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ASSESSING QUALITY OF A REGENERATED PRAIRIE USING FLORAL AND FAUNAL INDICES

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ABSTRACT

In today's highly fragmented landscape, prairie communities are becoming rarer as they give way to woody species encroachment, land development and agriculture. In response to this loss, many prairie restorations have taken place implementing seeding, planting and a number of management techniques. However, there is a possibility that natural successional processes may result in quality prairie communities. The goal of the present study was to determine the quality of a late successional prairie located in Macoupin County by its resemblance to several restored and remnant prairies. It was hypothesized that the site would have low floristic quality since no restoration or management practices have been routinely implemented. Vegetation cover data were collected in early and late summer of 2011 and 2012. Species richness (*S*), antilog of Shannon diversity (N_1), average coefficient of conservatism (*mean C*) and Floristic Quality Index (*FQI*) were calculated for each sampling season. These were also calculated for a comprehensive species list from the study site and four sites of known quality for comparison (N_1 was not included in these calculations due to a lack of abundance data). Bray-Curtis dissimilarities between sites were also calculated based on the species composition of these sites. Auchenorrhyncha (Insects: Hemiptera) were collected on four 40 m transects within the site. Auchenorrhyncha Quality Index (*AQI*), *S*, N_1 and *mean C* were calculated based on count data from these transects and others located in eight comparison sites. Indicator species were also found for the study site and each comparison site. Analysis of the abundance data showed that from 2011 to 2012 there was a decline in all four diversity index scores, likely due to drought. Comparisons between the study site and other high quality restorations and remnants revealed that the site does indeed have high floristic integrity, contrary to the hypothesis, though it was not very similar to any of the other sites in species composition. Results from the Auchenorrhyncha survey were consistent with those of the vegetation survey. When compared with those of other high quality sites, diversity index scores from the study site were all very similar and in some cases higher. This, again, was contrary to the hypothesis. Indicator species analysis revealed that the study site supported lower quality species than comparison sites.

It is estimated that prior to European settlement and the establishment of agricultural practices prairies, plant communities comprised mainly of grasses, forbs and some shrubs, covered more than 68 million hectares within the Great Plains (Robertson et al. 1997). Part of this area extended eastward covering parts of Iowa, Missouri, Indiana and Illinois to form the Prairie Peninsula. Due to a combination of geography, topography, glacial history, climate, and fire frequency, Illinois was dominated by this prairie while in other parts of the peninsula it existed as a mosaic of prairie and forest (King 1981; Brugam & Patterson 1996). In today's highly fragmented landscape, prairie communities are becoming rarer as they give way to woody encroachment, land development and agriculture. Illinois now retains less than 0.01% of its original prairie habitat and many of these remnants have been reduced to small, widely spaced inclusions surrounded by woodland (Brand & Dunn 1998; McClain et al. 2002). This loss has led to attempts at prairie restorations, all of which have the similar goal of reestablishing native plant species via plantings or seeding and different management techniques such as controlled burning (Camill et al. 2004). A number of these reconstructions have been successful, however, it has been demonstrated that species diversity in these sites can decline over a period of time, even with continued management (Kindscher & Tieszen 1998). In contrast, other studies have shown that there is a possibility for natural successional processes to result in quality prairie communities (Edgin & Ebinger 2000). The present study was an attempt to determine the resemblance of an old, wet-mesic field in late stages of succession to a prairie remnant using floral and faunal quality indices.

History of restoration

Although preservation of natural areas began as early as 1919, restoration of natural areas did not become a concern until the 1930s when ecologists realized that the majority of tallgrass prairie once occurring in the prairie peninsula had been converted to agricultural fields (Smith 1998). This realization spurred plantings throughout the Midwest, beginning with Curtis Prairie in Wisconsin, which is the world's oldest prairie restoration (Jordan 1983). Since the 1930s many restorations have taken place, most with the final goal of replicating the area's historic ecosystem (Allison 2002). This goal is a difficult one to reach as many sites do not have a history complete with quantitative data on species that were present (Westman 1991). It has also been criticized for being one-dimensional as restoration ecologists tend to focus solely on vegetation, ignoring faunal ecosystem elements (Betz 1986). In recent years, a number of studies have been conducted to determine how restoration practices affect prairie fauna including insects (Brand & Dunn 1998; Panzer 2002), birds (Herkert 1994), and small mammals (Diffendorfer et al. 1995). These are all steps in the right direction, but there is still much work to be done in understanding how to make restorations successful.

