Ecological Indicators 34 (2013) 260–267

Contents lists available at SciVerse ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Floristic quality assessment signals human disturbance over natural variability in a wetland system

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ARTICLE INFO

Article history:
Received 1 October 2012
Received in revised form 19 February 2013
Accepted 16 May 2013

Keywords:
Conservatism
Ecological condition
Human disturbance
Monitoring
Seasonal wetlands
Vascular plants

ABSTRACT

A common concern regarding the popular Floristic Quality Assessment (FQA) method is whether the site floristic quality scores change with natural temporal and site-specific variability. The more ignored question is whether this background variability will confound the index of human disturbance. Using non-forested seasonal wetlands in the northeastern United States, we tested if two common indices of site floristic quality (FQAI, Mean CoC) provide clear signals of site condition relative to gradients of wetland area and surface water depth, and consistent signals across time of year (early vs. late growing season), geomorphic setting (connected vs. isolated), and vegetation community type (pine barrens vernal pond, wet sedge meadow, shrub swamp). Mean CoC is the coefficient of conservatism (a qualitative estimate of species' sensitivity to human disturbance) averaged across the native and exotic taxa observed at a given site, and FQAI is the traditional Floristic Quality Assessment Index where Mean CoC is multiplied by square root of taxa richness. The FQAI did not linearly correspond to the site condition gradient and thus it could not be evaluated. Mean CoC was clearly associated with site condition, with no interference from wetland area or water level (based on computer-intensive resampling of linear model fit). Mean CoC also varied consistently with site condition between the surveys, geomorphic settings, and community types (based on computer-intensive resampling of linear model slope). However, connected wetlands showed inherently greater Mean CoC than isolated wetlands, suggesting a comparison of floristic quality between these categories would not be prudent. Overall this study suggests that FQA in the form of Mean CoC may withstand natural variability in certain non-forested wetland systems. Instead of assuming FQA is overly sensitive to natural variability, we recommend further efforts to identify situations in which FQA is robust.

1. Introduction

Floristic Quality Assessment (FQA) was originally developed in the late 1970s to assess prairies and open, undeveloped lands in the Lower Lake Michigan region of the United States (Swink and Wilhelm, 1994; Wilhelm and Ladd, 1988). Now FQA is used in over two-thirds of the United States (Medley and Scozzafava, 2009; Bried et al., 2012) in a variety of ecosystems, and has been applied in Canada (Francis et al., 2000) and in other countries (Landi and Chiarucci, 2010; Tu et al., 2009). It is an especially popular tool for wetland condition assessments and has been applied in wetland systems ranging from Virginia bottomland forests (Nichols et al., 2006) to Louisiana coastal marshes (Cretini et al., 2012) to North Dakota prairie potholes (Mushet et al., 2002). Several wetland FQA studies have rigorously evaluated index properties and performance (Bourdaghs et al., 2006; Ervin et al., 2006; Matthews et al., 2005), and a growing number of wetland monitoring programs are using FQA as part of multi-metric condition assessments (e.g., Hargiss et al., 2008; Mack, 2007; Reiss et al., 2010).

Accuracy of FQA depends on experienced botanists who assign a coefficient of conservatism (CoC) meant to capture a species' ecological amplitude (sensu Packham and Willis, 1976) and sensitivity to human disturbance. The coefficients range from 0 to 10 with tolerant weedy species receiving the lowest scores and conservative species (sensitive to habitat degradation, "remnant-dependent") the highest (Swink and Wilhelm, 1994; Taft et al., 1997). A complete or representative checklist of vascular flora is required for the area of interest, and the index is often the Mean CoC weighted by species richness or the Mean CoC alone (Rooney and Rogers, 2002; Spyreas et al., 2012; Taft et al., 2006).

A basic principle of FQA is that it reflects human disturbance or site condition with minimal ecological interference (Taft et al., 1997). Indeed, a challenge for any wetland assessment method is

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1470-160X/$ – see front matter
http://dx.doi.org/10.1016/j.ecolind.2013.05.012

to ensure that measurements are related to functioning or condition of the wetland and not natural temporal or spatial variability (Brazner et al., 2007; Rader et al., 2001; Stein et al., 2009; U.S. EPA, 2002). However, in practice such factors as habitat type, site area, and seasonality of vegetation could bias the index or reduce its precision (Bourdaghs et al., 2006; Cohen et al., 2004; Ervin et al., 2006; Matthews, 2003; Matthews et al., 2005; Miller and Wardrop, 2006). This is often due to the inclusion of species richness as a component of FQA (Matthews, 2003; Matthews et al., 2005; Miller and Wardrop, 2006). Users of FQA may avoid or mitigate such problems by grouping sites according to environmental similarity and restricting surveys within narrow time frames of the growing season. Of course doing so may reduce a project's flexibility, increase its complexity, and force practitioners to make subjective judgments about perceived environmental categories and gradients. We think the issue deserves further scrutiny.

Several studies have found FQA metrics to vary across natural environmental categories or gradients (e.g., Andreas et al., 2004; Bried and Edinger, 2009; Euliss and Mushet, 2011; Taft et al., 2006), leading to a general assumption that FQA is overly sensitive to natural variability. An important question is whether this background variability overwhelms the disturbance signal. For environmental gradients (i.e., continuous variables such as area or depth), a signal to noise ratio could be measured as strength (model fit) of the expected linear relationship between floristic quality and increasing human disturbance (Andreas et al., 2004; Cohen et al., 2004; Lopez and Fennessy, 2002; Miller and Wardrop, 2006; U.S. EPA, 2002) relative to strength of the relationship between floristic quality and the underlying gradient. Alternatively, the floristic quality residuals from a regression over disturbance could be analyzed for a linear relationship to the natural gradients. For environmental categories (e.g., habitat type), the slope and not just the intercept of the relationship between floristic quality and disturbance is of interest and could be compared among categories. If the signal strength is weak or if the slopes intersect, then inference pooled over gradients or categories may indeed be biased or imprecise as generally assumed.

Wetlands are a good study system for this problem because they are extremely dynamic and variable (Mitsch and Gosselink, 2007), and because FQA is popular for wetland assessments. The goal of this paper is to evaluate the influence of natural variability on wetland FQA. Specifically, we test if two common indices of site floristic quality provide clear signals of site condition relative to gradients of wetland area and surface water depth, and consistent signals across time of year (early vs. late growing season), geomorphic setting (connected vs. isolated), and vegetation community type (pine barrens vernal pond, wet sedge meadow, shrub swamp).

