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Source: Natural Areas Journal, 40(3) : 237-244

Published By: Natural Areas Association

URL: https://doi.org/10.3375/043.040.0320

Research Article

Assessing Conservation Value and Restoration Priorities in an Agroecological Landscape

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Associate Editor: Donatella Cogoni

ABSTRACT

The integration of agricultural and natural ecological systems is increasingly viewed as an essential step toward achieving conservation goals, from local to global scales. For this study, we assessed the ecological conditions and conservation value of upland habitats on six Wisconsin potato farms participating in an ecolabel program that requires the implementation of ecological management plans on non-crop lands. Our objective was to determine how natural and restorable-to-natural elements of the landscape related to adjacent intensively managed agricultural fields, in order to prioritize restoration targets and activities. We compared the plant species richness, floristic quality, and vegetation structure between three dominant upland habitats: woodlands, pine plantations, and weedy fields. We recorded 205 native plant species across all sites (approximately 25% of the regional native flora), indicating that habitat patches surrounding agricultural lands can offer substantial conservation value. Woodlands had the highest average conservation value (mean of 43.8 native species per site) and weedy fields the lowest (mean of 6.3 native species per site). Habitat edges were characterized by a higher frequency of both exotic and prairie–savanna indicator species, representing a unique assemblage of species warranting special consideration for conservation and restoration. We recommend that restoration efforts on this and similar agroecological systems prioritize woodland edges and weedy corners, where prescribed fire, native plant seeding, and invasive species removal could produce significant conservation gains while reducing agricultural weed colonization of cultivated fields.

Index terms: agroecology; ecolabelling; edge effects; native plant restoration; weed management

INTRODUCTION

Agricultural development is a major global threat to biodiversity (Green et al. 2005; Emmerson et al. 2016). In the midwestern United States, agricultural fields and feedlots have all but completely replaced the once extensive, diverse grassland and savanna biomes. The loss and degradation of natural ecosystems in agricultural landscapes is only expected to worsen in the near future as global demand for agricultural commodity production continues to increase (Lanz et al. 2018).

By definition, agroecosystems are subject to extreme human modification. In the central Wisconsin agricultural landscape, upland plant communities have diverse land use histories and varying degrees of fragmentation and disturbance, leading to highly variable floristic composition. Fragmentation in these landscapes leads to spillover edge effects stemming from irrigation, fertilizer runoff, pesticide drift, frequent mechanical disturbance, higher light conditions, and the colonization of weedy species (Boutin and Jobin 1998; Gove et al. 2007). The impacts of these processes are likely to manifest in variation in the composition and structure of vegetation between the edges and the interior areas of habitat patches, promoting the invasion of exotic species and the loss of regionally rare species (Ries et al. 2004; Collins et al. 2017). Smaller habitat patches are especially vulnerable to edge effects because of the high edge-to-core ratio (Leach and Givnish 1996; Fahrig 2003). Quantifying edge effects is therefore vital for assessing the conservation potential of an agricultural patchwork landscape.

Despite the many ecological consequences of agricultural development, agricultural lands can offer significant opportunities for biological conservation and restoration (Freemark et al. 2002; Wade et al. 2008; Geertsema et al. 2016), but with several unique challenges of implementation (Fischer et al. 2006; Egli et al. 2018). Rather than viewing agricultural landscapes as conservation "lost causes," there is thus increasing interest in determining which conservation actions can recover the most ecological function without diminishing the economic viability of farming. One prominent approach is that of "ecolabelling" agricultural products. Ecolabels can provide information for consumers that will allow them to recognize commodities produced using more ecologically friendly practices, and increase their willingness to pay a premium to purchase them.

The farms on which this study was conducted have been active participants in such an ecolabel project: the Wisconsin Healthy Grown label (healthygrown.com). Although motivated initially by the need to make adjustments required after a commonly used pesticide was banned, the growers readily and enthusiastically endorsed the idea that specific conservation actions should be a part of the requirements to be certified for the Healthy Grown ecolabel (Zedler et al. 2009). The research reported here was part of a larger project that had as its objective improving our understanding of how best to accommodate biological conservation and restoration efforts into the land use and operations of working farms.

