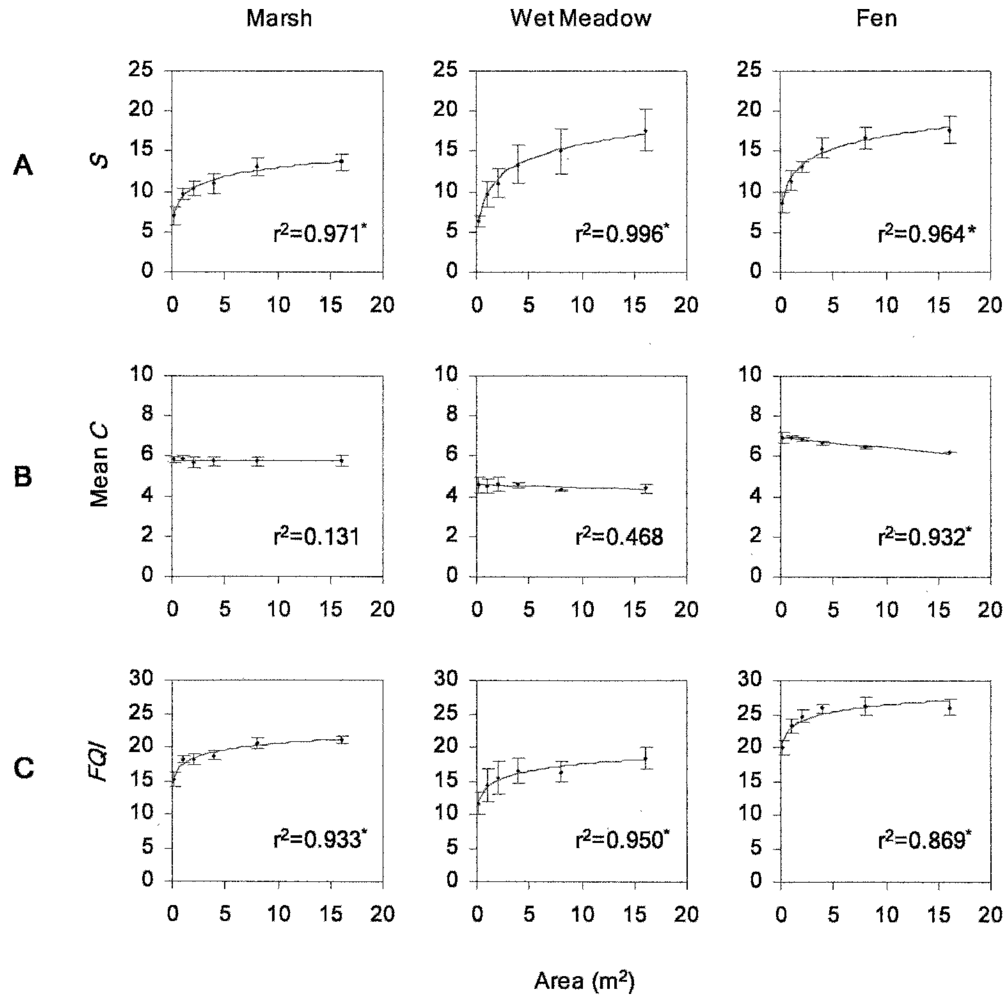

Figure 2. Species-area (A), mean Coefficient of Conservatism-area (B), and *FQI*-area (C) curves for three wetland community types in Allouez Bay, WI. * = Significant at $P < 0.01$, error bars represent one standard error.

with a simultaneous 20 percentile increase in each of the predictors.

