

A Method for Evaluating Outcomes of Restoration When No Reference Sites Exist

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Abstract

Ecological restoration typically seeks to shift species composition toward that of existing reference sites. Yet, comparing the assemblages in restored and reference habitats assumes that similarity to the reference habitat is the optimal outcome of restoration and does not provide a perspective on regionally rare off-site species. When no such reference assemblages of species exist, an accurate assessment of the habitat affinities of species is crucial. We present a method for using a species by habitat data matrix generated by biodiversity surveys to evaluate community responses to habitat restoration treatments. Habitats within the region are rated on their community similarity to a hypothetical restored habitat, other habitats of conservation concern, and disturbed habitats. Similarity scores are reinserted into the species by habitat matrix to produce indicator (*I*) scores for each species in relation to these habitats. We apply this procedure to an open wood-

land restoration project in north Mississippi (U.S.A.) by evaluating initial plant community responses to restoration. Results showed a substantial increase in open woodland indicators, a modest decrease in generalists historically restricted to floodplain forests, and no significant change in disturbance indicators as a group. These responses can be interpreted as a desirable outcome, regardless of whether species composition approaches that of reference sites. The broader value of this approach is that it provides a flexible and objective means of predicting and evaluating the outcome of restoration projects involving any group of species in any region, provided there is a biodiversity database that includes habitat and location information.

Key words: community similarity, disturbance, fire suppression, indicator species, invasive species, Mississippi, oak woodland restoration, off-site species, thinning.

Introduction

A key component of any restoration project is the ability to measure the success of that restoration (Aronson et al. 1995; Block et al. 2001), and this measure usually involves an assessment of the resulting species assemblage. There has been considerable attention paid to the question of how one determines the appropriate assemblage to strive for in ecological restoration (Jordan et al. 1987; Pickett & Parker 1994; Aronson et al. 1995; Palmer et al. 1997; White & Walker 1997; Swetnam et al. 1999; Block et al. 2001). Some have argued that establishing a reference assemblage for restoration is not necessary and that the perceived need for such a reference is based on outdated ecological principles such as steady-state conditions or ecosystem health (Pickett & Parker 1994). Others, however, have argued that identifying at least one and preferably several reference sites encompassing a sufficient range of historically relevant environmental variation within the region is crucial to evaluating the success of restoration

projects (Aronson et al. 1995; White & Walker 1997). A practical limitation of evaluating restoration success using extant reference sites is that such sites may be extremely rare or in some cases simply may not exist (Clewell & Rieger 1997).

One way in which the value of a species assemblage resulting from ecological restoration could be quantified is through an assessment of how local changes in species composition affect regional biodiversity. Efforts to protect biodiversity have typically focused on species deemed most worthy of protection (Spellerburg 1992; Groves 2003). The species that tend to garner the greatest attention by conservationists are those with limited distributions. These endemic species are usually specialists as well, restricted to certain habitats because of their rare qualities (Cody 1986; Lawton & May 1995). If the preservation of biodiversity requires the protection of species that are indicative of or even restricted to certain habitats, then perhaps one of the more urgent goals of ecological restoration is to increase the abundance and frequency of species that are indicative of rare and relatively undisturbed habitats.

Although some knowledge of reference conditions is essential, and having at least one extant reference site is certainly desirable when conducting restoration projects, similarity to reference conditions is not always the most desirable outcome of restoration. A narrow focus on

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achieving similarity to reference conditions does not provide a regional perspective on off-site species (i.e., species present at the degraded site that were not present or abundant at the site before degradation), including regionally rare off-site species that might be adversely affected by restoration. One scenario in which rare off-site species could be negatively affected by local restoration efforts is when these species have shifted their distributions away from their original habitat (which was destroyed) to another habitat, which itself was altered and is now targeted for restoration. For example, it is conceivable that some fire-sensitive species that were historically restricted to floodplains or mesic terraces (e.g., some spring ephemeral perennial herbs) shifted their distributions into upland areas as a result of the combined effects of fire suppression in upland areas and the conversion of mesic forests to agricultural fields in floodplains. In these cases, fire-suppressed upland forests may serve as the last remaining refugia for these species, refugia that might be lost following restoration of fire to these ecosystems.

Here, we present a method for evaluating responses of indicator species to habitat restoration treatments using a simple species by habitat data matrix, such as those generated by biodiversity surveys. Habitats within the region are rated on their community similarity to a habitat that no longer exists but is hypothesized to have existed historically, as well as to other rare habitats of conservation concern and disturbed habitats. These similarity scores are then reinserted into the species by habitat matrix to produce indicator scores for each species in relation to all habitats of conservation interest, not just the hypothetical habitat to be restored. Success is evaluated not in terms of similarity to specific reference sites but rather in terms of positive net increases in the abundance and frequency of species that are indicative of habitats of interest or conservation concern (i.e., rare habitats that have not been severely disturbed).

