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Floristic Quality Index for woodland ground flora restoration: Utility and effectiveness in a fire-managed landscape

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ABSTRACT

Monitoring is a critical component of ecological restoration and requires the use of metrics that are meaningful and interpretable. We analyzed the effectiveness of the Floristic Quality Index (FQI), a vegetative community metric based on species richness and the level of sensitivity to anthropogenic disturbance of individual species present (Coefficient of Conservatism (CC)), using ground layer vegetation data from forests and woodlands with different fire histories in the Missouri Ozarks, USA. Specifically, we used total species richness, mean CC, and FQI to quantify differences in ground layer vegetation between burned and unburned sites, determine if relationships between richness and mean CC were consistent at local and landscape-scales, and evaluate the influence of richness and mean CC on FQI values using empirical data. Concerns regarding FQI identified in previous studies were also observed in this study, including a negative relationship between richness and mean CC. However, we observed this negative relationship using data from all study plots (landscape-scale) but not within discrete site types (local-scale). Relationships among mean CC, richness, and FQI were complicated because species richness was strongly correlated to FQI values across plots in which richness was low, whereas mean CC was only correlated with FQI values across plots in which richness was high. We conclude that the interpretation of the Floristic Quality Index may be challenged by: (1) the possibility of obtaining the same FQI value through different combinations of mean CC and richness; and (2) the dominating effect of richness on FQI. Although the FQI metric appears responsive to prescribed fire effects on plant communities in the Missouri Ozarks, the inclusion of species richness and mean CC provide more complete indication of plant community response than FQI alone.

1. Introduction

Ecological restoration has become an increasingly common goal for public and private land managers, and monitoring the outcomes of restoration activities is important for evaluating success (Hobbs and Harris, 2001; Jordan et al., 1990). In terrestrial wooded ecosystems, characteristics of plant communities are often used to evaluate the response to restoration treatments and to compare restoration sites to reference communities (Apfelbaum et al., 2000; Barrioz et al., 2013; Elliott et al., 2009; Kinkead et al., 2013). The metrics developed to describe plant communities are numerous but are commonly based on quantifying plant abundance (e.g., estimates of cover or density), numbers of species present (e.g., species richness), and the distribution of plant abundance among the species present (e.g., evenness, Shannon–Wiener Index of Diversity). Each metric provides different information and is subject to limitations to their use (Taft et al., 2006). For example, Alatalo (1981) argued that evenness measures that include richness in their calculation are limited by sampling biases, and Wilsey et al. (2005) found that diversity metrics that include measures of abundance more accurately captured biodiversity in grasslands than simple species richness.

Many commonly used metrics, such as diversity and abundance measures, describe important characteristics of plant communities but are unweighted by species composition (Taft et al., 1997). Plant community composition (i.e., which species are present) may be an indicator of ecosystem functional diversity (Cadotte et al., 2011) or may reflect site conditions or disturbance history. Metrics weighted by species composition have been proposed for restoration monitoring. Wilhelm (1977) introduced a method for quantifying the

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Table 1
Coefficient of Conservatism (CC) value ranges and descriptions.

CC values	Disturbance tolerance	Site fidelity
0–3	Common in disturbed sites	Widely distributed
4–6	Resilient to moderate disturbance	Matrix species of plant communities
7–10	Resilient to minor disturbance	Site-specific to mature communities

Source: Adapted from Taft et al. (1997).

sensitivity of each region's native flora to anthropogenic disturbance (Coefficient of Conservatism (CC)), and Swink and Wilhelm (1994, 1979) and Taft et al. (1997) combined the vulnerability measure (CC) with native species richness to create the Floristic Quality Index (FQI). In this system, each native plant species in a region is assigned a CC score from 0 to 10, based on the species' association with habitats that are relatively unaltered as compared to pre-European settlement (Swink and Wilhelm, 1994; Taft et al., 1997). Species that proliferate with anthropogenic disturbances (i.e., ruderal species) are ranked with CC values from 0 to 3 (Table 1). Matrix species, those that are somewhat tolerant of post-industrial human activity and occur in common plant communities, are ranked with CC values from 4 to 6, while species dependent on largely undisturbed sites (conservative species) are ranked with CC values from 7 to 10 (Taft et al., 1997). The FQI value for a particular site is calculated by multiplying the mean CC value for a site by the square root of native species richness for the same sampling unit (Taft et al., 1997), thus integrating measures of plant diversity with weighted measures of species composition.

The use of FQI for monitoring restoration outcomes has increased among land managers (MTNF, 2010, 2011), despite several criticisms recently discussed in the scientific literature. For example, there is no consensus concerning the effectiveness of the subjective Coefficients of Conservatism when utilized to calculate a mean value to describe plant community integrity. Some scientists accept that CC values capture a species' fidelity and resilience to disturbances (Chamberlain and Ingram, 2012; Cohen et al., 2004; Cretini et al., 2012; Francis et al., 2000), while others argue against the subjectivity of assigning CC values (Bowles and Jones, 2005; Landi and Chiarucci, 2010). Bried et al. (2012) found that while botanists confidently assigned extreme rankings (0–1 and 9–10) in New York and New England, they did not concur as readily regarding matrix species. Bourdaghs et al. (2006) computed mean CC and FQI values for wetland communities sampled in Wisconsin and Michigan with each state's independently-assigned CC values, finding that CC values were significantly higher in Wisconsin than in Michigan. Thomas (2013) argued against using anthropogenic boundaries as CC scoring boundaries, and stated that scoring by ecologically meaningful units may diminish some of the subjectivity relating to CC value assignation. However, Matthews et al. (2015) showed that a CC value of an individual species could be used to predict the values of co-occurring species in Illinois forests and wetlands, leading to the conclusion that the CC system has relevant ecological value.

Previous authors have also discussed concerns with the interpretation of FQI. Because richness is incorporated in the FQI calculation, factors such as sampling intensity and sampling area will affect FQI values (Bried et al., 2013; Francis et al., 2000; Matthews et al., 2005; Taft et al., 1997). In contrast, mean CC has been observed to remain fairly constant with increasing sampling area (Bourdaghs et al., 2006; Taft et al., 1997). Therefore mean CC may exhibit scale-independence, and it has also been shown to be more sensitive than FQI for ranking anthropogenic disturbance in wetlands in New York (Bried et al., 2013). An additional concern is that a single FQI value may be derived from numerous

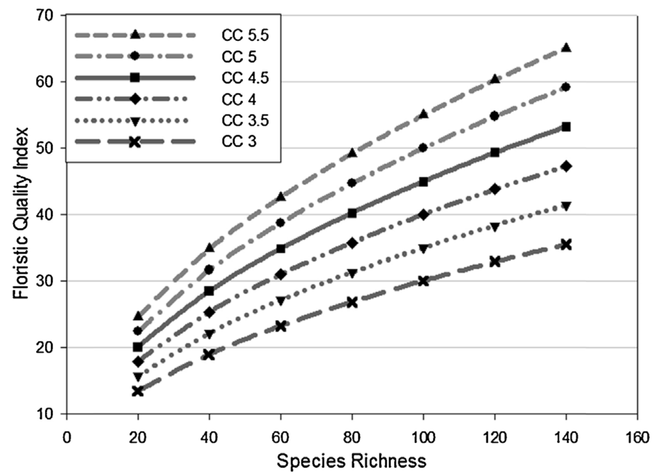

Fig. 1. Relationship between species richness and FQI for different levels of hypothetical mean CC. Similar FQI values can be calculated from differing combinations of richness and mean CC, and the relationship graphs as a power function when richness is on the *x*-axis.

combinations of richness and mean CC values (Fig. 1) (Taft et al., 1997). Moreover, the interpretation of FQI values across ecological communities may be confounded by the presence of different local species pools (DeBerry and Perry, 2015; Nichols et al., 2006; Rooney and Rogers, 2002; Spyreas and Matthews, 2006; Taft et al., 1997). Not only are CC values assigned independently within each state or region (Bried et al., 2012), but also communities and vegetative composition differ across landscapes (Turner, 1989). Due to these differences, FQI may not be comparable across scale or community gradients. Finally, previous studies have found negative correlations between richness and mean CC (Bowles and Jones, 2005; Thomas, 2013), suggesting that plant communities with high richness tend to have relatively low mean CC values. If this correlation is consistent across ecosystems, interpreting FQI could be challenging due to its constituents exhibiting an inherent negative relationship. The consistency of these patterns across sites or ecological communities (both at the landscape and local scales) has not been established, and the effectiveness of the FQI metric has not been evaluated in all vegetative systems and may be limited by community type (Taft et al., 2006).

