STATE-OF-THE-SCIENCE REPORT
Trends in Floristic Quality Assessment for Wetland Evaluation

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Over the past two decades, much has been written about the use of bioassessment tools to evaluate wetland condition. Interest in bioassessment has originated from a need to establish parameters for "biological integrity" in wetland ecosystems, whether for scientific research, natural areas assessment, inventory and monitoring, or in response to regulatory mandate. On the latter point, the need has been, in part, a reaction to Clean Water Act directives to "restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (33 U.S.C. §1251). For wetland scientists and managers, identifying a sampling focus for chemical or physical integrity (e.g., dissolved oxygen, temperature) has been a much more straightforward task than finding adequate methods for measuring biological integrity, an ambiguous concept that defies precise definition (Cronk and Fennessy 2001). This puts scientists and managers in the difficult position of attempting to express a *qualitative* construct (biological integrity) in *measurable* or *quantitative* terms. Even more challenging for wetland practitioners is the decision about which model organisms to use among the diversity of biota that inhabit wetland systems.

In wetlands, vegetation is one component of the biota that is frequently studied to evaluate wetland condition. Metrics describing biological integrity in terms of *in situ* vegetation are desirable for several reasons (U.S. EPA 2002): 1) plants are ubiquitous in wetland environments; 2) vegetation is a defining characteristic of wetland systems both from an ecological and a regulatory context; 3) sampling protocols for vegetation are well known; 4) plant communities express sensitivity to ecological disturbance and environmental stressors in measurable ways; and, 5) plants are not motile. To this end, Floristic Quality Assessment (FQA) has been identified as a potentially useful tool for wetland assessment. Proponents have cited FQA as a suitable approach for this purpose because its quantitative outputs – the Floristic Quality Index (FQI) and related metrics – are calculated from "Coefficients of Conservatism" (*C*-values) that are assigned by an independent panel of botanical experts knowledgeable about the flora of a particular region (see Boxes 1 and 2). The *C*-value list for a given region provides a foundation for the FQA approach, which is regarded by many as a non-biased analog for biological integrity in wetlands that is "dispassionate, cost-effective, and repeatable" (Swink and Wilhelm 1994). However, others have cited some concerns with traditional application of FQA to wetland assessment (Francis et al. 2000; Matthews 2003; Cohen et al. 2004; Miller and Wardrop 2006; Bried et al. 2013; DeBerry and Perry 2015). Our objective is to provide an overview of this broad spectrum of scientific opinion on FQA research, specifically with the intent of summarizing the benefits and challenges of the FQA approach to wetland assessment.

FQA RESEARCH IN WETLANDS: REGIONAL TRENDS

Table 1 provides a somewhat comprehensive list of published research on the use of FQA in North American wetland studies. We say "somewhat comprehensive" because in our attempts to include all relevant published literature some studies may have been inadvertently left out. Notwithstanding such an oversight, the list in Table 1 is provided as a resource for the reader who wishes to examine this topic in more detail. Although there is a much wider literature base on use of FQA in general, the studies cited in this table are specific to scientific research in which FQA was tested in wetlands directly, or in which it was used indirectly as part of a larger study in wetland environments. The remainder of this review focuses on the former (i.e., studies where FQA was tested directly to determine its suitability for use in wetland evaluation), which are identified in boldface type in Table 1. Note that this table does not include published studies on development of regional *C*-value lists, but we provide an overview of the listing process in Box 2.

As explained in Box 1, FQA originated in the Chicago region, so it is not surprising that an inordinate amount of the research listed in Table 1 has occurred in the north-central U.S. states and adjacent Canadian provinces (e.g., Illinois, Michigan, Wisconsin, Ohio, Indiana, North Dakota, South Dakota, and Ontario). The *C*-value lists for many of these regions have been in place longer, and some state agencies have developed regulatory programs incorporating FQA into wetland evaluation programs dating back to the late 1990s (e.g., Ohio, see Fennessy et al. 1998a,b). Other

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"hot spots" for FQA research in wetlands have included Mid-Atlantic states (e.g., Pennsylvania, Virginia, West Virginia), the Southeast (e.g., Florida, Mississippi, Louisiana), and the Northeast (e.g., New York). There has also been some recent research originating from the Midwest, the Eastern Intermountain Region (e.g., Oklahoma, Colorado, and Montana) and western Canada (e.g., Alberta).

Across the geographic domain of FQA application, the approach is being used in a variety of ways to evaluate wetland condition. Examples include ambient monitoring and assessment, targeting and prioritizing sites for conservation, assessment for impact analysis in wetland regulatory programs, performance evaluation for wetland mitigation sites, identification of reference sites for functional assessment, and incorporation into larger assessment models such as IBIs (Cronk and Fennessy 2001; Miller et al. 2006; Medley and Scozzafava 2009; Chamberlain et al. 2013). Further, FQA methods have been developed for a broad and growing geographic range engaging a diversity of wetland habitats types within which the method has been tested for research purposes.

The following summary is outlined in a format that will allow quick access to the primary findings from the literature. As mentioned above, the focus is specifically on research that has tested the efficacy of FQA as an evaluative tool in wetland ecosystems. Readers interested in reviewing the subject further are encouraged to read the primary literature in more detail, particularly the studies denoted in boldface type in Table 1.