We apply this procedure to an open oak woodland restoration project in north Mississippi (U.S.A.) by evaluating preliminary plant community responses to restoration treatments. Determining an appropriate assemblage of plant species for a restored Mississippi oak woodland presents a unique challenge because there is no remaining example of this habitat in Mississippi and therefore no extant reference sites (Brewer 2001). The closed-canopy hardwood forest found in upland portions of the landscape in north Mississippi today is the result of several decades of fire suppression in that area (Brewer 2001; Surrlette et al. 2008), a scenario repeated throughout much of eastern North America (Abrams 1992). The open oak woodlands of north Mississippi were nearly identical in terms of tree species composition and structure to communities such as barrens and open woodlands in adjacent regions (e.g., Kentucky, southern Illinois, Arkansas, southern Missouri, western Tennessee; Robertson & Heikens 1994; Batek et al. 1999; Fralish et al. 1999; Surrlette et al. 2008). Many of the ground cover plant species that were likely

indicative of open oak woodlands in north Mississippi still persist in the region today but are associated with some moderately disturbed communities with similar environmental characteristics, such as forest edges, thinned stands, and clearings in forests. Extant oak woodlands in adjacent regions (e.g., Kentucky, Arkansas, and Tennessee) are also likely to share a significant number of species with historic oak woodlands in Mississippi, but these would provide less than ideal reference sites because of environmental differences associated with soils and weather, which would invariably contribute to differences in composition. Therefore, rather than choose any one of these as reference sites, we chose to compile a list of native species found in upland forests and associated forest edges and canopy openings in north Mississippi, determine which of these also occurred in oak woodlands and structurally similar habitats in Kentucky (which contains open oak woodlands or "barrens" and for which there is a good up-to-date species list that contains habitat information; i.e., Jones 2005), and then combine these habitats to generate a composite hypothetical oak woodland habitat type for northern Mississippi. Additional details about this procedure are given below.

One possible result of restoration might be to increase plant diversity by favoring declining species that are open woodland indicators. On the other hand, restoring these forests to open woodland may actually lead to a reduction in plant diversity by reducing or eliminating species indicative of closed-canopy forests in floodplains or mesic terraces, which, like open woodlands, have also been degraded or destroyed over a significant portion of the landscape. Also, the disturbances that are associated with restoration efforts, namely controlled burning and thinning, may benefit some invasive non-native plant species such as Japanese stiltgrass (*Microstegium vimineum*; Glasgow 2005) or native widespread disturbance indicators such as Dog fennel (*Eupatorium capillifolium*) and Fireweed (*Erechtites hieraciifolia*). The uncertain outcome of restoration efforts with respect to plant species makes a comprehensive approach to predicting and evaluating responses of species from the regional species pool essential.

Methods

Quantifying Habitat Indication for a Regional Pool of Species

A goal of a current upland restoration project in north Mississippi is to restore species composition indicative of open oak woodlands or savannas. Unfortunately, no open oak woodlands or savannas currently exist in the uplands of north Mississippi today, despite their dominance of the landscape in the early 1800s (Brewer 2001). We therefore generated a composite open woodland habitat type by grouping species we encountered in our surveys that were also associated with the following open habitats identified by Jones (2005) in Kentucky: barrens, mesic open

woodlands, open woodlands, open dry woodlands, savannas, and woodland margins.

To quantify habitat indication for each species, we constructed a binary species by habitat matrix derived from a pool of 187 plant species encountered in surveys of thirty-four 10 × 30-m ground cover vegetation plots in mature upland, oak-dominated forests and associated edges and canopy openings (the habitat types targeted for open woodland restoration) throughout north-central Mississippi (Surrette et al. 2008). Overall, these species were associated with 53 different named habitats, as defined by Jones (2005). A comparable, up-to-date list of plant species, complete with location and habitat information, is not yet available for Mississippi or adjoining states.

An open woodlands similarity score, $S_{\text{open woodlands}}$, was generated for each habitat relative to open woodlands, based on the plant species present in that habitat as follows:

$$S_{\text{open woodlands}} = 1 - \frac{\sum |P_{\text{open woodlands}(i)} - P_{\text{habitat}(x)(i)}|}{\sum P_{\text{open woodlands}(i)} + P_{\text{habitat}(x)(i)}}$$

where P is equal to the presence (1) or absence (0) of plant species i in habitat x . The similarity scores then replaced the presence or absence value in the species by habitat matrix. Those scores were then used to produce an open woodland indicator score ($I_{\text{open woodlands}}$) for each plant species where:

$$I_{\text{open woodlands}} = \frac{\sum S_{\text{open woodlands}(x)(i)}}{N_i}$$

and N_i is the number of habitats where species i occurred. Under this method, those species that occurred in fewer habitats but were found in habitats that were most similar in species composition to open woodlands had the highest open woodland indicator values. Species that occurred in many habitats but were absent from habitats that were similar to the restored habitat had the lowest value.