Despite the criticisms regarding FQI, several studies have reported it to be effective for describing floristic sensitivity to anthropogenic disturbance, particularly in wetland ecosystems (Bried et al., 2014; Lopez and Fennessy, 2002; Matthews et al., 2015; Rothrock and Homoya, 2005). For example, in forested wetlands in Virginia, herbaceous FQI was negatively correlated with increasing anthropogenic disturbance (Nichols et al., 2006), and FQI was found to outperform mean CC at ranking locations based on anthropogenic disturbance in wetlands in the Great Lakes region (Bourdaghs et al., 2006). In Illinois tallgrass prairies, Taft et al. (2006) reported that FQI and mean CC were more effective at differentiating between restoration and reference sites than more traditional metrics of diversity and richness. Although most commonly used in wetlands and open ecosystems such as prairies, the use of FQI has recently been extended to applications in monitoring restoration of woodland and forest ecosystems (DeBerry et al., 2015; MTNF, 2010).

Restoration of woodland, savanna, and prairie plant communities has increasingly become an important management goal in the central hardwood region of the United States, and prescribed fire is commonly used to meet those objectives (Yaussy et al., 2004). Few studies have assessed the utility of FQI in regards to woodland communities and prescribed fire, despite the conflicting results

Table 2
Number of plots within Ecological Land Type and treatment. Soil parent materials include Roubidoux sandstone (RO), Upper Gasconade dolomite (UG), Lower Gasconade dolomite (LG), and Eminence–Potosi dolomite (EM). Values within treatment by ELT combinations are the number of plots.

Ecological Land Type (ELT) name	Annual unit	Periodic units	Control units	Total
RO/UG summit	0	4	11	15
RO/UG shoulders, shoulder ridges, and high benches	2	16	13	31
Exposed RO/UG backslope	9	27	41	77
Protected RO/UG backslope	8	39	25	72
Exposed LG/EM backslope	4	8	8	20
Protected LG/EM backslope	4	25	8	37
Exposed variable depth to dolomite	12	14	6	32
Protected variable depth to dolomite	5	7	6	18
LG/EM benches or shoulder ridges	13	28	10	51
Upper and lower reaches/upland waterways	5	19	7	31
Fens	1	0	0	1
Total	63	187	135	385

previously published about the interpretation of FQI in wetland and prairie ecosystems. We used data from a landscape-scale prescribed fire study in the Missouri Ozarks, USA to determine:

(1) How does prescribed burning affect species richness, mean CC, and the Floristic Quality Index in Missouri woodlands and forests?
(2) Is a negative relationship between richness and mean CC predictable and consistent across ecological site types and burn treatments?
(3) How well do species richness and mean CC each predict FQI based on empirical data from Missouri woodlands?
(4) Are relationships among FQI, species richness, and mean CC consistent at the local and landscape scales?

2. Methods

2.1. Study areas

This study was conducted at the Chilton Creek Management Area (CCMA) and the nearby Missouri Ozark Forest Ecosystem Project (MOFEP), located in the Current River Oak-Pine Woodland/Forest Hills Landtype Association of the Current River Hills Subsection (Nigh and Schroeder, 2002). Common over-story species encountered included oak (*Quercus* spp.) and hickory (*Carya* spp.), with less frequent shortleaf pine (*Pinus echinata* Mill.) occurrences. The elevation of the landscape is primarily between 150 and 300 m above sea level and most slopes are less than 25%. Mean annual temperature is 13.3 °C, and mean annual precipitation is 1120 mm (Chen et al., 2002). Dominant soils are Ultisols and Alfisols derived from highly weathered hillslope sediments, loess, and residuum (Meinert et al., 1997; Zenner et al., 2006).

The Missouri Department of Conservation's Ecological Classification System (ECS)[1] was developed to describe ecological units based on soils, geology, landform, and vegetation. Ecological land types (ELTs) are fine-scale designations within the ECS structure and characterize Ozark forest communities based on landform position, soils, the vegetation community, and climate (Meinert et al., 1997). We used ELTs to stratify study plots into common site types to determine site effects on response variables (Table 2).

2.2. Experimental design and treatments

In 1996, five management units (ranging from 160 to 240 ha in size) were established at CCMA to determine the effects of landscape-scale burning on plant communities. Beginning in 1998, four of the units were periodically burned (each was burned on independent and random 1–4 year fire return intervals with an overall mean fire interval of 2.4 years) and one unit was burned annually (mean fire interval 1.1 years). Prescribed fire at CCMA was primarily applied in the spring dormant season. Units were ignited both through interior ignition patterns and at the boundaries, with no attempts to achieve 100% blackening (Hartman and Heumann, 2003). Fuels included patchy areas of warm-season grass within the larger matrix of hardwood litter, and fires were considered to be of low to moderate intensity (Hartman and Heumann, 2003). The CCMA study was designed as a companion study (using prescribed burning) to MOFEP, a landscape-scale forest management experiment that does not use any prescribed burning. Focused primarily on timber harvest, MOFEP evaluates the effects of even-aged forest management, uneven-aged forest management, and no-harvest management on a variety of biotic and abiotic responses. The first harvest entry to the 280–480 ha units of MOFEP took place in 1996; however, the untreated control units have not received prescribed fire or timber harvest since 1950 (Shifley and Brookshire, 2000).

2.3. Data collection

At both CCMA and MOFEP, each management unit was divided into ecological land types (CCMA) or stands that approximated ecological land types (MOFEP). One or more sampling plots were randomly located within ecological land types or stands. A total of 250 plots were established across CCMA. For this study, we only used sampling plots from the MOFEP control units that occurred on ELTs (assigned to control plots after the ECS system was finalized) that were common to those at CCMA, resulting in a total of 135 unburned control plots.

Vegetation was sampled using a nested plot design with circular, 0.2 ha plots used to sample all overstory trees greater than 11.4 cm dbh (diameter at breast height, 1.37 m) and subsequently smaller units for sampling other vegetation (see Shifley and Kabrick, 2002). Each of four sub-plots located at cardinal directions 17 m from plot center included four 1-m^2 quadrats located at NE, SE, SW, and NW compass points 6 m from subplot center. Vascular ground flora data were collected at these 16 1-m^2 quadrats. Percent cover of each vascular species with any vegetation ≤1 m in height occurring within each quadrat was recorded to the nearest percentage (even if the entire plant was taller than 1 m or was rooted outside the quadrat), providing presence and abundance data for each species in the ground layer. Data were collected from May to September 2013. Unknown species were collected from outside quadrats whenever possible and identified in the laboratory. Of the >67,000 occurrences encountered in the study, less than 0.5% were

[1] The MO Ecological Classification System is now incorporated into the NRCS system and called Ecological Site Descriptions (ESDs).

not identified to species. Nomenclature follows Ladd and Thomas (2015).