2. Materials and methods

2.1. Study area

Study sites were located in three unique areas of upstate New York (map in Bried and Edinger, 2009): Albany Pine Bush Preserve (hereafter "Pine Bush"), Rome Sand Plains, Wilton Wildlife Preserve & Park (hereafter "Wilton Preserve"). The Pine Bush is located in an urbanized landscape matrix whereas Rome Sand Plains and the Wilton Preserve reside in less modified exurban settings. The region is characterized by a cold-temperate humid climate with approximately 100 cm of mean annual precipitation and elevation ranging from about 100 to 400 m above sea level.

The Pine Bush and Wilton Preserve are located on the Glacial Lake Albany sand plain of east-central New York State. The Pine Bush (42°42′ N, 73°52′ W) is a highly fragmented dune field landscape between the cities of Albany and Schenectady. It is best known for the federally endangered Karner blue butterfly (*Lycaeides melissa samuelis* Nabokov) and globally rare inland pitch pine (*Pinus rigida* Mill.) – scrub oak (*Quercus ilicifolia* Wang., *Quercus prinoides* Willd.) communities (Barnes, 2003). Pine Bush wetland types include pine barrens vernal pond, woodland vernal pool, red maple hardwood swamp, shallow emergent marsh, rich sloping fen, and wet sedge meadow, which collectively support over 300 plant species in the preserve (Mattox, 1994).

The Wilton Preserve (43° 13′ N, 73° 68′ W) is located approximately 50 km north of the Pine Bush near the city of Saratoga Springs. This area features flagship Karner blue populations and is intensively managed for overabundant white pine (*Pinus strobus* L.). Major wetland types include red maple hardwood swamp, hemlock hardwood swamp, pine barrens vernal pond, shallow emergent marsh, shrub swamp, wet sedge meadow, and woodland vernal pool. Collectively these wetlands support over 200 vascular plant species (New York Natural Heritage Program, unpubl. report).

Rome Sand Plains (43° 14′ N, 75° 34′ W) is located on the Glacial Lake Iroquois sand plain near the city of Rome approximately 150 km west of Glacial Lake Albany. Much of this dune and swale landscape is dominated by white pine forest. Major wetland types include pitch pine peat swamp, pine barrens vernal pond, black spruce – tamarack bog, highbush blueberry bog thicket, hemlock hardwood swamp, and red maple hardwood swamp. Collectively these wetlands support at least 325 vascular plant species, including 11 listed as state rare, endangered, or threatened (Kurczewski, 1999; NYSDEC, 2006).

2.2. Floristic survey and CoC

We selected 32 non-forested wetland basins (12 Pine Bush, 14 Wilton Preserve, 6 Rome Sand Plains) representing a range of habitat types and surrounding land cover (Table 1). We excluded wetland types lacking site replication and types not found across all three landscapes. We conducted floristic surveys at each site on two occasions in 2011, first during 03–30 June and again from 25 August to 17 September. The goal of our sampling approach was to record as many species as possible for each site within a reasonable amount of time.

For well-vegetated sites (>50% emergent cover) an observer and recorder team walked parallel lines (10–20 m apart depending on site area and logistics) oriented perpendicular to the site's longest axis and extending from one edge of the site to the other, with "edge" defined by upland, forest, road, etc. All vascular species detected from the lines were recorded. Additionally the team stopped approximately every 15 m to inspect the submerged, floating, and emergent vegetation within a 0.25 m^2 quadrat. At sites with emergent vegetation confined primarily to the littoral margin, the team meandered along the entire margin recording all vascular species observed and stopping every ca. 15 m for a closer inspection within the quadrat. In this manner the search effort was prorated by the emergent wetland area, with 21–69 quadrats per site per occasion (2350 total quadrats for the project). We sampled proportional to area under the assumption that variability and potentially richness would be greater across a larger area. Second author SKJ, an experienced field botanist and plant taxonomist, was the primary observer throughout the study.

All taxa were identified to species level, whenever possible. Specimen identifications were completed by SKJ using a variety of keys (Crow and Hellquist, 2000; Gleason and Cronquist, 1991; Magee and Ahles, 2007; Rhoads et al., 2007); nomenclature follows Kartesz (2011). All exotic taxa were given a CoC of zero, consistent with many FQA studies. For native taxa we used the mean of coefficients assigned by two of New York's leading botanists (see Bried et al., 2012). Discrepancies in CoC values between the two botanists were minimal (for this study the max difference = 2, mode

Table 1
Sample information listed in order of decreasing mean coefficient of conservatism (CoC).

Site	Geomorphic setting	Community type	Wetland size (ha)	Water level (cm)	Condition score	Rich	FQAI	Mean CoC
1	Connected	PBVP	0.20	80.9	9.4	10	18.50	5.85
2	Connected	PBVP	0.06	55.7	9.7	23	27.94	5.83
3	Connected	SM	0.10	51.1	10	10	18.18	5.75
4	Connected	SM	0.12	40.8	9.7	19	24.55	5.63
5	Connected	PBVP	0.14	21.3	9.7	14	20.18	5.39
6	Connected	SS	0.36	65.0	9.7	8	15.03	5.31
7	Connected	SS	0.16	84.2	9.7	15	20.53	5.30
8	Connected	SS	0.07	11.4	10	32	29.79	5.27
9	Connected	PBVP	0.30	71.6	9.4	18	22.04	5.19
10	Connected	SS	0.63	74.0	8.5	14	19.24	5.14
11	Connected	SM	0.30	62.1	9.7	17	20.62	5.00
12	Connected	PBVP	0.22	94.1	9.4	14	18.44	4.93
13	Connected	SM	0.32	33.6	8.8	21	22.59	4.93
14	Isolated	SM	0.08	48.8	8.8	22	22.60	4.82
15	Connected	SM	0.05	64.6	9.7	18	20.39	4.81
16	Connected	PBVP	0.17	52.3	7.9	24	23.47	4.79
17	Isolated	VP	0.54	88.6	8.8	21	21.93	4.79
18	Isolated	SS	0.06	53.5	8.8	18	19.45	4.58
19	Connected	SS	0.19	42.0	9.1	28	23.62	4.46
20	Connected	SS	0.08	42.4	8.2	8	12.39	4.38
21	Isolated	SM	0.27	50.0	9.4	5	9.62	4.30
22	Isolated	PBVP	1.37	61.3	8.5	25	21.30	4.26
23	Isolated	SS	0.16	45.6	8.5	29	22.40	4.16
24	Connected	SM	0.17	45.2	8.8	19	18.01	4.13
25	Isolated	VP	0.06	88.5	9.4	23	19.50	4.07
26	Isolated	VP	0.31	58.1	8.8	23	18.66	3.89
27	Isolated	PBVP	0.53	96.1	7.6	18	16.38	3.86
28	Isolated	VP	0.11	59.8	7.3	34	22.47	3.85
29	Isolated	VP	0.27	64.0	8.8	38	22.44	3.64
30	Isolated	SM	0.11	48.1	7.3	20	15.88	3.55
31	Isolated	VP	0.02	83.8	8.2	17	14.07	3.41
32	Isolated	SM	0.12	34.1	6.4	35	17.92	3.03