Two more similarity scores were derived for each habitat using the same method but with the focal habitat being in one case shady mesic forests and in the other disturbed habitats in north Mississippi. The shady mesic forest habitat type was a collection of the following habitats described by Jones in Kentucky: alluvial woods, beech woods, lowland woods, mesic slopes, mesic woods, sandy lowland woods, and wet woods. The shady mesic forest historically was common in floodplains and alluvial terraces in the early 1800s in north Mississippi before extensive agriculture (Brewer 2001; Surrette et al. 2008). Today, it is much less common and thus is a habitat of conservation concern. Hence, even if shade-tolerant forest herbs historically were not common in uplands, closed-canopy forests in the uplands today could serve as important refugia for these species, provided they could tolerate the drier soils in the uplands. On the other hand, floodplain forests appear to have been the ultimate sources of species that have

expanded their distributions into drier upland areas following fire suppression (Brewer 2001; Surrette et al. 2008). Most of these species are quite common and are not of conservation concern (e.g., *Liquidambar styraciflua*). The disturbed habitat type included two habitats described by Jones (2005) as “disturbed areas” or “weedy areas,” which we equated to areas characterized by relatively intense anthropogenic disturbance (e.g., waste areas or areas with tilled or otherwise disturbed soils).

These similarity scores were then used to generate $I_{\text{shady mesic forest}}$ and $I_{\text{disturbance}}$ values for each species. A reduction in species having high $I_{\text{shady mesic forest}}$ scores would indicate a loss of plant species associated with shady mesic forests, some of which may be of conservation concern. Increases in species with high $I_{\text{disturbance}}$ scores during restoration would indicate an undesirable influence by disturbance on the resulting species assemblage.

The Oak Woodland Restoration Project

In 2004, we initiated a long-term open oak woodland restoration project within a mature, fire-suppressed, upland hardwood forest at Strawberry Plains Audubon Preserve in north Mississippi (Brewer 2007). Data derived from a combination of Public Land Survey records of bearing trees and eyewitness accounts corroborate that these upland areas were open woodlands dominated by self-replacing stands of *Quercus velutina*, *Q. stellata*, *Q. marilandica*, and *Q. falcata* and a ground cover containing *Andropogon* species during the early to middle 1800s, before much of the forest was cleared for cotton (*Gossypium* L.) agriculture and before fire suppression (Surrette et al. 2008). In contrast, the mature forests that exist today in these areas have converted to mixed closed-canopy stands of upland oaks (e.g., *Q. stellata*, *Q. falcata*) in the overstory and species that historically were largely restricted to floodplains and alluvial terraces (e.g., *Nyssa sylvatica*, *Acer* spp., *Fraxinus* spp., *Juglans* spp., *Liquidambar styraciflua*, and *Q. alba*) now sharing the overstory and dominating the mid- and understory (Surrette et al. 2008).

The design of the entire restoration project will not be described here. Rather, for the purposes of describing the utility of this method, we will focus our attention on a portion of the experiment, namely a comparison of changes in ground cover plant species composition (from 2005 to 2007) in adjacent treated and untreated 10 × 30-m plots. Both plots were oriented lengthways along a 7% east-facing slope on opposite edges of a 10-m-wide periodically mowed powerline clearing, which bisected a 10-ha mature upland oak-dominated forest. A firebreak was established down the center of the powerline clearing. The plot on the north side of the clearing received the restoration treatments, which involved cutting or girdling stems of all “off-site” tree species (e.g., *N. sylvatica*, *L. styraciflua*, *Ulmus alata*) followed by treating the cuts or girdle wounds with 8% Triclopyr. This method resulted in about 70% topkill

and a reduction in tree canopy density from 86 to 62%. In addition to the thinning treatment, this plot was burned in late September of 2004 and early October of 2006 during the peak wildfire season within the thunderstorm season (Brewer & Rogers 2006). Treated and unmanipulated plots were also established within the forest interior, but these results are not presented here. Efforts to expand the restoration experiment to other sites were initiated in 2007, and burns are scheduled there in 2008.

Ground cover vegetation responses were quantified by counting stems or clumps (in the case of bunch-forming species) of all herb species, along with all woody stems less than 1 m tall within each 10×30 -m plot.