2.4. Data analysis

Although data collected in each plot included over-story and mid-story species layers, we analyzed solely within the ground flora strata (DeBerry and Perry, 2015). For each of the 0.2 ha vegetation sampling plots ($n = 16$ quadrats), we calculated total species richness, mean Coefficient of Conservatism, and the Floristic Quality Index, as follows:

$$\text{Species richness}\quad (S) = \text{Number of species}\quad(\text{native} + \text{adventive})\quad \text{per}\quad 16\,\text{m}^2$$

$$\text{Mean}\quad \text{CC} = \frac{\Sigma \text{CC}}{S}$$

$$\text{FQI} = \text{Mean}\quad \text{CC} * \sqrt{N}$$

where CC is the Coefficient of Conservatism for each unique species encountered in a plot (16 m^2), mean CC is the mean Coefficient of Conservatism for the plot, N is native species richness, and FQI is the Floristic Quality Index. CC values were assigned to each native species in the state of Missouri by Ladd and Thomas (2015), and we included non-native species in mean CC calculations by assigning them a CC of 0 (Francis et al., 2000). Of the 67,365 occurrences in this data set, 57 were of adventive species (0.085%).

Data analyses were conducted using the full dataset with plots from all ELTs (representing the landscape-scale) and using plots from specific ELTs (representing within-community patterns). We selected exposed (azimuth range 135–315°) Roubidoux sandstone (RO)/Upper Gasconade dolomite (UG) Backslopes and protected (azimuth range 315–135°) RO/UG Backslopes (hereafter referred to as exposed or protected RO/UG) in the site type analyses because these ELTs: (1) occurred most frequently across the study units (Table 2); (2) are similar to each other in slope position and parent material but differ primarily in aspect; and (3) are common site types that should be representative of the region (Nigh and Schroeder, 2002).

Data from the annually burned unit were pooled with data from the periodically burned units to compare burned plots (CCMA) to unburned plots (MOFEP) (Table 2). We used Generalized Linear Models to determine effects of prescribed burning on each response variable (richness, mean CC, and FQI) using PROC MIXED in SAS (SAS Software, Cary, NC). Fixed effects included treatment and ELT, and unit was the random effect. Data were checked for normality and transformed if needed using PROC TRANSREG. Post-transformation tests of normality were confirmed using PROC UNIVARIATE on the residuals generated by the linear models. Degrees of freedom were from Kenward and Roger (1997), who showed that their method consistently performed better than other methods when sample sizes were small and/or unequal.

Pearson's correlations were conducted using plot-level data to determine the strength of the correlations between richness and mean CC, richness and FQI, and mean CC and FQI for each treatment at the landscape and local scales. Patterns in the data were observed by creating scatterplots. If the scatterplots indicated a non-linear fit, non-linear regression was used to determine relationships. For relationships between richness and FQI, data were fit with the increasing form of an exponential decay function:

$$y = a(1 - e^{-bx})$$

where y = FQI, x = species richness, and a and b were parameters estimated by the model.

Analysis of Covariance (ANCOVA) was conducted using PROC GLM to determine if relationships between variables were affected by treatment. ANCOVA was also conducted on plots with richness values >54 after the relationship between richness and FQI was observed to differ when values of richness were relatively high. If results from ANCOVA included significant treatment effects, we plotted best fit lines for each treatment separately. Statistical significance was determined if $p > 0.05$ for all tests.

3. Results

3.1. Linear models

At the landscape-scale, plots that received prescribed fire had greater richness, mean CC, and FQI than unburned plots (Table 3). Richness was 78% greater on the burned treatment than the unburned treatment, whereas FQI was 37% greater and mean CC was 4% greater on burned sites than on unburned sites. Among the metrics tested, FQI showed the strongest treatment effect. On the exposed RO/UG and protected RO/UG ELTs, the burn treatment did not significantly affect mean CC. Richness was 68% and 76% greater on burned sites than unburned sites for the exposed and protected sites, respectively, and FQI was 36% and 42% greater on burned sites for the exposed and protected sites, respectively (Table 3).

3.2. Pearson's correlation

On burned treatments, mean CC and richness were significantly, negatively correlated at the landscape-scale and on the protected RO/UG plots but were not significantly correlated on exposed RO/UG plots (Table 4). In contrast, mean CC and richness were significantly, positively correlated on exposed RO/UG control

Table 3
Results from ANOVA of treatment differences for richness, mean Coefficient of Conservatism, and Floristic Quality Index. The uppermost table includes all plots across all ELTs, Con = control, $\bar{Y}$ = Mean, SE = standard error, Num = numerator, and Den = denominator.

	Burn $\bar{Y}$	Burn SE	Con $\bar{Y}$	Con SE	Num DF	Den DF	F	Pr > F
Landscape-scale (all plots)								
Richness	64.07	1.64	36.98	1.48	1	4.73	18.60	0.009
Mean CC	4.59	0.02	4.41	0.02	1	6.78	7.12	0.033
FQI	35.92	0.43	26.14	0.49	1	4.97	23.28	0.005
Exposed Roubidoux/upper Gasconade Backslope								
Richness	46.97	2.49	28.00	1.00	1	3.17	19.46	0.019
Mean CC	4.79	0.04	4.53	0.04	1	4.05	4.62	0.097
FQI	32.43	0.87	23.89	0.56	1	3.34	24.14	0.013
Protected Roubidoux/upper Gasconade Backslope								
Richness	48.21	1.60	27.44	1.62	1	3.14	75.94	0.003
Mean CC	4.64	0.03	4.32	0.05	1	4.24	6.59	0.059
FQI	32.00	0.50	22.47	0.70	1	3.82	65.34	0.002

Table 4
Pearson's correlation coefficients by treatment. For each community response variable, strength of relationship (*r* value) is above each *p*-value. Annual and periodic treatments are lumped due to small sample size within the annual treatment.

	Burned plots		Control plots	
	Mean CC	FQI	Mean CC	FQI
Landscape-scale	($n=250$)		($n=135$)	
Richness	−0.345	0.937	−0.142	0.935
	<0.001	<0.001	0.102	<0.001
Mean CC		−0.027		0.188
		0.668		0.029
Exposed RO/UG Backslope	($n=36$)		($n=41$)	
Richness	−0.084	0.947	0.420	0.946
	0.624	<0.001	0.006	<0.001
Mean CC		0.225		0.686
		0.187		<0.001
Protected RO/UG Backslope	($n=47$)		($n=25$)	
Richness	−0.311	0.884	0.145	0.922
	0.033	<0.001	0.491	<0.001
Mean CC		0.159		0.503
		0.285		0.011

