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Validating the Use of Coefficients of Conservatism to Assess Forest Herbaceous Layer Quality in Upland Mesic Forests

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ABSTRACT: Monitoring is a key activity in management and restoration, yet practitioners' choices among methods may be limited by available resources. Coefficients of Conservatism (*C* values) have been widely used among practitioners because they are easy to apply. To determine *C*, expert botanists assign species in a flora to one of 11 categories, from 0 to 10, with "0" assigned to species most tolerant of human disturbance, and "10" to species of highest-quality native habitats. Although first proposed over 30 y ago, *C* values are criticized as subjective, creating a need for external validation using independently derived metrics. Our study corroborates earlier validations of *C* values for forest understory species. Our work was made possible by consistent collection of herbaceous layer data from 126 plots at 38 sites across Iowa, USA, spanning five types of forests common in the Midwest: secondary, grazed, urban, managed (timber), and preserved. We used PCA to develop an independent metric of human disturbance (LH_i) based on five plant attributes (exotic, annual, biennial, closed-canopy specialist, mesic-site specialist, and fern) that are determined apart from species' *C* values. We found high correlation between mean *C* and LH_i ($r = 0.837$) and an identical pattern of means for these indices between the five forest types, with secondary forests having lowest and preserved forests highest values. This evidence supports earlier assertions that mean *C* provides a valid, simple, and inexpensive means to assess qualitative differences in the forest herbaceous layer of upland mesic forests due to human disturbance.

Index terms: anthropogenic disturbance, Floristic Quality Assessment, forest monitoring, forest understory, habitat specialist species

INTRODUCTION

Monitoring is a key activity in restoration and management (Sutter 1996; Seastedt et al. 2008). Two central questions that arise are "What?" and "How?" to monitor. The answer to the second question may be challenging because a fairly wide range of monitoring protocols has been proposed, from comprehensive multifactor protocols to simple, single measures of diversity such as species richness. For monitoring related to vegetative species composition, ideally a protocol would provide both qualitative and quantitative information valuable to assess plant community status at a given time, as well as trends over time in response to management actions.

The Society for Ecological Restoration International (SER; Ruiz-Jaen and Aide 2005) provides a comprehensive example. This system, which applies specifically to restoration, recommends use of nine ecosystem attributes for measuring restoration success, ranging from diversity and community structure to ecosystem function, resilience, and integration with the surrounding landscape (Ruiz-Jaen and Aide 2005). However, comprehensive protocols, while well thought out and scientifically sound, present challenges to restoration practitioners and land managers. Two of the biggest challenges are lack of time and money for comprehensive monitoring (Ruiz-Jaen and Aide 2005; Schulte et

al. 2006; Dettman and Mabry 2008). In addition, monitoring for some attributes can only be achieved through detailed long-term studies, while the monitoring phase for most on-the-ground projects is much shorter—on the order of 5 y or less for most restoration projects (Ruiz-Jaen and Aide 2005). Thus, practitioners seek single metrics that are simple, reproducible, scientifically accurate, inexpensive, and easily interpreted (Oliver 2005; Taft et al. 2006).

When a single, comparable index is desired, species richness is often the metric of choice. It is widely used in both conservation and restoration, and more generally in ecological and biogeographical studies (Wilsey et al. 2005). It is relatively easy to measure species richness (a simple count of species) and, therefore, it is often used as a convenient surrogate for diversity, which is complex and includes species richness as well as evenness, dominance, and rarity (e.g., Wilsey et al. 2005). In a study using six grassland sites, Wilsey and others (2005) found that richness was an incomplete surrogate though, and that in their case it was necessary to include relative abundance as an additional metric in order to capture the full range of site diversity.

Further, richness and related metrics may mask information that is particularly relevant for restoration and management.

For example, species richness may remain relatively constant over time, while the flora itself can shift toward exotic species (Dolan et al. 2011) or toward native species that are generalists in their habitat requirements (Rooney et al. 2004). It may also be the case that richness and/or diversity are not well aligned with conservation or restoration goals, which often focus on species identity and community composition (Brudvig et al. 2007; Landi and Chiarucci 2010; Spyreas et al. 2012).