Statistical Analysis of Community Responses to Restoration Treatments

We used simple linear regression to compare community-level changes in the treated plot versus the control plot. Densities in 2005 and 2007 were first log transformed to normalize residuals. For each species i present in at least one of the two plots in either 2005 or 2007, we calculated the difference between the plots in relative changes in abundance over a 2-year period, $D_{r(i)}$, as follows:

$$D_{r(i)} = \ln\left(\frac{N_{t2007} + 1}{N_{t2005} + 1}\right) - \ln\left(\frac{N_{c2007} + 1}{N_{c2005} + 1}\right)$$

where N_t and N_c are the abundances of species i in the treated plot and the control plot, respectively.

To examine how the community as a whole responded to the treatment, we regressed $D_{r(i)}$ against the species indicator scores separately for open woodland indication, shady mesic forest indication, and disturbance indication. Because increases in most non-native species are typically considered to be undesirable from a conservation standpoint, we assumed that non-native species were not indicative of any native habitat of conservation concern (in this case, open woodlands and shady mesic forests). Accordingly, these species were assigned $I_{\text{open woodland}}$ and $I_{\text{shady mesic forest}}$ scores of 0 when regressing $D_{r(i)}$ against these indicator scores. On the other hand, because habitats that have experienced intense anthropogenic disturbances are generally considered undesirable from a conservation standpoint, the calculated $I_{\text{disturbance}}$ score for each non-native species (rather than 0) was used in the regression of $D_{r(i)}$ against $I_{\text{disturbance}}$. Because changes in the abundance of species were positively correlated with initial differences in abundance between the two plots ($r = 0.25$, $p = 0.04$), we included the latter as a predictor variable in all three regression analyses. This multiple regression approach in effect gave equal weight to rare and common species, which enabled us to integrate the responses of numerous indicator species that might have been represented by one or a few plants. All linear regression analyses were done using JMP version 5.0.1.

Results

A comparison of changes in ground cover plant species composition in treated and control plots revealed greater changes in the treated plot than in the control plot. Species richness increased in the treated plot (from 46 to 60) but not in the control plot (42 in both years, with one gained and one lost species). More important, the increase in open woodland indicators was greater in the treated plot than in the control plot, as indicated by a significant positive correlation between D_r and $I_{\text{open woodland}}$ ($r = 0.46$; $p < 0.0001$; Fig. 1A). Much of the response could be attributed to dramatic increases in open woodland indicators such as *Schizachyrium scoparium*, *Chamaecrista fasciculata*, *Helianthus microcephalus*, and *Strophostyles umbellata* (Fig. 1A) and decreases in species that were not indicative of open woodlands, including *Prunus serotina*, *Parthenocissus quinquefolia*, and *Liquidambar styraciflua*. The y-intercept of the regression was not significantly different from 0 (-0.03 ; $p = 0.88$), indicating that the two plots were initially similar in species composition and that most indicator species increased at a greater rate in the treated plot than in the control plot.

Although not statistically significant at the 0.05 level, there appeared to be a negative relationship between D_r and $I_{\text{shady mesic forest}}$ values ($r = -0.23$; $p = 0.058$; Fig. 1B). Close inspection of this relationship, however, revealed that much of the negative relationship was due to dramatic increases by species with very low mesic forest indicator scores in the treated plot (e.g., *S. scoparium*, *Crotalaria sagittalis*, *Chamaecrista fasciculata*). Indeed, some species that were indicative of mesic forests (e.g., *Dichanthelium laxiflorum*, *Carex swanii*) appeared to respond positively to the treatments. The predicted value of D_r for a perfect indicator of shady mesic forest was close to 0 (0.04, well within the 95% confidence interval around 0). Some species that were moderately indicative of shady mesic forests, however, appeared to decline to a greater extent in the treated plot (e.g., *Prunus serotina*, *Liquidambar styraciflua*, and *Parthenocissus quinquefolia*) but so did some non-native species (e.g., *Lonicera japonica*), which, by definition, were assumed not to be indicative of any native habitat and thus assigned $I_{\text{shady mesic forest}}$ scores of 0. When non-native species such as *L. japonica* and *Trifolium dubium* were assigned their calculated $I_{\text{shady mesic forest}}$ scores (0.46 and 0.63, respectively), the negative relationship between D_r and $I_{\text{shady mesic forest}}$ scores was highly significant ($r = -0.33$; $p = 0.005$).