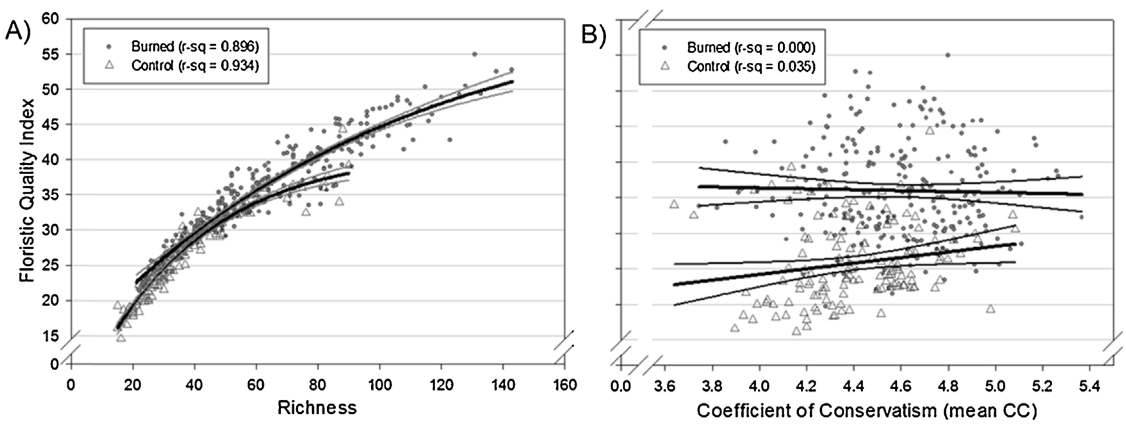

Fig. 2. Relationship between (A) richness or (B) mean CC and FQI (with 95% confidence intervals) for burned and control plots. If results from ANCOVA included treatment effects, we plotted best fit lines for each treatment.

plots and were otherwise not significantly correlated within controls. Richness and FQI were significantly, positively correlated on all treatment and ELT combinations. FQI and mean CC were not significantly correlated on burned treatments but were positively correlated on control treatments (Table 4).

3.3. ANCOVA and bivariate scatterplots

3.3.1. The landscape-scale

Based on the 95% confidence intervals for the non-linear regression lines generated using all the data, FQI values for burned plots were greater at a given value of richness than for control plots when richness was greater than about 54 species per 16 m^2 (Fig. 2A). From the Analysis of Covariance, we found that treatment did not affect the slope for the relationships between mean CC and FQI ($p=0.183$), but intercepts for the treatments were significantly different ($p=0.046$). Mean CC had no statistical relationship with FQI values ($p=0.699$), suggesting that richness was the stronger contributor to FQI (Fig. 2B). For plots that had richness > 54 species, mean CC was statistically related to FQI ($p=0.020$). Burn treatment had no effect on the slope for the relationships between mean CC and FQI ($p=0.678$), and there was no significant difference in the intercepts by treatment ($p=0.637$) (Fig. 3).

Burn treatment did not affect the slope of the relationships between mean CC and richness ($p=0.055$; Fig. 4), which demonstrated a significantly negative relationship ($p<0.001$) with significant differences in the intercepts for burn treatments ($p=0.009$) (Fig. 4). When plots were grouped by ELT, we observed a pattern that suggests a general negative relationship between mean CC and richness based on characteristics of heterogeneous plant communities across the landscape (Fig. 5).

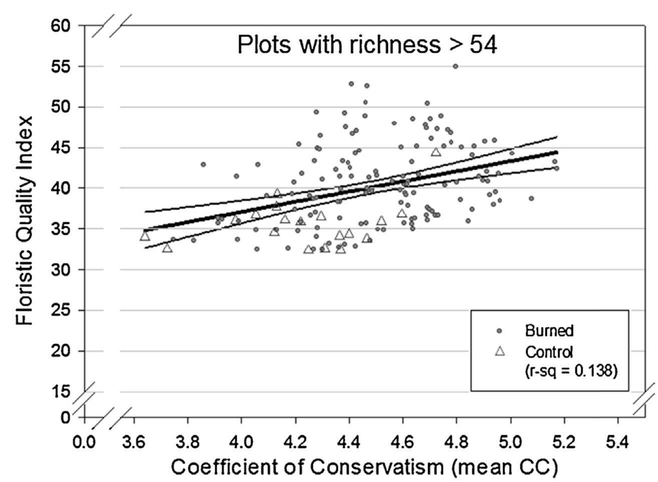

Fig. 3. Relationship between mean CC and FQI (with 95% confidence intervals) for burned and control plots.

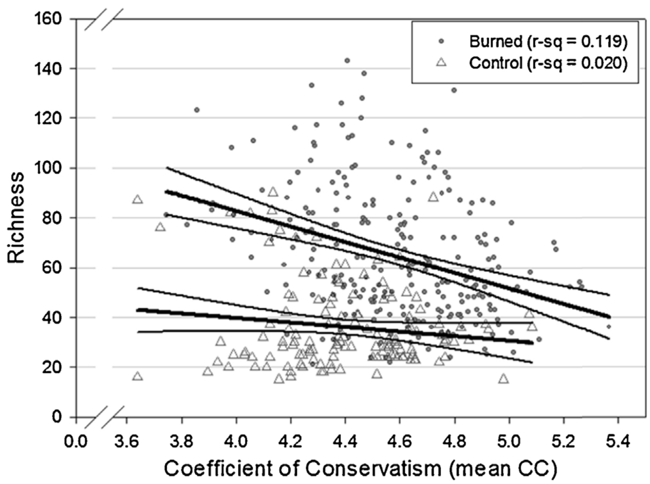

Fig. 4. Relationships between mean CC and richness (with 95% confidence intervals) for burned and control plots. If results from ANCOVA included treatment effects, we plotted best fit lines for each treatment.

3.3.2. Ecological land types

Comparisons of richness and FQI values showed that burned and control plots exhibited little overlap on exposed and protected RO/UG sites, and both ELTs indicated strong linear relationships between richness and FQI (Fig. 6). For both ELTs, there were significant effects of burn treatment on the slopes for the relationships between richness and FQI (p-values ≤ 0.007). However, we found no significant effects of burn treatment on the slopes for the relationship between mean CC and FQI for either ELT (p-values ≥ 0.177). On exposed and protected RO/UG sites, burn treatment did not significantly affect line intercepts ($p = 0.126$ and $p = 0.063$, respectively) but mean CC was statistically related to FQI (exposed $p < 0.001$, protected $p = 0.008$).

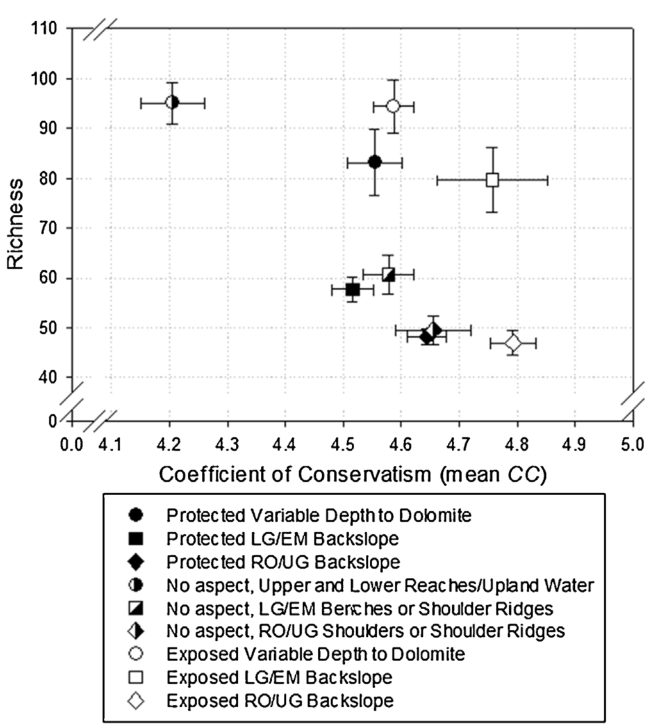

Fig. 5. Mean CC and mean species richness by ELT and elevation. Fill denotes aspect and error bars are one standard error. Lowest elevation ELTs are signified by circles, mid-slope ELTs are squares, and highest elevations are diamonds.