In contrast, Coefficients of Conservatism (*C* values) and related floristic quality metrics (e.g., FQI, FQA) have been more widely used by practitioners (e.g., Landi and Chiarucci 2010 and references therein). In fact, in 2013 the Universal Floristic Quality Assessment (FQA) Calculator was created as an interactive and freely available online tool, and user sessions have increased steadily since its introduction (Freyman et al. 2016). These metrics are popular among managers and restoration practitioners because they are easy to use, provide insights beyond species-level data (Brudvig et al. 2007), and avoid the too-simplistic division between native and exotic species (Dettman and Mabry 2008). These metrics have been applied to a wide variety of plant community types, including wetlands (Lopez and Fennessy 2002; Cohen et al. 2004; Matthews et al. 2009), prairies (Taft et al. 2006; Brudvig et al. 2007; Wallner et al. 2013), and forests (Nichols et al. 2006; Spyreas and Matthews 2006; Dettman and Mabry 2008; Spyreas et al. 2012), and are now also relatively easy to access for a number of flora via the Universal Calculator (Freyman et al. 2016).

To use *C* values, developed originally for plants of the Chicago region (Swink and Wilhelm 1979, 1994), each species in a flora is placed in one of 11 categories ranging from 0 to 10. Species assigned a “0” are able to persist in low-quality habitats, are relatively tolerant of human disturbance, and have general distributions across an array of sites. Species least tolerant of human disturbance and with an affinity for high-quality native habitats are placed in category “10.” In Iowa, nonnative species are not assigned and are, therefore, not included as part of the assessment. Expe-

rienced botanists by state or region make the assignments of species to categories. For Iowa, in the Midwest, USA, for example, coefficients were assigned by highly experienced botanists drawn from different agencies, including the Iowa Department of Natural Resources and the US Fish and Wildlife Service, and from colleges and universities.

However, in our experience and that of other authors (i.e., Mushet et al. 2002; Taft et al. 2006; Matthews et al. 2015), the use of *C* values has been criticized due to their apparently subjective assignment and potentially circular reasoning to justify their use. Work to validate the coefficients using independent objective metrics is an important next step to support their use. Most validation studies documented in the literature apply to the use of mean *C* for North American wetlands, including Ohio (Lopez and Fennessy 2002), North Dakota (Mushet et al. 2002), and Florida (Cohen et al. 2004). More recently, use of mean *C* for forest ground-layer vegetation has been validated for forested wetlands in Virginia (Nichols et al. 2006) and for forests and wetlands in Illinois (Matthews et al. 2015).

We also note that not all attempts to validate *C* have been successful (Landi and Chiarucci 2010), or were of mixed success depending on canopy layer (Nichols et al. 2006). Because such validation is not universal, each flora must be independently validated and different community types or structural elements within each flora must also be treated separately (Nichols et al. 2006).

Our goal in this study was to determine if mean *C* for the forest herbaceous layer could be validated using an independently derived metric based on species distribution in relation to human disturbance. To do this we characterized plant species in our data set by a number of independent attributes that previous research (Mabry 2002; Mabry and Fraterrigo 2009) indicated were important determinants of how species respond to different degrees of human disturbance. We then created an index combining these attributes and determined the correlation between mean *C* and this new independent metric. We

hypothesized that the two metrics would be positively related. Confirmation of this hypothesis would enable us to more confidently recommend the use of mean *C* to woodland managers as a simple, reliable metric to assess the effects of human activities on the herbaceous layer. Furthermore, if successful, the technique used to develop this independent metric could itself be used in future efforts to validate mean *C* for other floras.

METHODS

Study Area

In typical hardwood forest ecosystems, the herbaceous layer represents about 80% of the floristic diversity present (Gilliam 2007) and the characteristics of this layer serve as a strong indicator of floristic quality compared to other canopy layers (Nichols et al. 2006). In addition, substantial effort has been devoted to understanding the ecological significance, resilience, and patterns of recovery of the herbaceous layer from historical human activity (Gilliam and Roberts 2003; Whigham 2004), including the lasting legacy of past settlement and agriculture on its composition (McLachlan and Bazely 2001; Mabry 2002; Flinn and Vellend 2005; DeCandido et al. 2007), and the impact of human disturbance on its capacity to perform ecosystem services (Hooper et al. 2005; Gerken Golay et al. 2013).