BENEFITS OF USING FQA IN WETLAND EVALUATION

FQA Reflects Ecological Condition Of all the aspects of FQA cited in the literature, the consistent finding that FQA reflects ecological condition is perhaps the most compelling rationale for its use in wetland evaluation. This has been tested in various ways: 1) through a "dose-response" analysis that plots the FQA metric against a pre-determined anthropogenic disturbance gradient and tests for significant correlations (e.g., Fennessy et al. 1998a; Miller et al. 2006; Bried et al. 2013); 2) through ecosystem modeling using community ordination techniques (e.g., Miller et al. 2006; Bowers and Boutin 2008; Cariveau and Pavlacky 2009; DeBerry and Perry 2015); 3) through comparisons with other biological integrity metrics such as species richness, diversity, evenness, percent native species, or related community indices (e.g., Matthews 2003; Ervin et al. 2006; Matthews et al. 2009b); or 4) through comparison with ecosystem condition variables such as soil physiochemistry, site age, biomass, etc. (e.g., Nichols 1999; Lopez and Fennessy 2002; DeBerry and Perry 2015).

The majority of the studies that tested FQA in wetlands (Table 1) cited a significant correlation with wetland condition using one or more of the approaches outlined above, concluding that FQA was a useful tool for wetland evaluation. The primary point of departure among these studies is

TABLE 1 List of studies that used FQA in wetlands. References in bold face type tested the performance of FQA metrics in wetland evaluation.

Author(s)	*Wetland Type*
Ahn and Dee 2011	mitigation wetlands
Alix and Scribailo 1998	lacustrine fringe (emergent/aquatic)
Allain et al. 2004	wet prairie
Balcombe et al. 2005	mitigation and reference wetlands (emergent/scrub-shrub)
Boughton et al. 2010	wet pasture (agricultural)
Bourdaghs et al. 2006	coastal wetlands (lacustrine)
Bowers and Boutin 2008	streambanks (riparian)
Bried and Edinger 2009	pine barrens vernal ponds
Bried et al. 2013	non-forested vernal pond/sedge meadow/shrub swamp
Bried et al. 2014	"reference" emergent/scrub-shrub
Cariveau and Pavlacky 2009	playas
Chamberlain et al. 2012	review
Chu and Molano-Flores 2013	wetlands in pre- and post-development landscapes
Cohen et al. 2004	isolated depressional marsh
Cretini et al. 2011	coastal marshes
DeBerry 2006	created wetlands, natural forested wetlands
DeBerry and Perry 2015	created wetlands, natural forested wetlands
DeBoer et al. 2011	mitigation wetlands
Dee and Ahn 2012	mitigation wetlands
DeKeyser et al. 2003	prairie wetlands (seasonal, non-forested)
Ervin et al. 2006	depressional, lacustrine fringe, riverine
Euliss and Mushet 2011	prairie pothole wetlands
Fennessy et al. 1998a	riparian wetlands (forested, scrub-shrub, emergent)
Fennessy et al. 1998b	depressional wetlands (forested, scrub-shrub, emergent)
Forrest 2010	created stormwater wetlands
Francis et al. 2000	"woodlands" (non-tidal, forested)
Hargiss et al. 2008	prairie wetlands (temporary, seasonal, semi-permanent)
Hartzell et al. 2007	natural and created depressional wetlands (non-forested)
Herman 2005	natural and created emergent wetlands
Herman et al. 1997	review (FQI development and application)
Johnson et al 2014	floodplain forest
Johnston et al. 2008	open-coast, riverine, protected (predominantly emergent)
Johnston et al. 2009	open-coast, riverine, protected (predominantly emergent)
Johnston et al. 2010	open-coast, riverine, protected (predominantly emergent)
Kowalski and Wilcox 2003	sedge fen

BOX 1: FLORISTIC QUALITY ASSESSMENT EXPLAINED

Floristic Quality Assessment (FQA) is the term given to the calculation and subsequent analysis of weighted metrics originally developed in the Chicago region for evaluating the "quality" of native plant communities (Swink and Wilhelm 1979, 1994). Quality is a relative term used to approximate similarity of a particular plant species assemblage to pre-settlement conditions in a similar habitat type (Maser 1990). Implicit in its application is the notion that areas with species assemblages closer to those of pre-settlement times (i.e., prior to European colonization of North America) are more reflective of high quality habitat (Swink and Wilhelm 1994; Nichols 1999), and the assumption that anthropogenic disturbance represents a mode of introduction for "non-conservative" (e.g., invasive or cosmopolitan) species. It is important to note that "disturbance" is in itself a relative term that could be used to describe the types of disturbances known to occur during pre-settlement times, such as incendiary fires set by Native Americans to clear patches of ground – activities that would also be categorized as "anthropogenic" (Noss 1985). However, the concept of disturbance as it relates to FQA is most often associated with post-settlement; that is, anthropogenic disturbance following European occupation of the North American continent.