Success of a restoration is dependent upon the goals set by the manager prior to restoration taking place. As stated before, the most common goal for restorations is to return a site to its historic state, which is difficult due to lack of information (Westman 1991; Allison 2002). As a result, restoration managers turn to remnant prairies as guides and try to replicate them as closely as possible (Schramm 1990). This still is a hard goal to achieve since no two remnants are exactly alike and no ecosystem is capable of remaining void of change over time (Howell and Jordan 1991). The Society of Ecological Restoration (2004) outlined a number of criteria to be met for successful restorations, including species composition similar to that of a reference system, existence of necessary functional groups, and resilience in the face of normal stressors. Some of these criteria are difficult to measure and not all factors of ecosystem function can be assessed within temporal and monetary constraints, but these should be considered when outlining the goals of a restoration (Society of Ecological Restoration 2004).

In recent years, ecologists and restoration managers have attempted to quantify restoration success using a number of measurements. Some suggest using measurements such as proportion of native species, net primary productivity, and plant and animal diversity as indicators of success (Martin et al. 2005). More simple measurements, such as species richness, are used for determining quality, though they are commonly accused of not being truly sensitive to habitat integrity (Bowles & Jones 2006; Taft et al. 2006). Most recently, the Floristic Quality Index, a weighted index that is said to measure biological integrity, has become one of the most widely used measurements of both restoration success and remnant quality, with multiple variations for different regions (Taft et al. 1997; Bowles & Jones 2006).

While plants are considered by many to be among the best biological indicators of habitat integrity (Panzer & Schwartz 2000), some argue that focusing solely on plants can lead to incomplete goals and management strategies, especially within restorations (Pressy et al. 1993; Bomar 2001). Such a narrow scope can have adverse effects on other wildlife populations that take refuge in managed areas, as has been demonstrated in taxa such as birds (Swengel & Swengel 2001), insects (Panzer & Schwartz 2000) and some species of mammals (Erwin & Stasiak 1979). In response to this, some are turning to animal based assessments to determine habitat quality. Schorr et al. (2007) investigated using survival of rodents as a determinant of xeric habitat quality, though numerous variables made it difficult to produce robust models. Karr (1981) developed an Index of Biological Integrity (IBI) using fish metrics to assess quality of streams in the Midwest, which was found to be sensitive to differences in habitat degradation in a variety of situations (Karr et al. 1986; Leonard & Orth 1986; Karr et al. 1987). In Australia, a similar index was also developed for ants (Majer and Beeston 1986) which are considered to have many characteristics that are useful for monitoring development of restorations (Andersen 1990).

Arthropods have shown themselves to be very useful in the assessment of habitat quality (Kremen et al. 1993). Characteristics including large population numbers, diversity of occupied niches, variety of ecological roles, and quick developmental rates make them particularly good candidates for this use (Longcore 2003). Until recently, difficulties with identification and limited documentation of life histories have prevented widespread use of arthropods as indicators of ecosystem integrity (Andersen & Majer 2004; Dietrich 2009). Wallner (2010) developed an index similar to *FQI* using Auchenorrhyncha, or hoppers (Insects: Hemiptera [DeLong 1948]) called the Auchenorrhyncha Quality Index (*AQI*). This group displays the useful characteristics of most arthropods, with the added attraction of being well documented taxonomically (DeLong 1948). *AQI* uses the same principle as *FQI* of weighting species according to their dependence on natural habitat. Each species is designated a coefficient of conservatism that can range anywhere from zero to 18, and is based on a number of measureable, physical characteristics and life history traits. Although some may subject it to the same criticisms as *FQI*, *AQI* has been statistically validated and in time may prove to be a very useful tool in grassland management (Wallner 2010).

Restoration through succession?