The increase in indicators of intense disturbance was not significantly greater in the treated plot than in the control plot ($r = 0.05$; $p = 0.66$; Fig. 1C). Because residuals were highly skewed, the relationship was also investigated using Spearman rank correlation but still was not statistically significant ($\rho = 0.1$; $p = 0.38$). To be sure, some disturbance indicators appeared to increase in response to the restoration treatments (e.g., *Crotalaria sagittalis* and *Chamaecrista fasciculata* (the latter being highly indicative

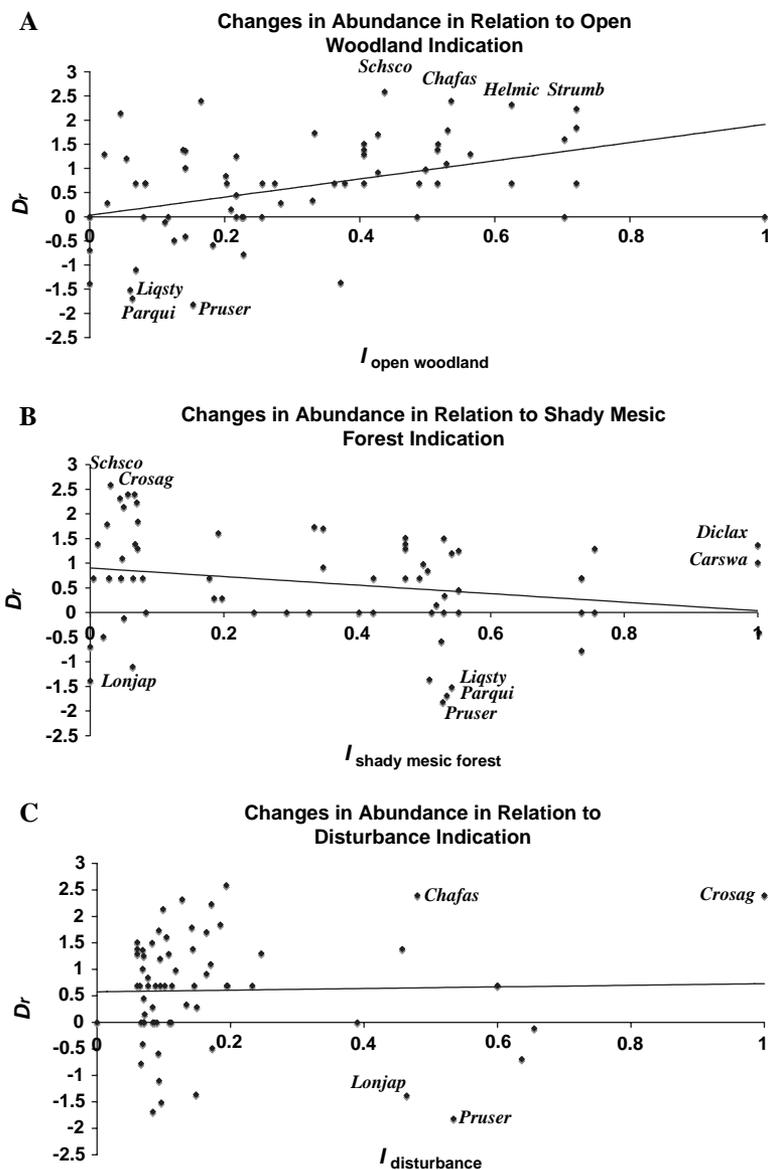


Figure 1. Scatter plots and linear regressions of D_r (i.e., relative increase in density in the treated plot between 2005 and 2007 minus the relative increase in density in the control plot between 2005 and 2007) versus (A) open habitat indication, (B) shady mesic forest indication, and (C) disturbance indication. Abbreviations of selected species: *Carswa*, *Carex swanii*; *Chafas*, *Chamaecrista fasciculata*; *Crosag*, *Crotalaria sagittalis*; *Diclax*, *Dichanthelium laxiflorum*; *Helmic*, *Helianthus microcephalus*; *Liqsty*, *Liquidambar styraciflua*; *Lonjap*, *Lonicera japonica*; *Parqui*, *Parthenocissus quinquefolia*; *Pruser*, *Prunus serotina*; *Schsco*, *Schizachyrium scoparium*; and *Strumb*, *Strophostyles umbellata*.

of open woodlands also), but increases in these species appeared to be countered by decreases in other disturbance indicators, including seedlings of Black cherry (*P. serotina*) and a couple of non-native species, *L. japonica* and *T. dubium*.

