EVALUATING NORTH AMERICAN TALLGRASS PRAIRIE QUALITY USING THE AUCHENORRHYNCHA QUALITY INDEX

BY

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DISSERTATION

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ABSTRACT

Auchenorrhyncha (i.e., leafhoppers, treehoppers, spittlebugs, and planthoppers) represent some most diverse groups of herbivorous insects in the tallgrass prairie, they have close associations with many native prairie grasses and forbs, and respond in predictable ways to changes in grassland degradation. These attributes make Auchenorrhyncha ideal candidates in the development of a habitat quality index to measure tallgrass prairie integrity. Chapter 1 provides a detailed introduction of ecological integrity, methods used in measuring tallgrass prairie integrity, and the usefulness of insects in assessing prairie integrity. Chapter 2 describes the development of the Auchenorrhyncha Quality Index (AQI), and examines how changes in robustness (vacuum and sweeping) and sensitivity (changes in time of year and prairie community) affected two versions of the AQI (with abundance [$AQI_{w/N}$] and without abundance [$AQI_{w/outN}$]). The AQI was computed by assigning each auchenorrhynchan species a Coefficient of Conservatism (CC) value, which ranged from 0 (habitat-generalist/tolerant to degradation) to 18 (prairie-dependent/intolerant to degradation). These CC values are averaged and combined with species richness producing the $AQI_{w/outN}$ or these values are summed and weighted with abundance and combined with species richness producing the $AQI_{w/N}$. The robustness and sensitivity of both versions of the AQI were analyzed by collecting Auchenorrhyncha from 35 sites in 4 states (Illinois, Wisconsin, Iowa, and Missouri) over 3 years (2004, 2005, and 2008) using a combination of sweeping and/or vacuuming from transects and/or 5x5m plots from wet-mesic, loess, glacial-drift, gravel hill prairies, and sand prairies. ANOVAs showed that both versions of the AQI were insensitive to changes in time of year and prairie community at the landscape level when sampling from four transects using a vacuum but both versions of the AQI exhibited variation on individual sites when vacuum or sweep sampling throughout the growing season and on wet prairies. Chapter 3 examined the ability of the AQI and related measures of Auchenorryncha integrity, and vegetation-based measures of integrity and diversity in discriminating glacial-drift hill prairie quality on 14 sites in Illinois; and examined the relationships between Auchenorrhyncha integrity and diversity and vegetation integrity and diversity. Both Auchenorrhyncha and vegetation integrity discriminated quality in similar ways and prairie Auchenorrhyncha diversity was positively associated with native prairie grasses. Chapter 4 examined the effects of prescribed burning on the AQI and related measures of Auchenorrhyncha integrity and diversity on 22 loess hill prairies in Illinois. These results showed that recently burned and frequently burned sites exhibited lower Auchenorrhnyncha integrity and diversity values than unburned prairies.

dedicated to my parents

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Chapter 1: The Usefulness of Inscects in Assessing Tallgrass Prairie Integrity

Ecological integrity (i.e. quality) of a habitat or ecosystem can be defined as the ability of that habitat or ecosystem to maintain its native biotic (e.g. species) and abiotic components and processes in the presence of disturbance (Karr 1991). These disturbances may include natural (e.g., tree falls, wind, and natural fires), anthropogenic or human-induced disturbances (e.g., construction of roads and dams), and habitat fragmentation (Pickett and White 1985, Karr 1991, Fore *et al*. 1996). Also, these disturbances can vary in intensity, magnitude, and frequency and may lead to the alteration in hydrology, elimination of species, and habitat loss or isolation of habitat remnants (Pickett and White 1985, Odum 1987).

Measuring ecological integrity has conservation implications. For example, land managers could identify areas that support a large proportion of native flora and fauna. These areas can be set aside as nature preserves. Assessing integrity can also be useful in evaluating the success of restoration and land management practices of threatened and endangered ecosystems. One ecosystem that could potentially benefit in measuring ecological integrity is the tallgrass prairie which, with only 1% of the original biome remaining, represents the most threatened ecosystem in North America (Vickery *et al*. 2000).

Two methods have been successfully used in measuring tallgrass prairie integrity. One of these methods is conventional measures of diversity, such as species richness, the Shannon-Weiner diversity Index and the Simpson's Dominance Index (Weaver and Shannon 1949, Whittaker 1975, Good 1953, McInosh 1967, Margurran 2004). Although, these measures provide rapid and repeatable assessments of tallgrass prairie integrity, they ignore species composition giving equal weight to a conservative species, such as *Aflexia rubranura*, a leafhopper specialist on prairie dropseed, in contributing to prairie integrity as an adventive species, such as *Doratura stylata*. As a consequence, inaccurate assessments of tallgrass prairie integrity may be produced.

To correct for this bias a second group of methods has been developed, commonly referred to as biotic indices. A biotic index is a metric value that is based on qualitative or quantitative data on a particular group of organisms (i.e., bioindicators) that reflects the relative condition of a habitat (Margurran 2004). These biotic indices are advantageous over conventional measures of diversity because they treat a species such as *Aflexia rubranura* as contributing more to prairie integrity than an adventive species, such as *Doratura stylata*. An example of a biotic index is the Floristic Quality Index, which combines both qualitative (e.g., life history or biological information) and quantitative data (i.e., species richness) into a single measure for plants (Taft *et al*. 1997). The Floristic Quality Index has been successfully used in assessing tallgrass prairie quality in Illinois (Taft *et al*., 2006).

However, evidence has shown that some diverse groups of prairie organisms, such as terrestrial arthropods respond to disturbance differently than plants (Harper *et al*. 2000, Swengel and Swengel 2007). As a consequence, indices based only on vegetation may not be able to detect negative impacts for either land management practices (e.g. prescribed fire) or other degrading factors (e.g. construction of roads) on arthropods. Thus, additional biotic indices based other diverse prairie organisms, such as terrestrial arthropods, are needed to compliment vegetation based measures of integrity in providing a broader assessment of prairie integrity.

Terrestrial arthropods, in particular insects, are ideal candidates in measuring prairie integrity because they represent some of the most diverse groups of organisms found in terrestrial ecosystems with current estimates of species richness ranging from 2 million to as many as 12 million species (May 1989, Hodkinson and Casson 1991, Hammond 1992, Gaston

1992, Samways 2005). These numbers indicate that arthropods significantly contribute to Earth's biodiversity (Hammond 1992), and in many cases represent a major proportion of the biota found in many terrestrial biomes, including the tallgrass prairie (Kremen *et al*. 1993, Samways 1993, 2005, Wiles and Chazdon 2006).