Multiple linear regressions of all ten species richness, Coefficient of Conservatism, and Floristic Quality indices (Table 1) 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 then used to predict 'reference' and 'degraded' conditions to compute power probabilities. Reference index values were computed by first finding the tenth percentile of the distribution of each of the predictors and then plugging those values into the regression models. Index values corresponding to more degraded conditions were computed by incrementing all three of the predictors simultaneously in one percentile increments and plugging those values into regression models. Power was then computed for a t-test comparing index means for a sample of reference wetlands and more degraded wetlands. The population means and variances were based on the regression models' predicted values and mean squared errors. The power computations were based on $n = 20$ wetlands from reference and degraded conditions. 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

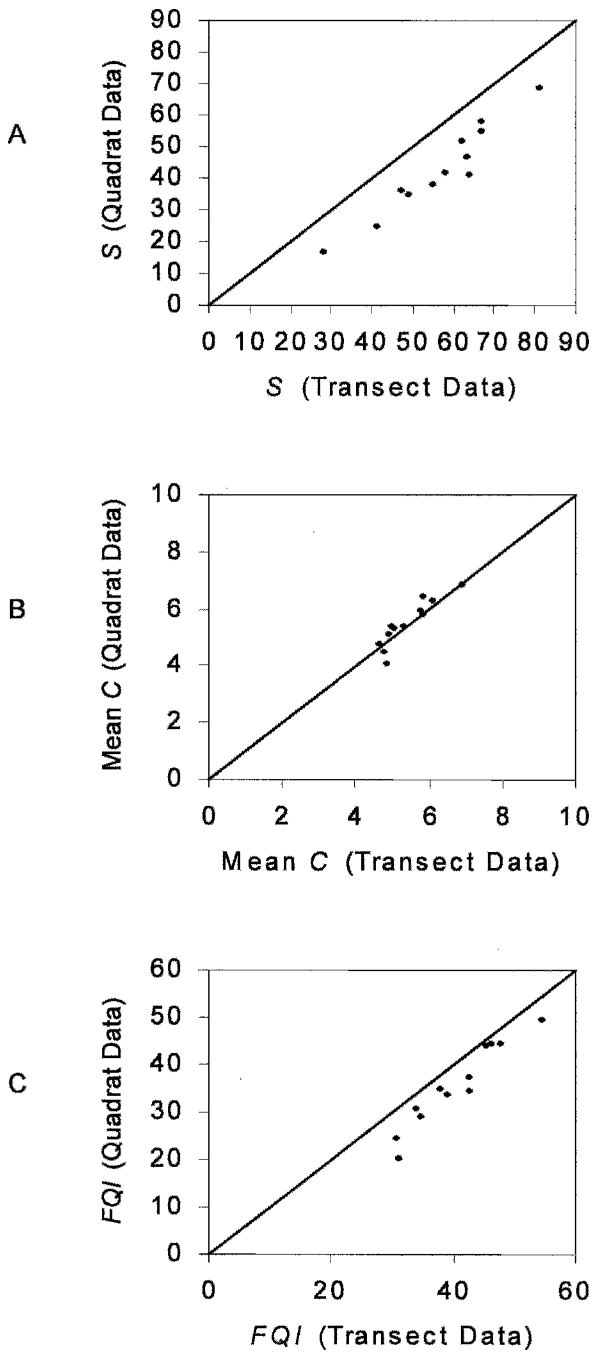

Figure 3. Scatter diagrams comparing (A) species richness, (B) mean Coefficient of Conservatism, and (C) *FQI* derived from transect and quadrat sampling from twelve sites. Each diamond represents one site. A one-one line has been added as a reference of equality.

in all three (marsh, wet meadow, and fen) community types (Figure 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 (Figure 2B). *FQI* increased with sampling area according to the ln-ln transformation of equation 2 in all three community types (Figure 2C), although the increase was somewhat dampened compared to the increase in species richness because the square root of species richness is an index parameter.

There was a similar pattern of *S*, $\overline{C}$, and *FQI* behavior at a larger sampling scale. *S* and *FQI* were both greater when computed from data collected along transects (larger sampling area) versus quadrat data (Figure 3A, 3C) in 12 sites in Michigan (Figure 1). $\overline{C}$ scores were very similar when computed from both data sets (Figure 3B).