Though major loss of prairie habitat has led to attempts at reconstructions via plantings and invasive management techniques, there is a possibility that natural successional processes may also result in quality prairie communities (Edgin & Ebinger 2000). This means that if a piece of land is left undisturbed, it has some potential to regenerate to a more natural state. This strategy has been observed in a number of different habitats. Galatowitsch and Van der Valk (1996) found that in prairie wetlands succession did not lead to plant communities similar to those of remnant wetlands, though there was evidence that, given time, the similarity could increase. Hodacova and Prach (2003) surveyed spoil heaps left after mining, and found that spontaneous succession favored higher species richness and resulted in dense herb layers that were better suited to prevent erosion than those produced by active restoration. Tropek et al. (2010) studied abandoned limestone quarries and found that the extreme heterogeneity in successional sites provided habitats for many endangered species of vascular plants and arthropods. On the other hand, technical reclamation established sites dominated by fast growing, ruderal species. These are only a few examples and much more research is required to determine whether succession can regularly give rise to more natural landscapes. It is well established that successional processes will occur and must be incorporated into restoration practices for them to be successful (Harker et al. 1997), but it remains to be seen if it can stand alone.

If areas can be left to themselves to regenerate, this may have major implications for future restoration efforts. If there are no concerns that require technical reclamation techniques, succession is a less invasive, cost effective, simple approach for restoration of natural areas (Tropek et al. 2010). These benefits could be used in persuading land owners to allow fallow fields to be incorporated into the natural landscape, rather than developing or leasing them to farmers. If monetary incentives could be linked with these benefits, it is possible that natural area reclamation would not be such a rare and difficult occurrence.

The objective of this study was to determine similarity of an old field in later stages of succession to remnant and restored prairies. Comparisons were made to a number of sites using the following biodiversity variables: species richness (*S*), the antilog of Shannon's diversity index (*N1*), average coefficient of conservatism (*mean C*), the Floristic Quality Index (*FQI*), and the Auchenorrhyncha Quality Index (*AQI*).

Study site

This study was conducted at a site in Macoupin County, Illinois, located 3.4 km ENE of Benld (39.06392° N - 89.45570° W; S28 T8N R6W). The site is a 1.4 ha field surrounded by oak-hickory forest to the south, east and west, and by agricultural fields to the north. The site is located in the Cahokia Creek Watershed and is on the boundary line between two natural divisions of Illinois, the Western Forest-Prairie Division (Carlinville Section) and the Southern Till Plain Division (Effingham Plain Section) (Schwegman1973). The field is dominated by forbs, with scattered grasses and sedges. The current owner purchased the property in 1977, at which point the field was in early stages of secondary succession. Since that time, the site has not undergone plowing or grazing by domestic animals, though it is believed to at one time have been used for agricultural purposes. Since 1986, the site has experienced occasional fires, some spontaneous, some managed, and very little mowing to prevent establishment of trees and shrubs. During the present study, the site was neither mown nor burned.

Soil is in the Marine silt loam series (type number 517B) which is somewhat poorly drained and is derived from parent material of loess over silty pedisediment. Climate of the region in which the site occurs is humid and temperate, with a mean annual temperature of 12.2°C. Total mean annual precipitation is about 980.4 mm, 57% of which falls between April and September.

METHODS AND MATERIALS

Vegetation field methods

A preliminary survey was carried out in 2010 in order to become acquainted with species present in the site. Each species encountered was collected, pressed and identified according to Mohlenbrock (2002), as were all species collected throughout this study. In addition, type specimens of all taxa were brought to the University of Illinois and verified by Paul Marcum and Rick Phillipe of the Illinois Natural History Survey.

Vegetation was surveyed during the 2011 and 2012 growing seasons. Stratified sampling was used to insure that the site was sampled fairly evenly, rather than heavily in a few areas and not at all in others. As such, the site was split into six sections, sections A through F, (Fig. 1) and random plot centers were generated within each section in proportion to the section's size. Constraints on plot placement were that a plot could not be closer than 0.50 m to a boundary line, and that all plots would be separated by $\geq$0.50 m. In total, 200 plots were generated for the entire site, 102 of which were used in this study (Fig. 2). Plots were located within the site using a handheld GPS (Garmin eTrex Legend, ± 15 m) and were staked and marked with a section letter and number for later relocation.