Not only do arthropods represent a significant component of biodiversity but they provide numerous ecological services, such as predators of other terrestrial arthropods (Boswell *et al*. 1998), food for birds and other vertebrate taxa (Willis and Oniki 1978, Wilson 1992, Samways 2005), parasitize other terrestrial arthropods (Prévost *et al*. 1989, Combes 1996, Magagula and Samways 2001), and recycle nutrients, making them more readily available to other organisms, such as plants (Abbadie *et al*. 1992, Humphreys 1994, Samways 2005). Terrestrial arthropods also aid in pollination of flowing plants in natural and agricultural ecosystems (Mahy *et al*. 1999, Johnson and Steiner 2000) and reduce reproduction potential of flowering plants in natural ecosystems (Molano-Flores 2009). Terrestrial arthropods also play major roles in the transfer of energy (Brown and Gange 1992, Milton 1995, Belovsky, Samways 2000a, 2005)

Terrestrial arthropods are also highly sensitive to environmental perturbations at various spatial and temporal scales. In fact, they often respond to perturbations; (e.g., mowing, haying, and prescribed burning) in predictable ways, making them ideal candidates in reflecting environmental change (Rosenberg *et al*. 1986, Kremen *et al*. 1993, McGeoch 1998, Nickel and Hilderbrandt 2003). Because of this sensitivity, these taxa have been readily used in measuring stream and river integrity (Fore *et al*. 1996, Andersen *et al*. 2004, Bonada *et al*. 2006). However, their use as indicators of terrestrial ecosystem integrity has been far less enthusiastically embraced (Majer and Andersen 1996, Kimberling *et al*. 2001, Karr and Kimberling 2003, Andersen and Majer 2004). This reluctance is mainly attributed to the fact that many terrestrial arthropods are difficult to identify, their populations seem to fluctuate dramatically from year to year, the distribution and life history of many groups are virtually unknown, and few researchers in land management are familiar with terrestrial arthropods, in particular insects (Kendeigh 1979, Andersen *et al*. 2004, Dietrich 2009).

Only a few indices based on insects have been developed to assess tallgrass prairie integrity. For example, the Illinois Butterfly Significance Index (Ruesink and Jeffords 2004) was developed to assess tallgrass prairie, forest, and wetland integrity in Illinois. Similar to the Floristic Quality Index (Taft *et al*. 1997), the Illinois Butterfly Significance Index combines biological information, such as a species' tendency to be restricted to a prairie and tolerance to habitat degradation (i.e., conservative species) with species richness and abundance. Other indices used to measure prairie integrity include rarity coefficients (Panzer *et al*. 2010). Rarity coefficients assign each prairie-remnant dependent species a score from 1 – 5 based on the number of individuals present on a site/total sites surveyed. A high score (e.g., 5) reflects low abundance of a species, indicating that the species is threatened whereas lower scores (e.g., 1, 2, 3, 4) reflect an abundant species, indicating that the population of that species is secure.

However, there are limitations in using the Illinois Butterfly Significance Index and rarity coefficients. For example the Illinois Butterfly Significance Index relies on habitat generalist butterflies, which may not be sensitive to changes in prairie integrity; consequently any assessment based on this index may be inaccurate. Conversely, rarity coefficients rely on remnant-dependent species which may yield inflated assessments of prairie integrity. Thus, other groups of prairie insects are needed in the development of biotic indices that respond to prairie degradation in predictable ways and include species that span a wide range of conservatism (i.e., habitat dependence/sensitive to degradation) to the tallgrass prairie yielding a more accurate assessment of prairie integrity.

One such group that is numerically dominant in abundance and species richness in native grasslands is Auchenorrhyncha (Insecta: Hemiptera [DeLong 1948, Waloff 1980, Hamilton 1995, Nickel 2003]). These insects include leafhoppers (Cicadellidae), planthoppers (Fulgoroidea), treehoppers (Membracidae), spittlebugs (Cercopoidea), and cicadas (Cicadoidea) (Hamilton 2005). These species display a wide range of conservatism, from strong affinities to a few species of prairie plants to habitat generalists (Whitcomb *et al*. 1987, Hamilton and Whitcomb 1993) and these species respond in predictable ways to grassland degradation (Harper *et al*. 2000, Hilderbrandt and Nickel 2003). These insects are also well documented in the taxonomic literature (DeLong 1948) allowing for rapid identification by professionals and non-professionsals. Because of these attributes, Auchenorrhyncha are ideal candidates in developing a biotic index to measure prairie integrity.

The main objectives of this study are to 1) develop a habitat quality index using Auchenorrhyncha to measure prairie integrity (i.e., Auchenorrhyncha Quality Index or AQI), 2) examine the robustness (changes in sampling methods/design) and sensitivity (e.g., changes in habitat, growing season) of the AQI, 3) investigate the ability of the AQI in discriminating between differences in glacial drift hill prairie quality in Illinois and whether these differences are correlated with vegetation-based measures of integrity and diversity, 4) examine the relationships between Auchenorrhyncha and vegetation integrity and diversity from glacial-drift hill prairies in Illinois, and 5) examine the effects of prescribed burning on Auchenorrhyncha-based measures of integrity and diversity on loess hill prairies in Illinois. Ultimately, the development of the AQI can provide an additional tool that can be used in combination with other indices of prairie integrity, for example the FQI, in providing land managers and policy makers with a more complete assessment of prairie integrity.

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Chapter 2: Auchenorrhyncha Quality Index - A New Method in the Assessment of North American Tallgrass Prairie Quality

ABSTRACT

In this study a species-based habitat quality index (Auchenorrhyncha Quality Index or AQI) was developed incorporating six ecological characteristics of prairie auchnenorrhynchan insects and other components of their diversity, such as species richness and abundance. Two permutations of this index were developed, the $AQI_{w/outN}$ and $AQI_{w/N}$. The robustness and sensitivity of both versions of the AQI were tested on 35 sites by sampling on hill, sand, and wet-mesic prairies under different temporal and spatial conditions using vacuum and sweep-sampling techniques on transects and/or plots. The majority of Auchenorrhyncha collected from these sites were leafhoppers and delphacid planthoppers, spanning a range of dependence to prairie remnants, although other less diverse groups of Auchenorrhyncha were also found. Results showed that at the landscape level vacuum sampling consistently yielded higher values of both versions of the AQI than sweep sampling, these values were not sensitive to differences in prairie vegetation and changes in the growing season. Both versions of the AQI calculated from vacuum sampling also stabilize after the fourth transect. However, both versions of the AQI showed variation on individual sites at different times of the year, and wet-mesic prairies required different amounts of sampling in calculating stable AQI values when using a vacuum and sweep net. To reduce seasonal variation on individual sites it is recommended that land managers and researchers sample twice a year using a vacuum and four transects and AQI profiles are needed to determine the number of transects required in calculating stable AQI values on wet-mesic prairies. Ultimately, the AQI can be used as an additional tool in the assessment of prairie integrity.

INTRODUCTION

The tallgrass prairie is the most endangered ecosystem in North America (Robertson *et al*. 1997). Of this once vast grassland, only 3 - 5% remains, most of which is highly fragmented and restricted to nature preserves, parks, and along railroad right-of-way (Whiles and Charlton 2006). Vegetative data has been used almost exclusively in measuring changes in quality of the remaining habitat patches of this dwindling ecosystem. However, recent evidence (Panzer *et al*. 1995, Harper *et al*. 2000, Swengel and Swengel 2007) indicates that some groups of prairie arthropods respond to management differently than plants. Thus, vegetation-based measures of habitat quality (e.g., species richness, Shannon Wiener Index, Floristic Quality Index) may not be able to detect negative impacts for either management or other degrading factors on arthropods.