PBVP, pine barrens vernal pond; SM, wet sedge meadow; SS, shrub swamp; VP, woodland vernal pool; water level, mean surface water level recorded biweekly from mid May to mid August at the deepest point (up to a max of 1 m); condition score, land use metric (decreasing stress from 1 to 10) from the Rhode Island Rapid Assessment Method (Kutcher, 2011); rich, number of native and exotic taxa combined; FQAI, Floristic Quality Assessment Index; Mean CoC × sqrt(rich).

difference = 1, mean difference < 1) because they worked collaboratively to compile the final CoC list (available by registering at http://www.neiwpcc.org/nebawwg/login.asp).

2.3. Disturbance gradient

We used a metric from the Rhode Island Rapid Assessment Method (Kutcher, 2011) for measurement of site condition (Table 1). This approach follows the federal guidelines for establishing reference conditions for wetlands (Faber-Langendoen et al., 2009; U.S. EPA, 2002) and provides a relative index of condition based on estimation instead of interpretation. Scoring follows the idea that diverse human disturbances additively contribute to the degradation of general wetland condition (Fennessy et al., 2007). This tool was designed specifically for freshwater wetlands and is relevant to our study area because Rhode Island has many glacial outwash plains and Aeolian sand deposits (T. Kutcher, pers. comm., 14 February 2013).

We estimated the degradation of buffers (% cultural land cover within 100-foot buffer) and intensity of surrounding land use (weighted average within 500-foot buffer) from visual observations on site and using Google Earth (© 2011 Google Inc.); see Kutcher (2011) for stress definitions and scoring. Substantially more variation in floristic quality was explained by the land use metric alone (simple linear regression $r^2 = 0.63$) than by the combined buffer and land use metrics ($r^2 = 0.41$). We therefore used only the land use scores (continuous range from 1 to 10, where 10 indicates lowest stress) to represent condition for analyses. A highly accurate disturbance measurement was not required because we were not interested in the index-condition association per se, but rather the association relative to environmental categories and gradients.

2.4. Environmental categories and gradients

We used environmental variables of geomorphic setting, primary community type, approximate wetland area, and maximum surface water level (Table 1). Some of the sites are isolated depressions fed by precipitation and potentially groundwater, whereas others grade into a larger wetland complex of red maple swamp, shallow emergent marsh, or other communities. We refer to the latter category as "connected" wetlands following Kutcher (2011); a few of these sites are associated with perennial streams and floodplain forests. We therefore broadly classified sites into isolated vs. connected geomorphic settings; see Brinson (1993) for information on wetland hydrogeomorphic classification.

Referring to natural community descriptions by Edinger et al. (2002), we categorized each site as pine barrens vernal pond, sedge meadow, shrub swamp, or vernal pool. The pine barrens vernal pond community is of special concern in New York State with only 11 documented occurrences (Bried and Edinger, 2009). Characteristic vegetation includes *Carex canescens* L., *Cephalanthus occidentalis* L., *Chamaedaphne calyculata* (L.) Moench., *Dulichium arundinaceum* (L.) Britt., *Scirpus cyperinus* (L.) Kunth, *Spiraea alba* Du Roi, *Thelypteris palustris* Schott, and *Vaccinium corymbosum* L. Wet sedge meadows have organic, permanently saturated soils and are dominated by tussock-forming sedges such as *Carex stricta* Lam. Associated vegetation may include *Calamagrostis canadensis* (Michx.) Beauv., *C. canescens*, and *Carex vesicaria* L., along with occasional shrubs (*C. calyculata*, *S. alba*, *Spiraea tomentosa* L.). Shrub swamps are broadly defined and variable across the region but tend to have a mineral to mucky substrate and in this study consisted of dense *C. occidentalis*, *Ilex mucronata* (L.) M. Powell, Savol. & S. Andrews, or *V. corymbosum*. Vernal pools are hydric soil depressions often with high levels of allochthonous input from surrounding woodlands. Examples of plant taxa associated with vernal pools in the study area include *Glyceria* R. Br., *Lemna minor* L., and *Ludwigia palustris* (L.) Ell.

All study sites have seasonal hydrology, most maintain standing water until late summer, and in wet years (including the year of this study) hold water throughout the year (J. T. Bried, pers. observations). Surface water levels were recorded biweekly from mid-May to mid-August off a stream gauge placed at the approximate deepest point of the basin or up to a max of 1 m water depth (due to gauge height and accessibility); depths greater than gauge height (1 m) were recorded at only six sites on select surveys. Area was estimated by walking the perceived wetland boundary with a GPS unit at high water level (i.e., during spring thaw). For isolated depressions we used the land–water interface as the boundary, and for connected sites we measured the area bound by upland, mature forest, or man-made edge features.

2.5. Analysis

Study sites were not randomly chosen from the wetland populations in each landscape, and the sample sizes within categories were often small (<10 sites). We therefore relied on a randomization framework to analyze the data. This framework does not rely on a random sampling assumption, does not require that data are sampled from a specified probability distribution (e.g., normal), and may allow valid statistical inference from small or unbalanced data sets (Bried and Ervin, 2011; Edgington and Onghena, 2007; Manly, 1997). We used two common indices of site floristic quality for the analyses (Spyreas et al., 2012; Taft et al., 2006): (1) mean coefficient of conservatism (Mean CoC), and (2) traditional Floristic Quality Assessment Index (FQAI) defined as Mean CoC multiplied by square root of taxa richness (including natives and exotics); some researchers have presented the FQAI by the algebraically equivalent sum of CoC divided by square root of taxa richness.

