



A null model test of Floristic Quality Assessment: Are plant species' Coefficients of Conservatism valid?



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ARTICLE INFO

Article history:

Received 3 August 2014

Received in revised form 6 November 2014

Accepted 17 November 2014

Keywords:

Ecological indicator

Human impact

Non-native species

Species occurrences

Vascular plant

Natural area quality

ABSTRACT

Floristic Quality Assessment (FQA) was developed as a tool for quantifying the conservation value of natural areas based on their plant species composition and richness. Floristic Quality Assessment is based on Coefficients of Conservatism (C values) assigned to each plant species in a region or state. Each species i , is assigned a value C_i , on a scale of 0–10 by expert botanists, based on its fidelity to undegraded natural areas. A criticism of Floristic Quality Assessment is the subjective nature of these C values. Our objective was to determine if C values of individual species are indicative of the C values of species with which they co-occur. If subjectively assigned species' C values carry meaningful information about plant assemblages and the conservation value of particular habitats, then individual species should tend to co-occur with species of similar C. We tested this hypothesis using occurrences of 1014 species in 388 forests and wetlands across Illinois, USA. Using a null model approach, we found that species co-occurred with species of similar C far more often than would be expected by chance; affirming the predictive ability of subjectively assigned C values. Furthermore, we quantified the extent to which each species was under- or overvalued relative to its co-occurring species assemblages to assess if any species C values were mis-assigned. Woody plants and perennial herbs, as groups, were undervalued as ecological indicators, i.e. their C values were too low. Several non-native species, which, by convention, are assigned a C of zero, were over- or under-valued relative to native species with a C of zero. Based on species occurrences across hundreds of sites, our results indicated that, despite their subjective basis, C values carry considerable ecological information, such that a given species can be used to predict the C values of its co-occurring assemblage. However, some species C values appeared less accurate than others. Our methodological approach could be applied in other states or regions to validate and refine C value assignments.

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1. Introduction

Floristic Quality Assessment (FQA) was developed as a tool for rapidly assessing the conservation value of natural areas. First developed for the Chicago, Illinois (USA) region (Swink and Wilhelm, 1979, 1994), it has since been adopted in several regions within North America (Medley and Scozzafava, 2009) and beyond (Landi and Chiarucci, 2010; Malik et al., 2012; Tu et al., 2009). Floristic Quality Assessment involves the calculation of ecological indicators based on the richness and composition of the vascular plant taxa within an explicitly defined assessment area (e.g., a habitat patch or sample plot). The primary FQA metrics are the Mean Coefficient of Conservatism (Mean C) and the Floristic Quality

Index (FQI). Coefficients of Conservatism (C values), numeric scores assigned to each plant species in a region or state, are the basis for both metrics. Botanists with regional expertise assign each species its C value, on a scale of 0–10, based on its likelihood of being found in or restricted to undegraded, “remnant” natural areas in that region (Andreas and Lichvar, 1995; Taft et al., 1997). Species restricted to remnant areas are assigned higher numbers, and species that can occur in degraded or anthropogenic habitats are assigned lower numbers. Thus, C can be defined as a score assigned to a species based on its degree of exclusivity to sites without recent anthropogenic disturbances. Non-native plants may be ignored in the metric calculation (e.g., DeKeyser et al., 2003; Swink and Wilhelm, 1994) or included with a C=0 (e.g., Taft et al., 2006).

The presence or absence of an individual plant species may have limited value as an ecological indicator. Therefore, C values are used to calculate community-level FQA metrics such as Mean C and FQI. Mean C is the average C value of all vascular

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plant species observed at a site, whereas FQI is the product of the Mean C and the square root of plant species richness (Swink and Wilhelm, 1994; Taft et al., 1997). Mean C and FQI have been used to identify and monitor natural areas, select areas for conservation purchase or management, monitor the progress of restorations, and set standards for compensatory wetland mitigation, such that these metrics have gained considerable influence on land conservation (Fennessy et al., 2007; Herman et al., 1997; Matthews and Endress, 2008; Swink and Wilhelm, 1994). Because they are the fundamental components of FQA metrics, the C values assigned to individual species are therefore of considerable importance. We evaluated the C values assigned to Illinois, USA, plant species based on species co-occurrence patterns, and we present a methodology that can be used to evaluate C values in other regions.