Terrestrial arthropods are the most diverse group of organisms in the tallgrass prairie and play crucial roles in ecosystem function, such as pollination, nutrient cycling, and energy flow (Whiles and Charlton 2006). Many of these species are restricted to a few remnant prairie communities (Panzer *et al*. 1995, Hamilton 1995, Reed 1996). Yet, despite their contribution to the maintenance of ecosystem integrity and their affinity to native prairie remnants, only a few studies have used arthropods to track changes in native grassland quality over time and space (Hamilton 1995, Harper *et al*. 2000, Panzer 2002, Nickel and Hildebrandt 2003). This dearth in literature may be attributed to the lack of ecological information or life history of arthropods that are associated with environmental disturbance (duration, frequency, intensity) and /or stress (habitat favorability) (Novotny 1995). Some evidence has also indicated that populations of some arthropod species fluctuate dramatically from year to year in response to short term changes in climate (e.g., rainfall, temperatures, humidity, etc.) and this makes it difficult to distinguish management response from natural fluctuations in populations (Waloff 1980, Schowalter 2000). Another factor that has hindered the selection of this group of bioindicators includes availability of taxonomists (Majer *et al*. 2007). Ultimately, which arthropod taxa are the most practical, inexpensive to handle, and the most suitable for measuring prairie quality must be chosen with care (Nickel and Hilderbrandt 2003).

The following criteria may be useful in selecting indicator taxa: 1) groups need to be selected that are diverse in both species and individuals so as to comprise a significant component of biodiversity, allowing them to be easily sampled for quantitative studies; 2) they should be treated thoroughly in the taxonomic literature to allowed for rapid identification by both professionals and non-professionals; and 3) they should include conservative (i.e., habitat specific and intolerant to degradation) and non-conservative (i.e., habitat generalist and tolerant to degradation) species, allowing for a more accurate assessment of prairie quality. Conservative species are needed because they may reflect the original condition of the prairie. Including non-conservative (i.e., adventives and generalist species) species are also important because they are a source and response of habitat degradation (Bourdaghs *et al*. 2006), and thus their inclusion will reduce the chance of generating inflated assessments of prairie quality. One group of arthropods that meets these criteria is Auchenorrhyncha. These are phytophagous (plant feeding) insects in the order Hemiptera. They include planthoppers (Fulgoroidea), treehoppers (Membracidae), spittlebugs (Cercopoidea), and cicadas (Cicadoidea), with the majority of these species being leafhoppers (Cicadellidae [Hamilton 2005]).

Auchenorrhyncha are abundant in grasslands; in some cases exceeding 1000 ind. /m^2 (Waloff 1980, Hamilton 1995). Because of their large population sizes they are likely to form an important component of grassland food webs (Waloff 1980, Curry 1994, Nickel 2003, Nickel and Hidelbrandt 2003, and Biederman 2005). For example, they serve as hosts to many insect parasitoids such as Pipuncluidae (Diptera), Dryinidae (Hymenoptera) and Strepsiptera, and they are consumed by ants, spiders, and some vertebrates (e.g., birds) (Waloff and Jervis 1987). Moreover, as primary consumers, they can damage plant tissue through oviposition and transmit plant pathogens, and thus can potentially influence plant species composition (Brown 1985, Nickel and Hilderbrandt 2003).

Auchenorrhyncha represent a significant component of the North American prairie biota, with over 1100 species associated with native grasses, sedges, and forbs (Breakey 1928, Doering 1940, 1941, Kopp and Yonke 1970, Hamilton 1995, 1999, 2005). More than half of these are endemic to prairies alone (Hamilton and Whitcomb 1993, Wilson *et al.* 1993), with current estimates of 300 species in Illinois alone.

Despite their high diversity in species and individuals, sampling of both their relative and absolute population density, as well as the full range of species at a site can be done rapidly on two or three dates a year (Stewart 2002, Nickel and Hilderbrandt 2003). This is achieved with moderate effort and inexpensive equipment, such as modified suction devices, sweep-nets, pitfall traps and/or a combination of these techniques (Wilson *et al.* 1993, Stewart 2002). These sampling procedures can be easily standardized allowing for comparisons of species diversity, abundance, and composition across time and space.

Auchenorrhyncha taxonomy is complete in the Midwest region (DeLong 1948). However, various keys, monographs, and other taxonomic literature are needed for accurate identification as the determination of these species requires careful dissection and examination of the male genitalia. For example, DeLong's (1948) monograph of the Illinois fauna of

Cicadellidae describes most species are described in detail, together with drawings of the genital apparatus, and information provided on their host plants, and habitats.

One of the most prominent advantages in using Auchenorrhyncha as an indicator of grassland quality is their well documented life history traits or ecological characteristics (Brown and Southwood 1983, Brown 1985, Denno 1994, Novotny 1994a, b, 1995, Nickel 2003). For example, leafhoppers feeding on ruderal plants have wide host plant ranges and often occur in early successional stages of grasslands (Whitcomb *et al*. 1986, 1987, 1988). Conversely, monophagous Auchenorrhyncha tend to be denizens of relatively undisturbed grasslands (Brown 1985, Denno 1994), feeding almost exclusively on perennial plants, like grasses and sedges (Whitcomb *et al*. 1988).

Other life history traits, such as the number of generations or voltinism are strongly correlated with the integrity of a habitat. Novotny (1995) and Nickel (2003) observed that Auchenorrhyncha with two (bivoltine), three or more (polyvoltine) generations a year are dominate in heavily or recently disturbed habitats, such as agriculture or recently abandoned agricultural fields. Species that produce one generation (univoltine) a year are dominant in undisturbed and natural habitats, such as fens (Novotny 1994b, 1995, Nickel 2003). Additional traits associated with variation in grassland quality include changes in wing length and dispersal strategies. Variation in these traits is largely attributed to trade-offs between flight activity and reproduction, which is strongly influenced by changes in habitat quality (Roff 1986). For example, Roff (1986, 1990) and Denno *et al*. (1990, 1991) observed that Auchenorrhyncha with long-wings (macropterous) are dominant in ephemeral grasslands or when host plant quality begins to deteriorate. These macropterous individuals have lower rates of fecundity, because more resources are being allocated to the development of wing apparatus rather than reproduction (Denno and Roderick 1990, Denno *et al*. 1991, and Roff 1990), allowing for rapid location of host plants. On the other hand, short-winged (brachypterous and submacropterous) Auchenorrhyncha often occur in habitats where host plant quality is not deteriorating and relatively long-lasting host plants, such as perennials are dominant. Thus, selective pressure for higher reproduction rates in brachypterous and submacropterous individuals is stronger than for macroptery and dispersal ability (Nickel 2003), except if habitat or host plant quality begins to deteriorate.

Another important trait is diapause or hibernation, which allows species to escape environmental perturbations, as well as synchronize with their host plants (Nickel 2003). This trait can be classified into four main strategies, with successive strategies being associated with a greater degree of habitat quality. These strategies are as follows: 1) highly migratory species, often overwintering in fallow fields and other non-native grassland ecosystems; 2) species that overwinter in ruderal plant species, are often native, but are more common in early successional stages of grassland communities (Whitcomb *et al*. 1987); 3) species that overwinter in the soil or dead vegetation, such as cicadas and spittlebugs; and 4) species that conduct their entire life cycle in native grassland communities.

Grassland Auchenorrhyncha that are found over a wide geographical range tend to be found in degraded grasslands. In fact, species that are found over broad geographical ranges generally have traits that are advantageous in colonizing new habitats and recolonizing old ones (Novotny 1994, 1995). Some of these traits may include high dispersal ability and a wide host plant range (Novotny 1991). In fact, Novotny (1995) found that leafhopper assemblages exhibiting strong migratory behavior have strong preferences for host plants on disturbed habitats, and those producing several generations per year also have large geographic ranges.