As with any bioindicator, Mean CoC and FQAI are expected to show a negative linear response to increasing human disturbance. We used simple linear modeling with a randomly generated error distribution for the model fit (r^2) and slope parameter. We assumed that an uncertainty range derived from computer-intensive resampling is more likely to capture reality than a single fit value or parameter estimate. Using Resampling Stats v4.0 (written by S. Blank, ©2012 statistics.com, Resampling Stats Inc., Arlington, VA), the data were sampled once with replacement, keeping the index and condition scores together for each site (i.e., shuffling rows as units). We then invoked the program's simple linear regression analysis and resampled the output of interest (model fit, slope parameter) for 1000 iterations. Assuming we representatively sampled the non-forested wetland populations in each landscape and habitat category, the 2.5th and 97.5th percentiles of the resampled distribution were interpreted as an uncertainty range for the true model fit and slope.

We first checked for differences in the early vs. late season surveys to determine if the data could be pooled for evaluating the remaining environmental variables. For this analysis we resampled both the model fit (r^2) and slope parameter from the regression of index over condition in each survey. For continuous gradients (area, depth) we used the model fit range as a performance metric, seeing how the fit of index over condition compared to the fit of index over area and index over depth. We also checked if residuals from the regression of index over condition linearly corresponded to the area and depth gradients. Site 22 was more than twice the area of the next largest site (see Table 1) and was thus removed from the area analysis as an obvious outlier. Also for area, we repeated the test retaining only the 13 sites perceived to have relatively discrete boundaries (i.e., the isolated depressions, with site 22 removed), and used the same analysis to check for species–area relationships before and after site removal.

For categorical variables (geomorphic setting, community type) we compared slope uncertainty ranges for the index-condition relationship. If ranges overlapped by >50% among the two geomorphic settings or any pair of the three community types, we assumed the index-condition relationship was more consistent than not among categories and that the index responds to wetland condition independent of category (i.e., parallel slopes). We then looked at differences among category means using randomization tests analogous to one-way ANOVA (for community type, because of three classes) and the unpaired *t*-test (for geomorphic setting, because of two classes). The randomization *t*-test was done in the same manner as described in Bried and Edinger (2009) and the randomization ANOVA as described in Bried and Hecht (2011), running 1000 iterations to evaluate significance.

3. Results

A cumulative total of 153 vascular plant taxa were found across the sites and survey occasions. Of these, 128 taxa were identified to specific or infraspecific levels, which included 11 exotic species. Two additional taxa identified only at the generic level could also be used (i.e., 130 taxa for analysis) because one was exotic (*Sonchus*), thus CoC = 0 automatically, and the other was keyed out to two species possibilities (in *Verbena*) with the same CoC.

Using a Mantel correlation test with the Sørensen distance measure, we found that community composition was not highly correlated (r = 0.67) between the early and late season surveys. Nevertheless, site floristic quality varied with site condition similarly between the two surveys (Fig. 1). We therefore decided to pool the surveys for analysis of the remaining variables. The Mean CoC provided a better relationship (higher model fit, narrower uncertainty) than FQAI.

The FQAI did not show linear variation with site condition. Point estimated model fit in the full data set was only 0.06, with an uncertainty (2.5th and 97.5th percentiles) from zero to 0.30 based on computer-intensive resampling. The fit was also poor within the separate geomorphic settings and community types ($r^2 < 0.1$, except 0.24 in the shrub swamp community). Additionally, slope for the FQAI-condition relationship in each habitat category showed wide uncertainty and potentially negative direction (Table 2). In light of these problems, subsequent analyses were done without considering the FQAI.

Results involving Mean CoC suggest a clear relationship with site condition and no interference from wetland area or mean water level (Fig. 2). There was possible area interference using only the 13 isolated sites (with site 22 removed as an area outlier), but the wide uncertainty range (r^2 = 0.00–0.55) makes this unclear. Similar results were found in regressions of the Mean CoC residuals over area (all sites: r^2 = 0.00–0.21, isolated sites: r^2 = 0.00–0.52) and depth (r^2 = 0.00–0.15). There was little evidence of a species–area

Table 2
Linear modeling of the Floristic Quality Assessment Index over increasing wetland condition within each geomorphic setting (connected vs. isolated sites) and community type, giving the point estimate for model fit (r^2) and a likely range for the slope parameter (based on computer-intensive resampling).

Habitat category	Slope range	r^2
Connected	−1.36 to 6.00	0.08
Isolated	−3.19 to 3.02	0.00
Pine barrens vernal pond	−2.32 to 5.02	0.09
Wet sedge meadow	−2.05 to 2.31	0.04
Shrub swamp	−3.58 to 9.26	0.24
Woodland vernal pool	−3.30 to 12.19	0.00