Sampling plots were 0.25 m^2 quadrats centered on each point Each species found within the square was recorded along with its percent coverage which was based on a modified Daubenmire (1959) coverage class. The classes were as follows: 1, <1; 2, 1-5; 3, 5-25; 4, 25-50; 5, 50-75; 6, 75-90; 7, 90-100 (ranges are percentages of the square). The site was sampled in two successive years (2011 and 2012). Each plot was sampled twice each year, in early summer (June 28 and July 6-7, 2011, and June 4-6, 2012), and late summer (August 23, 25, 30, and September 1 in 2011, and August 25, 26, and 30 in 2012).

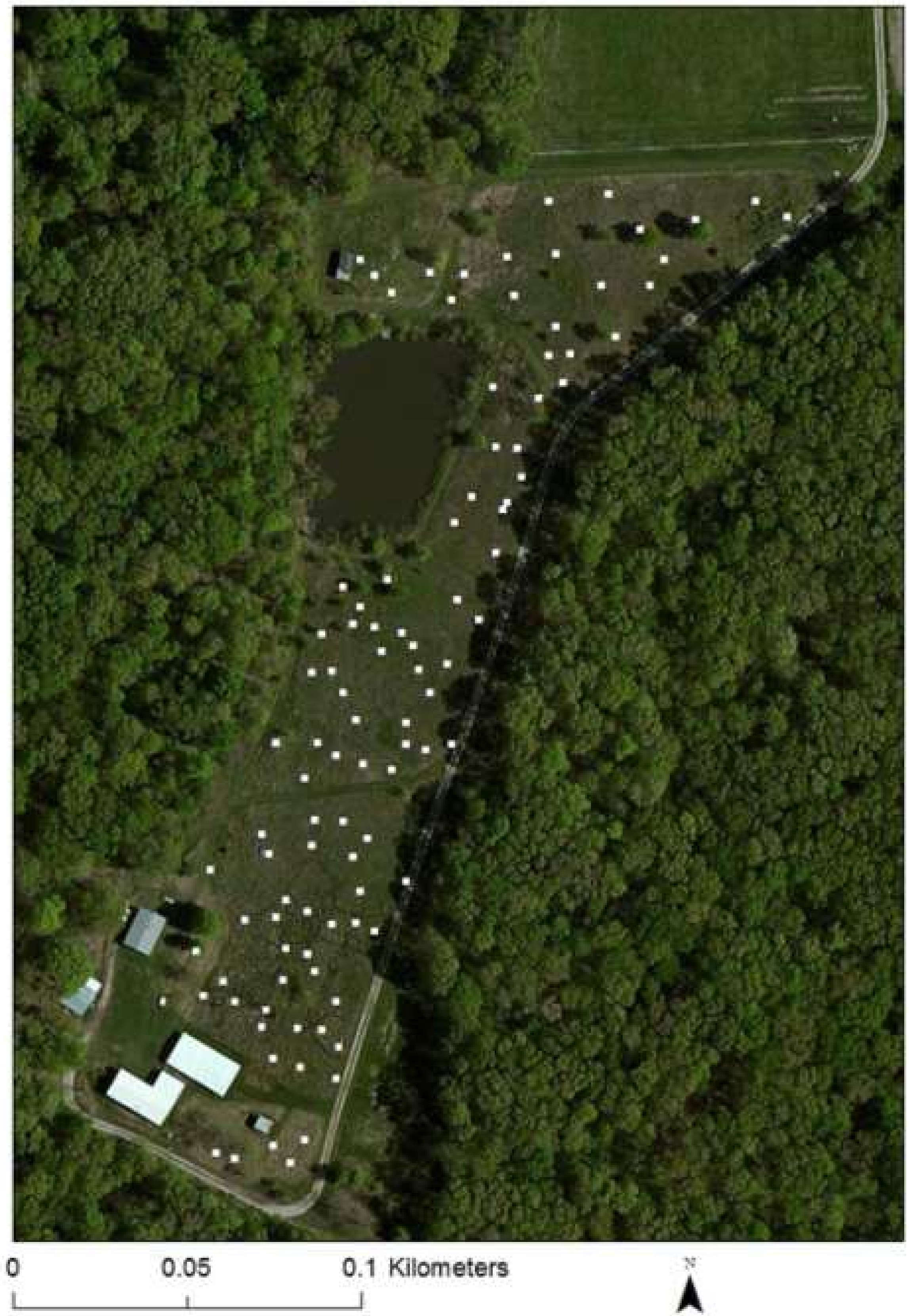

Figure 1. Satellite image of study area with surveyed plots shown in white.