Conversely, Hamilton (1995, 1997, and 2004) discovered that many auchenorrhynchan species that are endemic or geographically restricted to a few areas tend to exhibit traits such as low mobility, narrow host plant range, and producing one or two generations a year. Species that exhibit these traits tended to be dominant on undisturbed grassland ecosystems (Hamilton 1995).

Key life history traits (voltinism, overwintering preference, geographic range, wing length, and host plant affinity) can be combined to provide an objective measure of the expected conservatism of a particular species. The conservatism scores for individual auchenorrhynchan species occurring on a particular site can be summed or averaged, providing an overall measure of habitat quality, because higher quality, less biologically degraded prairies of higher conservation value are expected to harbor a higher proportion of conservative species. This measure can be computed with other components of diversity, such as abundance and species richness producing an index of prairie quality based on this group, which may compliment other measures of grassland quality (e.g. Rarity Coefficients; Panzer *et al*. 2010). If such a tool is found to be robust (i.e., statistically verified and sensitive) it can provide land managers with another method for assessing the relative condition of grassland ecosystems.

The present study is to develop an index of habitat quality based on Auchenorrhyncha life history traits combined with abundance and species richness data (i.e., Auchenorrhyncha Quality Index or AQI) that is robust under different sampling designs and sampling methods, insensitive to changes in prairie communities, and time of year (seasonality), and is stable with different amounts of sampling. I will test the following research questions in relation to creating the AQI: 1) is the AQI robust to changes in sampling method (sweep and modified leaf blower vacuum) and sampling design (transects and plots); 2) is the AQI robust to changes in sampling method and does this depend differ depending on the kind of habitat sampled?; 3) is the index sensitive to changes in sampling method (plot and transects) and the kind of habitat sampled?; 4) is the this index sensitive to sampling techniques (sweep and vacuum) and time of year sampled?; 5) are vacuuming and sweeping adequate methods in collecting all possible species at a site?; and 6) is the index sensitive to sampling intensity and if so, does this sensitivity differ depending on the kind of habitat sampled?

I predict vacuum sampling will yield higher index values than sweeping, regardless of the sampling technique, and habitat. This prediction is based on evidence that vacuuming is more effective in collecting Auchenorrhyncha than sweeping because a vacuum is not hindered by vegetation, and thus could potentially collect both conservative species occurring near the base of plants, as well as non-conservative species that inhabitat the top of vegetation (Wilson *et al.* 1994, Stewart 2002, Hilderbrandt and Nickel 2003). Since non-conservative (habitat-generalist/tolerant to degradation) Auchenorrhyncha are dominant early in the growing season when their cool season grass hosts are dominant, I predict index values will be lower early in the growing season compared to later in the season (Blocker *et al.* 1972, Waloff 1980). Finally, I predict that both versions of the AQI and Auchenorrhyncha species richness based on vacuum sampling will stabilize as sampling increases, but AQI values will not remain stable and species richness will not approach an asymptote when sweep sampling is used. This prediction is based on evidence that vacuuming is more effective in collecting Auchenorrhyncha than sweeping because a vacuum is not hindered by vegetation, and thus can collect both conservative species occurring near the base of plants, as well as non-conservative species that inhabitat the top of vegetation (Wilson *et al.* 1994, Stewart 2002, Hilderbrandt and Nickel 2003).

METHODS

Developing the Auchenorrhyncha Quality Index

The Auchenorrhyncha Quality Index (AQI) generates a metric that measures the relative biological integrity of North American Midwestern tallgrass prairies. Two permutations of this index were developed. The first permutation (**$\mathbf{AQI_a}$;** Equation 1) is the calculation of the mean coefficient of conservatism (CC) multiplied by the square-root transformation total species richness of per sampling unit (e.g., site, transect, plot, etc.). A species richness component was included because it provides quantitative information on Auchenorrhyncha biodiversity and may increase the ability of the AQI in discriminating between differences in prairie quality when species composition is similar.

The second permutation (**$\mathbf{AQI_b}$:** Equation 2) is the relative proportion of individuals for each species encountered multiplied by its CC value. These values are summed and multiplied by the square-root transformation of all Auchenorrhyncha encountered at the sampling unit. The relative abundance of each species was included because it provides additional information on the population dynamics of each species encountered. This information may be useful in discriminating site quality when both sites have similar species composition but differ in species abundances. Examples of diversity measures weighted with abundance are the Shannon Wiener and Simpson's Diversity indices (Magurran 2004).

Equation 1. The calculation of the AQI without abundance data:

$\mathbf{AQI_a = meanCC * \sqrt{Spp.}}$, in which:

meanCC = Mean coefficient of conservatism value for all species encountered per sampling effort (e.g., site, transect, plot)

$\sqrt{Spp.}$ = Square root transformation of the total number of species encountered at the site sampled.

Equation 2. The calculation of the AQI with abundance data:

$\mathbf{AQI_b = \sum [(n_i/N) * CC_i] * \sqrt{Spp.}}$, in which

n_i = The total number of individuals for species i

N = The total number of individuals for all species

CC_i = Coefficient of Conservatism for species i

$\sqrt{Spp.}$ = Square-root transformation of the total number of species encountered at the site sampled.

Terminology and Concepts

<u>Coefficient of Conservatism (CC)</u>. In calculating both versions of the AQI, each auchenorrhynchan species is assigned a value from 0 to 18, termed a coefficient of conservatism (CC). Lower values indicate taxa that lack affinity to prairie. Higher values reflect a species that tends to occur in prairies with minimal amounts of degradation (e.g., road construction, presence of exotic plants).

Each of these CC values is the sum of values for six individual life history traits: number of generations per year or voltinism, host plant specificity, overwintering preference, wing length, origin or geographic range, and habitat dependence (Table 2.1). Each of these traits ranges in value from 0 (non-conservative) to 3 (conservative). Exotic species were automatically given a value of 0 since they have not evolved with the native grassland habitat. Negative values were not assigned to these species since it is unknown if they replace native taxa over time or interfere in the recovery process of the native grassland community.

I assigned values to life history traits based on the following information: 1) scientific (Novotny 1994a, Panzer *et al*. 1995, Nickel and Achtzier 1999, Nickel and Remane 2002, Nickel 2003), 2) museum specimens, 3) personal communication from the following authorities on North American prairie Auchenorrhyncha - Andy Hamilton (Eastern Cereal and Oilseed Research Centre, Agriculture and Agri-Food Canada, Ottawa, ON), Charles Bartlett (University of Delaware, Department of Entomology and Wildlife Ecology), and Stephen Wilson (Central Missouri State University, Department of Biology), and 4) personal observations in the field.

Quantifying Life History Traits. Voltinism was quantified using the following scale: (0) more than two generations per year (Table 1), (1.50) 2 generations per year, and (3.00) only one generation per year. Data on voltinism was taken from Ossiannilsson (1978 – 1983), Hamilton (1983), Whitcomb *et al*. (1987a, b, 1988). Museum specimens and field observations were also used when information on voltinism was absent in the literature. It should be noted that voltinism is often strongly dependent upon latitude, whereby herbivorous insects with more northern distributions produce fewer generations a year because of the shorter growing season of their host plants (Nickel 2003). For example, *Aflexia rubranura*, a specialist on prairie dropseed (*Sporobolus heterolepis*) produces two generations a year in northern Illinois, but in the northern parts of the Midwest and Canada they only produce one generation (Hamilton 1995). As a consequence, managers may need to adjust some of these voltinism scores depending on where these insects are collected (see Appendix 1).