The goals of this study were to (1) assess the ecological condition and conservation value of non-crop lands in an

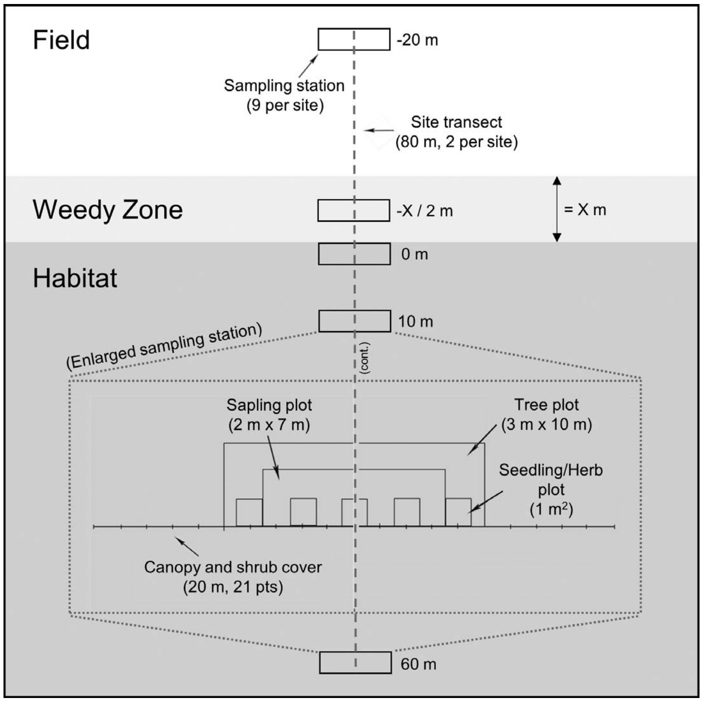

Figure 1.—Transect sampling design, with two 80 m transects per site, each encompassing 9 sampling stations: one each in the field and weedy zone, and one every 10 m (7 total) in the habitat zone. The dotted line indicates an overlaid enlarged view of an individual sampling station and the sub-plots within.

agroecological landscape, including oak woodlands, pine plantations, and weedy fields; (2) describe how plant species composition, vegetation structure, and abiotic conditions change across the boundaries between agricultural fields and adjacent non-crop lands; and (3) determine restoration priorities and best practices for achieving conservation goals.

METHODS

Study Area

Our study area included six potato farms within two sampling nodes in the central sands region of Wisconsin (Figure 1), all of which participate in the Healthy Grown ecolabel program. The region encompasses the eastern edge of glacial Lake Wisconsin, characterized by extensive dry sandy glacial outwash and shallow wet depressions (Martin 1965; Clayton and Knox 2008), with soils classified as sandy or sandy loams. The climate is continental with a mean annual temperature of 7.2 °C and mean annual precipitation of 801 mm. Prior to settlement (ca. 1840), the upland vegetation was a mosaic of oak barrens, pine barrens, dry prairies, and dry oak woodlands, characterized by frequent, extensive fires (Curtis 1959). Post-settlement, this region has undergone dramatic agricultural intensification, with once-frequent fires now virtually absent. One technological shift of importance was the introduction of center-pivot irrigation, which resulted in the creation of weedy "pivot corners" within rectangular fields (Figure 1). At present, the vegetation is composed of a patchwork of cultivated fields, ruderal fields, old fields, pine plantations, oak and pine woodlands, and degraded oak and pine barrens.

Study Sites

We focused on three dominant upland plant communities found in areas adjacent to cultivated fields: woodlands, pine plantations, and weedy fields. The sites in our study represent a range of vegetation states reflecting diverse management histories and encompass gradients in canopy openness and exotic species dominance—a surrogate for the intensity of anthropogenic disturbance.

Woodlands are predominantly oak or pine barrens that have undergone afforestation in the absence of fire. They have relatively dense canopy cover (>70%) dominated by a complex of Hill's oak (*Quercus ellipsoidalis*) and black oak (*Quercus velutina*), with a smaller contribution by spontaneously established white pine (*Pinus strobus*), jack pine (*Pinus banksiana*), white oak (*Quercus alba*), red maple (*Acer rubrum*), and black cherry (*Prunus serotina*).

Pine plantations, as defined for the purposes of this study, are sites in which mature (>10 cm dbh) pines are in cultivation. In this landscape, pure monoculture plantations intergrade with seminatural plantations where pines are planted into a portion of an afforested woodland. Therefore, the percentage of total basal area comprised by planted pine in plantations ranged from approximately 10% to nearly 100%. Pine plantations have relatively dense canopy cover (>70%) dominated by red pine (*Pinus resinosa*) and white pine, with a smaller contribution by jack pine, Hill's oak, black oak, bigtooth aspen (*Populus grandidentata*), and red maple.

Weedy fields are typically situated in the corners and on the edges of cultivated fields. Most cultivated fields are primarily arranged in 0.5 × 0.5 mile squares to accommodate the 0.25 mile radius of the center-pivot irrigation systems. The circular path of the irrigation boom precludes cultivation in the corners and extreme margins of the agricultural fields. These areas are typically open weedy habitats that receive variable management, and range from ruderal communities that are periodically mowed to sites that are planted and managed more intensively for forage crops. Ruderal pivot corners are dominated by exotic grass and forb species. More intensively managed pivot corners are planted with alfalfa (*Medicago sativa*) and forage grasses.