Criticisms of Floristic Quality Assessment include the unclear and inconsistent definitions of the term “conservatism” in the literature, the absence of grounding in ecological theory, and the subjectivity and lack of validation of individual C values (Bowles and Jones, 2006; Bried et al., 2012; Landi and Chiarucci, 2010; Spyreas, 2014). Users of FQA often assume that although individual C values may be imprecise and untested, problems are remedied when C values are averaged to calculate Mean C and FQI at the community level (e.g., Wilhelm and Ladd, 1988). Once assigned for a region by expert botanists, individual species’ C values are rarely validated, and C values have not been refined or reassigned based on field observations. Furthermore, beyond imprecision in the C values of individual species, the potential for systematic biases in C value assignments (e.g., unwarranted higher values given to certain groups of species) has not been investigated. Evaluation of individual C values is clearly needed.

Ecological indicators are often evaluated for their effectiveness in discriminating among sites based on degree of human influence or ecological integrity (Karr and Chu, 1999; Mack, 2006; Rooney and Bayley, 2010). Ecological attributes that reliably increase or decrease along a gradient of environmental degradation are considered to be appropriate indicators (Karr and Chu, 1999). This dose–response approach to identifying effective indicators has been used to evaluate FQA metrics, and several studies have demonstrated that Mean C and FQI decrease consistently as human impacts increase (Bourdagh et al., 2006; Bowers and Boutin, 2008; Bried et al., 2013; Cohen et al., 2004; Ervin et al., 2006; Lopez and Fennessy, 2002; Miller and Wardrop, 2006; Nichols et al., 2006). Evaluations of the community-level metrics Mean C and FQI, however, do not provide feedback that could be used to validate or refine the C values of individual species.

The dose–response approach to evaluating indicators could be extended to individual species. Individual C values might be tested by relating the abundance or presence of each species to an independent gradient of human impact (e.g., Bowers and Boutin, 2008; Cohen et al., 2004). However, this approach has serious limitations. Most species, especially conservative species, occur too infrequently to make this approach feasible. Furthermore, human impact is multidimensional and difficult to quantify. For example, wetlands are impacted by alterations to hydrology, nutrient deposition, grazing, sedimentation, and invasive species (Zedler and Kercher, 2004). Species respond individually to these very different human influences, so that each species is likely to have a unique distribution along any chosen stressor gradient (Ehrenfeld, 2008). Testing dose–response relationships for individual species is therefore unworkable for more than a handful of species that occur frequently enough to provide reliable empirical data.

An alternative approach to evaluating C values that avoids the aforementioned problems would be to evaluate patterns of C value co-occurrences. Therefore, we have chosen to test each species based on its co-occurring species; thus treating species as bio-indicators for the C values of other species. If a species’ C value

carries meaningful information about the sites at which it occurs, then sites with more severe, ongoing or recent anthropogenic impacts will predictably support assemblages of species with low average C values. Therefore, on average, species with low C values should co-occur with other species with low C values, and species with high C values should co-occur with other species with high C values. If not, then subjectively assigned C values are inconsistent with expectations, and a species’ C value would convey little information about the habitats that it occurs in or the assemblages that it occurs with. Analysis of co-occurrence patterns therefore provides an empirical validation of individual C values for large numbers of species. In this study, we evaluated whether C values carry meaningful information about the C values of the species with which they tend to associate. In addition, we determined whether certain species or groups of species had C values that were higher or lower than expected, which might suggest systematic bias in C value assignments.

2. Materials and methods

We analyzed species occurrence data from the Illinois Critical Trends Assessment Program (CTAP). CTAP botanists have sampled plant species composition in randomly selected upland and bottomland forest and herbaceous wetland sites throughout Illinois since 1997 (Carroll et al., 2002). Additionally, CTAP botanists selected and sampled reference forests and wetlands, which were representative of the least degraded forests and wetlands in Illinois. For this analysis, we included 388 sites sampled by CTAP botanists between 1997 and 2012. Sites included 157 randomly selected forests, 189 randomly selected wetlands, 25 reference forests and 17 reference wetlands. Forests were sampled from south to north from mid-May through June each year, and herbaceous wetlands were sampled during July. Forest sampling was confined within sites to areas that were homogeneous with respect to aspect, hydrology, topography and forest type, and was generally done at least 50 m from the forest edge. In forests all vascular plant species in thirty 0.25-m² ground layer quadrats were recorded. Quadrats were distributed along three 50-m transects (10 quadrats per transect) that radiated out from a randomly selected center point in randomly selected, nonoverlapping directions. Ground layer quadrats were nested within larger tree (10-m × 50-m) and shrub (2-m × 50-m) plots, within which woody plants were recorded. Additionally, a 0.1-ha plot encompassing one transect was searched for additional species not detected in quadrats. In herbaceous wetlands, vascular plants were recorded in twenty 0.25-m² quadrats located along a single transect placed on the edge of the wetland and oriented to span its hydrological gradient from the upland inward, and a larger (40-m × 50-m) plot encompassing the quadrats was surveyed for additional species. We compiled a total species list for each of the 388 sites by combining lists from herbaceous, shrub and tree plots, and we used species presence–absence data at the site level for all analyses. We used C values from the statewide list published by Taft et al. (1997). All non-native species were assigned $C = 0$.