Michigan and Wisconsin *C*-Value Comparison

Average *C*-values for the 403 species observed in this study were greater in Wisconsin (6.01) than in Michigan (5.42). Michigan *C*-values had greater variance (8.70) than Wisconsin (7.18). There were 220 *C*-value differences between the two states. Over half of the differences (115) were equal to 1. The largest difference was 6 for *Symphyotrichum pilosum* var. *pilosum* (Willd.) Nesom (Michigan = 1 and Wisconsin = 7). 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.). A partial *C*-value list is given in Appendix 1. A complete list of all the species observed in the study and their *C*-values can be found in Bourdaghs 2004.

There were also differences between the two states in classifying species as introduced or native. Seven species were observed that 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.).

To examine the effect of the differences between the two sets of *C*-values on index scores, $\overline{C}$ 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. On average, $\overline{C}$ calculated with Wisconsin *C*-values was significantly greater ($P = 0.04$; one-tailed *t*-test).

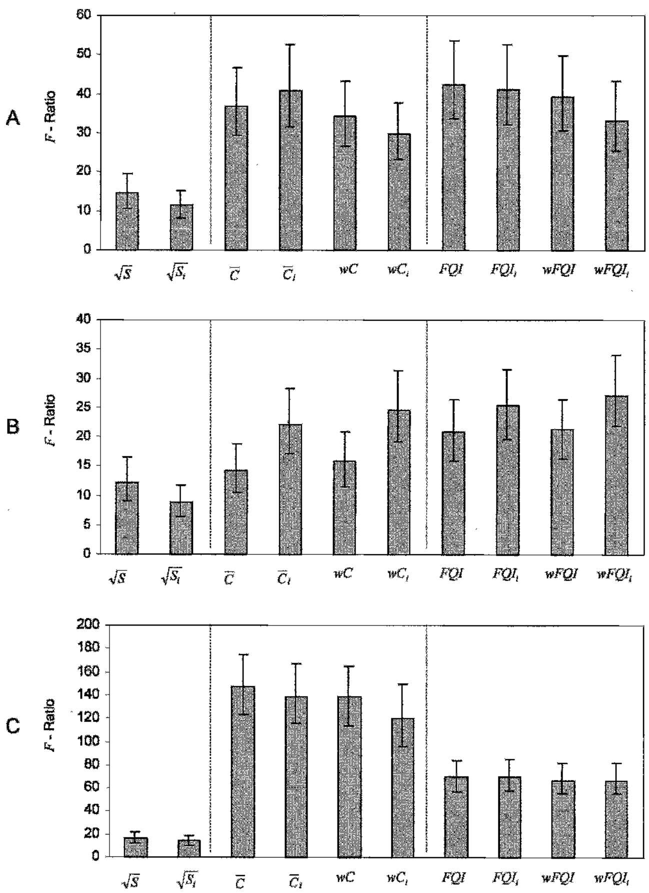

Figure 4. Bootstrap mean F-ratios, with 95% confidence intervals, for all species richness, Coefficient of Conservatism, and Floristic Quality indices in (A) open-coast, (B) riverine, and (C) protected wetlands. Dashed lines delineate index categories.

However, there was no significant difference with FQI ($P > 0.05$) between the two sets of C-values.

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 (Figure 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, 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

In open-coast and protected wetlands, all of the species richness, Coefficient of Conservatism, and Floristic Quality indices decreased significantly ($P < 0.05$) with declines in condition (Table 2). In riverine wetlands, only $\overline{C}_i$, wC_i, FQI, FQI_i, and $wFQI_i$-condition relationships were significant.

The strength of condition relationships decreased in the order Floristic Quality > Coefficient of Conservatism > species richness indices in open-coast wetlands (Table 2). This general pattern was also found in the riverine and protected wetland types but with 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. Condition relationships were also weaker for wC and wC_i than either S or S_i in protected wetlands.