Our study was made possible by consistent collection of herbaceous layer data from 126 plots located at 38 sites across Iowa between 2003 and 2011 during a series of studies examining the relationship between human disturbance and forest herbaceous layer change (Figure 1; Gerken 2005; Gerken Golay 2013). The plots were located in five types of remnant forests that are common in the Midwest: secondary, grazed, urban, managed (for timber harvest), and preserved forests. Sites were originally selected based on predominance of a single historical land use—for example, forests used for timber harvest did not also have a history of grazing.

Secondary forests constitute one end of

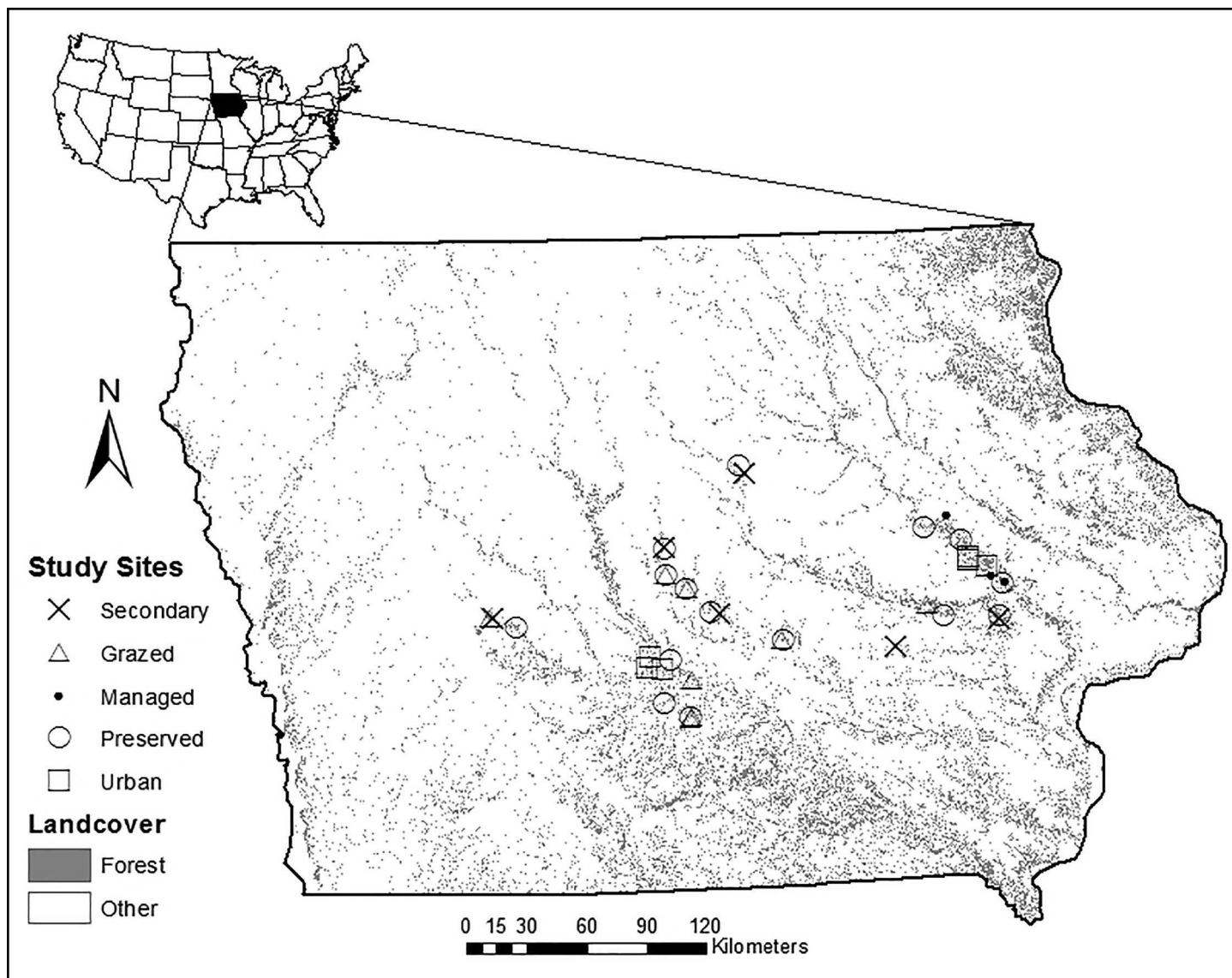


Figure 1. Locations of 38 study sites for five distinct forest land use types (secondary, grazed, urban, managed, and preserved) in central Iowa, USA. Sites were sampled from 2003 to 2011.

the human-disturbance continuum, in areas where total removal of vegetation occurred (typically for conversion to agricultural land use) and subsequent regrowth of forest vegetation followed agricultural abandonment. In Iowa, this type of land conversion led to a major decline in forest area, to a low of approximately 0.92 million ha in the 1970s (Jungst et al. 1998). Since that period, subsequent secondary succession has increased forest area in the state to an estimated 1.5 million ha (of Iowa's total of 14.6 million ha; Nelson et al. 2011). Reestablishment of woody plants can occur rapidly (Flinn and Vellend 2005), although many herbaceous layer species

are very slow to recolonize (Bierzychudek 1982; Flinn and Vellend 2005). Instead, the herbaceous layer in this type of forest is dominated by generalist plant species that disperse readily and expand quickly (McLachlan and Bazely 2001; Spyreas et al. 2012).

Forests that have been grazed by cattle are another type of remnant forest. Like much of the agriculturally dominated landscape of the Midwest, 80% of Iowa's forests have been grazed (Mabry 2002). Grazing directly impacts existing herbaceous vegetation and, in addition, compacts and dries soils (Kucera 1952), resulting in legacy