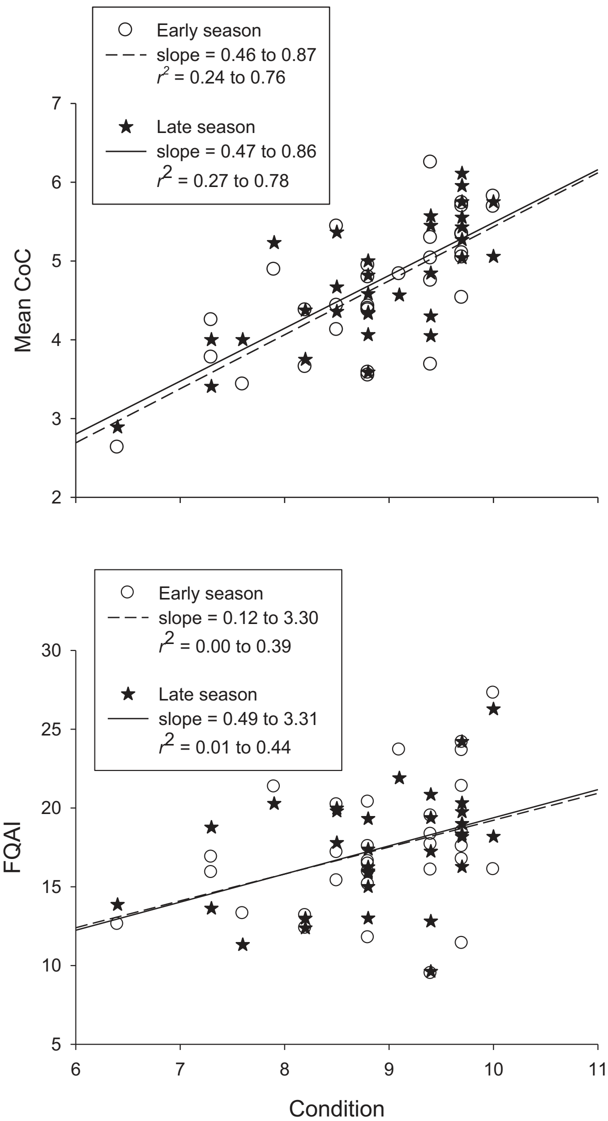

Fig. 1. Relationship of the mean coefficient of conservatism (top graph) and traditional Floristic Quality Assessment Index (bottom graph) to wetland ecological condition (rapid assessment land use metric) and seasonal variability as defined by survey periods occurring early (03–30 June) and late (25 August–17 September) in the growing season. Computer-intensive resampling was used to estimate an uncertainty range (2.5th and 97.5th percentiles) for the linear model fit (r^2) and slope parameter; best-fit lines correspond to the initial regressions (prior to randomizations).

relationship with (n = 31 sites, r^2 = 0.00–0.12) and without (n = 13 sites, r^2 = 0.00–0.30) the connected sites.

Although connected wetlands contained higher Mean CoC than isolated wetlands (5.12 vs. 4.01 on average, P < 0.001 using a randomization t-test), the slope of Mean CoC over condition was similar for both categories (Fig. 3A). There was no evidence that Mean CoC varied among the community types (P = 0.549 using randomization ANOVA) and the slopes were generally similar among categories, with greater than 50% overlap among ranges (Fig. 3B). The six vernal pools were excluded from Fig. 3B due to weak model fit (r^2 = 0.01) and potentially negative slope (range of −0.22 to 1.34, based on computer-intensive resampling).

4. Discussion

Natural variability and ecological interference is a significant concern for application of the FQA approach to site quality assessment, including regulatory assessment of wetland condition. An important question is whether the relationship between floristic quality and human disturbance is stronger than the influence of underlying natural variability. Our study finds that FQA in the form of average species conservatism (Mean CoC) can signal human disturbance over natural variability. It further suggests the signal can remain constant even as floristic quality varies among habitat categories, as demonstrated by the geomorphic setting analysis. Ideally, however, both the slopes and means should vary minimally. On average the connected wetlands showed inherently greater floristic quality than isolated wetlands, suggesting floristic quality comparisons between these categories should be avoided. This case study does not preclude consideration of stratified sampling designs or analyses to filter out heterogeneity, especially in high-stakes conservation efforts such as comparing restoration trajectories against reference standards (Balcombe et al., 2005; Matthews et al., 2009).

To properly evaluate FQA (or any bioindicator) it is important for the study design to capture both potential ecological interference (i.e., natural variation) and a gradient of human disturbance. In a prairie pothole wetland complex (Euliss and Mushet, 2011), plant-based metrics (including FQA metrics) varied among sites and years but the anthropogenic influence remained constant, precluding a test of the signal to noise ratio. Additionally, Euliss and Mushet (2011) declared high variability in the individual metric scores when in fact the Mean CoC was more or less constant (3.6–4.4 among sites, 3.8–4.7 among years), especially compared to the richness-based metrics (including FQAI). Apparently few studies have combined FQA, varying disturbance, and varying environmental conditions in the same analysis. For 75 isolated depressional marshes in Florida, Cohen et al. (2004) found the Mean CoC to strongly associate with disturbance independent of ecoregions. For 53 primarily riverine and depressional wetlands in northern and central Mississippi, Ervin et al. (2006) reported no interaction effect between geomorphic class and anthropogenic activity on the FQAI (Mean CoC was not used). Similar to our findings with Mean CoC, only the FQAI in Ervin et al. (2006) and not the FQAI-disturbance relationship differed between geomorphic classes. Contrary to expectations it seems FQA is proving robust against high background variation within certain wetland study systems.

The FQAI did not linearly correspond to the condition gradient and thus it could not be evaluated. Computer-intensive resampling in each habitat category even suggested the possibility for a negative slope (see Table 2), indicating that FQAI is an ineffective measure. Many researchers have found stronger performance with Mean CoC than FQAI, and have warned of problems with using the richness multiplier in FQAI (Bourdaghs et al., 2006; Cohen et al., 2004; Ervin et al., 2006; Matthews, 2003; Matthews et al., 2005; Miller and Wardrop, 2006; Taft et al., 2006). Richness can overwhelm the CoC component and many factors unrelated to human disturbance can influence richness, such as wetland area, seasonality, sampling effort, and ecological phenomena (e.g., intermediate disturbance hypothesis). Apparently there is misconception that the richness multiplier reduces assessment bias by accounting for species–area relationships (e.g., Medley and Scozzafava, 2009). On the contrary, a species–area relationship could serve as the primary source of bias in a condition assessment. If richness controls the FQAI and area drives richness, then transitively the FQAI may reflect area instead of condition.

Both the index scores and the index-condition relationship held among the pine barrens vernal pond, wet sedge meadow, and shrub swamp community types. However, it was difficult to qualitatively group these wetlands into single categories if they occurred in larger wetland complexes or contained multiple sub-communities (Bried and Edinger, 2009). For example, at Site 16 the mat-forming *Drosera rotundifolia* L. was restricted to an opening within a dense