For each focal species, i , a frequency histogram of the C values of its co-occurring species can be plotted (Fig. 1). Although species i can co-occur with species that have wide range of C values, if i is relatively conservative (i.e., if C_i is large) then i should tend to occur in sites with other conservative species. As species C_i increases, so should the average C of co-occurrences (Fig. 1). In other words, if C values reflect species assemblages, then across a large number of focal species, C_i should correlate positively with the average C value of co-occurring species. Under the null hypothesis that species’ C values are devoid of information about species assemblages, C_i should be unrelated to the average C values of co-occurring species.

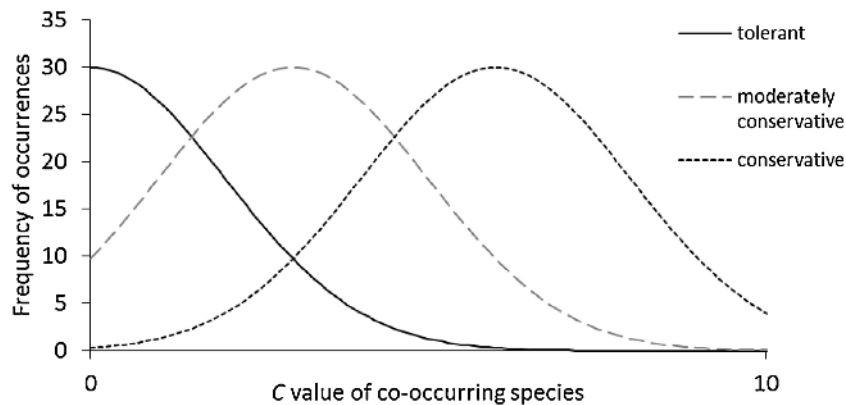


Fig. 1. Hypothetical frequency histograms of the Coefficients of Conservatism (C values) of all species that co-occur with three target species: a relatively tolerant, an intermediately conservative, and a conservative species.

For each of the 1014 species in our dataset, we determined the average C value of its co-occurrences. We counted each occurrence, rather than each co-occurring species. For example, if target species *Acer saccharinum* L. co-occurred with *Ulmus americana* L. at 10 sites we included the C value of *U. americana* 10 times in our average. Likewise, if *A. saccharinum* co-occurred with *Laportea canadensis* (L.) Weddell at eight sites we included the C value of *L. canadensis* eight times. We then calculated the Spearman rank correlation (r_s) between the C value of focal species (C_i) and the average C values of their co-occurrences. Next, we shuffled C values at random 1000 times among species, reassigning C values but keeping the species \times site matrix intact. After each shuffle we recalculated the rank correlation between C_i and the average C values of the co-occurrences. As a test of whether C values carried meaningful information about co-occurrence patterns we compared the observed r_s value with the distribution of r_s values generated under the null hypothesis and derived from the reshuffling of C values. It is important to note that for this analysis we were not concerned with the specific identity of co-occurring species, only their C values, meaning that our analysis was neutral with respect to any particular species associations or communities. We conducted this analysis using the R Language and Environment for Statistical Computing (R version 3.0.1, R Foundation for Statistical Computing Vienna, Austria). The R script is available in Appendix A to this article.

We next determined, for each species i , the residual error from a linear regression of the average C of co-occurrences on C_i . Species with large negative residuals are potentially overvalued, tending to occur with species less conservative than expected. Conversely, species with large positive residuals are potentially undervalued. If groups of species are consistently overvalued or undervalued, this may indicate systematic bias in the assignment of C values in these groups. To explore potential biases we compared the means of residuals between groups of species classified by growth form (woody vs. herbaceous), life span (perennial vs. annual/biennial) and origin (native vs. non-native) using two-sample t -tests. We restricted these comparisons to species that occurred in at least two sites to reduce the influence of infrequent species.