We identified potential sites in each of the three dominant vegetation classes using GIS databases and ground surveys. We selected sites that met criteria for size (>1 ha), conformance to one of our three identified vegetation cover types, adjacency to agricultural fields (all sites had ≥200 m of edge bordering cultivated fields), and distance from other sampled sites of the same community type (>1 km; Table 1).In total, we included 33 sites (15 woodlands, 9 pine plantations, and 9 weedy fields) in our analysis.

Vegetation Sampling

For each of the 33 sites, we established two 60 m site transects, oriented perpendicularly to the edge of the potato field–habitat boundary, and separated from each other by 50–75 m. We defined the potato field–habitat boundary as being 1 m inside the edge of the canopy (for woodlands and pine plantations), or

Table 1.—Summary of average species richness, conservation value, vegetation structure, site area, and landscape context by plant community type. Table reports the mean and standard deviation for each variable, with overall significance level for a one-way ANOVA indicated next to the variable name, and significance levels for pairwise comparisons between wooded sites and the other community types indicated next to values. FQI represents floristic quality index, and BA represents basal area.

	Wooded (n = 15)	**Pine (n = 9)**	**Weedy (n = 9)**
Richness ***	61.7 ± 11.2	49.9 ± 17.9 *	23.4 ± 6.1 ***
Native richness ***	43.8 ± 8.1	32.67 ± 14.5 *	6.33 ± 3.3 ***
% native spp ***	71.18 ± 7.2	63.93 ± 7.75 *	26.09 ± 10.2 ***
Savanna richness ***	13.73 ± 3.03	8.78 ± 6.96 **	1.33 ± 1.7 ***
% savanna ***	22.99 ± 7.08	16.28 ± 6.29 *	5.1 ± 6.0 ***
FQI ***	2.77 ± 0.62	1.95 ± 0.68 **	0.08 ± 0.09 ***
Total BA (cm^2) *	9606 ± 4353	19,238 ± 15,197 *	0
Tree BA (cm^2) *	9132 ± 4542	18,468 ± 15,110*	0
Sapling BA (cm^2)	474 ± 413	771 ± 624	0
% BA saplings	7.9 ± 11.2	5.9 ± 7.6	0
Saplings < 2.5 cm	87.13 ± 54.9	67.11 ± 77.7	0
Seedling count *	254.4 ± 158.7	108.9 ± 101.8 *	0
Canopy cover (%)	72 ± 11	68 ± 6	0
Shrub cover (%) **	52 ± 20	21 ± 17 **	0
Area (ha) **	40.1 ± 52.1	38.0 ± 47.4	2.31 ± 0.89 **
Natural landscape % *	34.7 ± 16.4	49.3 ± 15.4 *	30.6 ± 11.8

Significance codes: *** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$.

1 m from the edge of the disturbed annual vegetation (for weedy fields). At each site, we established sampling stations every 10 m along each of the two transect lines, with 7 stations per transect. Our sampling effort thus comprised 33 sites, with 66 transects encompassing a total of 462 sampling stations.

Each sampling station (Figure 1) contained nested plots oriented parallel to the main habitat boundary, and consisted of the following: five seedling/herb plots (1 m^2, each separated by 1 m) in which we recorded the number of tree seedlings (<1 m tall) and estimated the percentage of herbaceous cover (non-tree plants <1.5 m tall); one sapling plot (2 × 7 m) in which we recorded the number and diameter at breast height of all medium-sized woody stems (dbh >2.5 cm and <10 cm); and one adult tree plot (3 × 10 m) in which we recorded the number and dbh of all adult trees (>10 cm dbh). Additionally, we recorded canopy and shrub cover every 1 m along a centered 20 m sampling transect (Figure 1) using point estimated presence–absence of cover by species. We sampled all stations twice during the growing season to capture the spring and late-summer flora, and kept the larger of the two measurements for each plot metric.

At each sampling station, we took hemispherical photographs at 1.5 m above the first, third, and fifth 1 m^2 quadrant using a CI-110 Plant Canopy Digital Imager with a 150° lens. We analyzed the photographs using the CI-110 Plant Canopy Analysis Software, Version 3.1.0.0, to calculate leaf area index (LAI) and transmission coefficient for diffuse penetration (TCDP).

Statistical Analyses

We compared the three community types for their average species richness, Floristic Quality Index (FQI, see below), vegetation structure, site area, and landscape context. We characterized all species as native or exotic (using Wetter 2001), and determined their status as indicators of prairie or savanna flora based on community dominance (Curtis 1959; Cochrane and Iltis 2000). We excluded all taxa not identified to the species level in our analyses.