Discussion

The method described here can be of significant value to local restoration efforts and to ecologists and conservation biologists in general. It provides a means of evaluating the impact of open woodland restoration on plant species

assemblages in north Mississippi. Restoration of open woodlands in north Mississippi has only just begun in the last 3 years. Therefore, considerable uncertainty exists as to the outcome of restoration projects. A regional conservation perspective dictates that we be concerned not only with effects of local restoration treatments on indicators of open woodlands but also on indicators of other rare habitats of conservation concern. Furthermore, treatments involving disturbances such as fire and thinning require critical evaluation in these times of increased exotic species invasions (Keeley 2006). Although prescribed burning can be used to reduce invasive species in some cases

(MacDonald et al. 2007), it can benefit invasive species in other cases (Hobbs & Huenneke 1992; King & Grace 2000; Keeley 2006). We should be concerned not only with non-native invasive species but also with the responses of any widespread species that could respond positively to disturbances, including natives. Without a quantifiable means of evaluating the impacts of restoration treatments on entire communities, it is impossible to know whether restoration treatments will exacerbate an already growing problem, diminish that problem, or have no net effect.

Fortunately, in this particular case, preliminary evidence suggests that thinning and prescribed burning of edges of a fire-suppressed upland hardwood forest will increase open woodland indicators without greatly decreasing indicators of other habitats of concern (e.g., shady mesic forests) or increasing invasive species. Of the herbaceous plants indicative of shady mesic forests that were present at this site, most either appeared to respond positively to the restoration treatments (e.g., *Dichanthelium laxiflorum*, *Carex swanii*) or showed little change in abundance. The increase in the abundance of open woodland indicators is consistent with the results of similar restoration projects in some oak-dominated ecosystems (e.g., Nielsen et al. 2003). The results differ somewhat from those of a study of ground cover vegetation responses to frequent burning alone in oak forests, which showed increased diversity but not an increase in open woodland indicators relative to mesic forest indicators (Hutchinson et al. 2005). These differences may be related in part to the lack of thinning of the overstory in Hutchinson et al., and in part to the fact that our study was conducted at a forest edge instead of within the forest interior.

Although the method proposed here can be employed without an extant reference site, our approach benefited greatly from some knowledge of the historical reference conditions of the sites to be restored. Partial historical reconstructions of community types and structures, along with an examination of species lists associated with extant habitats in other regions, were crucial to defining a hypothetical reference assemblage. Our decision to implement restoration treatments to benefit open woodland ground cover plant species was based on historical reconstruction of the dominant tree species and canopy structure of early nineteenth-century upland communities in north Mississippi. Without a perspective on historical community types in this region, it probably would have never occurred to us that open woodland habitats such as fire-maintained savannas or barrens could be viable community types in this region because none exist today.

The approach used here provides a means of evaluating changes in species composition within a regional context and thus is relevant to conservation. By providing region-wide, community similarity-based indicator (*I*) scores for every species in a community, this method provides more information about compositional changes relevant to conservation (i.e., changes in habitat specialists) than do simple analyses of species richness or evenness. At the same

time, the analysis goes beyond a focus on legally protected species and can examine responses of common species that are nonetheless indicative of rare and/or declining habitats. The indicator score is somewhat analogous to the coefficient of conservatism (*CC*), whereas the sum of the indicator scores weighted by abundance for an assemblage is analogous to a floristic quality index (Wilhelm & Masters 1995). The method of calculation of community similarity-based indicator scores is more objective, however, and does not require consultation with expert taxonomists. Our approach, by itself, cannot be used to assess improvements in ecosystem “health” or “integrity” (sensu Leopold 1949; see also Winterhalder et al. 2004). If desired, it could be combined with an analysis of changes in ecosystem properties to see if increases in species indicative of uncommon and relatively undisturbed habitats are also associated with desirable ecosystem processes. This would be a departure from the recent focus by ecologists on the relationship between local species richness or evenness and ecosystem functioning (Tilman et al. 1996; Hooper & Vitousek 1997).

In addition to evaluating the outcome of restoration projects, this method could be used to predict in advance how some species might respond to restoration treatments. By using similarity among communities in a region to generate habitat indicator scores for species, this method allows one to predict the likelihood of a species occurring in a particular habitat, even if regional surveys (which typically are not as intensive as site-specific surveys) do not list it as occurring in that habitat. Hence, our approach provides a refinement of regional surveys and could enable one to predict the success of species in particular habitats that do not currently exist, that is, restoration end points. Other approaches designed to predict the success of species in particular habitats, including restored habitats, employ analyses of functional traits (Keddy 2000; Block et al. 2001; McGill et al. 2006). The method described here is meant to complement rather than supplant those means of evaluating a restoration project. Although our method does not require identification of functional traits in advance, there is no question that a more detailed understanding of the responses to restoration treatments will require an examination of functional traits (Clewell & Rieger 1997; McGill et al. 2006).