To further evaluate C values, we enlisted a group of 10 botanists from the Illinois Natural History Survey, with an average of 30 years of field experience with the Illinois flora. None of these botanists comprised the original expert group that assigned Illinois C values, and therefore they were not involved in the discussions behind original value assignments. We provided each botanist with a list of all species in our dataset that occurred in at least two sites, along with those species' C values. We highlighted the 80 species with the largest positive and negative residuals and asked each botanist to mark each highlighted species as either overvalued (assigned C value too high) or undervalued (assigned C value too low) based

on their knowledge of the species' natural history and distribution in Illinois. Botanists were not told if a highlighted species was overvalued or undervalued. If botanists had little experience with a particular species, they were asked to skip that species. We used receiver operating characteristics (ROC) graphs (Fawcett, 2006) to evaluate the degree to which botanists' responses corresponded with our co-occurrence-based assignments of species as overvalued or undervalued.

3. Results

Species' C values were positively correlated with the average C of their co-occurring species (Fig. 2). This was not a 1:1 relationship due to the fact that C is bounded between 0 and 10. For example, a species with $C = 10$ will not co-occur exclusively with other species with $C = 10$; thus the average C of its co-occurring species will be less than 10. Likewise, a species with $C = 0$ will co-occur with species which have an average C greater than 0. The observed correlation coefficient ($r_s = 0.557$) was much larger than correlation coefficients calculated under the null model (mean $r_s = -0.003$; Fig. 3). If C values were meaningless, we would expect r_s to be as large as the observed value less than 1 in 1000 times ($p < 0.001$). Results

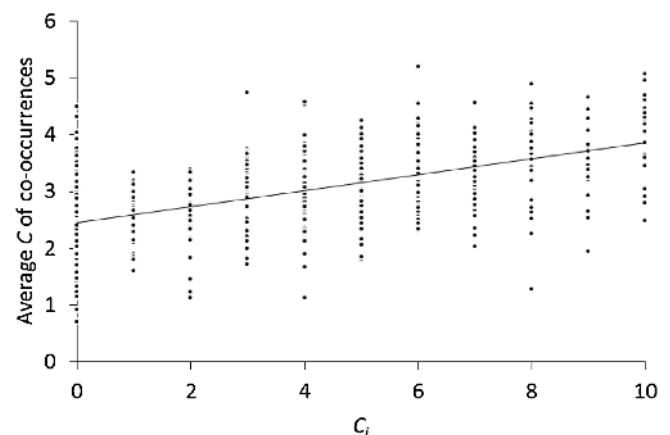


Fig. 2. Relationship between focal species' Coefficients of Conservatism (C_i) and the average C of the species with which they co-occur. Each point represents a single target species, i . The x-axis represents the C_i of the target species, and the y-axis represents the average C of all species that co-occurred with i . Species with large negative residuals from the linear regression of average C of co-occurrences on C_i are potentially overvalued, with a C_i that is high relative to co-occurring species, whereas species with large positive residuals are potentially undervalued, with a C_i that is low relative to co-occurring species (linear regression equation with non-native species included [shown]: $y = 0.141x + 2.45$; linear regression equation with non-native species excluded [not shown]: $y = 0.106x + 2.94$).

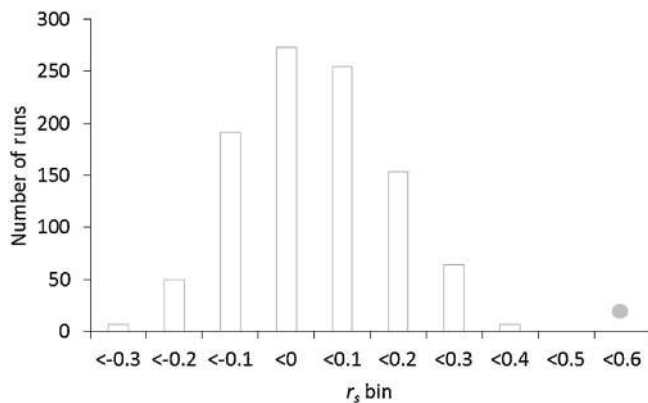


Fig. 3. Observed Spearman rank correlation (r_s) between focal species' Coefficients of Conservatism (C_i) and the average C of their co-occurrences (gray circle) and frequency histogram of r_s values derived under a null model in which C was shuffled randomly 1000 times among species. The observed correlation coefficient ($r_s = 0.557$) was significantly larger ($p < 0.001$) than correlation coefficients calculated under the null model (mean $r_s = -0.003$).