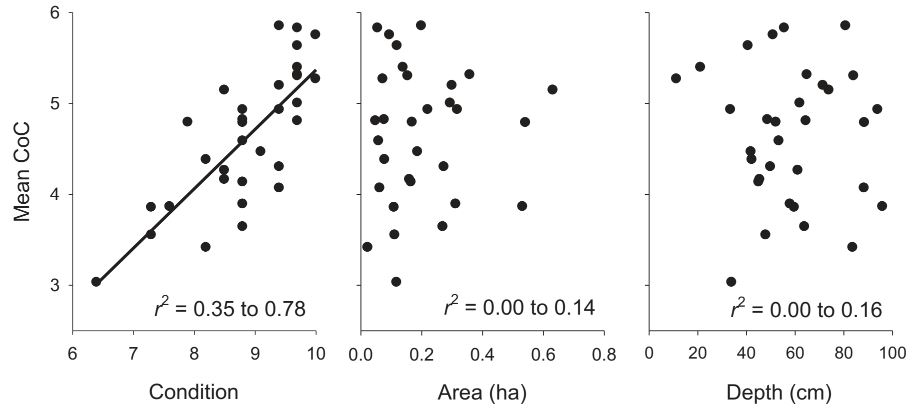

Fig. 2. Variation in the mean coefficient of conservatism (Mean CoC) over gradients in wetland condition (rapid assessment land use metric), area, and water level. Computer-intensive resampling was used to estimate an uncertainty range (2.5th and 97.5th percentiles) for the model fit (r^2); the best-fit line corresponds to the initial regression (prior to randomizations). Site 22 was removed from the area graph as an outlier (Section 2.5).

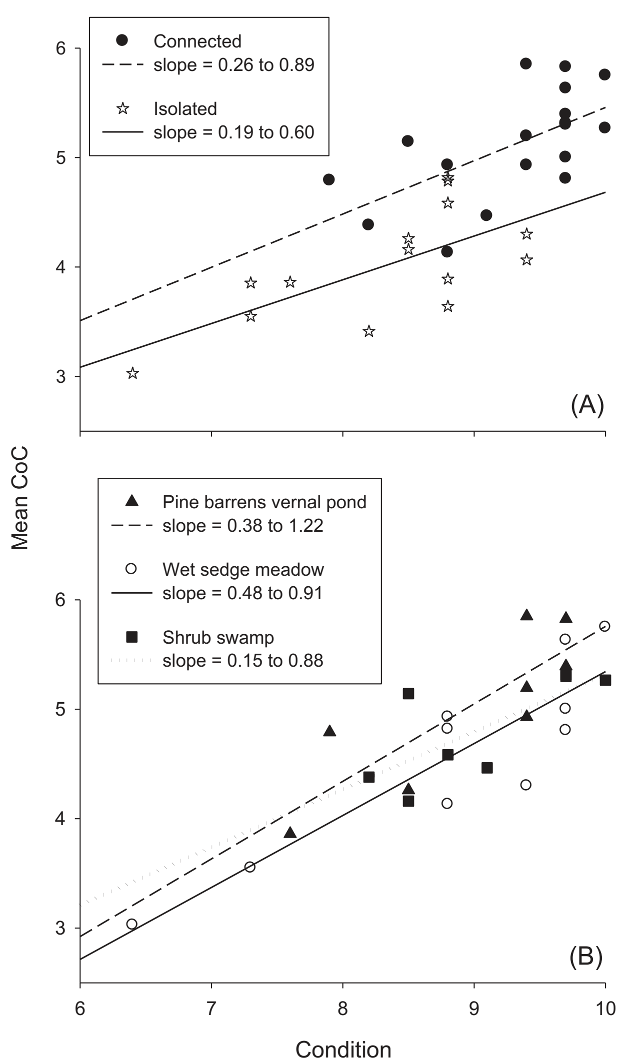

Fig. 3. Relationship between the mean coefficient of conservatism (Mean CoC), wetland condition (rapid assessment land use metric), and habitat categories defined by geomorphic setting (A) and primary community type (B). Computer-intensive resampling was used to estimate an uncertainty range (2.5th and 97.5th percentiles) for the slope parameter; best-fit lines correspond to the initial regressions (prior to randomizations). The six vernal pool sites were removed from panel B (see Section 3).

sub-shrub zone dominated by *C. calyculata*, which in turn was enveloped by a dense tall-shrub zone of *C. occidentalis* and *V. corymbosum*. Other species such as *C. canescens* dominated localized areas of the site and created additional sub-communities. As such, Site 16 and other alleged pine barrens vernal ponds can show floristic overlap with sedge meadows and shrub swamps and may lack diagnostic species (Bried and Edinger, 2009), making qualitative categorization difficult. Although the descriptions from Edinger et al. (2002) represent best available information, proper assessment and classification of these enigmatic wetlands may require a multi-scale, multivariate approach (e.g., Johnston et al., 2009).

Other caveats to our analysis include inherently low replication (<10) in select categories, some difficulty with determination of connected wetland size, and large annual variation in hydrology. Woodland vernal pools are quite common across the study region yet contained the fewest replicates (6 sites) in our sample and may have been underrepresented. With regards to area, the connected wetlands did not have spatially discrete surface waters and lacked prominent edges (forest, road, etc.). However, we analyzed area with and without these sites and did not find evidence for area interference on index performance. In terms of hydrology, our biweekly water level monitoring captured seasonal variation but was a single year snapshot. Hydrology of seasonal wetlands may change drastically from year to year (Brooks, 2004; Mitsch and Gosselink, 2007), potentially altering the depth gradient used for analysis in any given year. All study sites remained inundated during our 2011 field season due to above-normal rainfall, whereas in 2010 and 2012 many dried up by late summer (Albany Pine Bush Preserve Commission, unpubl. data).

Multiple sampling occasions during the growing season will help ensure a comprehensive plant checklist and improve floristic quality estimates. However, index scores may vary seasonally depending on recruitment, die off, and sampling error. It seems that FQA must withstand seasonal bias yet also separate disturbance or management effects from longer-term successional trends (Balcombe et al., 2005; Brudvig et al., 2007; Matthews et al., 2009; McIndoe et al., 2008; Spyreas et al., 2012). Temporal bias in FQA applications is often mentioned but seldom evaluated (Francis et al., 2000; Matthews, 2003). We found no evidence of seasonal bias even though some species turnover and/or sampling error was suggested between surveys (based on a Mantel correlation test). This suggests that species gained in the late season survey replaced the CoC values of species lost in the early season survey, or it simply shows that FQA is robust to negligible species turnover and sampling error. However, in a typical precipitation year standing water tends to disappear at many of these sites by late summer (J. T. Bried, pers. observations). This can lead to greater encroachment of tolerant annuals and other less conservative taxa (Euliss and Mushet, 2011), reducing floristic quality and potentially confounding the association with human disturbance. Most FQA development and evaluation has taken place in the colder and wetter regions of the United States (Medley and Scozzafava, 2009; Bried et al., 2012; Chamberlain and Ingram, 2012). We suspect that FQA applications in drought-prone regions may incur greater seasonal bias and interannual variation in performance.