effects long after cattle are removed and shifting herbaceous species composition toward exotic and generalist native species (Mabry 2002).

Urban and suburban forests are a third type of forest remnant. As urban areas have expanded in the Midwest, remaining forests have become small fragments preserved in city parks or neighborhood green spaces surrounded by networks of roads and developed areas. These remnants experience microclimate changes, such as increased insolation, wind, and temperature (Saunders et al. 1991) that may result in the loss of herbaceous species that depend

on the cool, mesic habitat of forest interiors (Robinson et al. 1994; Groffman et al. 2003) and an increase in exotic and generalist species (DeCandido et al. 2007).

Timber harvest is a fourth type of human disturbance. Twenty-nine percent of privately owned forests in Iowa are harvested to varying degrees (Leatherberry et al. 2005). In Iowa, most of these forests are managed according to single-tree harvest regimens that may conserve or even promote herbaceous species diversity (e.g., Franklin 1993; Jenkins and Parker 1999; Brosofske et al. 2001; Spyreas and Matthews 2006).

A fifth type of forest remnant in the Midwest includes areas relatively undisturbed by human activities. About 130,000 ha (or 16%) of Iowa forests are in public reserves, such as county- or state-managed natural areas (Nelson et al. 2011). While no forests in Iowa have been completely free of modern human activity, these preserved forests represent the best available proxy for reference forests that remain in the state. Although historically (in most cases more than 50 y ago) they may have been logged, they have had minimal recent human impact, and include an herbaceous layer with a diverse suite of spring ephemerals and other forest specialist species that do not tolerate grazing, trampling, and other intensive human uses (Eilers and Roosa 1994; McLachlan and Bazely 2001; Mabry 2002).