were similar when we restricted this analysis to wetlands only (observed mean $r_s = 0.590$; null mean $r_s = -0.003$; $p < 0.001$), forests only (observed mean $r_s = 0.392$; null mean $r_s = 0.004$; $p < 0.001$), or native species only (observed mean $r_s = 0.451$; null mean $r_s = 0.008$; $p < 0.001$). Therefore, because results were similar, we only discuss results from the combined species and habitat analysis.

Species with large negative residuals from a linear regression of average C of co-occurrences on C_i were classified as potentially overvalued, and those with large positive residuals were classified as potentially undervalued (Fig. 2, Appendix B). For example, the group of species with $C = 4$ co-occurred with species having an average $C = 3.1$ (Fig. 2), but *Symphytotrichum ericoides* (L.) G. L. Nesom, also with $C = 4$, co-occurred with species with an average $C = 1.7$, and we classified it as potentially overvalued. Based on this criterion, we also considered some species with $C = 0$ to be potentially overvalued relative to other $C = 0$ species, and some species with $C = 10$ to be potentially undervalued, despite that C values can only range from 0 to 10.

Of the 1014 species, 790 occurred in at least two sites. Among these 790 species, woody plants were significantly more likely than herbaceous plants to be undervalued (Fig. 4a). Among the herbaceous species, perennials were significantly ($\alpha = 0.05$) undervalued relative to annuals and biennials (Fig. 4b). Mean residuals of native species with $C = 0$ did not differ significantly from mean residuals of non-native species, suggesting that non-natives, as a group, were neither under- nor overvalued (Fig. 4c). However, individual non-native species of note were undervalued (Appendix B); in particular, invasive species such as *Typha × glauca* Godr., *Frangula alnus* Mill. and *Microstegium vimineum* (Trin.) A. Camus tended to co-occur with species with an average $C > 3.4$. In contrast, ruderal non-natives such as *Medicago lupulina* L. and *Polygonum aviculare* L. and crop species such as *Glycine max* (L.) Merr. and *Zea mays* L. were overvalued, co-occurring with species with an average $C < 2.7$. In addition to the invasive non-native species, our list of potentially undervalued species included several native species, such as *Galium aparine* L., *Prunus serotina* Ehrh. and *Calamagrostis canadensis* (Michx.) P. Beauv, which often occupy fairly undegraded sites despite their occasional occurrences in degraded sites like ditches and fencerows.

Botanists' opinions regarding which species were overvalued and undervalued corresponded with our co-occurrence-based assignments (Appendix B) with an accuracy of between 50 and 70%, depending on the botanist. We used ROC graphs to visualize botanists' abilities to correctly distinguish overvalued from

