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Adapting the floristic quality assessment index to indicate anthropogenic disturbance in central Pennsylvania wetlands

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Abstract

The floristic quality assessment index (FQAI) is an evaluation procedure that uses measures of ecological conservatism (expressed numerically as a coefficient of conservatism or *C* value) and richness of the native plant community to derive a score (*I*) that is an estimate of habitat quality. We evaluated the ability of the FQAI to indicate the level of anthropogenic disturbance in headwater wetlands in the Ridge and Valley physiographic province of central Pennsylvania. *I* scores were highly correlated with disturbance, with scores generally decreasing with increasing levels of disturbance. However, we found that *I* did not equally characterize sites with differing species richness. *I* scores were higher for sites with greater intrinsic native species, regardless of other influences on floristic quality. To eliminate sensitivity to species richness, we evaluated sites using mean conservatism values ($\bar{C}$) and a variant of the *I* score (adjusted FQAI, hereafter cited as I') that considered both the contribution of non-native species and the intrinsic low species richness of high quality forested wetlands. $\bar{C}$ values were more highly correlated with disturbance than *I* scores; however, site assessments based on $\bar{C}$ values alone were misleading. I' scores were also more highly correlated with disturbance than *I* scores and were robust to differences in native species richness. Therefore, we offer I' as an improved formulation of the index that, in addition to serving as a useful condition assessment tool, addresses two problematic issues that have plagued the FQAI since its conception: the overwhelming influence of the species richness multiplier and the role of non-native species in floristic assessment.
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Keywords: Biological indicator; Condition assessment; Headwater wetland; Ridge and Valley physiographic province; Floristic quality assessment index (FQAI); Species richness

1. Introduction

Under Section 305(b) of the Clean Water Act, state regulatory agencies and tribal entities are required to develop water quality standards for their aquatic systems and monitor these systems routinely for compliance. While stream monitoring programs are

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generally well established, similar initiatives for wetlands have been slow to emerge, primarily due to a scarcity of rapid and effective condition assessment methods. Although many states are moving toward developing standards for wetlands, there is concern that traditional monitoring programs will be largely inadequate. This is because these programs rely largely on levels of chemical constituents as indicators of impairment (Danielson, 1998) and chemical indicators are poorly suited to detect the types of stressors that typically impact wetlands, including non-point source runoff, changes in land use (alteration and fragmentation), invasion by non-native species, and hydrologic modifications (Karr and Chu, 1999; Danielson, 1998). Recently, there has been increased interest in developing biological criteria for wetland assessment (Carlisle et al., 1999; U.S. E.P.A., 2002; Mack, 2004). As biological monitoring becomes more widespread, there will be a concomitant need for assessment tools that can rapidly and effectively evaluate condition.

One assessment tool that may prove useful in measuring condition is the floristic quality assessment index (FQAI) developed by Swink and Wilhelm (1979, 1994). The FQAI uses measures of ecological conservatism and richness of the native plant community to derive an estimate of habitat quality (referred to as *I*). In the method, ecological conservatism is expressed numerically as a coefficient of conservatism or *C* value. Conservatism values range from 0 to 10 and are assigned a priori based on an individual plant species' fidelity to specific habitat types and its tolerance to both natural and anthropogenic disturbance (Taft et al., 1997; Andreas et al., 2004). In general, plants that are widespread with broad tolerances (generalist species) are given lower values than plants with more narrow distributions and tolerances (conservative species). As originally conceived, non-native species were assigned zero and not used to compute the index, however more recent studies have suggested the inclusion of non-natives as an alternative to the traditional approach (Fennessy et al., 1998a,b; Lopez and Fennessy, 2002; Rooney and Rogers, 2002; Bernthal, 2003; Andreas et al., 2004; Rothrock, 2004). Once devised, conservatism values are averaged and used to weight species richness. The FQAI, therefore, can be conceptualized as a variation on more conventional weighted averaging techniques (Andreas et al., 2004).

The FQAI was first proposed in the late 1970s as a method for assessing habitat quality in the Chicago area (Swink and Wilhelm, 1979). Since the mid-1990s, regionalized versions of the method have been developed for Missouri (Ladd, 1993), northern Ohio (Andreas et al., 2004), southern Ontario (Oldham et al., 1995), Michigan (Herman et al., 1997), Illinois (Taft et al., 1997), North Dakota (Northern Great Plains Floristic Quality Assessment Panel, 2001), Wisconsin (Bernthal, 2003), and Indiana (Rothrock, 2004). Some of these early studies provided anecdotal evidence to suggest that the FQAI may be a good predictor of condition and more recent studies have explored its utility in this regard. Studies from Ohio have demonstrated a strong correlation between *I* scores and relative disturbance rank for riparian and depressional wetlands (Fennessy et al., 1998b; Lopez and Fennessy, 2002) and emergent, scrub-shrub, and forested wetlands (Mack, 2004). Francis et al. (2000) tested the FQAI in deciduous woodlands in southern Ontario and reported a slight decrease in scores with increasing disturbance.

We evaluated the FQAI as a tool for characterizing disturbance among headwater wetlands in the Ridge and Valley physiographic province of central Pennsylvania. The Ridge and Valley encompasses 13,080 km^2 in Pennsylvania. There are 17,403 stream kilometers of which 13,089 or 75% are first and second order (Environmental Resources Research Institute, 1998). Headwater wetlands are defined as wetlands associated with first and second order streams, and therefore comprise a significant portion of the wetland resource.