Vegetation analysis

Both native and non-native vegetation data were used to calculate *S*, *N1*, *mean C*, and *FQI* for the study site. All calculations were done for species found within plots while *S, mean C*, and *FQI* were also calculated for the overall species inventory for comparison with inventories acquired from other sites (Table 1). Abundance measures were not available for all sites, thus *N1* was not included in this comparison as it was for the Auchenorrhyncha. Species composition was also used to calculate dissimilarity values between all sites using the Bray-Curtis dissimilarity index (Faith et al. 1987). This was done in the program DECODA (Minchin 1998).

Table 1. Name, status (restoration or remnant) and size (in hectares) of each site used in comparison study. All sites are located in Illinois and status for each (except for study site) was determined by either the Illinois Natural History Survey or the Illinois Natural Area Inventory.

Site	Designation	Size (ha)
Study Site	Restoration	1.5
Matanzas	Preserve	11.0
Midwine (South Patrol Road)	Restoration	190.2
Midwine (Exon Mobil Natural Area)	Preserve	15.8
Denby Prairie	Preserve	1.0

Auchenorrhyncha field methods

The layout for insect sampling differed from that for vegetation sampling because the *AQI* calculation was found to be more robust when transects were used rather than plots (Wallner 2010). Sampling took place along four 40 m transects (Fig. 3). Because the study site was long and narrow, transects were set up haphazardly rather than in parallel in order to keep them within the site and reduce edge effects (Wallner 2010). A leaf blower (Ryobi ™, South Pasadena CA) with a netted vacuum attachment was used for collection. The length of each transect was vacuumed for a total of five minutes, which was done at one minute increments to prevent plant material buildup in the collection net. After each minute, contents of the net were transferred to a Photo Tactic Utility Insect Extractor (PTUIE) and left for forty-five minutes to an hour for separation of insects from plant material. All Auchenorrhyncha collected in this fashion were separated and placed into small vials containing 70% ethanol and labeled with transect number and a letter to identify the individual. All specimens were identified to species by Chris Dietrich, systematic entomologist for the Illinois Natural History Survey.

Auchenorrhyncha were sampled on October 5, 2012. It is usually suggested that sampling for Auchenorrhyncha take place during late summer, around July or August (Blocker and Reed 1976). However, development of these insects is closely tied to host plant phenology (Dietrich pers comm.) and a long drought that occurred during the study period delayed plant maturation. On the advice of Chris Dietrich, sampling was carried out later in the year to permit plants and insects to mature.

Analysis of Auchenorrhyncha

Both native and non-native specimens were used to calculate *S*, N_1, and *mean C* (Table 2), as well as the Auchenorrhyncha Quality Index (*AQI*). The coefficients of conservatism for Auchenorrhyncha are, as with plants, indicators of each species' reliance on natural habitat for survival and reproduction, though their range is 0 to18 rather than 0 to 10. The calculation for *AQI* is the same as that for *FQI*.

All four biodiversity variables were calculated for the study site and a number of comparison sites known to be of mid to high quality. Comparison sites were sampled by Adam Wallner (2010) using the same methods described above, only some of the sites used three transects rather than four (Table 2). This comparison is necessary since *AQI* is not yet widely used and therefore does not have a standard rule of thumb for interpreting single scores like *FQI* does (Taft et al. 1997). Transect data were used to create a database in DECODA which was then transferred to PC-ORD (McCune & Mefford 2006) to find indicator species for each site.

Table 2. Name, abbreviation, size, number of transects sampled, and State location of each site used for comparison in Auchenorrhyncha survey. All sites have a designation of mid to high quality according to INAI grades.

Site	Abbreviation	Size (ha)	Number of Transects Sampled	State
Study Site	SS	1.5	4	Il
12 Mile	TM	38.0	4	Il
Cayler	CA	64.7	4	IA
Chloe	CH	61.0	4	MO
Hayden	HP	97.1	4	IA
Faville	FA	24.3	3	WI
Midewin	MI	6070.3	3	IL
Matanzas	MA	11.3	3	IL
Gooselake	GL	242.8	3	IL

RESULTS

Vegetation

Scores in all diversity indices remained fairly stable over the four sampling seasons though a drop occurred in N_1 for both 2012 seasons, as well as in the 2012 late season for *S* and *FQI* (Table 3). *Mean C* scores were consistent over all four sampling seasons. In all four seasons, *FQI* seems to be relatively low and not indicative of a quality area.