The FQA approach is based on the concept that different plant species have evolved varying degrees of tolerance to human-induced disturbance (Chapin 1991), exhibiting varying degrees of fidelity to specific habitat integrity (Mushet et al. 2002). This combination of tolerance and fidelity is parameterized in FQA through the concept of "species conservatism" (Swink and Wilhelm 1979, 1994), which is specified by the "coefficient of conservatism" (C), a numerical assignment between 0 and 10 applied to plant species by a panel of experts on the native flora of a particular region (Cronk and Fennessy 2001). A species with a C-value of 10 always occurs within high quality habitats (i.e., habitats most closely resembling "remnant" or pre-settlement conditions), and a species with a C-value of 0 is not found in high quality habitats and, in general, is highly tolerant of anthropogenic disturbance (Swink and Wilhelm 1994). On several state or regional C-value lists, the value of C=0 is arbitrarily assigned to non-native species. Below is an example of types of assignment categories used in creating a regional C-value list (Chamberlain and Ingram 2012):

0–3 Plants with a broad range of ecological tolerances that are found in a variety of plant communities
4–6 Plants with an intermediate range of ecological tolerances that are associated with a specific plant community
7–8 Plants with a narrow range of ecological tolerances that are associated with advanced successional stage
9–10 Plants with a high degree of fidelity to a narrow range of pristine habitats

Once C-values for a given region are assigned, they can then be used to generate the functional output of FQA – the Floristic Quality Index or "FQI" (also referred to as Floristic Quality Assessment Index, "FQAI", or simply "I"). As originally conceived by Swink and Wilhelm (1979, 1994), the index is calculated according to the following equation:

$FQI = \bar{C}\,(\sqrt{N})$

where $\bar{C}$ represents the average coefficient of conservatism for native species, and N is native species richness. Note also that $\bar{C}$ by itself can be used as an index of floristic quality. Further, because of the unitless property of both metrics (FQI and $\bar{C}$), several modified versions have been proposed. Examples include FQI and $\bar{C}$ calculated from all species present (i.e., native and non-native) (Rocchio 2007; Cariveau and Pavlacky 2009), FQI weighted by species abundance (e.g., Cretini et al. 2012; DeBerry and Perry 2015) similar to a prevalence index (see Tiner 1999), FQI as a percentage of a maximum attainable index score based on the species present (Miller and Wardrop 2006), FQI and $\bar{C}$ expressed as ratios between different vegetation layers in forested wetlands (Nichols et al. 2006), and FQI adjusted to account for changes due to latitude (Johnston et al. 2010). Details on the relative merits of these approaches are discussed in the text. Equations for some of the more commonly used FQA metrics are listed in the table below (see Swink and Wilhelm 1994; Cohen et al. 2004; Bourdaghs et al. 2006; Miller and Wardrop 2006; Cariveau and Pavlacky 2009).

FQA Metric	Equation	Coefficients and Constants
Mean Coefficient of Conservatism ($\bar{C}$) (native species only)	$\bar{C} = \frac{\sum_{i=1}^{n} C_i}{N}$	C_i = *C*-value for i^{th} species N = native species richness
Floristic Quality Index (FQI) (native species only)	$FQI = \bar{C}\,(\sqrt{N})$	
Mean Coefficient of Conservatism ($\bar{C}_{all}$) (all species)	$\bar{C}_{all} = \frac{\sum_{i=1}^{n} C_i}{S}$	S = species richness
Floristic Quality Index (FQI_{all}) (all species)	$FQI_{all} = \bar{C}_{all}\,(\sqrt{S})$	
[1]Abundance-weighted $\bar{C}$ ($\bar{C}_{adj}$)	$\bar{C}_{adj} = \frac{\sum_{i=1}^{n} x_i C_i}{\sum_{i=1}^{n} x_i}$	x_i = abundance value for i^{th} native species
[1]Abundance-weighted FQI (FQI_{adj})	$FQI_{adj} = \bar{C}_{adj}\,(\sqrt{N})$	
[2]Richness-corrected FQI (FQI')	$FQI' = \left(\frac{\bar{C}}{10}\,\frac{\sqrt{N}}{\sqrt{S}}\right) \times 100$	10 = maximum *C*-value correction factor

[1]Note that $\bar{C}_{adj}$ and FQI_{adj} may also be calculated for all species (not just natives) by substituting √S and coefficients from $\bar{C}_{all}$ into these equations.
[2]The richness-corrected factor calculates FQI' as a percentage of the maximum attainable FQI (Miller and Wardrop 2006).

in the mode that FQA should take for this type of analysis (i.e., FQI, $\bar{C}$, or modified index versions; see Box 1 and index discussion below). Irrespective of the specific index chosen, the general trends suggest that it is the conservatism concept itself that provides the basis for consistency in condition evaluation, the foundation of which is vetted through expert opinion in the *C*-value listing process (see Box 2) (Swink and Wilhelm 1994; Chamberlain and Ingram 2012; Chamberlain et al. 2013). Studies showing significant negative correlations between FQA metrics and a gradient of anthropogenic disturbance abound (e.g., Fennessy et al. 1998a; Cohen et al. 2004; Miller and Wardrop 2006; Bried et al. 2013), indicating that higher FQA index values routinely correspond to a lower incidence of disturbance in wetlands, and vice versa. Further, several researchers have noted correlations with soil chemical parameters, plant biomass, or aquatic fauna communities (Fennessy et al. 1998b; Lopez and Fennessy 2002; Miller et al. 2009; DeBerry and Perry 2015), interpreting these relationships as an indication that FQA is able to signal ecological differences among wetland sites as a reflection of relative habitat degradation. Still others have noted that FQA provides results that are consistent with ecological succession theory in regenerating or restored wetland sites (Matthews et al. 2009b; Spyreas et al. 2012; DeBerry and Perry 2015), suggesting that the approach has some practical application in wetland mitigation assessment.