Urbanization and agriculture are the primary types of anthropogenic, landscape-level disturbances affecting wetlands in the region (Cole et al., 1997). These activities degrade wetland systems by increasing sediment and nutrient inputs and altering hydrologic patterns. Plant community composition has been shown to respond to these stressors in predictable ways (Taft et al., 1997). For example, a decrease in both species richness (Jurik et al., 1994; Dittmar and Neely, 1999) and diversity (Dittmar and Neely, 1999) has been reported in response to sedimentation, while nutrient enrichment favors more tolerant non-native or weedy native species (Hobbs and Huenneke, 1992). Because the FQAI combines measures of richness with individual plant tolerances, the index should be responsive to these stressors and to the landscape-level changes that cause them.

The objectives of this study were to:

- evaluate the ability of the FQAI to characterize headwater wetlands along a gradient of human disturbance;
- examine the relationship between *I* scores and the landscape and individual site level measures of anthropogenic disturbance that comprise the disturbance gradient; and
- examine the relationship between *I* scores and two other independent measures of landscape fragmentation: road density and distance to nearest wetland.

We selected these latter two measures because the presence of roads and absence of neighboring wetlands, which are often converted to agriculture or other land uses in developed landscapes (Wickham et al., 2002), are both indicative of disturbance. Roads, in particular, have been shown to serve as conduits for plant dispersal, including non-native species (Spellerberg, 1998). The relationship between *I* scores and these variables, therefore, can provide an independent validation of the index as an evaluation tool.

2. Methods

2.1. Study area

The Ridge and Valley Physiographic Province encompasses almost 12,000 mi^2 (13,080 km^2) within the unglaciated portion of central Pennsylvania (Fig. 1). The region is characterized by a series of alternating ridges and valleys that arc across the state in a north-easterly direction (Rhoads and Klein, 1993). The climate is moderate, with an annual average temperature of 10 °C and monthly averages ranging from −3 °C in January to 22 °C in July (NOAA, 1991). Average annual precipitation is 102 cm and is evenly distributed throughout the year. In general, ridge tops are forested, with agriculture and urban development restricted to the valley floors. Wetlands are typically less than 20 ha in size for all wetland types (Environmental Resources Research Institute, 1998) and occur in association with streams (Brooks and Tiner, 1989). Although all or most wetlands within the Ridge and Valley have been subjected to anthropogenic disturbances in the past, those considered least disturbed have intact, forested buffers and occur in largely forested watersheds. In contrast, disturbed systems generally occur in highly cultivated or urban landscapes where the majority of forest cover has been removed.

2.2. Site selection, sampling protocol and index calculation

Since 1993, the Penn State Cooperative Wetlands Center (CWC) has collected data on 149 reference wetlands throughout the Ridge and Valley. Reference wetlands are selected to represent the range of wetland condition and classified into one of seven hydrogeomorphic (HGM) subclasses following Cole et al. (1997). Of these seven subclasses, four are common to the region: mainstem floodplains, headwater floodplains, riparian depressions, and slopes (Cole et al., 1997). Headwater floodplain, riparian depression and slope wetlands are associated with streams of second order or less and are fed by either surface or groundwater (Cole et al., 1997). As mentioned previously, streams of first and second order represent 75% (13,089 km) of the total stream length in the Ridge and Valley physiographic province (17,403 km). Thus, these headwater wetlands represent one of the most abundant wetland types. These three subclasses were subsequently combined into a single subclass (headwater complex) following a more recent HGM classification system developed for the Mid-Atlantic region (Brooks et al., in preparation) because they often occur in a mosaic without distinguishable boundaries.

Headwater wetlands are also a primary receptor for stressors in the surrounding landscape, occurring between valuable headwater streams and upland activities such as agriculture, mining, and development. Since we were interested in examining the FQAI as a method of detecting anthropogenic disturbance surrounding headwater wetlands, we selected sites across a range of human disturbance. For purposes of experimental design, we utilized a measure of disturbance that integrates information on surrounding land use, buffer characteristics, and an assessment of potential site stressors, and is described in detail in the