The power probabilities showed the same general pattern of index performance (Table 2, Figure 5). In each of the three wetland types, the power to detect a significant difference between predicted reference index values and predicted degraded index values when all three predictors were 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 power > 0.99 by a 20 percentile point 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 3). Standardized regression coefficients for Agriculture PC1 and Population

Table 2. Multiple regression results for each of the wetland hydrogeomorphic types.

Index	F	P	r^2	MSE	Intercept†	Agriculture PC1†	Pop. Dens. PC1†	Hydro. Mod.†	Power‡
Open-Coast Wetlands									
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									
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
Protected Wetlands									
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

† Significance levels for regression coefficients: * = $P < 0.05$, ** = $P < 0.001$.
‡ Power probabilities from model simulations when all three predictors are increased by 20 percentile points simultaneously.

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 show that *FQI* is sensitive is to sampling area due to the species richness parameter in the index (Figure 2C, Figure 3C). 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). Conversely, sampling area had little effect on $\overline{C}$ (Figure 2B, Figure 3B).

This observed behavior has broad implications concerning the sampling methodologies that are required for these indices. Proponents of Floristic Quality indices advocate that a complete site census is preferable for application (Taft et al. 1997, Herman et al. 2001). This methodology generally consists of a botanist walking the site multiple times

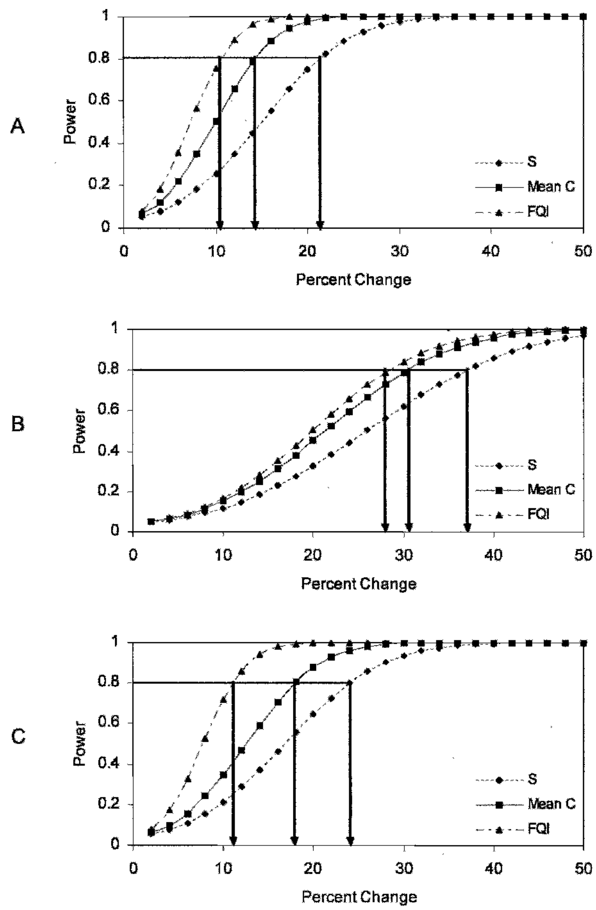

Figure 5. Power curves for detecting differences between predicted index reference values and percent predictor change values for *S*, $\overline{C}$, and *FQI* in all three wetland types.

during the growing season, recording plant species as they are observed, until a complete census is judged to be obtained. Our results show that this method of sampling is inappropriate for *FQI* because of the strong effect of sampling area on index scores and is not necessary for $\overline{C}$, as index scores are stable with a small sample. In other words, procedures that account for sampling area must be used to make unbiased comparisons with *FQI*, and the point of diminishing returns is reached with minimal sampling for $\overline{C}$. To be clear, all of the community types within a site should be sampled to be able to conclude confidently that an entire site is described by these indices, as there will be different suites of species between communities. Therefore, optimum sampling schemes should focus on controlling sampling area while minimally sampling all of the communities within a site. Controlling for sampling area can be done by either standardizing the site sampling area or by sampling sites with multiple randomly-placed plots and then using mean plot index scores as site index values, as was done in this study. The first sampling procedure has the disadvantage of being inflexible with respect to wetland size and may not characterize plant communities as accurately. 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 characterizing sites by a distribution of index scores, as opposed to a single value. Mean plot index scores are often normally distributed, which allows comparisons of different sites using parametric statistics (Magurran 1988).