We calculated FQI (Swink and Wilhelm 1979) as the average species coefficient of conservatism (the degree to which a species is restricted to a narrow range of habitats relative to other species in the region; Bernthal 2003) per species occurrence in seedling/herbaceous cover plots:

$$\mathrm{FQI}_j = \frac{\sum_{i=1}^{n} \mathrm{A}_{ij}\mathrm{W}_i}{\sum_{i=1}^{n} \mathrm{A}_{ij}}$$

where FQI_j is the floristic quality index for site j, A_{ij} is the abundance of species i at site j (measured as the sum of occurrences in all plots), and W_i is the coefficient of conservatism for species i. Scores range from 0 to 10, with a 0 representing generalist species that occur in a broad range of communities and a score of 10 representing species that are typically rare with narrow ecological tolerances.

We used single-factor ANOVAs to test for differences between community types, and independent two-sample t-tests to compare average values between wooded sites and the other two community types. When necessary, we log-transformed the data to meet the assumptions of normality and equal variance. To compare average species accumulation by site between community types, we computed sample-based rarefaction curves using EstimateS 8.0 (Colwell 2005). We calculated the pooled number of species per given sample size and accompanying 95% confidence intervals following Colwell et al. (2004). We compared the overall expected species accumulation curves produced from selecting sites randomly to an optimum curve produced by adding sites in order of the highest possible pooled species richness for a given sample size, in order to determine the minimum number of sites needed to capture all sampled species and to evaluate the complementarity of species assemblages between community types. To assess the contribution of site margins for harboring a unique subset of species, we compared sample-based rarefaction with and without the inclusion data from plots sampled at the site edges.

We used blocked multi-response permutation procedures (MRBP; McCune and Grace 2002) to determine whether species composition changed significantly between sampling stations at different positions along the crop-to-interior gradient for the afforested woodlands. The response variable in this analysis was the average total species cover for sampling stations at each position along the gradient. Site identity was treated as a blocking variable. The overall significance level of the analysis indicated whether species composition was grouped by sampling position. We made pairwise comparisons between sampling positions at different distances from edge moving from field to border, border to edge, edge to 10 m, from 10 m to 20 m, and so on. The approximate width of the field–habitat ecotone was defined as the zone in which pairs of neighboring sampling positions significantly explained species composition ($p < 0.05$). The collection of plots beyond the ecotone, where seedling/herbaceous composition was no longer explained by sampling position ($p > 0.05$) was defined as the habitat interior. We determined significance ($p < 0.05$) using Monte Carlo randomization procedures with 999 runs.

Based on the MRBP analysis, we divided the wooded sites into four "transition zones" where the species composition was distinct from other sampling positions. We used one-way ANOVAs and independent *t*-tests to compare responses by species groups, floristic quality (FQI), and woody vegetation structure between four transition zones. We used generalized linear models to explore relationships between variables within sites, with sampling position as a fixed effect nested within site identity as a random effect. For all analyses, the sample unit was the summed response of a variable in a sampling station. Therefore, average richness for a wooded edge is the average number of species found in a sampling station in that zone. We used indicator species analysis (PC-ORD 5) to determine the number of prairie–savanna indicator species, other native species, and exotic species that reached their maximum indicator value (IV) at each sampling position. We focused on prairie–savanna indicator species because they are often the focus for natural areas preservation in the region, and they signify conservation potential and the degree to which restoration activities can be successful. We used chi-square tests to determine whether the association between indicator species frequency and sampling position was nonrandom for each species group.

We performed all univariate statistical analyses using R 2.6 (R Core Team 2007), and all multivariate statistical analyses using PC-ORD 5. To estimate the percentage of the landscape surrounding each site (1.5 km radius) classified as natural vegetation, we used the Hawth's Tools Thematic Raster Summary in ArcMap 9.2 to analyze the 2001 "WISCLAND" land use GIS layer.

RESULTS

Site-Level Analysis of Three Community Types

Across all sampling stations at all sites, we recorded a total of 205 native species, representing 25% of the approximately 800 native upland plant species in the region. Of the 205 native species recorded, 75 (37%) were classified as indicator prairie–savanna species.

Of the three upland community types, afforested woodlands had the highest conservation value as measured by all indicators including overall, native, and prairie–savanna species richness, relative dominance by these species groups, and average floristic quality index (FQI). Pine plantations had intermediate values and weedy fields had the lowest values for all variables (Table 1).