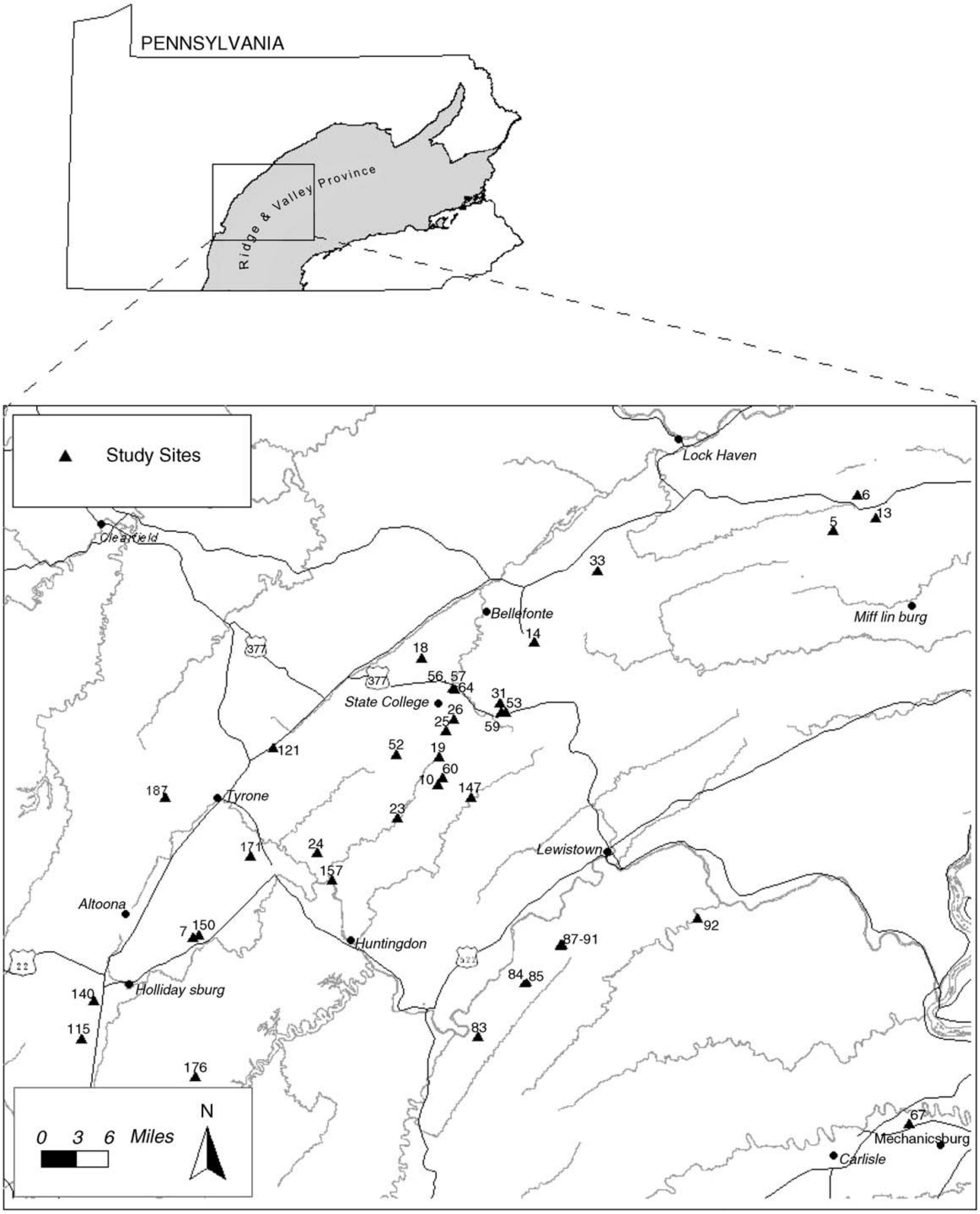

Fig. 1. Location of the 40 headwater complex reference sites within the Ridge and Valley physiographic province of central Pennsylvania. Reference sites were chosen randomly from National Wetland Inventory maps followed by a more directed search to fill in specific disturbance categories. They represent the range of condition for wetlands from highly disturbed to least impacted.

following section. Sites were selected across the entire range of disturbance, as expressed by the disturbance score, from least disturbed to most impacted. Our site classification resulted in a total of 40 headwater complex reference sites (15 headwater floodplains, 10 riparian depressions and 15 slopes) for analysis (Fig. 1).

To calculate I, we used C values devised for central Pennsylvania by Beatty et al. (2002). A species list for each site was compiled using dominance and richness data collected between 1993 and 2000 using a Rapid Assessment Protocol (RAP) (Brooks et al., 1999). The RAP is designed to sample a 1-ac area of wetland using a system of nested plots laid out along an evenly spaced grid of sampling locations. Herbaceous species cover is estimated within a 0.5 m × 2 m rectangular quadrat; herbaceous species richness, shrub species richness and shrub volume are measured within a 3 m radius circular plot; and tree species richness and dbh within an 11.6 m radius circular plot. Previous validation of this protocol showed that the sampling effort adequately captured the plateau point of the species–area curves for most sites in the Ridge and Valley physiographic province. Furthermore, we found no correlation between I scores and the amount of plot area sampled ($r = -0.131$, $P = 0.422$), as has been reported from other studies (Taft et al., 1997; Francis et al., 2000; Rooney and Rogers, 2002), indicating a uniform sampling effort was achieved. Plants were identified to the lowest taxonomic level possible and plants that could not be identified to species were not used to calculate I scores or its variants. All sites were sampled during the months of June through August.

The FQAI score (I) for each site was then calculated as:

$$I = \bar{C} \times \sqrt{N} \tag{1}$$

where $\bar{C}$ is the mean of the C values of native species and N is the number of native species.

2.3. Disturbance gradient and landscape measures of fragmentation

To examine the relationship between the index and disturbance, we plotted ecological dose–response curves. This type of plot provides a graphical way of interpreting the response of I scores to increasing "doses" of disturbance (U.S. E.P.A., 2002). Disturbance was quantified using a Level 2 rapid assessment method. The Level 2 method uses information on surrounding land use, buffer characteristics, and an assessment of potential site stressors (Brooks et al., 2004). Specifically, disturbance was calculated as:

$$100 - \mathrm{CF}\left\{\%\mathrm{FLC}\left[\frac{10 - \mathrm{STRESSORS}}{10}\right] + \left[\mathrm{BUFFERSCORE} - \mathrm{BUFFERHITS}\right]\right\} \tag{2}$$

where CF is a calibration factor (100/114) needed to standardize the scores to a scale of 0 to 100, %FLC is the sum of percent forested land cover and percent open water in a 1-km circle centered on the site, #STRESSORS is the number of stressors present on site, BUFFERSCORE is a value from 0 to 14 assigned to the buffer given its type and width and BUFFERHITS is the number of stressor indicators present that were likely to "puncture" the buffer.