Within certain non-forested wetland systems, the FQA method is proving effective despite high background variation (this study; Cohen et al., 2004; Ervin et al., 2006). Perhaps this is a function of how CoC are assigned, typically for each state as a whole by mentally “averaging” the species' behavior (Taft et al., 1997; Bried et al., 2012). Although statewide averaging may lose ecological precision, it accounts inherently for the varied growing conditions under which a species may be observed. As such, stratifying the design or analysis by ecologically meaningful units (e.g., ecoregions, vegetation types), which themselves are subject to human error, may not improve FQA. That is, not unless the CoC were originally designated to fit those units. Stratifying CoC development by, say, ecoregions may help capture natural variability up front and could extend FQA into states lacking CoC (Medley and Scozzafava, 2009), but this has seldom been attempted (but see Chamberlain and Ingram, 2012). Botanists may not have the experience, or recollection of their experience, to assign ecologically specific CoC. And with multiple CoC for a species, practitioners would have to be careful in using the correct one for their application. Rather than pursue ecologically stratified CoC or automatically assuming FQA is overly sensitive to natural variability, we recommend further efforts to identify situations in which FQA is robust.

Acknowledgments

We thank Amanda Dillon and Arianne Messerman for helping with vegetation surveys, and the many volunteers (especially Barb Hager and Grace Barber) who helped monitor water levels. We also thank the Albany Pine Bush Preserve Commission, the Wilton Wildlife Preserve & Park, and The Nature Conservancy Eastern New York Chapter for site access and their interest in the project. The manuscript has benefitted greatly from the insightful criticism provided by several anonymous reviewers. Financial support was provided by the New York State Department of Environmental Conservation, Environmental Protection Fund.

References

Andreas, B.K., Mack, J.J., McCormac, J.S., 2004. Floristic Quality Assessment Index (FQAI) For Vascular Plants and Mosses for the State of Ohio. Ohio Environmental Protection Agency, Division of Surface Water, Wetland Ecology Group, Columbus, OH, USA, pp. 219.

Balcombe, C.K., Anderson, J.T., Fortney, R.H., Rentch, J.S., Grafton, W.N., Kordek, W.S., 2005. A comparison of plant communities in mitigation and reference wetlands in the mid-Appalachians. Wetlands 25, 130–142.

Barnes, J.K., 2003. Natural History of the Albany Pine Bush: Albany and Schenectady Counties, New York. Albany, New York, USA, New York State Museum Bulletin #502.

Bourdaghs, M., Johnston, C.A., Regal, R.R., 2006. Properties and performance of the floristic quality index in Great Lakes coastal wetlands. Wetlands 26, 718–735.

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

Bried, J.T., Edinger, G.J., 2009. Baseline floristic assessment and classification of pine barrens vernal ponds. J. Torrey Bot. Soc. 136, 128–136.

Bried, J.T., Ervin, G.N., 2011. Randomized intervention analysis for detecting nonrandom change and management impact: dragonfly examples. Ecol. Indic. 11, 535–539.

Bried, J.T., Hecht, J.A., 2011. Comparison of herbicide treatments to control native aspen encroachment in pine barrens. Nat. Areas J. 31, 288–293.

Bried, J.T., Strout, K.M., Portante, T., 2012. Coefficients of conservatism for the vascular flora of New York and New England: inter-state comparisons and expert opinion bias. Northeast. Nat. Spec. Issue 19 (Special Issue 6), 101–114.

Brinson, M.M., 1993. A hydrogeomorphic classification for wetlands. In: Wetlands Research Program Technical Report. U.S. Army Corps of Engineers, Washington, DC.

Brooks, R.T., 2004. Weather-related effects on woodland vernal pool hydrology and hydroperiod. Wetlands 24, 104–114.

Brudvig, L.A., Mabry, C., Miller, J.R., Walker, T.A., 2007. Evaluation of central North American prairie management based on species diversity, life form, and individual species metrics. Conserv. Biol. 21, 864–974.

Chamberlain, S.J., Ingram, H.M., 2012. Developing coefficients of conservatism to advance floristic quality assessment in the Mid-Atlantic region. J. Torrey Bot. Soc. 139, 416–427.

Cohen, M.J., Carstenn, S., Lane, C.R., 2004. Floristic quality indices for biotic assessment of depressional marsh condition in Florida. Ecol. Appl. 14, 784–794.

Cretini, K.F., Visser, J.M., Krauss, K.W., Steyer, G.D., 2012. Development and use of a floristic quality index for coastal Louisiana marshes. Environ. Monit. Assess. 184, 2389–2403.

Crow, G.E., Hellquist, C.B., 2000. Aquatic and Wetland Plants of Northeastern North America, vols. 1 and 2. The University of Wisconsin Press, Madison, WI, USA.

Edgington, E.S., Onghena, P., 2007. Randomization Tests, 4th ed. Chapman & Hall/CRC, Boca Raton, FL.

Edinger, G.J., Evans, D.J., Gebauer, S., Howard, T.G., Hunt, D.M., Olivero, A.M., 2002. Ecological Communities of New York State. A Revised and Expanded Edition of Carol Reschke's Ecological Communities of New York State. (Draft for review), 2nd ed. New York Natural Heritage Program, Albany, NY, USA.

Ervin, G.N., Herman, B.D., Bried, J.T., Holly, D.C., 2006. Evaluating non-native species and wetland indicator status as components of wetlands floristic assessment. Wetlands 26, 1114–1129.

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

Faber-Langendoen, D., Lyons, R., Comer, P., 2009. Developing Options For Establishing Reference Conditions for Wetlands Across the Lower 48 States. NatureServe, Arlington, VA, USA.

Fennessy, M.S., Jacobs, A.D., Kentula, M.E., 2007. An evaluation of rapid methods for assessing the ecological condition of wetlands. Wetlands 27, 543–560.

Francis, C.M., Austen, M.J.W., Bowles, J.M., Draper, W.B., 2000. Assessing floristic quality in southern Ontario woodlands. Nat. Areas J. 20, 66–77.

Gleason, H.A., Cronquist, A., 1991. Manual of Vascular Plants of Northeastern United States and Adjacent Canada, 2nd ed. New York Botanical Garden, New York, USA.