Table 3. Diversity variables calculated for the study site over four sampling seasons.

Variable	Sampling Season			
	2011		2012	
	Early	Late	Early	Late
S	79	73	74	61
N_1	25.22	28.04	20.02	20.08
mean C	2.62	2.73	2.59	2.46
FQI	23.29	23.29	22.32	19.21

When indices were calculated for a more comprehensive species list and compared to those for other sites known to be of good quality, scores for all sites were very similar (Tables 4 and 5). An *FQI* of 35 or greater tends to suggest an area of high floristic integrity (Taft 1997). Although *FQI* can be influenced by site size *mean C* is considered an area-independent variable that can readily be used for comparisons.

Table 4. Diversity variables calculated based on comprehensive inventories of the study site and four comparison sites. N_1 was not included as no abundance measures were available for the comparison sites.

Variable	Site Name				
	Study Site	Matanzas	Midewin (SP)	Midewin (EX)	Denby
S	209	150	293	253	149
mean C	2.79	3.39	2.40	2.75	3.13
FQI	40.40	41.56	40.96	43.69	38.26

Table 5. 95% confidence intervals for comparison sites calculated for each variable.

Variable	95% CI
S	94.85- 327.65
mean C	2.23- 3.61
FQI	37.56- 44.68

Dissimilarity calculations resulted in high scores among all sites, showing that there is very little similarity among the sites based on species composition (Table 6). The lowest score, i.e. greatest similarity, occurred between the two Midwine sites. This of course is not surprising as they occur very close to one another and likely experience similar management strategies.

Table 6. Bray-Curtis dissimilarities based on species composition of the study site and four comparison sites. High scores correlate to low similarity between two sites.

	Study Site	Matanzas	Midewin (SP)	Midewin (EX)
Matanzas	0.65	-	-	-
Midewin (SP]	0.66	0.67	-	-
Midewin (EX	0.71	0.68	0.45	-
Denby	0.67	0.60	0.63	0.65

Auchenorrhyncha

As with the vegetation, scores for diversity indices were similar among the study site and comparison sites (Tables 7 and 8). Again these sites are of known quality and have INAI scores between A and B, indicative of high quality natural areas. Unlike FQI, site size is not a problem with the AQI calculations here since they are all based on sampling a limited number of transects.

Table 7. Diversity variables calculated based on transect surveys of the study site and eight comparison sites.

Variable	Site Name								
	Study Site	12 Mile	Cayler	Chloe	Hayden	Faville	Midewin	Matanzas	Gooselake
S	23	26	24	26	30	18	22	24	27
N_1	12.60	16.72	11.52	20.36	9.18	7.04	12.89	15.35	13.30
mean C	7.69	8.39	7.69	6.11	6.52	8.84	8.32	5.99	8.92
AQI	36.87	42.80	37.66	31.18	35.69	37.53	39.07	29.34	46.33

Table 8. 95% confidence intervals for comparison sites calculated for each variable.

Variable	95% CI
S	21.63- 27.62
N1	9.76- 16.83
mean C	6.58- 8.62
AQI	32.78- 42.12

Indicator species analysis revealed that while the sites received similar scores for all biodiversity variables, the study site is mostly characterized by hopper species that are less conservative than those that characterize the comparison sites (Table 9).

DISCUSSION

It is difficult to measure the time it takes for plant communities to respond and change when faced with disturbance, in part due to the short time frame in which most field studies occur (Tilman 1982). Adding to the difficulty is the fact that these communities exist in a loose equilibrium, meaning that while there may be high fluctuation in composition and abundance in response to disturbance on a short time scale, structure generally remains stable when viewed on a longer time scale (De Angelis et al. 1985). That being said, no community can remain the same over any period of time even with an apparent lack of disturbance (Collins 2000). Results of the present study concur with the idea that great variation occurs in brief amounts of time. Table 3 summarizes this variation within the study site, showing that even over the course of a single growing season there were changes in species composition and abundance. The change from 2011 to 2012 was even more drastic, with drops occurring in each of the diversity measurements. This was likely due to an extended drought period that occurred during the study. A longer time frame would be necessary to discern whether the diversity and quality would recover to its former state and persist as a stable entity as predicted by the loose equilibrium hypothesis. Although the *FQI* scores seem to be indicative of a low quality area, these scores are likely an artifact of low sampling intensity (plot data vs. species inventory). Thus they merely reflect temporal changes rather than overall site quality.