Host plant specificity was quantified based on the following scale: (0) polyphagous or feeding on non-prairie plants, (0.5) 2 families of prairie plants (e.g., grasses, sedges, forbs), (1.00) 1 family of prairie plants, (1.50) 2 or more genera of prairie plants, (2.00) 1 genus of prairie plants, (2.50) species within one genus of prairie plants, and (3.00) 1 species of prairie plants. Data on host plant preference were taken from the following sources: Breakey 1928, DeLong 1948, 1965, 1971, Blocker 1967, Genung and Mead 1969, Kopp and Yonke 1970, Blocker *et al*. 1972, Blocker and Reed 1976, Kramer 1976, Cwikla and Blocker 1981, Hamilton 1983, 1994, 1995, 1998, 2000, 2005, Whitcomb *et al* 1986, 1987, 1988, 1994, Whitcomb and Hicks 1988, Wilson *et al*. 1993, 1994, Bartlett and Dietz 2000, Bouchard *et al*. 2001, Panzer unpublished data.

Overwintering preference was quantified based on the following scale: (0) migrating and not overwintering locally, (1.00) non-prairie plants, (2.00) soil/duff (i.e., dead vegetation), (3.00) prairie forbs, grasses, and sedges. Data on overwintering was determined from host plant preferences, personal communication with Andy Hamilton (Eastern Cereal and Oilseed Research Centre, Agriculture and Agri-Food Canada, Ottawa, ON), and field observations.

Wing length was quantified as: (0) macropterous, (1.00) macropterous/brachypterous, (2.00) submacropterous, and (3.00) brachypterous. This trait was quantified from specimens sampled in the field. It should be noted that submacroptery can be difficult to assess, but it was standardized using Hamilton's (1995) definition, whereby the last 2 – 3 abdominal segments are not covered by the wings.

Origin was quantified as: (0) exotic, (1.50) native, occurring over a broad geographical range, and (3.00) restricted to a small geographical area. Data on origin was taken from Nielson (1961), Hamilton (1983, 1995, and 1997), and Whitcomb *et al*. (1994).

Habitat fidelity was quantified on the following scale: (0) remnant independent or adventives, (0.75) other-than-prairie associates or ecotonal, (1.50) prairie associates or found in prairie remnants, (2.25) remnant dependent species (more than one prairie ecotype), and (3.00) prairie-dependent species (one prairie ecotype). Data on habitat fidelity was taken from

Bouchard *et al*. 2001), Hamilton (1995, 2005), and Whitcomb *et al*. (1994). This trait was largely based on a hierarchical scheme developed by Panzer and colleagues (1997). When information on any of these traits was unavailable, I used field observations, and literature from closely related species in North America and Europe (Ossiannilsson 1978 – 1983, Nickel and Remane 2002, Nickel 2003). These traits were reviewed and approved by the following Auchenorrhyncha taxonomic authorities: Andy Hamilton (Eastern Cereal and Oilseed Research Centre, Agriculture and Agri-Food Canada, Ottawa, ON), Charles Bartlett (University of Delaware, Department of Entomology and Wildlife Ecology), Christopher Dietrich (University of Illinois at Urbana-Champaign, Illinois Natural History Survey), and Stephen Wilson (Central Missouri State University, Department of Biology).

Study Sites

Sampling was conducted on 35 sites within the eastern and central portions of the North American tallgrass prairie biome. Study sites extended from western Iowa (4) through south-central Wisconsin (9) to northwestern Missouri (2), and predominately Illinois (21) (Table 2). Study sites ranged in size from 4 to 15,000 ha, with most (18) falling within the 4 to 200 ha range, with a mean acreage of 847.70. Each site is known to support a major plant community which ranges along a hydrological gradient from hill prairie (gravel and loess) to wet-mesic prairie, with most (13) supporting wet and/or mesic prairie communities. These sites were ranked as high to mid quality sites.

Sampling Protocols used to examine robustness and sensitivity of the AQI

In determining if changes in sampling method (sweep net and modified leaf blower vacuum) and sampling design (transect and plots) effect AQI values (research Question 1), I sampled Auchenorrhyncha using a standard 38-cm diameter canvas net for 100 sweeps on three

5x5m plots and three 40m-linear transects and vacuum sampled for five minutes along three separate 40m-linear transects and on three separate 5x5m plots from five sites between the hours of 11:00 and 17:00, from 17 July to 22 August, 2005. A similar sampling protocol was used to determine if changes in sampling method vs. habitat effect AQI values (research question 2), with the exception that vacuum and sweep samples were taken from 10 wet-mesic prairies, six loess hill prairies, and three sand prairies and that sampling occurred between the hours of 11:00 and 18:00 from 16 June to 20 September 2004 - 2005. A variation in the sampling protocol used to test research question 2 was applied to test research question 3 (i.e., are both versions of the AQI sensitive to changes in sampling design and prairie community when using a vacuum), in that I vacuum sampled for five minutes on three 40m-linear transects and three separate 5x5m plots from seven wet-mesic, five loess prairies, and seven sand prairies between the hours of 11:00 and 18:00 from 16 June to 20 September, 2005. In determining whether both versions of the AQI were sensitive to changes in sampling method and time of year sampled (research question 4), I made three collecting trips in June, July, and August in 2008 between the hours of 11:00 and 18:00 on Des Plaines Conservation Area, Revis Hill Prairie, and Lost Mound Savanna Army Depot. On each of these sites I vacuum sampled for five minutes on three 20m-linear transects and conducted 100 sweeps on three separate transects.

In comparing sampling efficacy of sweep and vacuum sampling (research question 5) across prairie communities and within individual prairie communities (e.g., wet-mesic, loess/gravel hill, and sand) I collected Auchenorrhyncha from four wet-mesic, four hill prairies (three loess and one gavel), and two sand prairies (2). On each of these sites I collected Auchenorrhyncha by vacuum sampling for five minutes along four 40m-linear transects and conducted 100 sweeps from four separate 40m-linear transects between the hours of 11:00 and

18:00 from 16 June to 20 September 2004 – 2005. Lastly, in determining whether vacuum and sweep sampling produce stable AQI values on multiple prairie communities (e.g., hill, wet-mesic, and sand), within individual prairie communities, and on individuals sites (research question 6), I calculated both versions of the AQI from 10 sites used in addressing research question 5 (Table 2.2). Values were calculated from the first transect, then any new species detected during the second transect were added and AQI values were recalculated. This procedure was repeated for transects 3 and 4. Means and standard deviations were computed from species collected from 2, 3, and all 4 transects for each sampling technique and were compared. Similar calculations were generated for wet-mesic and hill prairies. However, means and standard deviations of both versions of the AQI were not calculated from sand prairies because only two prairies were sampled. Lastly, similar calculations were made in generating cumulative AQI curves for individual sites, except means and standard deviations were not computed.