To wit, the fact that disturbance indicators as a group did not respond positively to burning and thinning in the current study may seem counterintuitive, but close scrutiny of individual species and their associated traits provides a plausible explanation. Some disturbance indicator species responded positively to these treatments, whereas others responded negatively. Those that responded positively (e.g., *Chamaecrista fasciculata*) were also indicative of open woodlands. *C. fasciculata* is an annual forb that produces a persistent, fire- or disturbance-regulated seed bank and is frequently associated with open, fire-maintained habitats (e.g., savannas, barrens; Clewell 1985; Jones 2005). In contrast, those disturbance indicators that

appeared to respond negatively to the treatments (e.g., *Prunus serotina*, *Lonicera japonica*) were bird-dispersed mesophytic trees and vines that were not indicative of open woodlands and historically either were absent or very rare in upland communities in this region (Brewer 2001; Surrlette et al. 2008). Hence, *P. serotina* and *L. japonica*, though frequent invaders of recently cleared forests or forest edges (and thus appropriately regarded as disturbance indicators), nonetheless appeared to respond negatively to fire (or perhaps the combination of fire and thinning). By combining our species-based analysis with an analysis of traits, one could gain a more nuanced understanding of disturbance regimes and disturbance-adaptive traits.

In summary, the method described here uses readily available data to make reasonable predictions about restoration outcomes and can be applied to any group of species in any region, provided there exists a regional species list with habitat and location information. Currently, gathering and organizing habitat and location data will be time consuming for many species, but the trend toward the digitizing of collection records, the posting of collection records online, and the ever-increasing speed of computation and data transfer should give this kind of analysis increasing relevance over time.

Implications for Practice

- This method will allow restoration practitioners to use a regional species list with habitat and location information to generate a hypothetical reference assemblage or assemblages.
- This approach provides an objective and quantitative method for determining floristic (or faunistic) quality of assemblages based on practitioner-generated species indicator scores.
- This approach provides an objective and quantitative method for calculating the fidelity of a species to a particular habitat without resorting to consultation with expert taxonomists.
- This method will allow a practitioner to predict the likelihood of a species occurring in a particular habitat, even if the species is not listed as occurring in that habitat in a regional species database.

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LITERATURE CITED

- Abrams, M. D. 1992. Fire and the development of oak forests. *Bioscience* **42**:346–353.
- Aronson, J., S. Dhillon, and E. LeFloc'h. 1995. On the need to select an ecosystem of reference, however imperfect: a reply to Pickett and Parker. *Restoration Ecology* **3**:1–3.
- Batek, M. J., A. J. Rebertus, W. A. Schroeder, T. L. Haithcoat, E. Compas, and R. P. Guyette. 1999. Reconstruction of early nineteenth-century vegetation and fire regimes in the Missouri Ozarks. *Journal of Biogeography* **26**:397–412.
- Block, W. M., A. B. Franklin, J. P. Ward Jr., J. L. Ganey, and G. C. White. 2001. Design and implementation of monitoring studies to evaluate the success of ecological restoration on wildlife. *Restoration Ecology* **9**:293–303.
- Brewer, J. S. 2001. Current and presettlement tree species composition of some upland forests in northern Mississippi. *Journal of the Torrey Botanical Society* **128**:332–349.
- Brewer, J. S. 2007. Forest edges as a starting point for restoration of fire-dependent oak woodlands. Abstract, Joint Meeting of the Ecological Society of America and Society for Ecological Restoration, San Jose, California, 10 August 2007 (available from <http://eco.confex.com/eco/2007/techprogram/P8863.HTM>).
- Brewer, J. S., and C. H. Rogers. 2006. Relationships between prescribed burning and wildfire occurrence and intensity in pine-hardwood forests in North Mississippi, USA. *International Journal of Wildland Fire* **15**:203–211.
- Clewell, A. F. 1985. Guide to the vascular plants of the Florida panhandle. The Florida State University Press, Tallahassee, Florida.
- Clewell, A. F., and J. P. Rieger. 1997. What practitioners need from restoration ecologists. *Restoration Ecology* **5**:350–354.
- Cody, M. L. 1986. Diversity, rarity, and conservation in Mediterranean-climate regions. Pages 122–152. in M. E. Soulé, editor. *Conservation biology: the science of scarcity and diversity*. Sinauer Associates, Sunderland, Massachusetts.
- Fralish, J. S., S. B. Franklin, and D. D. Close. 1999. Open woodland communities of southern Illinois, Western Kentucky, and Middle Tennessee. Pages 171–188 in R. C. Anderson, J. S. Fralish, and J. M. Baskin, editors. *Savannas, barrens, and rock outcrop plant communities of North America*. Cambridge University Press, Cambridge, United Kingdom.
- Glasgow, L. S. 2005. The effects of prescribed burning on invasibility and understory composition in a southeastern Ohio forest. M.Sc. thesis. Environmental and Plant Biology, Ohio University, Athens.
- Groves, C. R. 2003. *Drafting a conservation blueprint: a practitioner's guide to planning for biodiversity*. Island Press, Washington, D.C.
- Hobbs, R. J., and L. F. Huenneke. 1992. Disturbance, diversity, and invasion: implications for conservation. *Conservation Biology* **6**: 324–337.
- Hooper, D. U., and P. M. Vitousek. 1997. The effects of plant composition and diversity on ecosystem processes. *Science* **277**:1302–1305.
- Hutchinson, T. F., R. E. J. Boerner, S. Sutherland, E. K. Sutherland, M. Ortt, and L. Iverson. 2005. Prescribed fire effects on the herbaceous layer of mixed-oak forests. *Canadian Journal of Forest Research* **35**:877–890.
- Jones, R. L. 2005. *Plant life of Kentucky: an illustrated guide to the vascular flora*. University Press of Kentucky, Lexington, Kentucky.
- Jordan, W. R. III, M. D. Gilpin and J. D. Abner, editors. 1987. *Restoration ecology: a synthetic approach to ecological research*. Cambridge University Press, Cambridge, United Kingdom.