Our previous herbaceous layer research has included all five forest types (Gerken 2005; Mabry et al. 2008; Gerken Golay 2013; Gerken Golay et al. 2013). Secondary sites used in this study were former crop fields (verified using historical aerial photos) that had reforested through natural secondary succession processes. Grazed sites had been subjected to cattle grazing within the past 20 y (verified via personal communication with forest landowners), with no additional forest management activities. Previous research in similar forests has documented long-lasting impacts of grazing on herbaceous layer composition that persist for longer than this minimum criterion (Mabry 2002). Urban forest remnants were located within the

corporate boundaries of Cedar Rapids or Des Moines, Iowa, and were in residential neighborhoods or city parks. Harvested sites had a history of single-tree selection harvests, such as regeneration cutting, timber-stand improvement, or crop-tree release (Nyland 2007) and no history of cattle grazing within the last 30 y (also verified with forest management professionals and forest landowners). Preserved forests were chosen very carefully to represent the highest quality forest sites available to study, and were owned by state, county, or private conservation organizations, with documentation indicating they were not disturbed by harvesting activities for at least the past 30 y, and in every case, they had not been disturbed by clearing or cattle grazing (verified using aerial photos and personal communication with land managers at each site). All plots were located on uplands and slopes; bottomland forests were excluded, based on previous analyses indicating that floristic composition and various ecological factors (i.e., hydrology, soil, disturbance regime) in bottomland forests are significantly different from upland forests (Pyle 1995; Lyon and Sagers 1998; Holmes et al. 2005).

Data Collection

Because most forest remnants were small, random or systematic plot placement was not feasible. Instead, we used topographic maps to a priori identify plot locations on uplands and slopes (as per Gerken Golay et al. 2013). We then demarcated 20-m × 20-m plots in the field. This method allowed us to avoid bias and to avoid placing plots close to forest edges or other potentially confounding landscape attributes. The same investigator (Gerken Golay) surveyed all plots using a consistent protocol. The entire plot was traversed once and each species observed was recorded. Species that could not be positively identified in the field were collected for later verification in consultation with botanical experts. Each plot was visited in spring and summer of a given year and not revisited in subsequent years. A subset of plots ($n = 27$) was also visited in fall. This additional survey accounted for 24 new species beyond those observed in spring and summer surveys.

Woody plants were not included in this analysis. We compiled inventories from the five forest types into a single list of all herbaceous species observed by plot for all plots, and included the committee-assigned C for each native species (Iowa State University-Ada Hayden Herbarium, 2004). We then calculated a mean C for the native herbaceous plants in each plot within each forest type (Gerken Golay et al. 2013). Importantly, in relation to the independent metric described below, the Iowa C value set does not include nonnative species (they are not assigned a C value for the Iowa flora).

Development of an Independent Floristic Quality Metric

To test whether mean C could be independently validated, we used a series of three steps. First, all species in the data set were categorized by a set of plant attributes (Table 1) chosen based on previous work showing the attributes were strongly and consistently associated with different degrees of human disturbance in both Iowa and New England (Mabry 2002; Mabry and Fraterrigo 2009). This earlier research included plot by species data from Iowa (using a previous and separate data set) and Massachusetts, representing a range of sites subject to both human and natural disturbance. These data were then converted into a matrix of plots by plant attributes. The subsequent ordination showed that habitat specialists and ferns were associated with the least disturbed sites in both Iowa and Massachusetts, while habitat generalists were associated with the most human disturbance in both states. Nonnative species and annuals/biennials were strongly associated with human disturbed sites in Iowa. We note that we do not explore the characteristics of the relationships themselves (for example, why ferns are most often associated with least-human-disturbed sites), only that the relationships are consistent and, therefore, have high predictive quality. In addition, our earlier work took into account both natural and human disturbance. The least-disturbed sites in both Iowa and Massachusetts were subject to a background of natural disturbances, such as wind and

Table 1. Plant attributes characteristically associated with the least and most human disturbance in Midwestern and northeastern forests, USA (Mabry 2002; Mabry and Fraterrigo 2009).

	Least human disturbance	Most human disturbance
Annual		x
Biennial		x
Exotic species (introduced since European settlement)		x
Life form a fern	x	
Closed-canopy habitat only (specialist)	x	
Rich mesic forest habitat only (specialist)	x	

ice storms. However, the results showed naturally disturbed sites were very distinct from human-disturbed sites, as these site types were clearly distinguished by ordination by both species and the attributes discussed above.