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

Johnston, C.A., Zedler, J.B., Tulbure, M.G., Frieswyk, C.B., Bedford, B.L., Vaccaro, L., 2009. A unifying approach for evaluating the condition of wetland plant communities and identifying related stressors. Ecol. Appl. 19, 1739–1757.

Kartesz, J.T., 2011. The Biota of North America Program: North American Plant Atlas (http://www.bonap.org/MapSwitchboard.html). Chapel Hill, N.C. [maps generated from Kartesz, J.T. 2010. Floristic Synthesis of North America, Version 1.0. Biota of North America Program. (in press)].

Kurczewski, F.E., 1999. Historic and prehistoric changes in the Rome, New York pine barrens. Northeast. Nat. 6, 327–340.

Kutcher, T.E., 2011. Integrating rapid assessment with biological and landscape indicators of freshwater wetland condition. In: Rhode Island Natural History Survey. University of Rhode Island, Kingston, Rhode Island, USA, 48 pp.+appendices.

Landi, S., Chiarucci, A., 2010. Is floristic quality assessment reliable in human-managed ecosystems? Syst. Biodivers. 8, 864–881.

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

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

Magee, D.W., Ahles, H.E., 2007. Flora of the Northeast: A Manual of the Vascular Flora of New England and Adjacent New York, 2nd ed. University of Massachusetts Press, Amherst, MA, USA.

Manly, B.F.J., 1997. Randomization, Bootstrap and Monte Carlo Methods in Biology, 2nd ed. Chapman & Hall, London, UK.

Matthews, J.W., 2003. Assessment of the floristic quality index for use in Illinois, USA, wetlands. Nat. Areas J. 23, 53–60.

Matthews, J.W., Tessene, P.A., Wiesbrook, S.M., Zercher, B.W., 2005. Effect of area and isolation on species richness and indices of floristic quality in Illinois, USA wetlands. Wetlands 25, 607–615.

Matthews, J.W., Spyreas, G., Endress, A.G., 2009. Trajectories of vegetation-based indicators used to assess wetland restoration. Ecol. Appl. 19, 2093–2107.

Mattox, J.E., 1994. Wetland Vascular Flora of the Pine Bush, Albany and Schenectady Counties, New York State, in the 19th and 20th Centuries. Bard College, New York, USA (M.S. thesis).

McIndoe, J.M., Rothrock, P.E., Reber, R.T., 2008. Monitoring tallgrass prairie restoration performance using Floristic Quality Assessment. Proc. Indiana Acad. Sci. 117, 16–28.

Medley, L., Scozzafava, M., 2009. Moving toward a national floristic quality assessment: considerations for the EPA National Wetland Condition Assessment. Natl. Wetlands Newsl. 31 (1), 6–9.

Miller, S.J., Wardrop, D.H., 2006. Adapting the floristic quality assessment index to indicate anthropogenic disturbance in central Pennsylvania wetlands. Ecol. Indic. 6, 313–326.
Mitsch, W.J., Gosselink, J.G., 2007. Wetlands, 4th ed. John Wiley & Sons, New York, USA.
Mushet, D.M., Euliss Jr., N.H., Shaffer, T.L., 2002. Floristic quality assessment of one natural and three restored wetland complexes in North Dakota, USA. Wetlands 22, 126–138.
Nichols, J.D., Perry, J.E., DeBerry, D.A., 2006. Using a floristic quality assessment technique to evaluate plant community integrity of forested wetlands in southeastern Virginia. Nat. Areas J. 26, 360–369.
NYSDEC, 2006. Rome Sand Plains Consolidated Unit Management Plan. New York State Department of Environmental Conservation, Division of Lands & Forests, City of Rome, Oneida County, NY, USA.
Packham, J.R., Willis, A.J., 1976. Aspects of the ecological amplitude of two woodland herbs, *Oxalis acetosella* L. and *Galeobdolon luteum* Huds. J. Ecol. 64, 485–510.
Rader, R.B., Batzer, D.P., Wissinger, S.A. (Eds.), 2001. Bioassessment and Management of North American Freshwater Wetlands. John Wiley & Sons, New York, USA.
Reiss, K.C., Brown, M.T., Lane, C.R., 2010. Characteristic community structure of Florida's subtropical wetlands: the Florida wetland condition index for depressional marshes, depressional forested, and flowing water forested wetlands. Wetl. Ecol. Manage. 18, 543–556.
Rhoads, A.F., Block, T.A., Anisko, A., 2007. The Plants of Pennsylvania: An Illustrated Manual. University of Pennsylvania Press, Philadelphia, PA, USA.
Rooney, T.P., Rogers, D.A., 2002. The modified floristic quality index. Nat. Areas J. 22, 340–344.
Spyreas, G., Meiners, S.J., Matthews, J.W., Molano-Flores, B., 2012. Successional trends in floristic quality. J. Appl. Ecol. 49, 339–348.
Stein, E.D., Fetscher, A.E., Clark, R.P., Wiskind, A., Grenier, J.L., Sutula, M., Collins, J.N., Grosso, C., 2009. Validation of a wetland rapid assessment method: use of EPA's level 1-2-3 framework for method testing and refinement. Wetlands 29, 648–665.
Swink, F., Wilhelm, G., 1994. Plants of the Chicago Region, 4th ed. Indiana Academy of Science, Indianapolis, IN, USA.
Taft, J.B., Wilhelm, G.S., Ladd, D.M., Masters, L.A., 1997. Floristic quality assessment for vegetation in Illinois: a method for assessing vegetation integrity. Erigenia 15, 3–95.
Taft, J.B., Hauser, C., Robertson, K.R., 2006. Estimating floristic integrity in tallgrass prairie. Biol. Conserv. 131, 42–51.
Tu, W., Gao, X., Wu, N., Liu, S., 2009. A novel method for floristic quality assessment using the vegetation of the Jiuding Mountain, Sichuan, China as an example. Nord. J. Bot. 27, 61–68.
U.S. EPA, 2002. Methods for Evaluating Wetland Condition. Office of Water, U.S. Environmental Protection Agency, Washington, DC, EPA-822-R-02-014.
Wilhelm, G., Ladd, D., 1988. Natural areas assessment in the Chicago region. Trans. N. Am. Wildl. Nat. Res. 53, 361–375.