- Keddy, P. A. 2000. Wetland ecology: principles and conservation. Cambridge University Press, Cambridge, United Kingdom.
- Keeley, J. E. 2006. Fire management impacts on invasive plants in the Western United States. *Conservation Biology* **20**:375–384.
- King S. M., and J. B. Grace. 2000. The effects of gap size and disturbance type on invasion of wet pine savanna by cogongrass, *Imperata cylindrica* (Poaceae). *American Journal of Botany* **87**:1279–1286.
- Lawton, J. H., and R. M. May, editors. 1995. Extinction rates. Oxford University Press, Oxford, United Kingdom.
- Leopold, A. 1949. A Sand County Almanac. Oxford University Press, Oxford, United Kingdom.
- MacDonald, N. W., B. T. Scull, and S. R. Abella. 2007. Mid-spring burning reduces spotted knapweed and increases native grasses during a Michigan experimental grassland establishment. *Restoration Ecology* **15**:118–128.
- McGill, B. J., B. J. Enquist, E. Weiher, and M. Westoby. 2006. Rebuilding community ecology from functional traits. *Trends in Ecology and Evolution* **21**:178–185.
- Nielsen, S., C. Kirschbaum, and A. Haney. 2003. Restoration of midwest oak barrens: structural manipulation or process-only? *Conservation Ecology* **7**:10 (available from <http://www.consecol.org/vol7/iss2/art10/>) accessed July 2007.
- Palmer, M. A., R. F. Ambrose, and N. L. Poff. 1997. Ecological theory and community restoration ecology. *Restoration Ecology* **5**:291–300.
- Pickett, S. T. A., and V. T. Parker. 1994. Avoiding the old pitfalls: opportunities in a new discipline. *Restoration Ecology* **2**:75–79.
- Robertson, P. A., and A. L. Heikens. 1994. Fire frequency in oak-hickory forests of southern Illinois. *Castanea* **59**:286–291.
- Spellerburg, I. F. 1992. Evaluation and assessment for conservation: ecological guidelines for determining priorities for nature conservation. Chapman and Hall, London, United Kingdom.
- Surette, S. B., S. M. Aquilani, and J. S. Brewer. 2008. Current and historical composition and size structure of upland forests across a soil gradient in north Mississippi. *Southeastern Naturalist* **7**:27–48.
- Surette, S. B., and J. S. Brewer. 2008. Inferring relationships between native plant diversity and *Lonicera japonica* in upland forests in north Mississippi, USA. *Applied Vegetation Science* **11**:205–214.
- Swetnam, T. W., C. D. Allen, and J. L. Betancourt. 1999. Applied historical ecology: using the past to manage the future. *Ecological Applications* **9**:1189–1206.
- Tilman, D., D. Wedin, and J. Knops. 1996. Productivity and sustainability influenced by biodiversity in grassland ecosystems. *Nature* **379**:718–720.
- White, P. S., and J. L. Walker. 1997. Approximating nature's variation: selecting and using reference information in restoration ecology. *Restoration Ecology* **5**:338–349.
- Wilhelm, G. S., and L. A. Masters. 1995. Floristic quality assessment in the Chicago region and application computer programs. Morton Arboretum, Lisle, Illinois.
- Winterhalder, K., A. F. Clewell, and J. Aronson. 2004. Values and science in ecological restoration—a response to Davis and Slobodkin. *Restoration Ecology* **12**:4–7.