The "Challenges" section presents some conclusions about limitations of *C*-value lists, FQA indices, and sampling-related issues, all important considerations when applying FQA to wetland evaluation. However, suffice it to mention that even in light of these challenges, the majority of the studies listed in Table 1 concluded that, in one form or another, FQA serves as a general analog for biological integrity in wetlands.

FQA is Robust Another characteristic of the FQA approach is the relative consistency of the results achieved by researchers over different sampling seasons and under various sampling regimes. This observation has been made in the context of season-to-season comparisons (i.e., spring vs. summer sampling; Fennessy et al. 1998a; Francis et al. 2000; Lopez and Fennessy 2002; Cariveau and Pavlacky 2009; Bried et al. 2013), species list generation versus plot-based data collection methods (DeBerry and Perry 2015), sampling using different plot sizes (DeBoer et al. 2011), and in some cases, when comparing different wetland community types (Bried et al. 2013; Spyreas 2014).

It is important to note that certain FQA metrics do not follow this trend in all circumstances. For example, some studies have noted a strong seasonal effect on species richness in wetland communities, which indirectly influences the FQI metric due to the square root of N transformation (see Box 1; Matthews 2003; Miller and Wardrop 2006). However, even with these effects, FQA has been shown to

TABLE 1, CONTINUED

Author(s)	*Wetland Type*
Larkin et al. 2012	emergent marsh (Typha dominant, Typha absent)
Laughlin 2001	various
Lishawa et al. 2010	coastal wetlands (emergent)
Lopez and Fennessy 2002	depressional wetlands (forested, scrub-shrub, emergent)
Matthews 2003	floodplain forest, wet shrubland, sedge meadow, marsh
Matthews 2015	restored wetlands
Matthews et al. 2005	floodplain wetlands (forested, shrub, emergent, pond)
Matthews et al. 2015	floodplain forest, herbaceous wetlands
Matthews et al. 2009a	mitigation wetlands
Matthews et al. 2009b	mitigation wetlands
Medley and Scozzafava 2009	status review for use in NWCA
Miller and Wardrop 2006	headwater complex (riparian wetlands)
Miller et al. 2006	headwater wetlands (riparian)
Miller et al. 2009	riparian
Mushet et al. 2002	prairie potholes (natural and restored)
Nedland et al. 2007	restored wetlands (emergent, scrub-shrub, aquatic bed)
Nichols 1999	lacustrine (aquatic macrophytes)
Nichols 2001	lacustrine (aquatic macrophytes)
Nichols et al. 2006	hardwood flats
Niemi et al. 2011	coastal wetlands (open coastal, riverine, barrier)
Raab and Bayley 2012	emergent marsh reclamation (oil sands)
Reiss 2006	forested depressional wetlands
Reiss and Brown 2007	palustrine depressional wetlands (emergent, forested)
Rocchio 2007	various
Rooney et al. 2012	shallow open-water marsh wetlands
Rothrock and Homoya 2005	various (FO, SS, EM, aq), also included upland habitats
Spieles et al. 2006	mitigation bank wetlands
Spyreas 2014	various
Stanley et al. 2005	coastal wet meadow (lacustrine)
Tulbure et al. 2007	coastal wetlands (lacustrine, non-forested)
Wardrop et al. 2007	various (predominantly forested)
Werner and Zedler 2002	sedge meadow
Wilcox et al. 2002	lacustrine fringe
Wilson and Bayley 2012	emergent and aquatic bed prairie wetlands
Wilson et al. 2013a	wet meadow
Wilson et al. 2013b	stormwater, reclamation, & reference marsh wetlands

provide the same relative differences between sites (i.e., consistent site ranks based on conservatism) irrespective of differences in absolute index values between seasons (Herman 2005). With respect to species richness, perhaps the more important consideration is that of the species-area relationship, the effect of wetland size on richness, and the associated effect of area on FQA metrics (see the "Challenges" section below for further discussion).

Some researchers have concluded that FQA metrics should only be used to compare wetlands with similar habitat classifications (Francis et al. 2000; Matthews et al. 2005; Rocchio 2007), citing inconsistency in results when FQA is applied across habitat types. This recommendation is best taken in the context of the purpose for which wetlands are being evaluated. If the intent is to use FQA to identify sites with high conservation value (Swink and Wilhelm 1994), then FQA can be applied in a "categorical" sense to identify wetlands with "high", "medium", or "low" quality across habitat types. Some regions have used this approach to established index thresholds for targeting natural habitats in the "high" category for preservation (e.g., FQI>45 or $\bar{C}$>4.5; Swink and Wilhelm 1994; Rothrock and Homoya 2005; see comments under "Challenges" regarding use of FQA thresholds for wetland regulatory purposes). However, if the intent is to draw direct comparisons between wetlands to make inferences about relative ecological condition, then just based on the differences in habitat-specific ecological tolerances of the inhabiting species alone, direct comparisons between wetlands of different community types (e.g., forested vs. emergent) could lead to false conclusions about functional similarities or differences derived from FQA index scores (Matthews 2003). Interestingly, the categorical approach can be used to index biotic integrity in a similar manner to that described above for conservation value. In several studies, FQA has been used effectively as a component of a vegetation-based Index of Biotic Integrity (IBI), which generally separates sites along similar "high", "medium", and "low" condition class lines (e.g., Miller et al. 2006; Euliss and Mushet 2011; Raab and Bayley 2012; Wilson and Bayley 2012). In such cases, IBIs are region-specific and generally developed for a particular wetland habitat type.