Estimated species accumulation curves for native species were markedly different between community types (Figure 2a). For a sample of nine sites, the afforested woodlands contained a total of 145 species, the pine plantations 118 species, and the weedy corners 24 species. A majority of native species encountered can be captured in a fraction of the 33 sites sampled (Figure 2b). Over 50% of all species are captured in 3 of 33 (9%) of sites. Over 75% of all species are captured in 6 of the 33 (18%) sites and all 205 native species can be captured in 24 of the 33 (73%) sites. Although wooded sites appear to make up a large proportion of the high-priority sites in the optimum accumulation curve, the result of a Wilcoxon signed rank test comparing median site FQI ranks between wooded and pine sites provided only weak evidence supporting this observation. Weedy sites, however, had a significantly lower median priority (larger rank) as compared to either wooded ($p < 0.001$) or pine sites ($p < 0.01$). Therefore weedy corners contribute few species not already found in the other upland community types.

Pine plantations and afforested sites did not differ significantly in most measures of woody plant dominance. On average pine plantations had significantly greater basal area and fewer seedlings than woodlands ($p < 0.05$). Average canopy cover was similar between the two community types and was negatively correlated with site species richness. Afforested woodlands had significantly greater shrub cover than pine plantations ($p < 0.01$). By definition, weedy sites had minimal woody vegetation and so are not included in these comparisons. Both native species and prairie–savanna species richness and dominance were negatively correlated with the percent dominance of *P. resinosa* by total tree basal area. When plantations with greater than 90% dominance by planted pine are excluded from the analysis, the remaining four sites had an average conservation value similar to the woodlands (mean richness = 61, 70% native, 21% prairie–savanna, mean FQI = 2.46)

Edge Analysis of Woodlands

Sampling stations in the interior of woodland sites captured 149 of 177 (84%) of the native woodland species sampled (Figure 2c), with woodland margins containing a substantial reservoir of species not found in site interiors. Measures of species richness and floristic quality were significantly different between the four vegetation zones in the transition between cultivated fields and wooded sites (field, border, edge, and interior; Table 2). Overall species richness, native species richness, and savanna species richness were highest at the edge compared to the other vegetation zones. Native species richness was negatively associated with the abundance of *Carex pensylvanica* ($p < 0.05$)

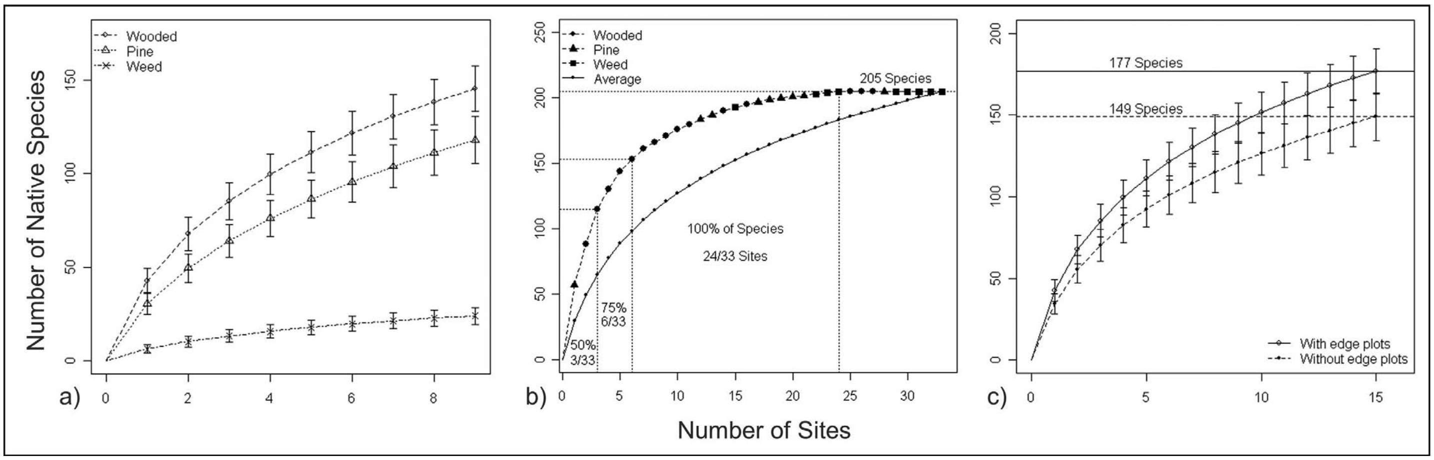

Figure 2.—(a) Estimated species accumulation curves for the three community types. (b) Comparison of optimum vs. estimated average accumulation curves for native species. (c) Average estimated accumulation curves for native species with and without the inclusion of sampling stations located in the habitat edge or in the border between the edge and cultivated field. Error bars represent 95% confidence intervals.