Michigan and Wisconsin *C*-Value Comparison

Comparisons of index scores and overall statistics of Michigan and Wisconsin *C*-values, as well as large individual species differences, may be indicative of subjective effects in *C*-value assignment. Although the behavior of plant species varies geographically, it is unlikely that Michigan and

Table 3. Normal stadardized multiple regression coefficients for each of the wetland types.

Index	Intercept	Agriculture PC1	Pop. Dens. PC1	Hydro. Mod.
Open-Coast Wetlands				
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				
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				
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

* = $P < 0.05$, ** = $P < 0.001$.

Wisconsin are very different, given similarities in their pre-European settlement vegetation. The difference of the overall mean *C*-values between the two states was not large (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. It is unknown, and likely cannot be determined, whether this difference is due to an artifact of *C*-value assignment methodology or whether plants in Wisconsin are naturally more 'conservative.' However, the former factor seems more probable given the reliance on best professional judgment in the *C*-value assignment process. The large differences observed with a handful of species (*C*-value difference $\geq$ 4) may also be indicative of subjective effects. A likely explanation for *C*-value differences between states is that a species may be at the edge of its range in one state and widespread in the other, leading to a difference in *C*. All of the species that have large *C*-value differences, however, are more or less widespread throughout both states and have natural ranges that extend beyond Michigan and Wisconsin (Voss 1972, Voss 1985, Gleason and Cronquist 1991, Voss 1996, Wisconsin State Herbarium; http://www.botany.wisc.edu/herbarium).

Assigning *C*-values by an ecologically meaningful unit, such as Ecological Province or Section (Keys et al. 1995), may improve accuracy and reduce subjective effects. These units are based on broad climate and vegetation patterns, and 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.

Relative Discriminant Ability

The ability of an index to discriminate between 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 as indicators. 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 differences between sites than species richness indices (Figure 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 likely due to the fact that many of the protected wetland sites were fen and bog communities. *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) found that different plant community types had different $\overline{C}$ in Wisconsin. The high *F*-ratios of Coefficient of Conservatism indices in protected wetlands are likely due to this classification error and may not be indicative of the performance of these indices.

Indicator Testing

In general, Floristic Quality indices performed better than Coefficient of Conservatism indices, which, in turn, outperformed species richness indices (Table 2, Figure 5). A possible explanation for the performance of Floristic Quality indices can be made by examining 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 (Bourdaghs 2004). 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, in turn, 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 2). None of the indices passed both of the performance criteria, although 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 summarized 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 (Danz et al. 2005). 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 encompasses more than 3,030 km^2. This scale may be too coarse to estimate local stressors accurately, which may have the largest effects on wetland condition (Mensing et al. 1998, Tufford et al. 1998). The log segment-shed area of riverine wetlands was greater than those for both open-coast and protected wetlands (one-tailed *t*-tests, $P < 0.05$). 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.

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. It was not a goal of this study to invert the research question and use these indices to test which stressors affect condition. Although this is an important research question, there is a subtle difference between the two, and therefore, they must be addressed independently to avoid problems with circular reasoning. Thus, the inconsistent predictor-contribution results (Table 3) between wetland types do not necessarily mean that particular stressors are either more or less important in a general sense, or in specific wetland types. To test for the effects of individual stressors on 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 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. Although these two PCs summarize multiple stressors indirectly, they do not summarize all of the primary stressors that can affect the extent and composition of coastal wetlands. Further study is therefore needed to test for the effect of individual stressors, which includes a more complete account of primary stressors.