In comparing the study site to other high quality natural areas, it was surprising to find that it was comparable based on the diversity indices measured (Tables 4 and 5). Although the comparison sites were much larger, in all three diversity indices the study site scored higher than one or more of the comparison sites. Based on the dissimilarities calculated it cannot be said that the study site is very similar to any of the comparison sites (Table 6). This is not surprising as each of the sites occurs in very different areas with varying environmental aspects. Management practices, grazing by herbivores, topography, soil type and many other variables can influence species composition and differences in these variables can lead to areas that look very different (Hartnett et al. 1996). This does not however mean that two areas with different species compositions cannot be of similar quality. Two sites could not share a single species in common and still have the same average coefficient of conservatism (Taft et al. 1997).

The results of this survey beg the question, where did this prairie come from? It was stated earlier that the landowner did not seed or plant the site and that very little was done in the way of management. Seed banks are one possible explanation. Species capable of producing seed banks have evolved mechanisms to prevent premature germination, so seeds will remain dormant but viable until proper conditions are met (van Baalen 1982). Seeds of some species are able to remain viable for very long periods of time, some up to 100 years (Kivilaan & Bandurski 1981). An important aspect of germination of banked seeds is their return to a desirable depth in the surface soil as

Table 9. Auchenorrhyncha indicator species for study site and comparison sites. Each species is shown with corresponding coefficient of conservatism and indicator value. Statistical significance was tested by randomizing group membership 10,000 times. Significant indicators (P< 0.05) are listed in descending order of indicator value within each group.

Site	Species	CC	IV	*P*
SS	*Latalus sayii*	5	97.3	0.0002
	Scaphytopius frontalis	4.75	90.9	0.0001
	Masamia sp.	11.875	75.0	0.0038
	Toya propinqua	0	75.0	0.0050
	Scaphytopius acutus	4.75	57.3	0.0203
TM	*Kelisia curvata*	13.25	100.0	0.0001
	Prosapia bicincta	6.25	75.0	0.0051
	Xerophloea peltata	8.5	54.9	0.0145
CA	*Extrusanus orrysus*	12.75	100.0	0.0003
	Commellus comma	11.75	75.0	0.0041
	Scolops sulcipes	11.5	75.0	0.0003
	Diplocolenus configuratus	12	70.0	0.0028
	Athysanus argentarius	0	55.9	0.0001
	Doratura stylata	0	48.2	0.0028
CH	*Bakerella rotundifrons*	10.25	100.0	0.0003
	Stirellus bicolor	6	88.9	0.0003
HA	*Agalliota constricta*	3	94.7	0.0002
	Agalliopsis novella	4.75	75.0	0.0040
	Empoasca fabae	1.5	75.0	0.0050
FA	*Erasmoneura tecta*	10	100.0	0.0009
	Megamelus metazeri	14.5	100.0	0.0009
	Memnonia panzeri	17	100.0	0.0009
	Kelisia pectinata	13.25	76.9	0.0003
	Cicadula melanogaster	9.5	76.3	0.0048
	Caenodelphax nigriscutelleta	13.5	71.4	0.0027
MI	*Flexamia prairiana*	11.75	100.0	0.0010
	Aflexia rubranura	16.5	66.7	0.0244
	Bruchomorpha dorsata	12.25	51.3	0.0255
MA	*Kelisia vesiculata*	9.5	100.0	0.0009
	Amplicephalus osborni	13	66.7	0.0251
	Delphacodes puella	5.5	66.7	0.0251
	Graminella nigrifrons	2.5	66.7	0.0251
	Pissonotus piceus	10	66.7	0.0251
	Forcipata loca	1.5	43.9	0.0321
	Xestocephalus desertorum	4	40.7	0.0417
GL	*Destria fumida*	16	100.0	0.0012
	Deltocephalus gnarus	11.5	66.7	0.0268
	Neohecalus magnificus	15.75	66.7	0.0241
	Opsius stactogalus	0	66.7	0.0268
	Xestocephalus pulicarius	4	64.0	0.0058
	Dorydiella kansana	13.5	49.0	0.0151