For each sampling protocol used in testing these six research questions, I pooled the following samples into one sample for each site: vacuum/transects, sweep/transects, vacuum/plots, and sweep/plots. Samples were pooled to examine the effects of sampling methods, sampling designs, seasonality, and a combination of these factors on both versions of the AQI at the landscape level. However, I did not pool samples collected from transects from individual sites when testing seasonality vs. sampling method (research question 4) and sampling intensity vs. sampling method (research question 6). Transects and plots were sampled on homogeneous vegetation to reduce the influence of plant variation within each site which could affect Auchenorrhyncha species composition. Transects and plots were placed at least 5m apart to reduce the chance of pseudoreplication. Transects were parallel to one another and perpendicular to a 50m-baseline for research questions 1, 2, 3, 5, and 6 and their placement was determined using a random numbers table. The placement of plots was determined by throwing a Frisbee in a random direction. GPS coordinates were recorded for each transect and plot.

Data Analysis

Kolmogorov-Smirnov test was used to test for normality and the Levene's test was used to examine if the data had equal variance. Two-way Analysis of Variance followed by Holms-Sidak *post-hoc* tests were performed to determine if significant differences ($P \leq 0.05$) occurred between sampling design and sampling method (research question 1), sampling method and prairie community (research question 2), sampling design and prairie community (research question 3), and seasonality or time of year and sampling method (research question 4). Holms-Sidak *post hoc* test was selected over more conventional *post hoc* tests, such as Tukey's because it is a more conservative *post hoc* test, requiring a lower alpha level (Ott and Longnecker 2001). Statistical tests were performed using Sigma Stat version 3.1. Means ± SE are reported.

Species accumulation curves were generated to compare the efficacy of sweep and vacuum across all prairies and on individual prairie communities (research question 5) using the statistical software EstimateS 8.0.0 (2006). When a species accumulation curve approaches an asymptote for sweeping or vacuuming then sampling effort has been sufficient in collecting most of the species using either method (Colwell and Coddington 1994). Species observed (actual number of species found) and Chao2 non-parametric species richness estimator were selected from EstimateS. Chao2 was selected over other estimators because it produces accurate predictions of species richness from small numbers of samples and is insensitive to moderate levels of patchiness found in the species abundance distributions (Magurran 2004).

Processing of Auchenorrhyncha

After each vacuum and sweeping sampling event, specimens were transferred into Photo Tactic Optimal Insect Extractors (PTOIE) for approximately 45min – 1 hour and stored in 95% ethanol. All adult Auchenorrhyncha were identified to species, individuals of each species were counted, and specimens were deposited at the Illinois Natural History Survey Entomology Collection. Each species was given a coefficient of conservatism integer value based on the methods described above (Table 2.1). Both versions of the AQI (Equation 1, 2) were calculated for each of these tests. Auchenorrhyncha nomenclature followed DeLong (1948), Wilson and McPherson (1980), and Whitcomb and Hicks (1988).

RESULTS

Overall Diversity and Conservatism. In total 274 species were collected. Most of these were cicadellids (176), followed by delphacids (42), membracids (11), cercopoids (12), small groups of Fulgoromorpha (23) and cicadas (3; Figure 2.1). Coefficient of conservatism values were assigned to each of these taxa (Appendix A.1). The frequency distribution of coefficients of conservatism (0 – 18) is right-skewed indicating that species are adventives and moderately conservative (Figure 2.2).

Cicadellids (176 species) and delphacids (42 species) represented ~80% of the diversity found on these prairie remnants, displaying a wide range of conservatism to native grasslands (Figures 2.3a, b). Seventy-five out of the 176 (42%) cicadellids encountered were adventives and habitat-generalist species with low CC values ranging from 0 – 6 (Figure 2.3a), whereas 22 delphacid species (53%) were conservative with CC values ranging 11 - 18 (Figure 2.3b). Conversely, membracids and cercopoids (Figures 2.3c, d) reflect the lower end of the conservatism spectrum, with only a few exceptionally conservative species being represented in each group (Appendix 1). Small groups of non-delphacid Fulgoromorpha, although not contributing as much to Auchenorrhyncha diversity as cicadellids and delphacids, are highly conservative in grassland ecosystems (Figure 2.3e). Of these groups the family Caliscelidae represents some of the most conservative species in this biome, the least speciose Auchenorrhyncha group encountered are cicadas.

Robustness and Sensitivity of the AQI

Vacuum sampling yielded significantly higher values than sweep sampling for both versions of the AQI (research question 1; Table 2.3), but AQI values were insensitive to sampling design and there was also no significant interaction between sampling method and sampling design. Vacuum sampling also yielded significantly higher values than sweeping for both versions of the AQI (research question 2; Table 2.3), but the AQI was insensitive to habitat and there was also no significant interaction between sampling method and habitat. However, both versions of the AQI were marginally significant on loess hill prairies, followed by wet-mesic prairies, than sand prairies (research question 3; Table 2.3), but the AQI was insensitive to sampling design and there was no significant interaction between sampling design and habitat. Vacuum sampling also yielded significantly higher values than sweep sampling for both versions of the AQI (research question 4; Table 2.3), but the AQI was insensitive to growing season and there was also no significant interaction between sampling method and growing season.

On individual sites, vacuum sampling yielded significantly higher values than sweep sampling for both versions of the AQI (research question 4; Table 2.3) at Lost Mound and Des Plaines, but not at Revis hill prairie. $AQI_{w/N}$ values were significantly higher in June than in July at Des Plaines and were also significantly higher in August than in June at Lost Mound but were not significantly different at Revis, whereas $AQI_{w/outN}$ values were only significantly higher in

June than July but not in August at Lost Mound (Table 2.3). Lastly, no significant interactions were found between sampling method and seasonality for both versions of the AQI on Lost Mound, Des Plaines, and Revis prairies.

In comparing sampling efficiency of vacuum and sweeping (research question 5) from on all 10 sites, I found that vacuum sampling yielded 108 species, sweeping yielded 86 species, and chao2 predicted an additional 17 species from vacuuming and an additional 53 species when using a sweep net. Moreover, species accumulation curves showed that SOBs (species observed) approached an asymptote and converged with Chao2 (predicted number of species; Figure 2.4a) whereas SOBs did not approach an asymptote or converge with Chao2 when using a sweep net (Figure 2.4b).

A similar trend was observed at the community level, for instance, vacuum sampling collected 74 species on wet prairies, 65 species on hill prairies, and 36 species on sand prairies whereas sweeping yielded 53 species on hill prairies, 50 species on wet prairies, and 24 species on sand prairies. Chao2 predicted an additional 9 species from hill prairies, 14 species from wet prairies, and 19 species from sand prairies when using a vacuum whereas Chao2 derived from sweep sampling predicted an additional 18 species from hill prairies, 28 species from wet-mesic prairies, and 10 species from sand prairies. Moreover, species accumulation curves showed SOBs approaching an asymptote and converging with Chao2 on hill prairies (Figure 2.5a), and wet-mesic prairies (Figure 2.5b), but not on sand prairies (Figure 2.5c) when using a vacuum. However, SOBs does not approach an asymptote or converge with Chao2 on hill prairies, wet-mesic, and sand prairies when using a sweep net (Figure 2.6a, b, c).