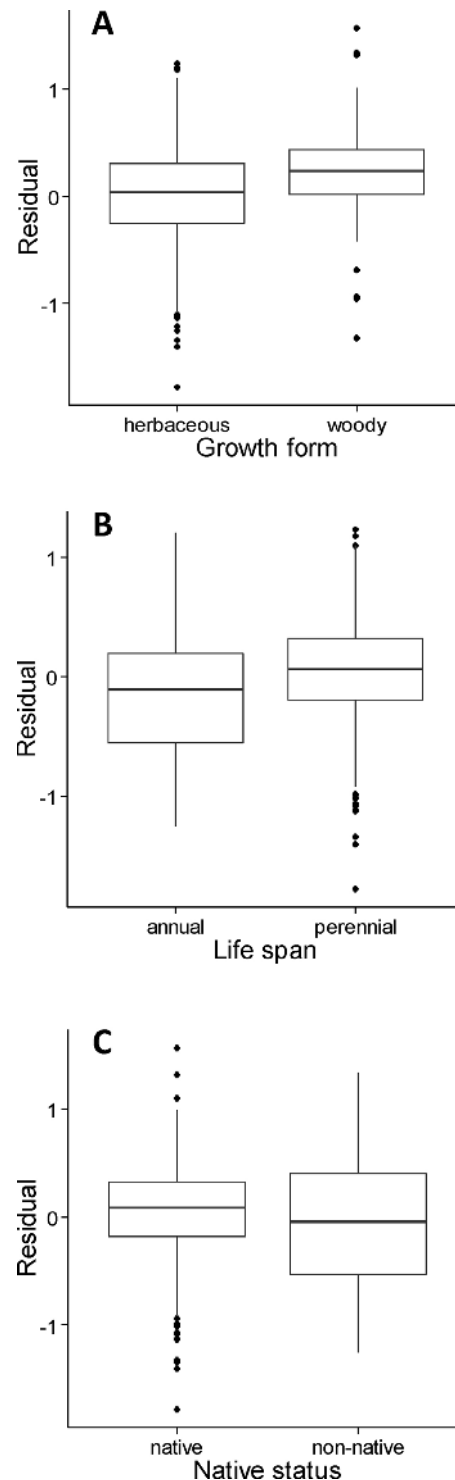


Fig. 4. Boxplots illustrating the distributions of residuals for herbaceous vs. woody species (A), herbaceous annuals/biennials vs. herbaceous perennials (B), and native species with $C = 0$ vs. non-native species (C). Residuals were derived from a regression of average C of co-occurrences on the C_i of focal species (Fig. 2). Group means were compared using Welch two-sample t -tests: (A) $t = -6.53$, $df = 312.74$, $p = 2.7 \times 10^{-10}$; (B) $t = -3.83$, $df = 175.68$, $p = 0.0002$; (C) $t = 1.70$, $df = 129.17$, $p = 0.09$.

undervalued species (Fig. 5). Based on their responses, 9 of 10 botanists fell above the 1:1 line in the ROC graphs, suggesting that botanists' knowledge, as demonstrated by their reappraisal of assigned C values, corresponded with our empirical approach to identifying over and undervalued species. Much of the inconsistency between our empirical results and the botanists' opinions

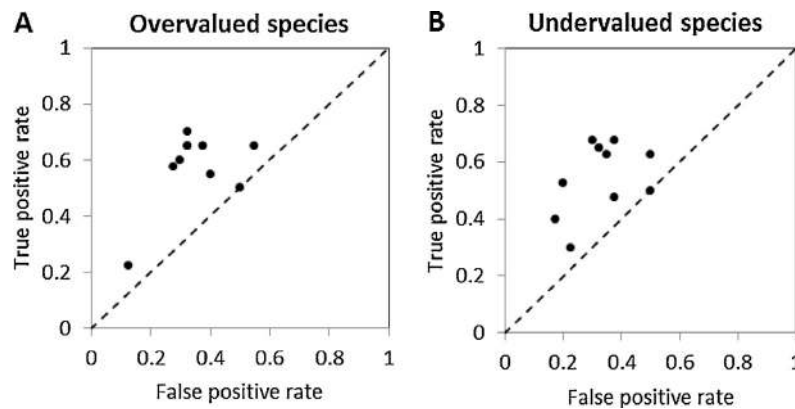


Fig. 5. Receiver operating characteristics (ROC) graphs showing botanists' performance at identifying overvalued (A) and undervalued (B) species. True positive rate is number of overvalued (or for B, undervalued) species correctly classified by a botanist as overvalued divided by the number of overvalued (or for B, undervalued) species. False positive rate is the number of undervalued (or for B, overvalued) species incorrectly classified as overvalued (or for B, undervalued) divided by the number of undervalued (or for B, overvalued) species. Each point represents a different botanist, and 9 of 10 fell above the 1:1 line in the ROC graphs, suggesting that responses were better than random guesses.

was due to several, often rare, species on the list for which botanists did not have enough experience to make an informed choice, or for which empirical results were imprecise due to the low number of occurrences.

4. Discussion

4.1. Performance of C values

Expert-assigned C values significantly outperformed randomly assigned C values in predicting the average C values of co-occurring plants. This finding lends credence to the original panel-based assignment of C values that underpins FQA. Although botanists' assignments of C values are subjective, they are grounded in years of expertise and intimate knowledge of the natural history of plant species within a state or region. Our study complements studies that have shown that conservative species tend to occur in sites that are species rich and occupied by other conservative species (Matthews, 2004; Spyreas and Matthews, 2006) as well as studies showing that the community-level metric Mean C responds consistently and predictably to human impacts—by decreasing with increasing site or landscape degradation (Bourdaghs et al., 2006; Cohen et al., 2004; Francis et al., 2000; Lopez and Fennessy, 2002; Miller and Wardrop, 2006), or increasing with time since site disturbance (Spyreas et al., 2012). Similarly, when Mean C and FQI have been included in multi-metric indices of wetland integrity, they typically outperform other plant-based metrics such as species richness (DeKeyser et al., 2003; Loughheed et al., 2007; Mack et al., 2000; Miller et al., 2006; Reiss, 2006; Wilson and Bayley, 2012).