The attributes selected (Table 1) were independently determined for each species by using published regional floras (primarily Gleason and Cronquist 1991 or Barkley 1986, when not listed in the former). The total number of species with a given characteristic was summed for each plot. We then determined mean occurrence of each attribute per plot within forest type.

The second step in our validation integrated the plant attributes using principal component analysis (PCA). Because variances were unequal (ranging from the smallest $Var(Fern) = 2.08$ to the largest $Var(Mesic-site specialists) = 43.33$), each variable was standardized with z-scores before performing the PCA. The PCA axes were centered at zero, and the standardized PCA values ranged between -4.5 and 3.5. To transform the resulting loadings (coefficients in the PCA) back to the original units, each loading was divided by the variable's standard deviation, which thereby reversed the standardization. The de-standardization process increased the PCA values, such that a constant of 3.087 was added to each PCA value to center the equation at 5 and allow for comparison on a scale similar to that for mean C .

Third, a life history index, LH_i , was calculated on a plot basis as:

$$LH_i = \sum_{k=1}^6 \frac{a_k}{s_k} X_{k,i} + Constant$$

where for each plot, i , k represents each variable, a_k represents the value from the first component of the standardized PCA, s_k is the standard deviation, and $X_{k,i}$ are the values for each variable k and each plot i . The result was the index:

$$LH_i = 3.087 - 0.137X_{E,i} - 0.119X_{A,i} - 0.330X_{B,i} + 0.176X_{F,i} + 0.760X_{Cl,i} + 0.054X_{M,i}$$

where for each plot, i , E is the number of exotics, A is the number of annual forbs, B is the number of biennial forbs, F is the number of ferns, Cl is the number of closed-canopy specialists, and M is the number of mesic-site specialists.

By generating the life history index in this way, the values fall between 0 and 10, with 0 indicating high human disturbance (based on attributes previously associated with high human disturbance in mesic forests) and 10 indicating low human disturbance (based on attributes previously associated with low human disturbance in mesic forests). The mean LH_i for sites in this dataset is exactly 5. Data were normally distributed and required no additional transformations.

Statistical Methods for Comparison of Metrics

We examined whether our new metric performed similarly to mean C in two ways. First, plants were assigned their Iowa C value (in the Iowa system, exotics are

not assigned; Iowa State University-Ada Hayden Herbarium 2004). We considered site as the sampling unit, and plots within sites as subsamples. We then examined the relationship between mean C and mean LH_i using Pearson product-moment correlation. We also used one-way analysis of variance (ANOVA) to determine whether mean C and LH_i (dependent variables) differed between the five forest types (the independent variable), and Tukey's HSD post-hoc test to determine whether they showed the same pattern of difference. A strong correlation between the two metrics would indicate mean C is validated by LH_i since LH_i is composed of attributes independent of species' C values and documented by previous research to be associated with forests either most or least disturbed by humans.

RESULTS

We recorded 280 herbaceous species across all plots. Mean C and mean LH_i ordered the sites identically (Table 2) and were strongly correlated with each other ($r = 0.837$). Both mean C and LH_i were lowest for secondary forests and highest for preserved forests, and mean rankings increased from secondary to grazed, urban, managed, and preserved forests (Table 2). One-way ANOVA showed that both mean C and LH_i were highly significantly different by forest land use type (Table 3). Post hoc tests revealed that ANOVA separated the sites identically for both mean C and LH_i , with secondary forests distinct from urban, managed, and preserved, and grazed forests also distinct from preserved.

Table 2. Mean (± 1 SD) of mean C and mean LH_i for herbaceous vegetation communities in five forest land use types (secondary, grazed, urban, managed, preserved) varying in degree of human disturbance in Iowa, USA. Means with the same letter are not different at $P = 0.05$ according to Tukey's HSD.