For determination of %FLC, the general approach was to overlay the wetland location maps with land cover maps, center a 1-km radius circle on each sample point, and calculate %cover of the land cover types. Land cover was defined according to Anderson et al. (1976). Forested land cover consisted of deciduous, coniferous and mixed upland forests, forested wetland and open water. Non-forested cover consisted of agricultural land cover (i.e., annual crop, perennial crop, pasture), and transitional and developed land cover (i.e., urban, suburban, and barren land). A 1-km radius was chosen because in the Ridge and Valley physiographic province a circle of this size "fits" into valleys or onto ridgetops. Land use follows the distinct topography with forest on the ridgetops, and non-forest in the valleys. Therefore, a circle with a radius larger than 1 km often creeps up ridge sides or down onto valley floors and could encompass the transition in both physical habitat and land cover.

In the field, indicators of stressors present were marked on a standard checklist and the number of stressor categories tallied. The assessment assumes that the higher the number of stressor categories present at a site, the lower the influence of the surrounding forest land cover. This is reflected in the first portion of the equation, which weights the amount of surrounding forested land cover used in the landscape-level assessment with the number of stressor categories present on the site. Type and width of any vegetated buffer present is also recorded. Buffers can help to ameliorate the effects of surrounding land use on wetland condition. Their potential to do so is a function of both their width and type, therefore, the presence of a buffer improves the condition score. Wide buffers (>100 m) of natural forest get the highest score (14); no buffer, the lowest (0). However, certain stressors cannot be effectively mitigated by the presence of a buffer and are represented in the formula by BUFFER HITS. These are types of stressors, like culverts, that allow the effects of surrounding land use to affect the wetland despite the presence of a vegetated buffer.

Thus, the Level 2 rapid assessment is an attempt to quantify the degree of human disturbance at the site. Ecological condition is assumed to be negatively correlated with the extent of human disturbance and in a linear fashion. Sites with a score of zero are expected to be in excellent condition and sites with a score of 100 to be in extremely poor condition. Earlier studies using variations of this approach have shown the disturbance gradient to be highly correlated with plant (Wardrop and Brooks, 1998), macroinvertebrate (Bennett, 1999) and bird communities (O'Connell et al., 2000).

The linear distance (m) between each site and the nearest wetland was estimated using National Wetland Inventory maps. Road density (m/ha) was determined as the number of paved roads within a 1-km circle centered on each site.

2.4. Data analysis

Since all of the variables we examined violated one or both of the necessary assumptions of normality, we used Spearman rank correlation coefficients to assess the relationship between FQAI and its variants to disturbance variables. All statistical analyses were performed using Minitab version 13.2 (Minitab Inc., 2000).

3. Results

Reference headwater floodplain, riparian depression and slope wetlands are predominantly forested systems and generally fall into one of four palustrine forest categories as described by Fike (1999): red maple-black gum, hemlock-mixed hardwood, red spruce-mixed hardwood, or hemlock palustrine forest. Those with low disturbance scores occur in watersheds that are largely forested (>90%) and have extensive, intact, forested buffers and few identifiable stressors. In contrast, sites at the higher end of the disturbance gradient are located in watersheds that support both urban and agricultural development and the proportion of watershed comprised of forest is generally small, around 30% or less. Buffer areas are narrow (generally less than 30 m) and potential stressors are prevalent, particularly those associated with hydrologic modification and increased sedimentation.

The characteristics of each site in terms of floristic quality are listed in Table 1. For comparison, we divided sites into three disturbance categories (high, moderate, and low) based on data from O'Connell et al. (2000) that indicate ecological condition is closely associated with forest cover. In studying 200 sites within the Mid-Atlantic Highlands Area, they found those in good or excellent condition had an average forest cover of 87%, while sites in poor ecological condition had an average forest cover of less than 28%. Our three categories of high, moderate, and low disturbance, therefore, reflect percent forested cover values of <28, 29–86, and >87%, respectively.

Sites in the least disturbed category had the highest $\bar{C}$ values and I scores. Non-native species comprised only a small percentage of the flora (less than 30% on average) at these sites compared to sites with moderate to high disturbance scores, in which over one-half of the flora was represented by non-natives (38–55% on average). Native species richness was greatest at sites with moderate disturbance scores, but sites with low disturbance scores had the highest proportion of native species. Table 2 shows the mean number of species per

Table 1
Floristic quality data for headwater wetlands in the Ridge and Valley of central Pennsylvania