The most useful condition indicators possess the following attributes: 1) are conceptually relevant to assessment questions and to ecological function; 2) can be implemented in a straightforward and cost effective manner; 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 (Jackson et al. 2000, NRC 2000, Dale and Beyeler 2001, Fore 2003). Results from this study show that, in Great Lakes coastal wetlands, both Coefficient of Conservatism and Floristic Quality indices have most of these attributes. *C*-values are conceptually simple to understand and have an 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. However, voucher specimens should be collected and questionable identifications verified. Sampling requirements are also likely within the normal range of the duties performed by regulating agency personnel, thereby making implementation cost effective. The ability of an indicator to discriminate sites along a known condition gradient is the most important property of an indicator. Both 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 assessed 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 (Kercher et al. 2003). Spatial variability was somewhat controlled by classifying wetlands by geomorphic type and by bounding the study area within the Laurentian Mixed Forest Ecological Province. An unanticipated source of spatial variability was the effect of some protected wetlands (i.e., peatlands) on index scores. 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, especially since lake-level fluctuations can have substantial effects on wetland vegetation (Wilcox et al. 2002). 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 in indicating levels of condition, if different levels of condition were defined. Users could then apply these indices to appropriate management goals.

Conclusions

As the use of Coefficient of Conservatism and Floristic Quality indices increases, users should apply the 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 included or excluded introduced species (Table 2, Figure 4) were 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 computations.

Performance results between indices that were weighted or not weighted by abundance were also indistinguishable (Table 2, Figure 4). This suggests that weighting indices by abundance should be avoided because of the increased data and computational requirements needed to compute these types of indices. However, there can be applications where weighted indices outperform non-weighted indices. Poling et al. (2003) 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) were 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 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 depends on the application, as well as the sampling design. Results presented in this study (Figure 2, Figure 3) clearly show that Floristic Quality indices are affected by sampling area due to the species richness parameter. Another criticism is that because they are the product of two parameters, Floristic Quality indices can give misleading results. 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, though, of both types of indices was more or less the same (Figure 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 2, Figure 5) and, thus, should be used in that capacity. Therefore, users need to take into account the anticipated application for the index and the method of data collection, 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 calibration could be in the form of defining different levels of condition such as 'good,' 'fair,' and 'poor' or an impairment threshold that could signal managers to perform some sort of 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 of lake levels 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 effects 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 were effective indicators of condition in Great Lakes coastal wetlands and have excellent potential as ecological assessment tools.

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Appendix 1. Plant species observed in at least 5 study sites with Michigan (Herman et al. 2001), Wisconsin (Bernthal et al. 2001), and study area specific *C*-values. Null values indicate that the species does not occur in that state.