Burn treatment did not affect the slope for the relationships between mean CC and species richness on exposed ($p = 0.058$) or protected ($p = 0.060$) ELTs (Fig. 7) but significantly affected intercepts on both ELTs ($p \leq 0.049$), with greater species richness values on burned plots than on unburned plots across the range of mean CC values. Mean CC was not significantly related to species richness on exposed ($p = 0.630$) or protected ($p = 0.237$) RO/UG ELTs.

4. Discussion

4.1. Treatment effects on plant community metrics

Under the assumption that there are distinct and quantifiable plant community responses to prescribed fire (Apfelbaum et al., 2000; Burton et al., 2011; Glasgow and Matlack, 2007; Hutchinson et al., 2005), we tested for treatment effects on richness, mean CC, and FQI in Missouri Ozark woodland ecosystems. While FQI and richness were significantly different between treatments, mean CC differed only at the landscape-scale. Floristic Quality Index has outperformed mean CC at ranking locations with anthropogenic disturbance in wetlands (Bourdaghs et al., 2006). Lopez and Fennessy (2002) found that FQI could be useful for the assessment, monitoring, and tracking of wetlands and wetland restoration projects through time. Bowers and Boutin (2008) also supported the conclusion that FQI and mean CC both out-performed species richness in riparian corridors in Ontario, Canada, and Taft et al. (2006) found similar results in prairie ecosystems in Illinois.

Plots treated with prescribed fire had significantly greater species richness but not mean CC on both exposed and protected sites in our study. Prescribed fire has been repeatedly shown to increase richness in longleaf pine (*Pinus palustris* Mill.) ecosystems (Brockway and Lewis, 1997) and woodlands in Kentucky (Arthur et al., 1998), Ohio (Hutchinson et al., 2005), and Oklahoma (Burton et al., 2011). Wilhelm and Masters (1994) found woodland ground flora FQI response to be correlated with increasing richness values while mean CC values did not respond to prescribed fire. Increased richness could occur in conjunction with no change in mean CC through several community assembly scenarios, including addition of species with CC values equal to the mean or bi-modal increases in both ruderal and conservative species. Although we found significant differences in richness between treatments, this analysis did not consider temporal changes during the study period and therefore does not fully describe the effect of prescribed fire on species richness or composition in these sites.

4.2. Determination of Floristic Quality Index

Richness was highly correlated with FQI and was the dominant factor in calculated FQI values in this study. Francis et al. (2000) discussed FQI metric derivation as potentially being driven by narrow variation in mean CC scores and relatively greater variation in richness values. While the theoretical range of CC values is 0–10, we found that mean CC ranged from 3.6 to 5.4 in our dataset, whereas richness ranged from 15 ($15^{0.5} = 3.9$) to 143 ($143^{0.5} = 11.9$). Mean CC had the greatest correlation with FQI only within plots exhibiting relatively high levels of species richness, as well as within unburned plots. Although differences in mean CC within this range may have ecological significance, the greater amplitude of species richness values would contribute to the stronger statistical relationships between species richness and FQI. Given the broad spatial scale and high number of sample plots in our study, these findings provide empirical evidence that mean CC does not vary greatly, relative to its theoretical range, across the landscape in Missouri woodlands.

As a mathematical effect of using the square root of richness (Taft et al., 1997), FQI increases more rapidly when a new species

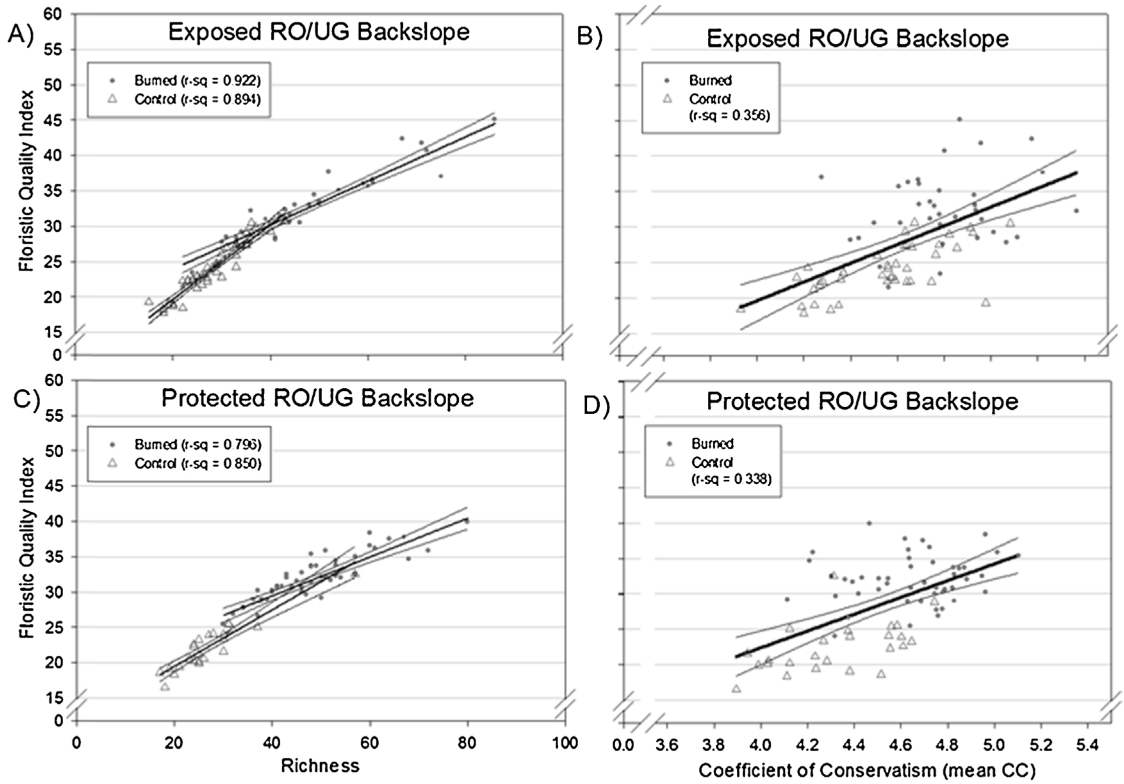

Fig. 6. Relationships between (A) and (C) richness or (B) and (D) mean CC and FQI in exposed RO/UG ($n = 77$) and protected RO/UG ($n = 72$) with 95% confidence intervals.

is added to a less-rich plot than to a more-rich plot (Table 5). Conversely, a change in mean CC has a greater impact on FQI values in plant communities with high richness than in plant communities with low richness (Table 5). Our data suggest that species richness of about 55 per 16 m^2 was the point at which mean CC values began to impact FQI. Thus, a given change in the plant community (i.e., an increase by x number of species or by y units of mean CC) can impact the resulting FQI value differently depending on initial species richness, making it potentially difficult to separate biologically meaningful changes in FQI from mathematical effects. Other studies have proposed alterations to the FQI calculation (DeBerry et al., 2015) to decrease the mathematical impact of richness on FQI values. For example, Miller and Wardrop (2006) used a scaled variant in relation to the maximum attainable FQI value in forested wetlands. Measures of abundance and evenness have been added to the calculation of the traditional FQI (Cretini et al., 2012; Filazzola et al., 2014; Poling et al., 2003), as in the original Floristic Quality Assessment (Taft et al., 1997) from which the FQI was derived.