Site number	Disturbance score	Disturbance category[a]	Total species richness	Native species richness (*N*)	%Native species	Non-native species richness (*A*)	%Non-native species	FQAI (*I*)	Adjusted FQAI (*I*′)	Mean *C* value
5	1	Low	40	29	73	11	28	30.6	48.4	5.69
88	1	Low	42	26	62	16	38	26.7	41.2	5.23
89	1	Low	46	32	70	14	30	29.2	43.0	5.16
91	1	Low	45	33	73	12	27	27.9	41.5	4.85
13	2	Low	39	30	77	9	23	33.0	52.9	6.03
84	2	Low	29	23	79	6	21	25.2	46.9	5.26
85	2	Low	32	25	78	7	22	27.6	48.8	5.52
6	3	Low	31	24	77	7	23	30.0	53.9	6.13
19	4	Low	32	26	81	6	19	30.8	54.4	6.04
60	6	Low	18	12	67	6	33	20.2	47.6	5.83
171	9	Low	48	28	58	20	42	30.8	44.5	5.82
90	10	Low	48	33	69	15	31	30.5	44.0	5.30
14	10	Low	34	21	62	13	38	25.1	43.0	5.48
10	21	Low	31	23	74	8	26	27.9	50.2	5.83
83	24	Low	62	43	69	19	31	32.8	41.6	5.00
87	27	Low	67	51	76	16	24	32.5	39.7	4.55
Mean			*40.25*	*28.69*	*72*	*11.56*	*28*	*28.80*	*46.35*	*5.48*
S.D./S.E.			*12.48*	*8.96*		*4.76*		*3.38*	*4.76*	*0.46*
23	30	Moderate	63	45	71	18	29	31.3	39.4	4.67
157	30	Moderate	143	99	69	44	31	41.1	34.4	4.13
24	34	Moderate	56	35	63	21	38	30.1	40.2	5.09
147	35	Moderate	102	71	70	31	30	41.2	40.8	4.89
150	36	Moderate	74	45	61	29	39	26.7	31.0	3.98
187	39	Moderate	46	30	65	16	35	27.2	40.1	4.97
7	48	Moderate	55	42	76	13	24	26.1	35.2	4.02
121	54	Moderate	33	22	67	11	33	24.7	43.1	5.27
115	56	Moderate	56	23	41	33	59	16.5	22.0	3.43
33	63	Moderate	83	44	53	39	47	23.8	26.1	3.59
140	67	Moderate	63	30	48	33	52	20.4	25.8	3.73
18	71	Moderate	74	38	51	36	49	17.7	20.6	2.87
92	72	Moderate	51	34	67	17	33	20.8	29.1	3.56
Mean			*69.15*	*42.92*	*62*	*26.23*	*38*	*26.73*	*32.90*	*4.17*
S.D./S.E.			*28.23*	*20.97*		*10.76*		*7.76*	*7.67*	*0.74*
53	76	High	44	22	50	22	50	12.8	19.3	2.73
64	79	High	18	9	50	9	50	14.3	33.8	4.78
57	82	High	27	15	56	12	44	16.3	31.3	4.20
31	83	High	64	29	45	35	55	16.0	20.0	2.97
56	84	High	32	17	53	15	47	17.9	31.7	4.35
26	86	High	41	17	41	24	59	12.4	19.3	3.00
25	88	High	34	14	41	20	59	11.5	19.7	3.07
176	90	High	66	22	33	44	67	18.8	23.1	4.00
67	91	High	19	7	37	12	63	7.9	18.2	3.00
59	93	High	12	5	42	7	58	4.5	12.9	2.00
52	96	High	31	16	52	15	48	11.8	21.1	2.94
Mean			*35.27*	*15.73*	*45*	*19.55*	*55*	*13.1*	*22.8*	*3.37*
S.D./S.E.			*17.56*	*7.06*		*11.34*		*4.3*	*6.6*	*0.25*

[a] For comparison, sites are divided into low, moderate, and high disturbance categories based on O'Connell et al. (2000).

Table 2
Mean number of species per conservatism class (±S.E.)

Disturbance category	Conservatism class			
	0–2	3–5	6–8	9–10
Low disturbance sites	1.29 ± 0.24	3.46 ± 0.34	3.90 ± 0.38	0.19 ± 0.09
High disturbance sites	2.36 ± 0.31	2.06 ± 0.36	0.70 ± 0.20	0.00 ± 0.00

Sites were divided into low and high disturbance levels based on O'Connell et al. (2000).

conservatism class for sites with both low and high disturbance scores. Plant species with high conservatism values decreased markedly as disturbance increased. Sites at the lower end of the disturbance gradient had a greater number of plant species with C values greater than 3, while at more disturbed sites, species with C values of 0–2 were most prevalent.

As predicted, I scores showed a strong negative correlation ($r = -0.75$, $P < 0.001$) to anthropogenic disturbance (as measured by the Level 2 rapid assessment method) with scores generally decreasing with increasing disturbance (Fig. 2). When plotted against native species richness, I scores were strongly correlated (Table 3). The index was also correlated with $\bar{C}$ value. We found no correlation between I scores and non-native species richness.

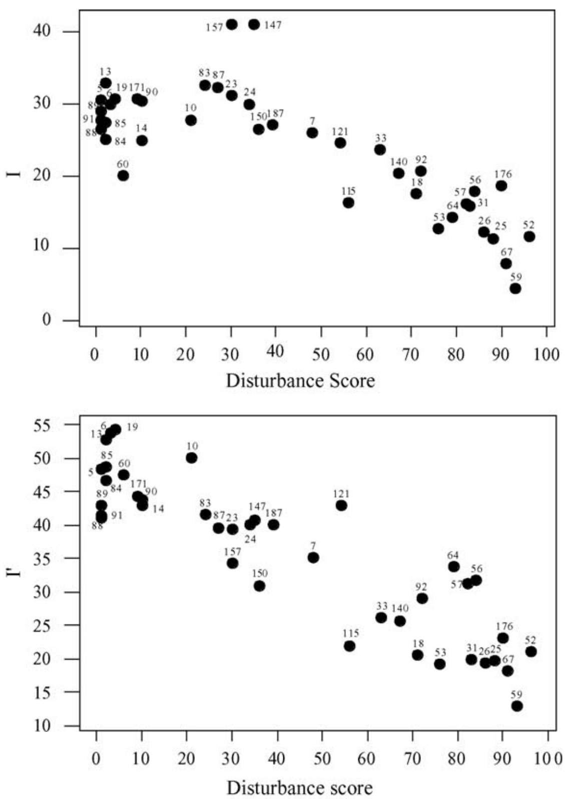

Fig. 2. Ecological dose–response curves for I and I' vs. human disturbance. Dose–response curves examine the response of these variables to increasing amounts of anthropogenic disturbance.