Scientific Name	MI	WI	Study Area	Number of Sites
Acorus calamus L.	6	7	6.5	14
Agrostis hyemalis (Walt.) B.S.P.	4	4	4	8
Alnus incana ssp. *rugosa* (Du Roi) Clausen	5	4	4.5	18
Alopecurus aequalis Sobol.	4	4	4	7
Andromeda polifolia var. *glaucophylla* (Link) DC.	10	10	10	10
Anemone canadensis L.	4	4	4	7
Argentina anserina (L.) Rydb.	5	4	4.5	10
Asclepias incarnata L.	6	5	5.5	5
Bidens cernua L.	3	4	3.5	9
Boehmeria cylindrica (L.) Sw.	5	6	5.5	8
Calamagrostis canadensis (Michx.) Beauv.	3	5	4	42
Calla palustris L.	10	9	9.5	13
Calystegia sepium (L.) R.Br.	2	2	2	7
Campanula aparinoides Pursh	7	7	7	37
Carex aquatilis Wahlenb.	7	7	7	24
Carex bebbii Olney ex Fern.	4	4	4	9
Carex comosa Boott	5	5	5	19
Carex diandra Schrank	8	9	8.5	5
Carex hystericina Muhl. ex Willd.	2	3	2.5	7
Carex lacustris Willd.	6	6	6	36
Carex lasiocarpa var. *americana* Fern.	8	9	8.5	36
Carex rostrata Stokes	10	10	10	5
Carex stipata Muhl. ex Willd.	1	2	1.5	11
Carex stricta Lam.	4	7	5.5	41
Carex utriculata Boott	5	7	6	18
Carex viridula Michx.	4	6	5	5
Carex vulpinoidea Michx.	1	2	1.5	11
Ceratophyllum demersum L.	1	3	2	17
Chamaedaphne calyculata (L.) Moench	8	9	8.5	10
Cicuta bulbifera L.	5	7	6	37
Cirsium arvense (L.) Scop.	*	*	*	18
Cirsium muticum Michx.	6	8	7	6
Cladium mariscoides (Muhl.) Torr.	10	10	10	7
Comarum palustre L.	7	8	7.5	29
Cornus sericea ssp. *sericea* L.	2	3	2.5	7
Dasiphora floribunda (Pursh) Kartesz, comb. nov. ined	10	9	9.5	6
Drosera rotundifolia L.	6	7	6.5	10
Dulichium arundinaceum (L.) Britt.	8	9	8.5	10
Eleocharis elliptica Kunth	6	7	6.5	9
Eleocharis erythropoda Steud.	4	3	3.5	17
Eleocharis palustris (L.) Roemer & J.A. Schultes	5	6	5.5	19
Elodea canadensis Michx.	1	3	2	15
Epilobium ciliatum Raf.	3	3	3	6
Epilobium coloratum Biehler	3	3	3	7
Epilobium leptophyllum Raf.	6	8	7	10
Equisetum fluviatile L.	7	7	7	24
Eupatorium maculatum L.	4	4	4	15
Eupatorium perfoliatum L.	4	6	5	21
Euthamia graminifolia (L.) Nutt.	3	4	3.5	14
Fraxinus pennsylvanica Marsh.	2	2	2	9
Galium trifidum L.	6	6	6	26
Glyceria borealis (Nash) Batchelder	6	8	7	5
Glyceria striata (Lam.) A.S. Hitche.	4	4	4	8
Impatiens capensis Meerb.	2	2	2	33
Iris versicolor L.	5	5	5	14

Appendix 1. Continued.