In examining the effects of sampling intensity on both versions of the AQI (research question 6) from all 10 sites, I found that AQI values stabilized after the fourth transect was added (Figures 2.7a, b), but a higher degree of variation around the mean was observed for $AQI_{w/N}$ values (Figure 2.7a) compared to $AQI_{w/outN}$ values (Figure 2.7b). Sweep sampling also produced AQI values that stabilize after the fourth transect was added, but these values were lower than AQI values calculated from vacuum sampling (Figures 2.8a, b). $AQI_{w/N}$ values calculated from sweep sampling also displayed a higher degree of variability (Figure 2.8a) than $AQI_{w/outN}$ (Figure 2.8b).

A similar trend was observed in examining the effects of sampling intensity on both versions of the AQI at the community level (i.e., hill prairies and wet-mesic prairies). For instance, vacuum sampling yielded AQI values that stabilized after the fourth transect was added for wet-mesic (Figures 2.9a, b) and hill prairies (Figures 2.10a, b), but $AQI_{w/N}$ values displayed a greater degree of variation around the mean (Figures 2.9a, 2.10a) than $AQI_{w/outN}$ values (Figures 2.9a, 2.10b). Conversely, sweep sampling on wet-mesic prairies yielded $AQI_{w/N}$ values that stabilize after the second transect was added (Figure 2.11a), but $AQI_{w/outN}$ values calculated from wet prairies (Figure 2.11b) and hill prairies (Figures 2.12a, b) did not stabilize. $AQI_{w/N}$ values calculated from sweep sampling (Figures 2.11a, 2.12a) also showed higher variation around the mean than $AQI_{w/outN}$ values (Figures 2.11b, 2.12b). On individual sites, I observed variation for both versions of the AQI when vacuuming and sweeping on wet-mesic and sand prairies (Figures 2.14, 2.15, 2.17, 2.18), but no variation was observed on hill prairies (Figures 2.13, 2.16). More specifically, on Oquawka sand prairie (Figure 2.14b), $AQI_{w/N}$ values derived from vacuum sampling decreased after the third transect was added as a result of collecting more non-conservative species, such as typhlocybinaes. But, these AQI values increased after the fourth transect was added because of collecting more conservative species, such as *Flexamia* spp (Figure 2.13). $AQI_{w/N}$ values gradually increased on Hayden prairie and 12-mile prairie (Figure

2.15b) as a result of encountered more conservative Auchenorrhyncha individuals (e.g., *Flexamia* spp.), but $AQI_{w/outN}$ values derived from vacuum sampling only increased on Hayden prairie (Figure 2.15a).

For sweep sampling, $AQI_{w/N}$ values decreased on Cayler prairie (Figure 2.17) after the second transect due to an influx of more non-conservative species being encountered, such as typhlocybines. AQI $_{w/outN}$ values also decreased on Oquawka prairie after the second and third transects were added as a result of encountering more non-conservative species (e.g., typhlocybine spp.), but increased after the fourth transect (Figure 2.18) after encountering more conservative species (e.g., *Flexamia* spp.). However, $AQI_{w/outN}$ values continued to increase at 12-mile prairie, Cayler prairie, and Hayden suggesting that additional conservative auchenorrhynchan species are being encountered after sampling from four transects (Figure 2.17).

DISCUSSION

Robustness and Sensitivity of the AQI

In examining the effects of sampling design vs. sampling method (research question 1), I found that both versions of the AQI are more sensitive to differences in sampling method (vacuum vs. sweep) than sampling design (transect vs. plot), whereby vacuum sampling yielded higher AQI values than those obtained by sweeping. A similar pattern was found when examining the sensitivity of both versions of the AQI to sampling method (vacuum and sweep) and prairie community (research question 2). These results support my prediction that vacuum sampling collects more conservative Auchenorrhyncha than sweeping regardless of the prairie being sampled. Two explanations may explain these results. First, literature has shown that Auchenorrhyncha are vertically stratified on plants, with more long-winged, adventives species (i.e., non-conservative) occurring near the tops of plants and more conservative species inhabiting the base of plants (Novotny 1991, Stewart 2002, Nickel 2003, and Nickel and Hilderbrant 2003). Second, studies have shown that vacuum sampling consistently yields more auchenorrynchan species and a higher proportion of conservative auchenorrhynchan species than sweeping (Wilson *et al.* 1993, Stewart 2002, Nickel and Hidlebrandt 2003).

Both versions of the AQI values were also insensitive to sampling design (plot vs. transect) and prairie community sampled (research question 3). However, $\mathrm{AQI}_{\mathrm{w/outN}}$ values were marginally higher on loess prairies, which had more conservative species than wet/wet-mesic sand prairies. Hamilton (1995, 2005) has found similar results where hill prairies harbor a disproportionally greater number of conservative Auchenorrhyncha species compared to other prairie communities. Because loess hill prairies have remained relatively inaccessible (i.e. occur on steep slopes) and have not been plowed or converted to agriculture (Robertson *et al.* 1995) they remain some the most numerous of all high quality prairie communities in Illinois (Taft *et al.* 2009), and thus may support more conservative auchenorrhynchan species than other Illinois prairies.

Both versions of the AQI were also insensitive to sampling method (sweep and vacuum) and seasonality (research question 4) over a wide geographical scale (i.e., multiple sites). These results support my prediction that vacuum sampling would yield higher AQI values than those obtained by sweeping regardless of time of year. These data suggest that one visit in late July or August may be sufficient in calculating reliable AQI values. However, sampling later in the growing season may yield a disproportionate number of conservative Auchenorrhyncha species (Blocker *et al.* 1972) that could inflate AQI values. To correct for this potential bias an additional visit in June may be required, yielding adventives species that are often more dominant early in the growing season (DeLong 1948, Hamilton and Whitcomb 2010). By including two visits in June and in late July/August a more accurate assessment of prairie integrity can be generated.

However, I observed variation on individual sites when examining the effects of sampling method and seasonality on both versions of the AQI (research question 4 on individual sites), with higher AQI values observed later in the growing at Lost Mound, but not Des Plaines and Revis hill prairie. This did not support my prediction that AQI values will only be higher when vacuum sampling later in the growing season. In the case of Lost Mound, low $AQI_{w/N}$ values observed early in the growing season may be attributed to the influx of exotic and adventives Auchenorrhyncha species that tend to synchronize with their non-native and/or weedy host plants that are dominant early in the growing season (Denno 1994). This influx may have affected the species richness component of the index, resulting in higher AQI values. The higher $AQI_{w/N}$ values in mid and late summer may be attributed to more individuals of conservative auchenorrhynchan species reaching developmental maturity as their native, warm season perennial host plants begin to dominate this prairie (Blocker 1976). These data also suggest that sampling on Lost Mound should occur in late June and late July/August. Collecting from both of these months would yield conservative and adventives Auchenorrhyncha, and thus generate AQI values that reflect a more accurate assessment of prairie integrity.

In the case of Des Plaines Conservation Area, significantly higher $AQI_{w/N}$ values were observed early in the growing season, but not later in the season, whereas $AQI_{w/outN}$ values did not show significant differences throughout the growing season. The peak in $AQI_{w/N}$ values early in the season may be attributed to a large population of *Athysanella balli* that was found at Des Plaines. This highly conservative leafhopper species (Appendix A.1) was the most dominant of all auchenorrhynchan species collected at Des Plaines. Its abundance gradually decreased throughout the season, which may have been caused by an increase in predators and parasitoids (Nickel 2003). These data suggest that sampling later in the growing season at Des Plaines may reduce the possibility of having a disproportional number of conservative Auchenorrhyncha individuals yielding inflated $AQI_{w/N}$ values.