We found that FQI and mean CC were positively correlated on unburned plots but were not correlated on burned plots within each ELT. The range in species richness observed on unburned plots (range: 90–15 = 75) was narrower than that on burned plots (range: 143–21 = 122). Similar to the observations by Francis et al. (2000) in mature forests with low richness, the narrower range of richness in our unburned plots could increase the mathematical contribution of mean CC to the final FQI value. However, ANCOVA results showed no treatment effects on intercepts or slopes and that mean CC had a significant positive effect on FQI values across both treatments for each local site types. This may be a function of the burned areas having higher mean CC and richness values, suggesting that when

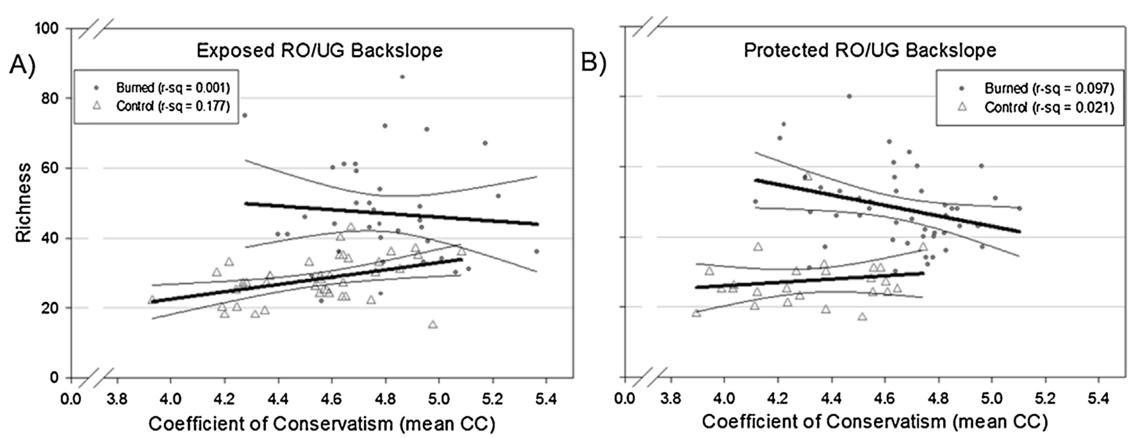

Fig. 7. Relationships between mean CC and richness on (A) exposed RO/UG ($n = 77$) and (B) protected RO/UG ($n = 72$) with 95% confidence intervals. If results from ANCOVA included treatment effects, we plotted best fit lines for each treatment.

Table 5
Example of mathematical relationship between low levels of richness and higher levels of richness with mean CC held constant and with richness held constant in hypothetical plots. Mean CC values have an increased effect on final FQI values at higher levels of richness.

	Richness	$Richness^{0.5}$	Mean CC	FQI	Difference in FQI values
Plot A	20.0	4.5	5.0	22.4	$B - A = 9.25$
Plot B	40.0	6.3	5.0	31.6	
Plot C	100.0	10.0	5.0	50.0	$D - C = 4.75$
Plot D	120.0	11.0	5.0	54.8	
Plot E	20.0	4.6	4.0	17.9	$F - E = 4.47$
Plot F	20.0	4.5	5.0	22.4	
Plot G	100.0	10.0	4.0	40.0	$H - G = 10$
Plot H	100.0	10.0	5.0	50.0	

grouped, the positive relationship may be driven by the differences between burned and control sites. It is not clear if prescribed burning would affect the relationship of FQI and mean CC if burned and unburned sites did not differ in their respective ranges of mean CC and richness.

4.3. Interpretation of Floristic Quality Index

A previously reported, negative relationship between richness and mean CC (Bowles and Jones, 2005; Thomas, 2013) has been discussed as a concern for the interpretation of FQI (Thomas, 2013), because concomitant shifts in opposite directions for richness and mean CC could result in no net change in FQI. For example, Francis et al. (2000) showed that forests in Ontario exhibited increased mean CC and decreased richness as stand age increased, but FQI was not different at different stand ages. In effect, this relationship could result in the inability of FQI to detect potentially ecologically significant changes to plant communities, indicating the importance of including species richness and mean CC in descriptions of plant community response to restoration treatments. However, although our results provide evidence of a significant negative relationship between richness and mean CC at the landscape-scale, these relationships were not consistent across burn treatment and site types. Similar to what has been reported in wetlands (Bourdaghs et al., 2006), these results indicate that this negative relationship is not a predictable property across plant communities in Ozark woodlands.

We observed the strongest negative relationship between richness and mean CC on burned sites analyzed at the landscape-scale, which may be driven in part by the variability in local plant communities across the landscape (Chen et al., 2002). Myers et al. (2013) discuss the importance of environmental variables in community sorting and their effect on large-scale vegetative diversity in the Missouri Ozarks. In general, we observed patterns in richness and mean CC associated with ELTs, such that ELTs with high richness generally exhibited lower mean CC. As a result, the significant negative relationship between richness and mean CC observed at the landscape-scale may be a function of between-community difference rather than processes that affect community assembly within communities.

The relationship between richness and mean CC was either positive or not significant on unburned plots. High mean CC in a plot may be due to a lack of ruderal species or an abundance of conservative species. Following fire, increases in species richness may be attributed to increased occurrence of annual species with ruderal ecology, such as *Erechtites hieraciifolius* (L.) Raf. ex DC., American burnweed (Darbyshire et al., 2012), which has a CC value of 1 in Missouri (Ladd and Thomas, 2015). Francis et al. (2000) found that higher mean CC in relatively mature forest stands than in younger stands was due to fewer generalist (ruderal) species rather than more specialist (conservative) species. In woodland ecosystems, expected increases in high CC species due to burning may be offset by the presence of ruderal species also associated with fire. The short-term net outcome is greater richness, somewhat greater FQI, but little change in mean CC. Species turnover may still occur in unburned plots, but richness and mean CC may exhibit little to no change.

The range of richness and mean CC combinations that can create the same FQI value (Fig. 1) raises the question of whether two plant communities with equal FQI but different richness and mean CC are comparable in ecological integrity. We used histograms of CC values from two pairs of plots, each from protected and exposed RO/UG ELTs, to demonstrate different distributions of CC-values and richness that can lead to similar Floristic Quality Indices (Fig. 8). Proponents of the CC system would argue that the community with a mean CC of 5.22 has higher floristic quality than one with a 4.27 (Fig. 8A) (Rooney and Rogers, 2002). Alternatively, a proponent of richness-based metrics could argue that the increased species redundancy and resilience potential of a richer site has higher floristic integrity (Vogel et al., 2012). However, one may regard a site with high plant species fidelity and low richness as comparable in floristic quality to a rich community dominated by species with ruderal ecology, in which case the Floristic Quality Index would be functioning as intended.

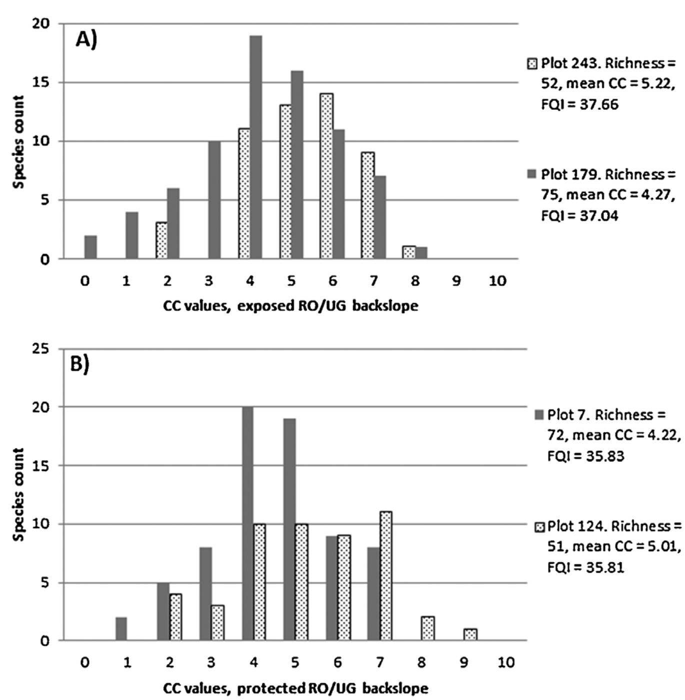

Fig. 8. Histograms of CC values for two plots from (A) exposed RO/UG and (B) two plots from protected RO/UG at CCMA. In each case, the plot-level FQI values were very similar but the richness and mean CC values were different.