Scientific Name	MI	WI	Study Area	Number of Sites
Juncus alpinoarticulatus Chaix in Villars	5	6	5.5	7
Juncus balticus var. *littoralis* Engelm.	4	5	4.5	8
Juncus dudleyi Wieg.	1	4	2.5	10
Juncus effusus L.	3	4	3.5	14
Juncus nodosus L.	5	6	5.5	22
Juncus pelocarpus E. Mey.	8	8	8	7
Larix laricina (DuRooi) K. Koch	5	8	6.5	6
Lathyrus palustris L.	7	5	6	13
Leersia oryzoides (L.) Sw.	3	3	3	26
Lemna minor L.	5	4	4.5	13
Lemna trisulca L.	6	6	6	6
Lycopus americanus Muhl. ex W. Bart	2	4	3	14
Lycopus uniflorus Michx.	2	4	3	30
Lysimachia thyrsiflora L.	6	7	6.5	28
Lythrum salicaria L.	*	*	*	17
Mentha arvensis L.	3	3	3	10
Menyanthes trifoliata L.	8	10	9	11
Mimulus ringens L.	5	6	5.5	11
Muhlenbergia glomerata (Willd.) Trin.	10	9	9.5	5
Myrica gale L.	6	9	7.5	22
Myriophyllum sibiricum Komarov	10	6	8	10
Najas flexilis (Willd.) Rostk. & Schmidt	5	6	5.5	12
Nuphar lutea ssp. *variegata* (Dur.) E.O. Beal	7	6	6.5	12
Nymphaea odorata Ait.	6	6	6	9
Onoclea sensibilis L.	2	5	3.5	8
Phalaris arundinacea L.	0	*	0	24
Phragmites australis (Cav.) Trin. ex Steud.	0	1	0.5	15
Pilea fontana (Lunell) Rydb.	5	7	6	11
Poa palustris L.	3	5	4	6
Pogonia ophioglossoides (L.) Ker-Gawl.	10	9	9.5	5
Polygonum amphibium L.	6	5	5.5	7
Polygonum amphibium var. *stipulaceum* Coleman	6	6	6	5
Polygonum hydropiperoides Michx.	5	6	5.5	5
Polygonum punctatum Ell.	5	5	5	7
Polygonum sagittatum L.	5	6	5.5	12
Populus deltoides Bartr. ex Marsh.	1	2	1.5	5
Potamogeton natans L.	5	5	5	7
Potamogeton richardsonii (Benn.) Rydb.	5	5	5	6
Rhynchospora alba (L.) Vahl	6	9	7.5	5
Rosa palustris Marsh.	5	7	6	7
Rumex orbiculatus Gray	9	8	8.5	18
Sagittaria graminea Michx.	10	9	9.5	13
Sagittaria latifolia Willd.	1	3	2	31
Salix candida Flugge ex Willd.	9	10	9.5	5
Salix discolor Muhl.	1	2	1.5	15
Salix pedicellaris Pursh.	8	8	8	17
Salix petiolaris Sm.	1	6	3.5	5
Salix serissima (Bailey) Fern.	8	8	8	5
Sarracenia purpurea L.	10	10	10	11
Schoenoplectus acutus var. *acutus* (Muhl. ex Bigelow) A. & D. Love	5	6	5.5	5
Schoenoplectus americanus (Pers.) Volk. ex Schinz & R. Keller	10		10	6
Schoenoplectus fluviatilis (Torr.) M.T. Strong	6	6	6	8
Schoenoplectus pungens var. *pungens* (Vahl) Palla	5	5	5	6
Schoenoplectus subterminalis (Torr.) Sojak	8	9	8.5	7
Schoenoplectus tabernaemontani (K.C. Gmel.) Palla	4	4	4	43
Scirpus atrovirens Willd.	3	3	3	7
Scutellaria galericulata L.	5	5	5	23

Appendix 1. Continued.

Scientific Name	MI	WI	Study Area	Number of Sites
Sium suave Walt.	5	5	5	14
Solanum dulcamara L.	*	*	*	7
Solidago canadensis L.	1	1	1	13
Solidago gigantea Ait.	3	3	3	6
Solidago uliginosa Nutt.	4	8	6	6
Sonchus arvensis L.	*	*	*	9
Sparganium eurycarpum Engelm. ex Gray	5	5	5	35
Sparganium fluctuans (Morong) B.L. Robbins.	10	10	10	5
Spiraea alba Du Roi	4	4	4	15
Spirodela polyrrhiza (L.) Schleiden	6	5	5.5	8
Stachys palustris L.	5	5	5	10
Stuckenia pectinatus (L.) Boerner	3	3	3	7
Symphyotrichum lanceolatum var. *lanceolatum* (Willd.) Nesom	2	4	3	5
Symphyotrichum puniceum var. *puniceum* (L.) A.& D. Love	5	5	5	9
Taraxacum officinale G.H. Weber ex Wiggers	*	*	*	6
Thelypteris palustris Schott	2	7	4.5	10
Thuja occidentalis L.	4	9	6.5	10
Triadenum fraseri (Spach) Gleason	6	8	7	15
Triglochin maritimum L.	8	10	9	7
Typha angustifolia L.	*	*	*	24
Typha latifolia L.	1	1	1	28
Typha X *glauca* Godr. (pro sp.)	*	*	*	18
Urtica dioica L.	1	1	1	15
Utricularia intermedia Hayne	10	9	9.5	10
Utricularia macrorhiza Le Conte	6	7	6.5	17
Vaccinium macrocarpon Ait.	8	9	8.5	6
Vaccinium oxycoccos L.	8	9	8.5	9
Verbena hastata L.	4	3	3.5	8
Zizania aquatica L.	9	8	8.5	8

* Species is considered introduced.