The average FQI and the percentage of total species richness represented by native species increased steadily along the gradient from the field to the habitat interior (Figure 3). The percentage of total species richness accounted for by prairie–savanna species showed no notable trend along this gradient. However, the average frequency of prairie–savanna species was significantly higher at the edge than either the border or interior ($p < 0.001$). The average frequency of exotic species declined steadily from the border to interior, while the average frequency of native non-prairie-savanna species increased between the border and edge, remaining constant between the edge and interior. A disproportionately large number of exotic species and prairie–savanna indicator species achieved their highest indicator value in sampling stations located at the edge and border.

DISCUSSION

The Conservation Value of Non-crop Lands

The non-crop lands sampled in this agroecological landscape contain a substantial proportion of the regional native plant diversity, representing 25% of the approximately 800 species in the region. Given that our analysis excluded all taxa not identified to the species level, the actual proportion of native flora contained on non-crop lands could be significantly higher. Of the 205 native species recorded, 75 (37%) were classified as indicator prairie–savanna species. As demonstrated elsewhere (e.g., Weibull et al. 2003; Boutin et al. 2008; Geertsema et al. 2016), dominant upland habitats surrounding agricultural land offer significant potential for conservation and ecological restoration.

The conservation value of sites differed significantly based upon management history. Weedy pivot corners—characterized by occasional mowing, frequent mechanical disturbance, and variable cultivation—offered little conservation value, with native species richness ranging from 2 to 12 species. Weedy sites contributed only 6 of the 205 native plant species, with most native species also found in woodlands and pine plantations (Figure 2a). Although the lack of any woody canopy in these fields approximates the open conditions found through the landscape pre-settlement, the legacy of plowing and other disturbances in the past has favored the dominance of mostly exotic weedy species and only a few prairie and savanna species (Table 1).

Table 2.—Summary of species richness, floristic quality, canopy cover, shrub cover, and light condition between the four vegetation zones in the transition between cultivated fields and wooded sites (field, border, edge, and interior). FQI represents floristic quality index, LAI represents leaf area index, and TCDP represents transmission coefficient for diffuse penetration. The overall significance level for a one-way ANOVA is indicated next to the variable name, with significance levels for pairwise comparisons between the interior and other vegetation zones indicated next to associated values.

Variable	Field	Border	Edge	Interior
Richness ***	1.4 ± 0.74 ***	14.0 ± 3.8	19.1 ± 6.0 ***	13.5 ± 5.4
Native richness ***	0.07 ± 0.27 ***	6.8 ± 2.64 ***	12.9 ± 4.1 *	11.29 ± 4.44
% native spp ***	3.57 ± 13.4 ***	49.8 ± 19.0 ***	68.84 ± 12.41 ***	84.9 ± 12.2
Savanna richness ***	0	2.37 ± 1.69	4.27 ± 2.13 ***	2.74 ± 1.65
% savanna spp	0	18.2 ± 15.51	23.6 ± 13.1	20.8 ± 12.1
FQI ***	0	1.16 ± 0.89 ***	2.42 ± 0.83 ***	3.29 ± 0.84
Canopy cover (%) ***	0	6 ± 17 ***	81 ± 18	87 ± 17
Shrub cover (%) ***	0	5 ± 17 ***	73 ± 25 *	60 ± 33
LAI ***	NA	0.59 ± 0.34 ***	1.07 ± 0.34	1.23 ± 0.43
TCDP ***	NA	0.65 ± 0.16 ***	0.42 ± 0.13	0.44 ± 0.12

Significance codes: *** $p < 0.001$, ** $p < 0.01$, * $p < 0.05$.

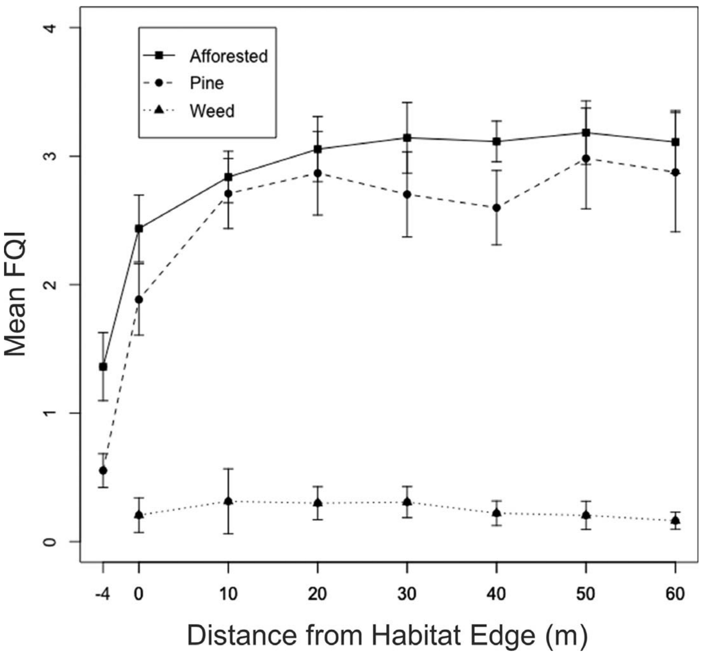

Figure 3.—Mean Floristic Quality Index (FQI) of habitat types with distance from the habitat edge (0 m). Error bars represent standard error.