Ours is not the first study to evaluate individual C values. Previous authors have investigated C values by relating the presence or abundance of individual species to gradients of human impacts. In a study of 204 wetlands in North Dakota, Mushet et al. (2002) found close correspondence between panel-assigned C values and C values derived from species occurrences along a qualitative gradient of perceived anthropogenic impacts to wetlands. Similarly, Cohen et al. (2004) and Bowers and Boutin (2008) found that the C values of individual species were corroborated by species occurrences along estimated human impact gradients. However, defining a gradient of human impacts is not straightforward. A species may respond positively to one type of impact but negatively to another. Our study differs from previous studies by evaluating species' C values based on co-occurrence patterns. Thus, we treat human impacts implicitly, avoiding difficulties associated with defining and quantifying a gradient of human impact, the so-called “x-axis problem”

in these types of analyses (Mack et al., 2000). Similarly, Fridley et al. (2007), in a study of plant niche widths, used species co-occurrence patterns to avoid the persistent problems associated with defining specific environmental gradients.

4.2. Overvalued and undervalued species

The null model approach used in this study could be applied elsewhere to reveal unintentional biases and refine C values based on observational data. It is unreasonable to expect that a panel of botanists could accurately assign C values for every one of the ~2500 native plant species in Illinois, and some need for refinement is not surprising. We identified several outlier species that tended to occur with species that were more or less conservative than expected. These species can be flagged as potentially having C values that are too high or too low relative to other species in the dataset. Our designation of species as potentially overvalued or undervalued corresponded with the impressions of the separate group of botanists whom we asked to review our list. For example, most botanists agreed that C values of species like *Carex frankii* Kunth and *Populus heterophylla* L. were too high, whereas those of *Calamagrostis canadensis*, *Juniperus virginiana* L. and *Nasturtium officinale* W. T. Aiton were too low. There is a strong case for refining the C values of species with occurrence patterns that conform to the consensus opinions of a second, independent group of field botanists regarding misassigned C values.

One objective of our research was to determine whether there may be systematic patterns or errors in these apparent misassignments. For example, woody plants tended to be undervalued, with positive residuals, relative to herbaceous plants. Similarly, herbaceous perennial plants tended to be undervalued relative to annual plants. These findings were somewhat surprising since, compared to short-lived species, woody species might be expected to be less responsive to recent anthropogenic impacts due to their longer turnover times. As a consequence, woody species may be less effective indicators of current ecological integrity (Moffatt and McLachlan, 2004; Nichols et al., 2006), and we might expect them to be overvalued as indicators. Perhaps botanists have overcompensated for this “ecological inertia” by tending to undervalue long-lived species. Examination of species co-occurrence patterns could be used to reveal unexpected biases in C value assignments in other regions as well.

We suspected that non-natives as a group might be undervalued as ecological indicators since non-native species are assigned C=0 as a rule, and not based on their occurrence patterns or their

relative likelihood of occurrence in high integrity communities. Compared to native species with $C=0$, however, we did not find any evidence to support that expectation. Although non-native species, as a group, were neither overvalued nor undervalued, notable individual species were. Overvalued species included ruderals and crop plants that only occupy severely and periodically disturbed habitats alongside similarly tolerant species (e.g., *Polygonum aviculare*, *Setaria glauca* (L.) P. Beauv. and *Zea mays*). Even with $C=0$, these species were overvalued relative to other species with $C=0$. In contrast, other non-native species may occur in relatively unimpacted natural communities, perhaps invading after minor natural or anthropogenic disturbances (Belote et al., 2008; Davis et al., 2000). These invasive non-natives (e.g., *Frangula alnus*, *Lonicera japonica* Thunb. and *Microstegium vimineum*), although undesirable, are interesting because they might indicate the presence of relatively conservative native species assemblages with considerable potential for response to ecological management. Recently invaded sites may retain conservative species as long-lived clones or in seed banks. Thus, compared to ruderal species, perennial invasive species are indicative of greater current and potential future ecological integrity. Of course, their indicative value may be fleeting because increasing dominance during an invasion may eventually lead to the extirpation of conservative native species (Spyreas et al., 2010; Mitchell et al., 2011).