FQA is Easy A common theme among wetland regulatory programs across the U.S. is the need for wetland assessment tools that are quick, easy to use, and reproducible (Medley and Scozzafava 2009; MPCA 2014). The authors of the FQA approach (Swink and Wilhelm 1979, 1994) identified this as a primary goal of the conservatism concept in their methodology, and by most researchers' standards that goal has been achieved in theory and in practice (Cohen et al. 2004; Bourdaghs et al. 2004; Rocchio 2007; Breid et al. 2013; Spyreas 2014). In fact, the most labor intensive aspect of FQA is the *C*-value listing process (Box 2); once this step is achieved, sampling and calculation of the FQA metrics are reasonably straightforward since all that is required is a species list for a given area (see Box 1). Some researchers have "complicated" the approach by applying different mathematical weights or adjustments to the FQA metrics to address specific research questions, with variable results (Cohen et al. 2004; Bourdaghs et al. 2006; Ervin et al. 2006; Miller and Wardrop 2006; Nichols et al. 2006; Cariveau and Pavlacky 2009; Cretini et al. 2012;DeBerry and Perry 2015). The implications of these approaches will be discussed further under "Challenges" below. An important point, however, is that the original FQA metrics (FQI, $\bar{C}$) are unitless, which means that they are easily incorporable into these types of modifications, an illustration of FQA's ease of use and versatility in evaluating wetland condition.

CHALLENGES OF FQA IN WETLAND EVALUATION

FQA Lacks Comparability across Regions A consistent criticism of FQA is the observation that results are not comparable across geographic regions. In other words, given absolute values for a metric like FQI that is calculated from two different *C*-value lists for two different geographic areas, some researchers suggest that there is minimal benefit gained by attempting to draw comparisons between the two, even if the community types are similar (Rothrock and Homoya 2005; Deboer et al. 2011). This has much to do with the *C*-value lists themselves. For example, some states have different listing criteria when compared to their neighbors (Medley and Scozzafava 2009). In addition, some states include non-native species in the listing process, whereas others do not (Matthews et al. 2015). Clearly, two or more lists that do not take a congruent approach to the *C*-value assignment process run the risk of producing different results just based on the potential for single species to have different *C*-values in different regions.

Most researchers that have addressed this problem suggest that FQA is best applied on a regional or statewide basis, and that comparisons between regions should be avoided (Rothrock and Homoya 2005; Bourdaghs et al. 2006; Reiss 2006). Others have advocated developing regional lists using ecoregions rather than state boundaries (Bourdaghs et al. 2006; Bried et al. 2013), an approach that has been undertaken in areas such as the Mid-Atlantic region (Chamberlain and Ingram 2012) and the Northeast region (Bried et al. 2012). Still others have evaluated the effect of latitude on FQA, suggesting that correction factors can be built into the method to account for natural variability across latitudinal gradients (Johnston et al. 2010; Spyreas 2014). Based on these observations, the regional specificity of existing and future *C*-value lists should be viewed as the *modus operandi* for FQA in wetland evaluation. Further, because the overall FQA approach generally provides the same relative results across boundaries (i.e., based on conservatism ranks), this should not be viewed as a disadvantage of the approach (Rothrock and Homoya

2005). Given the advent of the Regional Supplements to the Corps of Engineers Wetland Delineation Manual (Wakeley 2002) as well as the use of ecological regions to revise the National Wetland Plant List (Lichvar and Minkin 2008), the "regional paradigm" is also consistent with current trends in wetland regulation.

The inclusion or exclusion of non-native species in FQA bears mentioning because it is an important consideration that has been the subject of some debate in the literature. The authors of the FQA approach reject the notion of including non-native species, maintaining that the presence of non-natives will be measured indirectly by their negative effect on the abundance of native species through competition and habitat modification (Swink and Wilhelm 1994). Others have argued that accounting for non-native species in site evaluation provides a better overall understanding of ecosystem health, and results from several tests of FQA in wetland habitats suggest that FQA indices perform better when non-native species are included (Cohen et al. 2004; Herman 2005; Bourdaghs et al. 2006; Miller and Wardrop 2006; Rocchio 2007; Cariveau and Pavlacky 2009; Forrest 2010).

One problem with incorporating non-native species is the way in which they are treated in the *C*-value listing process. In some cases, non-native species are simply left off of the list, which precludes their use in FQA metrics. In other cases, non-natives are assigned an arbitrary value of *C*=0 – the lowest possible conservatism rank (see Box 1). The latter situation creates a problem in the calculation of the index when several non-native species are present, because the *C*-value for these species has been assigned based on nativity and not on degree of fidelity to natural areas *per se* (DeBerry and Perry 2015; Matthews et al. 2015). This is analogous to the "zero truncation problem" in ecological studies, where the mere absence of a species gives no information about how unfavorable the environment is for that species. Just as no negative abundance values are possible in a sample, there is no negative *C*-value scale to account for the relative differences of non-native species in a floristic quality sense, and the scale is "truncated" at zero (DeBerry and Perry 2015). Some authors have considered use of negative *C*-values, including an early version of FQA proposed by Swink and Wilhelm (1979), but to the best of our knowledge this approach has yet to be implemented effectively. In Virginia, DeBerry (unpublished data) has recently evaluated multiple data sets in a proposal to assign the values –5, –3, and –1 to non-native species on the Virginia *C*-value list corresponding to state-assigned categories of "high", "medium", and "low" invasion risk, respectively (Heffernan et al. 2014). Using this approach, negative values would be able to account for the relative differences in the degree to which different invasive species reflect ecological integrity without the need to modify the *C*=0 assignments for the remaining non-invasive exotic plants on the Virginia *C*-value list (see Matthews et al. 2015 for further discussion on negative values).