Sites classified as pine plantations represented a substantial reservoir of regional biodiversity, but with large variability in the condition and quality of individual sites, driven by differences in establishment history and the type and frequency of management. Sites intensively managed for timber production contributed few unique species to the regional upland flora. However, mixed woods of planted pines and naturally established trees contained a significant reservoir of native plant species similar in composition to woodlands. This variation in conservation value can be partly attributed to differences in the intensity of red pine cultivation. Excluding plantations with greater than 90% dominance by planted pine, the remaining four sites had an average conservation value similar to the woodlands (mean richness = 61, 70% native, 21% prairie–savanna, mean FQI = 2.46). Based upon the optimum combination of sites, pine plantations contributed a significant number of unique native species, with eight sites contributing 71 species, or 35% of the all native species sampled (Figure 2b). Our data suggest that pine cultivation is detrimental to the conservation of native plant species in this landscape, although less intensively cultivated plantations can serve as important reservoirs for many native species. Whether such reservoirs represent viable populations of these species may depend on their size and proximity to other suitable habitat.

Woodlands, on average, were the most intact natural communities and offered the highest conservation value. Historically, some of these areas have been partially cleared and grazed, although current landowners primarily manage woodlands as woodlots and hunting grounds. Native species richness was relatively high, but was negatively associated with the abundance of *Carex pensylvanica*, a species that is often associated with a history of severe grazing (Leach and Givnish 1999). In this landscape, it is likely that management history is the overriding factor determining the ecological conditions and species composition of upland communities. As site species richness was negatively correlated with average canopy cover, the greatest conservation gains will likely be achieved by utilizing restoration treatments that move sites toward more open canopy conditions.

Edge Effects between Cultivated Fields and Woodlands

Woodland edges were characterized by relatively high overall species richness, higher relative dominance by exotic species, and denser woody vegetation (Table 2). Unsurprisingly, floristic quality increased with increasing distance from the habitat edge (Figure 3). Although the disturbance along edges favored exotic species dominance, there was also a higher frequency of desirable prairie–savanna species along this ecotone. As an ecological transition zone where distinct plant assemblages intergrade, forest–field ecotones have long been recognized as regions of relatively high species richness (e.g., Ranney et al. 1981), but with high rates of colonization by invasive species that threaten native species' persistence (Ries et al. 2004; Collins et al. 2017). The field–woodland transition zones in this agroecological landscape harbor a unique assemblage of species warranting special consideration for conservation and restoration.

Implications for Conservation and Restoration near Agricultural Lands

This agroecological landscape harbors a large reservoir of native plant species with substantial conservation value, but it is characterized by fragmentation, frequent disturbance, variable management, and accompanying pressure from exotic species. Without informed management to achieve long-term conservation and restoration objectives, this landscape may face the threats of local extinction of native species and the homogenization of plant communities (Fahrig 2003).

Woodlands contain the highest average conservation value for ground layer species and woody vegetation, and thus should be the primary target for most management activities. In the transition zone between woodland patches and cultivated fields, three distinct vegetation zones deserve special consideration for adaptive management: field border, woodland edge, and woodland interior (Table 1, Appendix 1). Given the dominance by agricultural and ecological weeds in the field borders, and the higher frequency of prairie–savanna species in the adjacent edge, we recommend prioritizing field borders adjacent to woodlands for restoration plantings and invasive species removal. Even relatively narrow strips of restored land bordering agricultural fields could retain substantial proportions of native vegetation, with FQI values tapering off just 10–20 m in from the habitat edge in our study (Figure 3).

With appropriate management, growers can reduce the agricultural weed pressure in their fields, decrease the ecological weed pressure on adjacent woodlands, and increase the dominance of desirable native species. Based on restoration treatments in similar communities (e.g., Nielsen et al. 2003), canopy thinning and prescribed burning are likely to promote these species.