Although there was a strong correlation between I scores and disturbance, an examination of individual site rankings revealed an inherent bias in I scores toward sites with greater native species richness. Sites with higher $\bar{C}$ values, but lower native species richness (e.g., sites 6 and 60) scored lower than some sites with lower $\bar{C}$ values and higher native species richness (e.g., sites 147 and 157; Table 1, Fig. 2). To eliminate the sensitivity of the index to species richness, we determined FQAI as a percentage of the maximum attainable I score for each site. The maximum attainable I score is calculated by assuming that the $\bar{C}$ value is 10 (the highest possible $\bar{C}$ value) and all plant species are native. It, therefore, serves as a yardstick by which to measure optimal habitat quality and any departure from this optimum can be interpreted as a loss or diminishment of floristic integrity. The adjusted FQAI (I') score was calculated as:

$$I' = \left(\frac{\bar{C}}{10} \frac{\sqrt{N}}{\sqrt{N + A}} \right) \times 100 \tag{3}$$

Table 3
Spearman rank correlations between I and I' scores and components of each index

Component	I		I'	
	r	P	r	P
Native species richness	0.72	*	0.19	0.232
Non-native species richness	−0.05	0.753	−0.58	*
Mean C value	0.68	*	0.98	*

I was highly correlated with native species richness, while I' was highly correlated with non-native species richness. Both indices were highly correlated with mean C values.
* $P < 0.001$.

Table 4
Spearman rank correlations between I and I' scores and measures of disturbance

Disturbance measure	I		I'	
	r	P	r	P
%Forest	0.75	*	0.81	*
Buffer score	0.78	*	0.84	*
Stressor score	−0.60	*	−0.80	*
Distance to nearest wetland	0.16	0.340	0.11	0.499
Road density	−0.35	†	−0.36	†

Indices I and I' were highly correlated with components of the disturbance gradient and the road density index. Neither index was correlated with distance to nearest wetland.
* $P < 0.001$.
† $P < 0.05$.

where A is the number of non-native species. The I' scores re-ordered our sites and resulted in a stronger correlation ($r = -0.87$, $P < 0.001$) with disturbance (Fig. 2). I' scores were not correlated with native species richness, but were highly correlated with non-native species richness and $\bar{C}$ values (Table 3).

Both I and I' scores were highly correlated with the three components (%forest, buffer score, stressor score) of our disturbance score (Table 4). The indices were also significantly correlated with road density, but not with distance to nearest wetland. I' was more effective than I in differentiating among our high, moderate, and low disturbance categories (Fig. 3).

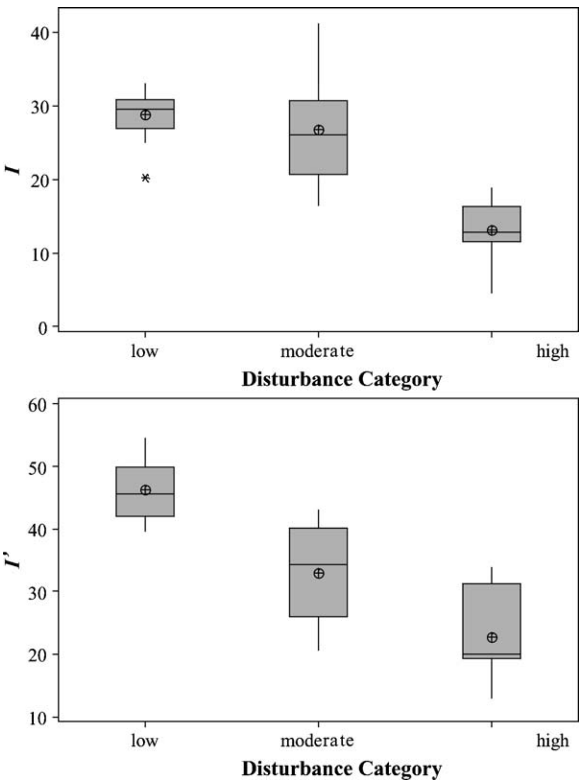

Fig. 3. Box plots of I and I' showing the efficacy of each approach in differentiating among three categories of human disturbance. Disturbance categories are low, moderate, and high and correspond to three levels of forest cover (<28, 29–86 and >87%, respectively) within a 1-km radius circle centered on each site. The bottom of the box and lower whisker represent the first quartile and lower data limit, respectively, while the top of the box and upper whisker are the third quartile and upper data limit. The median is represented by a solid line transecting the box. The mean (⊕) and any outliers (*) are also depicted. I' more neatly separates the low and moderate disturbance categories.

4. Discussion

While the FQAI is gradually gaining acceptance as an effective evaluation tool (Rooney and Rogers, 2002), two fundamental issues remain that have been problematic for the index since its conception: the overwhelming influence of species richness in the equation and the role of non-native species in assessing floristic quality. Our results are consistent with other studies (Taft et al., 1997; Francis et al., 2000) that have shown that the index scores sites with greater numbers of native species higher, regardless of other influences on floristic quality. In theory, the site with the greatest proportion of conservative species should receive the highest I score (Swink and Wilhelm, 1994; Herman et al., 1997). However, in practice, this is not always the case.

We observed four possible outcomes of the FQAI method in this study (Fig. 4). These scenarios represent realistic outcomes, since our sites had $\bar{C}$ values of 2–6, and native species richness values of 5–99. Scenarios 1–3 support the original intent of the method to rank a site with more conservative species higher or, in the case of Scenario 3, a site with greater native species richness. Of greater consequence to this and other studies, however, is the outcome shown in Scenario 4. In this scenario, a site with a lower $\bar{C}$ value and higher species richness can obtain a higher I score than a site with a higher $\bar{C}$ value, but lower species richness (Taft et al., 1997).