FQA Issues in Forested Wetlands A quick survey of the studies cited in Table 1 will show that the majority of the research on FQA in wetlands has been conducted in non-forested habitat. Studies that have evaluated FQA performance in forested wetlands have produced mixed results (Fennessy et al. 1998b; Francis et al. 2000; Nichols et al. 2006; DeBerry and Perry 2015). The primary concern with FQA in forested systems is that woody plants do not express the same type of responses to ecological disturbance as herbaceous species. Trees exhibit a property

BOX 2: CREATING A REGIONAL C-VALUE LIST – LESSONS LEARNED

The recent popularity of FQA in wetland monitoring and management has led to an increased desire to develop lists of coefficients for either regional or statewide floras. As those who have attempted such an endeavor can attest, the process of assigning coefficients can be challenging in the pre-planning, implementation, and post-assignment phases. We can learn much from our colleagues who have successfully navigated this process and emerged with an effective and informative product.

There are many planning considerations that must be addressed before assignment can take place. These include selecting a taxonomic authority and addressing nomenclature issues such as synonymy, hybrids, and whether to assign values to subspecies and varieties. In regions that cover large areas, there may be a need to address taxa that are native to only a part of the region. Assignment also involves selecting and vetting the botanists that will form the committee. Collectively, the botanical committee must provide sufficient expertise and coverage of the target geographical area. Equally important is the need to choose botanists that will work well together as a team to ensure the project is completed with minimal conflicts and delays.

When it comes to assigning coefficients, there are generally two models that have been followed. The first model is to allow botanists to assign values independently and then meet face to face to discuss the subset of taxa where disagreement falls above a set threshold. For example, taxa with coefficients that vary more than two standard deviations from the median would be tabled and reevaluated. The second model involves convening the committee and assigning values in situ by consensus. In both models, decision rules for the assignment of values are imperative to ensure consistency. The use of previously-assigned coefficients can serve to inform and expedite the process. Some project managers have also required their botanists to assign a confidence rating to each value as an added measure of validity.

Once values are assigned, there is typically more work to be done to finalize coefficient lists. During the assignment process, there may be issues with synonymy and nomenclature that require further review, taxa that are unfamiliar to the committee that need additional research, and disputed values that must be resolved. Such tasks may take an additional three to six months to complete and should be factored into project timelines and budgets. Another consideration is how to transfer the information to wetland managers so the values can be used. Some regions have developed online interactive calculators to facilitate calculation of FQA metrics.

Finally, there are logistical issues to contend with including where to hold committee meetings, whether to pay committee members for their participation, and how to follow-up with committee members after meetings are completed. Those considering embarking on FQA for their region should not only reflect on the observations presented here, but also reach out to those individuals who have successfully completed similar projects to ensure they achieve a positive outcome.

termed “ecological inertia” characterized by slower growth and a life history strategy focused on allocating resources to structural tissue for long-term survival (Chapin 1991; Lopez et al. 2002). By contrast, herbaceous species allocate resources differently, with a life history strategy that typically results in short-term survival in comparison with trees (Grime 1977). In this respect, herbaceous species are more likely to show the effects of short-term disturbance when compared to woody species (DeBerry and Perry 2015; Matthews et al. 2015). Some studies noted better performance of FQA when individual community layers were separated out in the analysis (e.g., herbaceous, shrub, sapling, and tree), emphasizing that the herbaceous layer indices were most often correlated with ecological condition, whereas tree layer indices provided limited information (Nichols et al. 2006; DeBerry and Perry 2015; Matthews et al. 2015).

One interesting consideration is the potential effect of these properties on FQA performance in regenerating forest communities like wetland mitigation sites. DeBerry (2006; DeBerry and Perry 2015) described a phenomenon referred to as “*C*-value inflation” in which younger mitigation sites were typically planted with highly conservative species (e.g., $C>5$) due to planting requirements imposed by regulatory agencies, whereas older sites followed a more natural successional trend characterized by dominance of tree species with lower conservatism values. A common observation on mitigation sites is that planting “late successional” (i.e., highly conservative) tree species on young mitigation sites results in high mortality and eventually a natural turnover in which the regenerating tree layer is replaced by “early successional” (i.e., lower *C*-value) species (McLeod et al. 2001; Matthews et al. 2009b; DeBerry and Perry 2012). Research in mitigation systems has emphasized the importance of species composition in success monitoring (DeBerry and Perry 2004, 2012; Spieles 2005; Matthews et al. 2009b), which makes FQA a desirable tool for assessment since composition is indirectly indexed through the each species’ unique *C*-value. However, when regulatory agencies impose FQA metric thresholds (e.g., FQI>25 or $\bar{C}$ >3.5; see DeBoer et al. 2011), they may be arbitrarily setting sites up for failure regardless of the target ecosystem (e.g., forested, scrub-shrub, or emergent wetlands) due to the combination of *C*-value inflation and the natural successional trajectories of wetland mitigation sites (Matthews et al. 2009b; DeBerry and Perry 2015). A better approach may be to simply evaluate FQA metrics for the herbaceous layer in mitigation sites (DeBerry 2006), or to set realistic target thresholds based on comparison to a large number of reference sites identified within a regional landscape setting (Matthews et al. 2009b).