Land management recommendations must balance practical and economic considerations for growers with biodiversity losses, which can differ depending on the particular context and developmental trends (Beckmann et al. 2019). Prairie planting and weed control in field borders may not be feasible if they interfere with agricultural production. Because weedy corners in their present state have little conservation value, planting these areas as prairie restorations would produce the largest net gains in species conservation among the communities sampled. The costs of establishing prairie plantings are high, but maintenance costs are low given ease of access and mowing.

A final consideration is the relevance of our findings for the support of the conservation component of the Healthy Grown certification and other ecolabelling programs. The advantage of the long-term commitment required of a viable certified ecolabel is that well-planned and consistent conservation actions can cumulatively lead to major improvements in the overall ecological health of a farm landscape. For example, the restoration of "pivot corners" to prairie can occur over time, with a small number being restored in any one year.

To implement ecological management plans that will maximize desirable conservation outcomes on participating farms, it is essential to (1) start with a clear picture of how regional biodiversity is distributed across non-crop lands on participating farms, (2) understand how the conservation value of sites is correlated with dominant environmental variables that can be significantly altered by management activities, and (3) understand how in-field production practices are affecting adjacent non-crop lands on a local scale through edge effects. We demonstrate here that there can be sufficient value in both economic and ecological terms to warrant targeted restoration activities, with the goal of maintaining maximal biodiversity reserves in agricultural landscapes. In order to maximize high-value natural lands, management plans should include elements of education, agricultural technician training, and research to establish the ecological and social contributions to agricultural enterprises (Rey Benayas et al. 2019). Ultimately, conservation goals will best be achieved through a combination of informed agricultural management and selected restoration interventions.

ACKNOWLEDGMENTS

We thank the Wisconsin Potato and Vegetable Growers Association (WPVGA) and the Healthy Grown project, as well as Wisconsin BioIPM department staff for their support of the research. We also thank Jason Croft and Matt Ziehr for their assistance in the field, Ted Cochrane for assistance in identifying plant collections, and Sara Hotchkiss, Bret Larget, and Tom Givnish for advice and feedback on the research, and the USDA National Research Initiative (USDA/CSREES Grant # 2005-55101-16238) and Davis Fund for their financial support. Lastly, we thank Eric Menges and one anonymous reviewer for their insightful comments.

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Appendix 1.—List of leading indicator species for the three vegetation zones along the transition from field to woodland. Growth forms follow Gleason and Cronquist (1991). Bold type indicates prairie–savanna indicator species status. Asterisks indicate exotic species. Plants recognized as agricultural weeds are denoted with "**a**" and plants that are recognized as ecological weeds are denoted by "**e**". Coefficients of conservatism (CofC) are listed for native taxa.

Taxa	I.V.	*p*	Growth form	CofC
Indicator species for the field border				
Ambrosia artemisiifolia **a**	76	0.0002	Annual	0
Digitaria sanguinalis *****a**	66.7	0.0002	Annual grass	
Elytrigia repens *****e**	75	0.0004	Perennial grass	
Lepidium densiflorum Schrad. *	38.1	0.0062	Annual	
Berteroa incana (L.) DC.***e**	37.6	0.02	Annual–perennial	
Portulaca oleracea L. ***a**	26.7	0.025	Annual	
Asclepias syriaca L.	37.8	0.082	Perennial	1
Setaria pumila ***a**	19.9	0.093	Annual grass	
Cyperus lupulinus	20	0.095	Perennial	3
*Vicia villosa****e**	20.1	0.11	Annual–biennial	
Indicator species for the woodland edge				
Solidago missouriensis	20	0.0958	Perennial	7
Quercus ellipsoidalis	44.5	0.1002	Tree	5
Celastrus orbiculata ***e**	16.2	0.1856	Vine	
Hieracium aurantiacum *	11.7	0.2793	Perennial	
Viola sororia	19.2	0.3043	Annual–perennial	3
Pteridium aquilinum	18.6	0.3097	Perennial	2
Spiraea alba	19.5	0.3101	Shrub	4
Houstonia longifolia	13.3	0.3153	Perennial	6
Comandra umbellata	12.2	0.3171	Perennial	6
Helianthus strumosus	14.1	0.3221	Perennial	4
Indicator species for the woodland interior				
Acer rubrum	58.9	0.0004	Tree	3
Toxicodendron radicans	40	0.0016	Vine	4
Pinus strobus	41.2	0.0024	Tree	5
Quercus alba L.	47.8	0.0028	Tree	7
Sambucus canadensis	37.6	0.0052	Shrub	3
Ilex verticillata	33.3	0.0068	Shrub	7
Vaccinium angustifolium	38	0.007	Shrub	4
Prunus serotina	52.9	0.0072	Tree	3
Galium triflorum	44	0.0082	Perennial	5
Corylus americana	46.1	0.0084	Shrub	5