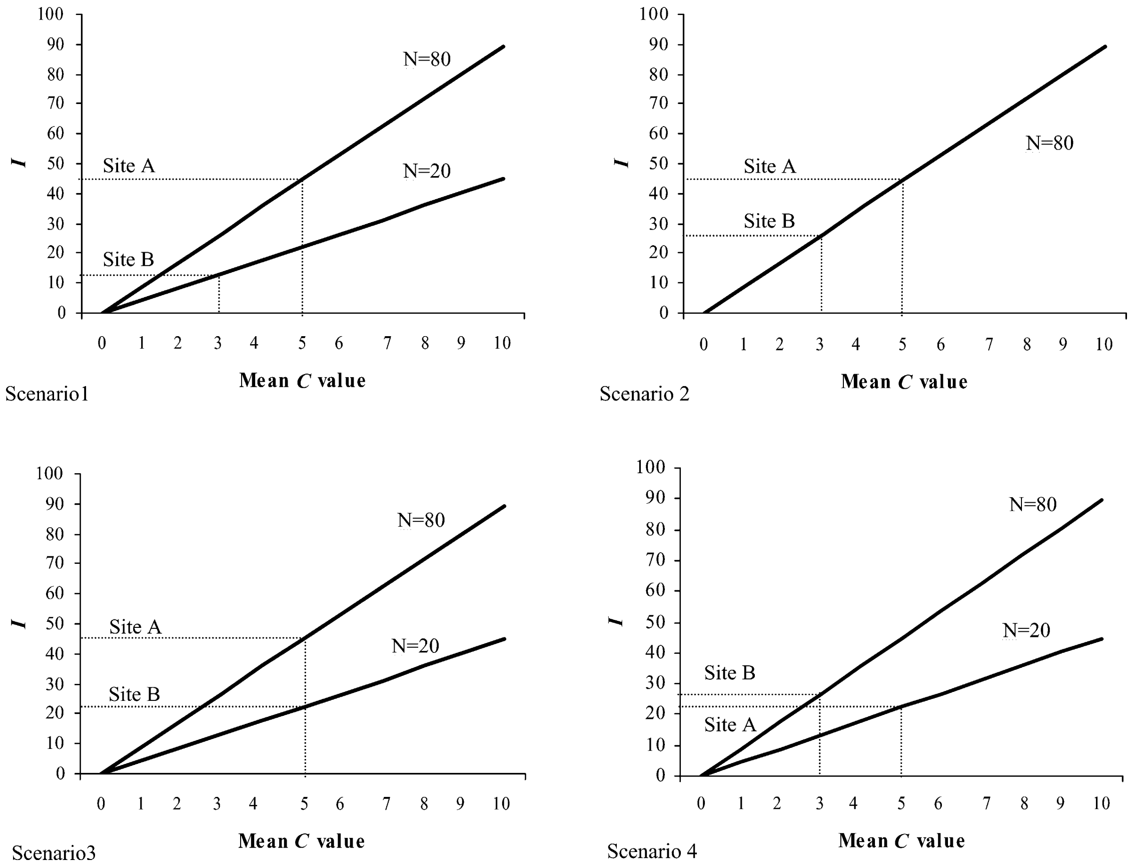

Fig. 4. Four hypothetical outcomes of I based on $\bar{C}$ value and species richness observed during our study (adapted from Taft et al., 1997). In the first three scenarios, Site A receives a higher I score than Site B because of greater species richness, a higher $\bar{C}$ value, or both. These three scenarios are typical and support the original intent of the index. In Scenario 4, Site B receives a higher I score than Site A due to increased richness, even though this site has a lower $\bar{C}$ value. This scenario illustrates a fundamental problem with I related to the overwhelming influence of the richness multiplier.

We observed this unintended outcome at some of our sites with moderate disturbance scores. These sites supported high numbers of native species with low to moderate C values. In fact, species richness peaked at these sites. Greater species richness, however, is not always equivalent to greater habitat quality. In fact, increased species richness is often associated with low to intermediate levels of disturbance, which may increase resource availability and decrease competition (Sousa, 1984; Dittmar and Neely, 1999). By taking the square root of this value, Swink and Wilhelm (1994) attempted to dampen the effect of high species richness on the index. Despite this transformation, species richness is still a driving factor in the FQAI and this sensitivity is often viewed as a fundamental flaw in the method (Francis et al., 2000).

Taft et al. (1997) and Rooney and Rogers (2002) advocate using $\bar{C}$ values rather than I scores when factors such as the amount of area sampled, heterogeneity of the plant community, or observer expertise may bias results. $\bar{C}$ provides a measure of aggregate conservatism that is unobscured by species richness (Bernthal, 2003). $\bar{C}$ values were strongly correlated to disturbance ($r = -0.85$, $P < 0.001$) and followed a similar trend with values generally decreasing with increasing disturbance largely due to a loss of conservative species at more disturbed sites (Table 2). While $\bar{C}$ values were effective at differentiating our highest quality sites from our most degraded, the use of $\bar{C}$ alone is not effective in making comparisons among disturbed sites. For example, sites 7 and 176 have similar $\bar{C}$ values, but differed greatly in disturbance rank, I score, and native species richness (Table 1). In cases like this, comparisons using $\bar{C}$ can be misleading (Taft et al., 1997).