The question of how to deal with non-natives in FQA has been raised by a number of authors (e.g., Ervin et al., 2006; Miller and Wardrop, 2006; Spyreas, 2014), but it has not been answered satisfactorily. Swink and Wilhelm (1979) initially assigned negative C values to invasive non-native species and other species that were deemed undesirable elements of the regional flora for different reasons (e.g., *Lonicera maackii* (Rupr.) Herder and *Cannabis sativa* L.). However, assigning negative values to the most invasive non-natives actually contradicts the concept of C as an indication of the likelihood of finding a species in an undegraded natural area. As our study suggests, the most invasive non-natives are often good indicators of communities with at least moderately high floristic quality, precisely because these species are able to invade native communities of high conservation value. Some authors, including Swink and Wilhelm in a later edition of Plants of the Chicago Region (1994), recommend leaving non-native species out of FQA entirely, based on the assumption that if non-natives are problematic, their presence will be accounted for indirectly through their competitive effects on the richness or composition of the native assemblage. Another argument for excluding non-natives from FQA is that since they did not evolve locally they are not informative about remnant, historical, natural communities (Masters, 1997; Swink and Wilhelm, 1994). The latter argument is based on a commitment to historical fidelity that is increasingly seen as untenable in an era of rapid global changes (Choi, 2007; Hobbs et al., 2006; Davis et al., 2011). The most commonly advocated approach is to assign $C=0$ to all non-native species (e.g., Bourdaghs et al., 2006; Cohen et al., 2004). However, since species occurrences are determined based on species traits other than their origin, this approach is inconsistent with a strict definition of conservatism as fidelity to undegraded natural areas. One solution would be to consider conservatism and invasiveness as separate indicators in Floristic Quality Assessment, for example by calculating both Mean C and a metric to describe invasiveness (e.g., Luken, 2004).

Coefficients of Conservatism for Illinois plants were assigned almost 20 years ago by Taft et al. (1997) based on the authors' combined 60 years of experience with the flora of the Midwestern USA. During the past several decades however, the growing season in the Midwestern USA has lengthened, and average annual precipitation and the frequency of extreme precipitation events have increased (Andresen et al., 2012). The relative abundance of non-native species has increased, and plant communities have become

increasingly homogenized (Rogers et al., 2008). As a consequence of these regional changes, species that would have been considered tolerant by previous panels of botanists may be considered conservative today, if they were assessed by botanists whose experience spans more recent years. This is an example of the "shifting baseline syndrome" that is latent in ecological assessment (Pauly, 1995). Shifting baselines will force future panels of botanists to grapple with questions about whether conservatism should be defined relative to historic, remnant communities, or if instead, it should be defined relative to the present landscape.

4.3. Conclusions

Despite their subjectivity, our analysis reveals that C values carry considerable information regarding species assemblages. However, botanists must sometimes assign C values to species based on limited knowledge of species' autecology. As a consequence, C value assignment is susceptible to imprecision and even bias. We recommend that future botanist panels be particularly cautious when assigning C values for long-lived and woody taxa. We also recommend that FQA users carefully consider how to interpret non-native species in FQA, in light of their management goals and their definition of conservatism.

We advocate an iterative framework for refining Floristic Quality Assessment and as suggested by Cohen et al. (2004), one that incorporates both data on plant occurrences and the subjective prior knowledge of experts. After an initial assignment of C values by expert botanist panels, C values of individual species should be validated using empirical data from large datasets or monitoring programs. Potentially misassigned C values could then be flagged and re-evaluated, and, in some cases, formally reassigned by the original or a secondary panel of botanists. This approach can be repeated periodically as species occurrence data accumulate.

Acknowledgements

Special thanks to C. Carroll-Cunningham, J. Ebinger, J. Ellis, W. Handel, S. Hill, D. Ketzner, P. Marcum, M. Murphy, L. Phillippe and E. Ulaszek for their participation. Forest and wetland sites were surveyed by Critical Trends Assessment Program (CTAP) personnel at the Illinois Natural History Survey with funding from the Illinois Department of Natural Resources. Anonymous reviews provided helpful comments on the manuscript.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at <http://dx.doi.org/10.1016/j.ecolind.2014.11.017>.

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