Land use	Number of sites (plots)	Mean C	SD	Mean LH_i	SD
Secondary	6 (24)	3.9 a	0.42	3.0	1.29
Grazed	8 (20)	4.2 ab	0.37	4.4	0.96
Urban	6 (22)	4.5 bc	0.35	4.8	0.58
Managed	3 (12)	4.7 cd	0.09	5.8	0.36
Preserved	15 (48)	4.8 d	0.29	6.0	0.75

The first component of the PCA captured 42.1% ($P < 0.0001$) of the variation in mean C from a linear combination of the six plant attributes, reflecting negative coefficients for exotics, annuals, and biennials, and positive coefficients for ferns, closed-canopy specialists, and mesic-site specialists, suggesting these attributes effectively captured the gradient of disturbance in upland mesic forests (Table 4).

DISCUSSION

Our goal was to test whether mean C could be validated for Midwestern USA forest herbaceous species by developing a new independent metric based on plant attributes and linked to human disturbance. The strong correlation we found between mean C and this new index (LH_i), and the consistent ordering and separation of land use types by ANOVA, validates use of mean C to characterize forest herbaceous layer flora and indicates that both metrics work well for detecting the impact of human disturbance on the herbaceous layer in mesic upland forests, particularly for secondary and grazed forests. The breadth of the data set we used, collected from 126 plots on 38 sites representing five predominant forest land use types for Midwestern forests, as well as the similarity in patterns among these sets of results, provides independent validation of the use of mean C and lends confidence to our conclusion as it applies to these forests.

There are at least four practical advantages of mean C for land managers and restorationists. First, because it is based on one easily referenced number per species, mean C is an easy and intuitive metric for

calculating a site mean that allows monitoring vegetative composition beyond the individual species level. Second, C values are becoming more widely used and accepted, with an online tool now available to calculate the closely related FQA (Freyman et al. 2016). Third, mean C has previously been reported in the scientific literature, allowing for use of a common metric to compare the impact of human disturbance on the flora of different states or regions. For example, floristic quality findings based on mean C between prairie pothole wetlands in North Dakota and wetlands in Ohio were parallel, despite the differences in climate and geography between the two regions (Lopez and Fennessy 2002; Mushet et al. 2002). Fourth, mean C can be used to examine floristic community change over time. For example, in one study of a recently urbanized area in Indiana, C values were applied to contemporary field observations and to historical herbarium specimens (Dolan et al. 2011). These investigators found that mean C decreased from a historic value of 3.4 to a contemporary value of 2.9, suggesting that this metric is indeed sensitive to compositional changes in response to human disturbance, and that it may also be useful to compare current floristic communities to historical

or previously published floras, expanding the metric's usefulness beyond rapid field assessments of current conditions.

The practical advantages of mean C are also reflected in increasing breadth of application. Mean C has been proposed as effective in assessing floristic quality in a wide range of habitats, including wetlands (Lopez and Fennessy 2002; Cohen et al. 2004; Matthews et al. 2009), prairies and grasslands (Jog et al. 2006; Taft et al. 2006; Brudvig et al. 2007; Wallner et al. 2013), and forests (Spyreas and Matthews 2006; Dettman and Mabry 2008; Spyreas et al. 2012; Matthews et al. 2015). Mean C values have also been useful for a wide variety of monitoring approaches, including distinguishing habitat quality differences among sites (Taft et al. 2006), identifying effects of grassland management on species of conservation concern (Jog et al. 2006), quantifying successional trends in forests over time (Spyreas et al. 2012), tracking restoration projects over time (Lopez and Fennessy 2002), and setting goals for managers and environmental decision-makers (Herman et al. 1997).

Our results corroborate other accounts validating the use of mean C , including studies of wetlands in three distinct North American regions (Lopez and Fennessy 2002; Mushet et al. 2002; Cohen et al. 2004) as well as more recent assessments of forest flora in Illinois (Matthews et al. 2015). In the latter study, Matthews et al. (2015) point out that much information about plant communities is described by this succinct metric. One objective of their study was to identify possible mis-assignments of the coefficients by the expert panel and to evaluate whether there was a systematic pattern of errors. Matthews and coworkers examined the C values of co-oc-

Table 3. One-way ANOVA for differences in mean C and LH_i for five forest land use types in Iowa, USA: secondary ($n = 6$), grazed ($n = 8$), urban ($n = 6$), managed ($n = 3$), and preserved ($n = 15$). Degrees of freedom for land use = 4, and error = 33.