Other studies have recommended using an alternate method for calculating FQAI that includes non-native species (Taft et al., 1997; Fennessy et al., 1998a,b; Francis et al., 2000; Lopez and Fennessy, 2002; Bernthal, 2003; Rothrock, 2004). The number of non-native species at a site can greatly influence quality, particularly if they are invasive and displace native species (Taft et al., 1997; Francis et al., 2000). Furthermore, disturbance has been shown to facilitate the establishment of non-native species (Pyle, 1995; Anderson et al., 1996), and their presence (or absence) from a site can, in itself, provide a strong signal of condition.

While most authors agree the inclusion of non-natives in I is important, the question of how to incorporate this variable so that it best represents a decrease in floristic quality remains unresolved. The standard method is to simply treat non-native species as if they were natives in calculating both $\bar{C}$and N. This approach, however does not address the inherent bias in the FQAI toward species rich sites because although a decrease in $\bar{C}$ will decrease the overall I score, this decrease is not large enough to overcome the overriding influence of increased richness.

I' addresses both of these issues simultaneously by using non-native species to dampen the influence of the richness multiplier. In the formula, the maximum attainable I score (which assumes all species have a C value of 10 and all plants are native) acts as a yardstick with which to measure habitat quality and any departure from this optimal habitat condition can be interpreted as a loss or diminishment of floristic integrity. Species poor sites with little or no non-native species will score higher using I' because they are closer to an optimal habitat condition. In contrast, species rich sites with a moderate to high proportion of non-natives will score lower because the presence of non-native species signals a decrease in habitat quality.

The sensitivity of I' to incremental increases in non-native species is illustrated in Fig. 5. For a site with a $\bar{C}$ value of 5, native species richness of 20 and no non-native species, the adjusted FQAI score is 50. As the proportion of non-native species increases, however, the I' score decreases accordingly. I', therefore, expresses this potential stress to the plant community and gives a more realistic assessment of site conditions.

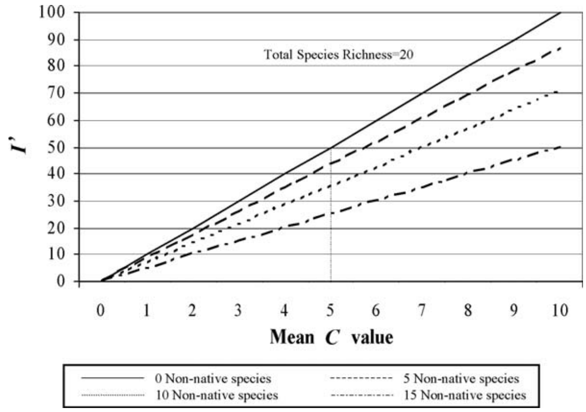

Fig. 5. Response of I' to incremental increases in the number of non-native species. For a site with a $\bar{C}$ value of 10 and zero non-native species, the I' score is 100 (the maximum attainable I). A site with a $\bar{C}$ value of 5 and zero non-native species would score 50. As the proportion of non-native species in the flora increases, the I' score decreases accordingly. This decrease represents the negative contribution of non-native species to floristic integrity.

The ability of I' to differentiate among different disturbance categories and its strong correlation to at least one independent measure of landscape disturbance (road density) further underscores its efficacy as a condition assessment tool. The identification of high quality wetlands, in particular, is extremely valuable since these sites are often used to develop mitigation performance criteria (Danielson, 1998). The lack of correlation between both indices and distance to nearest wetland was surprising, but may be an artifact of the method used to measure this variable. In the Ridge and Valley, National Wetland Inventory maps often inadvertently exclude smaller, headwater wetlands (e.g., depressions and slopes) that cannot be readily discerned from high altitude aerial photographs (Wardrop et al., in review).

The strength of the relationship between I' and both landscape and site-level measures of disturbance highlights its use as a validation and calibration tool for refining multi-level assessments of wetland condition. CWC has formulated a process in which the condition of wetlands can be assessed by employing one or more of three levels of effort (Brooks et al., 2004). The three levels can be generally described as a landscape assessment, a rapid assessment, and an intensive assessment of wetland condition. A *landscape-level assessment* (Level 1) can be accomplished in the office using only readily available digital data and a geographic information system (GIS) and requires a low level of effort compared to the site assessments. The *rapid-site assessment* (Level 2) refines the results of the landscape assessment by incorporating observational indicators of human disturbance to a site into the evaluation of ecological condition. The *intensive-site assessment* (Level 3) entails detailed data collection on each site assessed and produces the most complete evaluation of condition. Validation and refinement of landscape and rapid assessments requires a robust indicator of ecological condition, and I' is such an indicator. An indicator of condition that is strongly correlated with land cover measures allows informed decision-making to take place where it was previously absent, by using widely-available land cover data as a preliminary indicator of condition. Improved decision-making is of paramount importance to headwater wetlands, due to their unique role in providing high quality habitat and biodiversity-rich ecosystems.

Our study has demonstrated that I' is an effective tool for evaluating condition and one that, like the original FQAI equation, is relatively easy to use and produces objective results. However, as Bernthal (2003) cautions, regulatory decisions should not be based solely on I' scores alone. Additional data that could be used to supplement I' scores include other plant metrics, wetland functional assessments, wildlife surveys, and information on threatened and endangered species (Taft et al., 1997; Bernthal, 2003). In headwater wetlands in the Ridge and Valley, we have observed strong correlations between other plant metrics and disturbance (Miller et al., 2004). The use of these additional plant metrics, as well as other site-specific data would provide a more comprehensive assessment of condition.

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