opposed to being buried deeply (McRill & Sager 1973; Grant 1983). One would assume that if the site had a history of cultivation, plowing would have brought these seeds up towards the surface and allowed them to germinate, thus exhausting the seed bank over time. However, the history of the site is still under question as no records can be found of its previous use, so the presence and subsequent germination of a seed bank is still a plausible explanation for the source of species present. Another possibility is that seeds were brought in from surrounding sources. In driving around the area, it was noticed that several patches containing the same species found in the site occurred on the outskirts of wooded areas and fields and alongside roads. Though in most cases seeds move only very short distances on their own (Howe & Smallwood 1982) there are circumstances under which they can be dispersed over much greater distances. Birds and large herbivores can serve as vectors for seed dispersal (Collins & Uno 1985). Some species produce fruits that can be transferred via the fur of passing animals. Species in the genus *Desmodium* have flat, sticky fruits that can grasp onto fur, clothing and even bare skin. There are other species that produce seeds with appendages that enable them to remain air-borne for a long period of time. Many species in the family Asteraceae have seeds with a feather-like pappus for wind dispersal. Cain et al. (2000) emphasize the importance of long-distance seed dispersal and the furthering of our understanding of the topic through more experimentation.

Results from the Auchonrrhyncha survey reflect the findings from vegetation surveys. Wallner et al. (2012) found a similar correlation when FQI, AQI and other diversity indices were used to evaluate quality in hill prairie restorations and remnants. All index values used were capable of distinguishing among high, mid and low quality hill prairies.

In the present study, scores for all diversity indices were similar across the study site and all comparison sites, indicating that the study site is comparable in quality to the others (Tables 7 and 8). Because no specific management plans are currently in practice in the study site, it is not known how burning and/or mowing would affect these insects. Some studies have shown that prairie arthropods respond differently to management than plants (Swengel & Swengel 2007). This should be taken into consideration so as to not damage the existing communities. A survey would have to be conducted after the implementation of management practices in order to determine the effects on Auchenorrhyncha as well as an appropriate disturbance regime. Wallner (2010) observed that Auchenorrhyncha populations also corresponded closely to the types of vegetation present. He concluded that presence and abundance of conservative Auchenorrhyncha species was dependent on the presence of perennial C4 grasses, thus management for these species should include maintenance of these grass communities.

Although index scores showed the study site to be similar to the comparison sites, indicator species analysis revealed that not only are the sites characterized by completely different species, but also that the study site is characterized by species with lower conservatism scores than those of the comparison sites (Table 9). In fact, the study site seems to support mostly lower quality species, whereas high quality species are relatively abundant in many of the comparison sites. The inability of quality indices and diversity measurements to detect these types of differences poses a problem for ecologists and restoration managers. Index scores alone could lead one to believe that a natural site is maintaining populations of quality species, or that a restoration is successful, when in fact they may have very low or even decreasing numbers of conservative species. In order to prevent these misconceptions, indicator species analysis should be combined with quality indices and diversity measurements when surveying natural areas (Dufrêne & Legendre 1997).

In conclusion, the results of this study do not support the hypothesis that succession without the aid of management would result in a site with low floristic integrity. In fact, according to all the diversity indices scores the study site has very high floristic integrity. Studies conducted in multiple habitat types yielded similar results (Edgin & Ebinger 2000; Galatowitsch & Van der Valk 1996) and in some cases succession lead to areas of better quality than those restored mechanically (Hodacova & Prach 2003; Tropek et al. 2010). Of course, success of natural restoration is likely dependent on edaphic aspects of the site itself (e.g. moisture availability, soil nutrients) and the availability of local seed sources (Řehounková & Prach 2007). All sites will not have these attributes, so mechanical restoration and management would still be necessary. Also, eradication and prevention of encroachment by exotic and woody species would still be required over time, as is the case with the study site. Nonetheless, successional processes should be accounted for in any type of restoration attempt to increase the likelihood of success.

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