5. Conclusion

We explored the application of FQI for monitoring restoration outcomes using a landscape-scale prescribed fire study in the Missouri Ozarks, USA and found that FQI was effective at differentiating burned and unburned ground flora communities. Although FQI is derived from both mean CC and species richness, our empirical data show that FQI values were strongly related to species richness, whereas mean CC was statistically associated with FQI primarily when species richness was relatively high. As a result, these findings suggest that information on species richness would provide land managers with nearly comparable information as FQI, although sampling scale and total species richness values affected those relationships. In addition, we found that interpretation of FQI at the landscape-scale differed from interpretation within specific site types. Our data showed a negative relationship between mean CC and species richness at the landscape-scale but not within specific ELTs. Our results suggest that FQI can be useful for monitoring plant communities in woodland ecosystems, yet the component parts of mean CC and species richness offer valuable information that aid interpretation of fire effects on ground flora. Community assembly patterns of burned woodlands, especially those within site types as compared to the landscape-scale, require additional attention.

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References

Alatalo, R.V., 1981. Problems in the measurement of evenness in ecology. Oikos 37, 199, http://dx.doi.org/10.2307/3544465.

Apfelbaum, S., Bowles, M., O'Leary, M., Stoll, B., 2000. Vegetation change after thirteen years of fire management of a northeastern Illinois oak woodland. Appl. Ecol. Serv., 26.

Arthur, M.A., Paratley, R.D., Blankenship, B.A., 1998. Single and repeated fires affect survival and regeneration of woody and herbaceous species in an oak-pine forest. J. Torrey Bot. Soc. 125, 225, http://dx.doi.org/10.2307/2997220.

Barrioz, S., Keyser, P., Buckley, D., Buehler, D., Harper, C., 2013. Vegetation and avian response to oak savanna restoration in the mid-south USA. Am. Midl. Nat. 169, 194–213.

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

Bowers, K., Boutin, C., 2008. Evaluating the relationship between floristic quality and measures of plant biodiversity along stream bank habitats. Ecol. Indic. 8, 466–475, http://dx.doi.org/10.1016/j.ecolind.2007.05.001.

Bowles, M.L., Jones, M., 2005. Testing the efficacy of species richness and floristic quality assessment of quality, temporal change, and fire effects in tallgrass prairie natural areas. Nat. Areas J. 26, 17–30.

Bried, J.T., Jog, S.K., Dzialowski, A.R., Davis, C.A., 2014. Potential vegetation criteria for identifying reference-quality wetlands in the south-central United States. Wetlands 34, 1159–1169.

Bried, J.T., Jog, S.K., Matthews, J.W., 2013. Floristic quality assessment signals human disturbance over natural variability in a wetland system. Ecol. Indic. 34, 260–267.

Bried, J.T., Strout, K.L., Portante, T., 2012. Coefficients of conservatism for the vascular flora of New York and New England: inter-state comparisons and expert opinion bias. Northeast. Nat. 19, 101–114, http://dx.doi.org/10.1656/045.019.s608.

Brockway, D.G., Lewis, C.E., 1997. Long-term effects of dormant-season prescribed fire on plant community diversity, structure and productivity in a longleaf pine wiregrass ecosystem. For. Ecol. Manag. 96, 167–183.

Burton, J.A., Hallgren, S.W., Fuhlendorf, S.D., Leslie, D.M., 2011. Understory response to varying fire frequencies after 20 years of prescribed burning in an upland oak forest. Plant Ecol. 212, 1513–1525, http://dx.doi.org/10.1007/s11258-011-9926-y.

Cadotte, M.W., Carscadden, K., Mirotchnick, N., 2011. Beyond species: functional diversity and the maintenance of ecological processes and services. J. Appl. Ecol. 48, 1079–1087.

Chamberlain, S.J., Ingram, H.M., 2012. Developing coefficients of conservatism to advance floristic quality assessment in the Mid-Atlantic region. J. Torrey Bot. Soc. 139, 416–427, http://dx.doi.org/10.3159/TORREY-D-12-00007.1.

Chen, J., Huebner, C.D., Saunders, S.C., Song, B.,2002. Plant distribution and diversity across an Ozark landscape. In: Proceedings of the Second Missouri Ozark Forest Ecosystem Project Symposium: Post-treatment Results of the Landscape Experiment. General Technical Report No. NC-227. U.S. Dept. of Agriculture, Forest Service, North Central Forest Experiment Station, St. Paul, MN, pp. 45–64.

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

Cretini, K.F., Visser, J.M., Krauss, K.W., Steyer, G.D., 2012. Development and use of a floristic quality index for coastal Louisiana marshes. Environ. Monit. Assess. 184, 2389–2403, http://dx.doi.org/10.1007/s10661-011-2125-4.

Darbyshire, S.J., Francis, A., DiTommaso, A., Clements, D.R., 2012. The biology of Canadian weeds. 150 *Erechtites hieraciifolius* (L.) Raf. ex DC. Can. J. Plant Sci. 92, 729–746, http://dx.doi.org/10.4141/cjps2012-003.

DeBerry, D.A., Chamberlain, S.J., Matthews, J.W., 2015. Trends in floristic quality assessment for wetland evaluation. Wetl. Sci. Pract., 12–22.

DeBerry, D.A., Perry, J.E., 2015. Using the floristic quality concept to assess created and natural wetlands: ecological and management implications. Ecol. Indic. 53, 247–257, http://dx.doi.org/10.1016/j.ecolind.2015.02.003.

Elliott, K.J., Vose, J.M., Hendrick, R.L., 2009. Long-term effects of high intensity prescribed fire on vegetation dynamics in the Wine Spring Creek Watershed, western North Carolina, USA. Fire Ecol. 5, 68–85.

Filazzola, A., Tanentzap, A.J., Bazely, D.R., 2014. Estimating the impacts of browsers on forest understories using a modified index of community composition. For. Ecol. Manag. 313, 10–16, http://dx.doi.org/10.1016/j.foreco.2013.10.040.

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

Glasgow, L.S., Matlack, G.R., 2007. Prescribed burning and understory composition in a temperate deciduous forest, Ohio, USA. For. Ecol. Manag. 238, 54–64, http://dx.doi.org/10.1016/j.foreco.2006.08.344.

Hartman, G.W., Heumann, B., 2003. Prescribed fire effects in the Ozarks of Missouri: the Chilton Creek Project 1996–2001. In: Proceedings of the Second International Wildland Fire Ecology and Fire Management Congress, Orlando, FL, pp. 16–22.

Hobbs, R.J., Harris, J.A., 2001. Restoration ecology: repairing the earth's ecosystems in the new millennium. Restor. Ecol. 9, 239–246.

Hutchinson, T.F., Boerner, R.E., Sutherland, S., Sutherland, E.K., Ortt, M., Iverson, L.R., 2005. Prescribed fire effects on the herbaceous layer of mixed-oak forests. Can. J. For. Res. 35, 877–890, http://dx.doi.org/10.1139/x04-189.