The Species-Area Problem and Index Form The most commonly cited criticism of FQA is that the square root of N (native species richness) transformation in the equation for FQI (see Box 1) results in an index that focuses more on *area* than *condition* (Bried et al. 2013). This is due to the fact that species richness tends to increase with increasing wetland size (i.e., species-area relationship), so that a small wetland with a few highly conservative species (e.g., $\bar{C}>5$) could end up with a *lower* FQI than a large wetland with high species richness but low-ranking *C*-value species (e.g., $\bar{C}<2$). The relative conservation status of these two wetlands might be subject to debate, but few would deny the fact that there are many unique small wetlands supporting rare species that would be undervalued by a straight FQI comparison with larger wetlands just based on the species-area relationship and the dependence of FQI on richness (Mushet et al. 2002; Cohen et al. 2004; Matthews et al. 2005; Miller and Wardrop 2006; Chu and Molano-Flores 2013). For example, vernal ponds are small, often isolated wetland sites that tend to be low in species richness but high in habitat quality and species conservatism (Bried et al. 2013), whereas mineral flats can be large, expansive sites with high species richness but low conservatism (Nichols et al. 2006). Direct FQI comparisons between these two types of wetland habitats might result in the erroneous conclusion that the former (potentially lower FQI due to low richness) lacks conservation potential in comparison to the latter (potentially higher FQI due to high richness).

Several correctives have been proposed to minimize the species-area problem in FQA. Examples include standardization of sample area size within each wetland (Bourdaghs et al. 2006; DeBoer et al. 2011; DeBerry and Perry 2015), collecting data from standard plot sizes within each wetland (Rocchio 2007), focusing on $\bar{C}$ as the primary index rather than FQI since $\bar{C}$ is independent of species richness (Cohen et al. 2004; Rocchio 2007; Bried et al. 2013), introducing modifications that relativize the index to reduce the effect of species richness (Miller and Wardrop 2006), and calculating abundance-weighted versions of FQI to normalize the species-area influence (Cretini et al. 2012; DeBerry and Perry 2015).

Effective use of some of these approaches will depend on the type of analysis being performed. For example, when comparing natural wetlands within a specific habitat classification across a state or region, standardizing sampling area would be beneficial as it would ensure sampling balance across the domain of study sites. Further, index modifications like the one proposed by Miller and Wardrop (2006), or just using $\bar{C}$ for data analysis, can be applied in any situation where FQA is used. Plot-based sampling is typically a regulatory requirement for compliance monitoring in wetland mitigation sites, so a standardized plot sampling approach could be a easily incorporated into mitigation assessment (Herman 2005; DeBoer et al. 2011; DeBerry and Perry 2015). Along the same lines, abundance data are also usually required as a component of mitigation monitoring, so abundance weights can be easily integrated into a modified FQI for created and restored wetland sites

(DeBerry 2006). It should be noted that most researchers who tested an abundance-weighted FQI in natural habitats suggested that performance of the abundance-weighted index did not warrant the additional effort required to collected data on species cover, density, frequency, etc. (Francis et al. 2000; Cohen et al. 2004; Bourdaghs et al. 2006; Rocchio 2007; Cariveau and Pavlacky 2009). However, some have noted that abundance weights are useful because of their stabilizing effect on the species-area problem (Cretini et al. 2012), and also because abundance-weighted FQIs have been shown to preserve the conservatism ranks of wetland sites while providing more information about relative ecological condition based on quantitative measures of the inhabiting species (DeBerry and Perry 2015).

This leads to another challenge that wetland practitioners are faced with when attempting to apply FQA in wetland evaluation, namely, that the FQA approach does not produce a single index that is considered "best" in all circumstances. As previously stated, although both FQI and $\bar{C}$ were originally intended to be the sole product of FQA (Swink and Wilhelm 1994), the unitless property of these two metrics has allowed researchers to devise novel and creative modifications to answer specific research questions. For some, the answer to the question, "Which index should I use?" is straightforward: all of them. In other words, because metrics like FQI, $\bar{C}$, and related modifications are easy enough to compute, and because they are each intended to answer related but slightly different questions, some researchers are recommending that scientists and wetland managers should report all relevant FQA metrics (Rocchio 2007). Certainly for FQA testing within specific wetland community types or on a statewide or regional basis, researchers should evaluate the performance of all FQA-related metrics deemed appropriate for the research questions being addressed and make recommendations accordingly.