	Mean square	Mean square	F ratio	Probability
Parameter	Type	Error		
Mean C	1.16	0.11	10.61	≤ 0.0001
LH_i	10.0	0.75	14.6	≤ 0.0001

Table 4. Correlation coefficients for individual plant attributes with the life history index, LH_i , for forest herbaceous species, Iowa, USA.

Variable	<i>r</i>
Exotic species	−0.790
Biennial forbs	−0.778
Annual forbs	−0.486
Ferns	0.403
Mesic-site specialists	0.567
Closed-canopy specialists	0.716

curing plants and found that woody and perennial herbaceous plants tended to be undervalued, and some nonnative species were overvalued. As a result, they recommend caution when assigning *C* values to long-lived perennials and woody taxa. In addition, Matthews and others (2015) recommend nonnative species be given careful consideration if included with *C* values (although as previously indicated nonnatives are not assigned coefficients in Iowa and so were not included in the assessment of Iowa flora reported herein). Our use of an independent metric based on plant attributes other than *C* itself to validate mean *C* strengthens the case for further use of mean *C* for qualitative assessment of upland mesic forests.

Reports of the utility of mean *C* are not universal, however, suggesting that they may need to be evaluated and validated separately for each flora. For example, mean *C* was similar among several Midwestern USA prairie sites that had been disturbed in various ways by burning and grazing, and for these plant communities the impact of management was better captured by looking at the proportion of native versus exotic species (Brudvig et al. 2007). The relationship between mean *C* and human disturbance may depend on vegetation layer, or tree canopy versus herbaceous species, for example (Nichols et al. 2006). Another limitation may be related to temporal applications, in which mean *C* may be effective for characterizing early- and mid-successional stages but less effective for characterizing later successional communities—perhaps because the most conservative species are also the ones least likely to establish naturally (e.g., Bowles and Jones 2006; Spyreas et al. 2012). The

coefficients may also be less useful in parts of the world with a long history of human habitation and where it is, therefore, difficult to distinguish between natural and human-influenced habitats (Landi and Chiarucci 2010). Another caveat is that *C* values were not intended to be the only indicator used for assessing the natural quality of a site, but to complement and corroborate the results obtained with other methods (Herman et al. 1997). Thus, they may best be used in conjunction with additional measures such as richness (Francis et al. 2000), or with other taxa-specific evaluations such as bird community use of vegetation (Wilson 2012).

The protocol we developed for creating an independent index based on life history variables may be transferable (and/or easily modified) and could be used to validate mean *C* in other states and regions where species' life history characteristics are similarly associated with human disturbance. The LH_i calculation presented here could be used directly on analyses of floristic inventories, as long as species could independently be classified by relevant plant attributes.

The data set we used to develop the index and to validate mean *C* included forest community plots only on uplands and slopes. Bottomland forests were not included because they are different in composition (Lyon and Sagers 1998; Holmes et al. 2005) and disturbance regime (Pyle 1995). However, research in Illinois demonstrated similar patterns between bottomland and upland forests for species richness and mean *C* (Spyreas and Matthews 2006) suggesting that this metric is probably also a valid way to evaluate floristic integrity

among bottomland forest sites. Similarly, ecological integrity, as evidenced by mean *C* values, was quantified across upland (drier) to bottomland (wetter) sites in Indiana, indicating that the metric is likely to be applicable in a variety of habitat types (Rothrock and Homoya 2005).

CONCLUSIONS

Restoration practitioners and land managers need cost- and time-efficient methods to assess qualitative differences in vegetative composition among different sites and/or over time. Use of coefficients of conservatism developed independently for different flora have been proposed as a relatively simple-to-determine metric that can be used to describe vegetative composition more fully than other indices in widespread use (e.g., species richness). We developed an independent metric that was very highly correlated with mean *C*, and that resulted in an identical pattern of land use type differences. We conclude that mean *C* is a valid metric that can be used to assess human impacts on upland mesic forests in Iowa and potentially other Midwestern forests as well.

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