Jordan, W.R., Gilpin, M.E., Aber, J.D., 1990. Restoration Ecology: A Synthetic Approach to Ecological Research. Cambridge University Press, 348 pp.

Kenward, M.G., Roger, J.H., 1997. Small sample inference for fixed effects from restricted maximum likelihood. Biometrics, 983–997.

Kinkead, C.O., Kabrick, J.M., Stambaugh, M.C., Grabner, K.W., 2013. Changes to oak woodland stand structure and ground flora composition caused by thinning and burning. In: Proceedings, 18th Central Hardwood Forest Conference. General Technical Report NRS-P-117, U.S. Dept. of Agriculture, Forest Service, Northeastern Research Station, USA, pp. 373–383.

Ladd, D.M., Thomas, J.R., 2015. Ecological checklist of the Missouri flora for floristic quality assessment. Phytoneuron 12, 1–274.

Landi, S., Chiarucci, A., 2010. Is floristic quality assessment reliable in human-managed ecosystems? Syst. Biodivers. 8, 269–280, http://dx.doi.org/10.1080/14772001003770307.

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

Matthews, J.W., Spyreas, G., Long, C.M., 2015. A null model test of floristic quality assessment: are plant species' coefficients of conservatism valid? Ecol. Indic. 52, 1–7, http://dx.doi.org/10.1016/j.ecolind.2014.11.017.

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

Meinert, D., Nigh, T., Kabrick, J., 1997. Landforms, geology, and soils of the MOFEP study area. In: Proceedings of the Missouri Ozark Forest Ecosystem Project Symposium: an experimental approach to landscape research. General Technical Report No. NC-193, U.S. Dept. of Agriculture, Forest Service, North Central Forest Experiment Station, St. Paul, MN, pp. 56–68.

Miller, S.J., Wardrop, D.H., 2006. Adapting the floristic quality assessment index to indicate anthropogenic disturbance in central Pennsylvania wetlands. Ecol. Indic. 6, 313–326.

MTNF U.S.D., 2010. Proposal for Collaborative Forest Landscape Restoration Program [WWW Document]., pp. 4–5, http://www.r5.fs.fed.us/restoration/documents/cflrp/2010Proposals/Region9/MarkTwain/MTwain_NF_revised_MoPWR_CFLRP_proposal.pdf (accessed 22.03.15).

MTNF U.S.D.A., 2011. Missouri Pine-Oak Woodlands Restoration Project [WWW Document]., pp. 21, http://gis.fs.fed.us/restoration/documents/cflrp/2011Proposals/Region9/MarkTwain/revMoPWRCFLRPproposal20110217.pdf (accessed 23.03.15).

Myers, J.A., Chase, J.M., Jiménez, I., Jørgensen, P.M., Araujo-Murakami, A., Paniagua-Zambrana, N., Seidel, R., 2013. Beta-diversity in temperate and tropical forests reflects dissimilar mechanisms of community assembly. Ecol. Lett. 16, 151–157.

Nichols, J.D., Perry, J.E., DeBerry, D.A., 2006. Using a Floristic Quality Assessment technique to evaluate plant community integrity of forested wetlands in southeastern Virginia. Nat. Areas J. 26, 360–369, http://dx.doi.org/10.3375/0885-8608(2006)26[360:UAFQAT]2.0.CO;2.
Nigh, T.A., Schroeder, W.A., 2002. Atlas of Missouri Ecoregions. Missouri Department of Conservation, 212 pp.
Poling, T.C., Banker, M.G., Jablonski, L.M., Geiger, D.R., 2003. Quadrat-level floristic quality index reflects shifts in composition of a restored tallgrass prairie (Ohio). Ecol. Restor. 21, 144–145.
Rooney, T.P., Rogers, D.A., 2002. The modified floristic quality index. Nat. Areas J. 22, 340–344.
Rothrock, P.E., Homoya, M.A., 2005. An evaluation of Indiana's floristic quality assessment. Proc. Indiana Acad. Sci. 114 (1), 9–18.
Shifley, S.R., Brookshire, B.L., 2000. Missouri Ozark Forest Ecosystem Project: Site History, Soils, Landforms, Woody and Herbaceous Vegetation, Down Wood, and Inventory Methods for the Landscape Experiment. General Technical Report No. NC-208. US. Dept. of Agriculture, Forest Service, North Central Forest Experiment Station, St. Paul, MN, 314 pp.
Shifley, S.R., Kabrick, J.M., 2002. Proceedings of the Second Missouri Ozark Forest Ecosystem Project: Post-Treatment Results of the Landscape Experiment. General Technical Report (GTR) No. NC-227. U.S. Dept. of Agriculture, Forest Service, North Central Forest Experiment Station, St. Paul, MN, 227 pp.
Spyreas, G., Matthews, J.W., 2006. Floristic conservation value, nested understory floras, and the development of second-growth forest. Ecol. Appl. 16, 1351–1366.
Swink, F., Wilhelm, G.S., 1994. Plants of the Chicago Region. Indiana Academy of Science, Indianapolis, IN, USA, 921 pp.
Swink, F., Wilhelm, G.S., 1979. Plants of the Chicago Region (Revised and Expanded Edition With Keys). The Morton Arboretum, Lisle, IL, USA, 922 pp.
Taft, J.B., Hauser, C., Robertson, K.R., 2006. Estimating floristic integrity in tallgrass prairie. Biol. Conserv. 131, 42–51, http://dx.doi.org/10.1016/j.biocon.2006.02.006.
Taft, J.B., Wilhelm, G.S., Ladd, D.M., Masters, L.A., 1997. Floristic quality assessment for vegetation in Illinois, a method for assessing vegetation integrity. Erigenia 15, 3–95.
Thomas, J.R., 2013. Golden prairie floristic integrity report to the Missouri Prairie Foundation. Institute of Botanical Training, LLC, 41 pp.
Turner, M.G., 1989. Landscape ecology: the effect of pattern on process. Annu. Rev. Ecol. Syst., 171–197.
Vogel, A., Scherer-Lorenzen, M., Weigelt, A., 2012. Grassland resistance and resilience after drought depends on management intensity and species richness. PLoS ONE 7, e36992, http://dx.doi.org/10.1371/journal.pone.0036992.
Wilhelm, G., 1977. Ecological Assessment of Open Land Areas in Kane County, Illinois: A Checklist of the Kane County Flora With Numerical Evaluations – Its Basis, Rationale and Application. Kane County Urban Development, Geneva, IL.
Wilhelm, G., Masters, L., 1994. Floristic changes after five growing seasons in burned and unburned woodland. Erigenia 13, 141–150.
Wilsey, B.J., Chalcraft, D.R., Bowles, C.M., Willig, M.R., 2005. Relationships among indices suggest that richness is an incomplete surrogate for grassland biodiversity. Ecology 86, 1178–1184.
Yaussy, D.A., Hix, D.M., Long, R.P., Goebel, P.C., 2004. Proceedings: 14th Central Hardwood Forest Conference. General Technical Report No. NE-316. U.S. Dept. of Agriculture, Forest Service, Northeastern Research Station, Newtown Square, PA, pp. 162–210.
Zenner, E.K., Kabrick, J.M., Jensen, R.G., Peck, J.E., Grabner, J.K., 2006. Responses of ground flora to a gradient of harvest intensity in the Missouri Ozarks. For. Ecol. Manag. 222, 326–334, http://dx.doi.org/10.1016/j.foreco.2005.10.027.