The "Botanical Acumen" Problem The FQA approach is limited to some extent by the field experience of the wetland scientists and botanists collecting the data. The accurate identification of several wetland plant taxa, such as grasses and sedges, requires a high level of field botanical skill that is often not consistently represented across the population of scientists and wetland managers who routinely perform wetland evaluations (U.S. EPA 2002). This presents the problem of consistency – if many conservative species are "overlooked" due to difficulty of identification, then FQI values can be artificially lowered by sampling bias irrespective of the actual conservatism of the community being sampled. The research cited in Table 1 generally does not address this "botanical acumen" problem (DeBerry 2006), but it is a concern because of the importance of species composition in the FQA approach and the critical role that *species identity* plays in the application of *C*-values to the FQA metrics. Although restricting FQA to well-known or dominant taxa has been proposed as a rapid approach that most wetland practitioners would be qualified to perform (e.g., MPCA 2014), there is evidence that targeting only abundant taxa reduces the level of certainty in FQA indices (Cohen et al. 2004). Of course, the best approach would be to ensure that FQA assessment teams are comprised of competent field botanists, and that quality assurance measures (e.g., voucher submittals to herbaria) are included in the work plan for a wetland evaluation program. The extent to which this can be implemented in practice, though, is questionable.

CONCLUSION

In our review of FQA trends in wetland evaluation, we have been careful to include the broad range of opinion on the applicability of this assessment tool in a wide array of wetland habitats across North America. In doing so, we have discussed both the benefits and challenges of the FQA approach as interpreted by wetland scientists and practitioners who have "put FQA to the test" in wetland environments. At this point, it is important to reiterate that regardless of the various challenges and potential weaknesses noted above, in the majority of FQA studies conducted to date in wetland habitats, researchers have concluded that FQA is a useful tool for wetland evaluation. Below are some key summary points for consideration by those anticipating use of the method in future research, or for those actively engaged in using FQA to evaluate wetlands:

The conservatism concept, which is captured in the *C*-value assignments for a given region, is a powerful idea that lends itself to versatility in practice. The emphasis on state or regional applications seems most appropriate given the differences in ecological tolerances that even a single species can exhibit over different geographic areas. The regional approach is consistent with current trends in wetland delineation and regulatory programs, and the use of ecoregions in the *C*-value listing process may ultimately be the most ecologically-relevant approach to cataloguing conservatism.

Although the FQA approach does not produce a single index that is appropriate for all situations, ease of use allows wetland practitioners to calculate any number of FQA metrics with minimal effort. Researchers are encouraged to consider all potential FQA metrics that could be ecologically relevant within a given region, and to then test those metrics for applicability using the methods described above. As the research in Table 1 demonstrates, in some cases $\bar{C}$ "works better" than FQI and vice versa, and in other cases modified indices can provide a more consistent and reliable prediction of ecosystem condition.

Careful consideration should be given to the specific research questions being asked before study design and data collection methods are finalized for a typical FQA project. While a species list is all that is needed to compute $\bar{C}$ and FQI, researchers may want to minimize the species-area influence on richness by standardizing sample area or plot size, by introducing index modifications, or by accounting for relative abundance in the FQA metrics. Researchers may also want to control sampling season for multi-year research in which inter-site comparisons will be made, or in which time-dependent community changes will be evaluated (e.g., wetland mitigation monitoring).

Although FQA has been used to compare wetlands from different community types (e.g., forested vs. emergent), this approach should be discouraged even in studies that are designed to identify natural habitats for conservation. In practice, targeting habitats for conservation is more appropriately informed by establishing habitat-specific thresholds of conservatism value (e.g., FQI>45, $\bar{C}$>4.5) rather than making relative comparisons between different wetland types.

Non-native species should be regionally reviewed for use in FQA due to the additional ecological information provided by including non-natives in FQA metrics. For states or regions considering developing or revising a *C*-value list, we recommend avoiding the practice of indiscriminately assigning *C*=0 to *all* non-natives across the board. An arbitrary value of zero does not account for the relative differences among non-native species with respect to floristic quality (e.g., invasive and non-invasive plants are not equivalent in expressing ecological integrity). It is not apparent what the best approach would be to differentiate floristic quality for the non-native species within a geographic area, but research is ongoing. At a minimum, FQA studies should be clear in documenting how non-native plants are treated in the analysis.

Establishing absolute FQA metric thresholds for wetland mitigation success criteria (e.g., FQI>25 or $\bar{C}$>3.5) is discouraged. While this practice in theory should encourage wetland managers to maintain mitigation sites with high conservatism values, it does not account for normal successional trajectories or the influence of factors like *C*-value inflation (see text under "FQA Issues in Forested Wetlands"). FQA success thresholds have been described as unrealistic given the early successional state of the typical wetland mitigation site. This could result in large and unnecessary expenditures of time and money "fixing problems" on sites that don't meet their FQA criteria but that are actually just following normal successional patterns of vegetation development based on our current scientific understanding in these systems. A better approach may be to establish realistic thresholds that account for different stages of successional development as a site matures, or to set thresholds based on comparison to a large number of regional reference sites.

More research is needed to test the performance of FQA indices in forested wetlands. At a minimum, wetland practitioners are encouraged to calculate vegetation layer-based FQA metrics in addition to the overall community metrics, with an emphasis on the herbaceous layer due to its efficacy in differentiating ecological condition under FQA analysis.

Finally, one area where FQA is likely to gain additional use is in the development of regional, vegetation-based IBIs for specific wetland classes. This is a beneficial use of FQA because it incorporates the robust conservatism concept in a format that promotes rigorous testing and selects only metrics with significant correlations to *a priori* disturbance gradients (e.g., dose-response analysis). ■

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