Unlike Des Plaines or Lost Mound, Revis prairie did not show any significant difference in AQI values regardless of time of visit or sampling method, indicating that both indices are insensitive to seasonality and sampling technique on this site. These results also suggest that Revis has higher overall integrity than Lost Mound and Des Plaines because it retains relatively high integrity throughout the growing season. Since these other sites had influxes of adventives species at certain times of year, for example in early summer, that caused their integrity to drop temporarily, then perhaps they are more vulnerable to permanent degradation.

In comparing sampling efficiency (research question 5) between vacuum and sweep sampling, I found that vacuum sampling collects more auchenorrhynchan species than sweeping on hill prairies, wet-mesic prairies, and sand prairies. These results support my prediction that vacuum sampling will not only yield more species than sweeping, but a greater proportion of the species that actually inhabit a site. This agrees with results from other studies where insects approach an asymptote as vacuum sampling increases (Henderson and Whitaker 1977, Wilson *et al*. 1993).

These results (Figures 2.5, 2.6) also show that hill prairies supported more Auchenorrhyncha species and abundance, followed by wet-mesic, then sand prairies. However, the low diversity observed on sand prairies may be a sampling artifact as only two sites were sampled. In addition to the differences in diversity observed on these prairie communities, I also found differences in species composition, particularly conservative species. For instance, a unique Auchenorrhyncha fauna was found on hill prairies which include *Athysanella incongrua*, *Laevicephalus minimus*, and *Bruchomorpha jocosa*. Wet-mesic prairies also harbored unique Auchenorrhyncha fauna, which included *Flexamia inflata, Neohecalus magnificus*, and *Aphelonema simplex*. Sand prairies were also found to support unique assemblages of Auchenorrhyncha, such as *Philaenarcys killa*, *Flexamia grammica*, and *Scaphytopius abbreviatus*. These unique assemblages of Auchenorrhyncha have been well documented in literature (Whitcomb et al. 1987, Hamilton 2004, 2005, Hamilton and Whitcomb 2010). Thus, the presence of these species indicates that Auchenorrhyncha integrity is high on these hill prairies, wet-mesic prairies, and sand prairies and efforts to maintain and conserve Auchenorrhyncha integrity should focus on protecting these species.

In examining sampling intensity (research question 6) on both versions of the AQI when using a vacuum, I observed that AQI values stabilize after the fourth transect was added and these values were consistently higher compared to values calculated from sweep sampling. These data support my prediction that AQI values calculated from vacuum sampling will stabilize and will be higher than values computed from sweep sampling. The lower AQI values calculated from sweeping may be the result of the sweep net's inability to maneuver through the dense vegetation, resulting in collecting species near the top of the vegetation that are often dominated by adventives species (DeLong 1965, Wilson *et al*. 1993, Nickel and Hilderbrandt 2003). These data also show that $AQI_{w/N}$ values, based on either vacuum or sweep data, showed more variation across prairies, within prairies, and at individual sites than $AQI_{w/outN}$ values (Figures 2.7 – 2.18). This pattern was strongly exhibited on wet-mesic and sand prairies, indicating that Auchenorrhyncha abundance is highly variable from one transect to the next on these prairie communities. $\mathrm{AQI}_{\mathrm{w/outN}}$ values were also highly variable on wet-mesic prairies (Figure 2.18b) when using a vacuum or sweep net with some sites requiring more than four transects, other sites requiring three transects, and other sites only requiring two transects in order to collect a sufficient number of Auchenorrhyncha in the calculation of stable AQI values.

These results indicate that land managers and researchers could calculate stable AQI values from four transects at the landscape level when using a vacuum. However, additional sampling is needed when calculating stable AQI values on individual sites, such as wet-mesic prairies. To ensure that sufficient sampling has been implemented in generating stable AQI values on wet-mesic prairies it is recommended that AQI profiles are generated. These profiles display the total number of transects needed in calculating stabile AQI values.

Conclusions and Implications for Use

Since both permutations of the AQI displayed similar results, this then begs the question: What is the most appropriate permutation to use?

The $\mathrm{AQI}_{\mathrm{w/outN}}$ is preferred over the $\mathrm{AQI}_{\mathrm{w/N}}$ for two main reasons: 1) calculating the AQI with species abundance will add an additional step by counting individuals and 2) dramatic population fluctuations associated with Auchenorrhyncha from year to year in response to short term changes in climate and land management will affect species abundance (Waloff 1980, Panzer 2002). However, there can be applications where weighted indices (e.g., $\mathrm{AQI}_{\mathrm{w/N}}$) out-perform non-weighted indices (Bourdaghs *et al*. 2006). For example, Taft *et al*. (1997) and Polling *et al*. (2003) identified a situation where abundance weighting of the Floristic Quality Index improved its performance in a tallgrass prairie restoration in Ohio. Successional shifts from colonizing grasses to later successional grasses were detected only by using weighted indices because the abundance distribution of the community was changing over time but species composition was not. Results presented by Polling *et al*. (2003) suggest that weighted indices by abundance may be useful in detecting differences in quality when species composition is similar.

What is the optimal season for sampling?

Two visits may be sufficient in collecting most Auchenorrhyncha and generating an accurate assessment of prairie integrity. One visit should occur early in the growing season (mid to late June) to sample adventives species and the other visit should occur later in the summer (July and August) to collect conservative Auchenorrhyncha genera, such as *Paraphlepsius*, *Pendarus*, and *Prairiana*). Moreover, collecting later in the summer will allow these large-sized, conservative genera to reach development maturity, which is needed for accurate identifications (DeLong 1948). But, if land managers have financial and time constraints then one visit later in the growing (mid- to late July) may be sufficient in calculating relatively reliable and repeatable AQI values.

What is the optimal sampling strategy?

Overall vacuum/transects consistently yielded more Auchenorrhyncha than sweeps/plots. This sampling strategy can be implemented on hill, wet-mesic, and sand prairies and should be carried out in June and late July or August to capture both conservative and non-conservative Auchenorrhyncha (Blocker *et al*. 1972). Only four transects are needed in generating stable AQI values using a vacuum at the landscape level, but additional transects may be needed on individual wet-mesic prairies to calculate stable AQI values. To accurately determine the number of transects needed in calculating stable AQI values on wet-mesic prairies when using a vacuum or sweep net land managers and researchers may need to generate AQI profiles.

Further Considerations

Further research is needed to calibrate AQI values against a gradient of anthropogenic disturbance, such as the INAI grading criteria (Karr and Kimberling 2003). This calibration could be in the form of defining different levels of condition such as ‘good’, ‘fair’, and ‘poor’ that could signal managers to perform some sort of action (Bourgdahs *et al*. 2006). Once these different levels of condition are established federal, state, and local agencies can use the AQI as an additional assessment tool to establish conservation priority areas, determine the effects of prairie management practices, and evaluate the success of restoration activities (Davis and Simon 1995, Karr and Kimberling 2003). AQI should also be used in combination with other measures of diversity, based on other groups of prairie organisms (e.g. plants, bird, butterflies, wasps), which may provide land managers with a more complete picture of prairie integrity.

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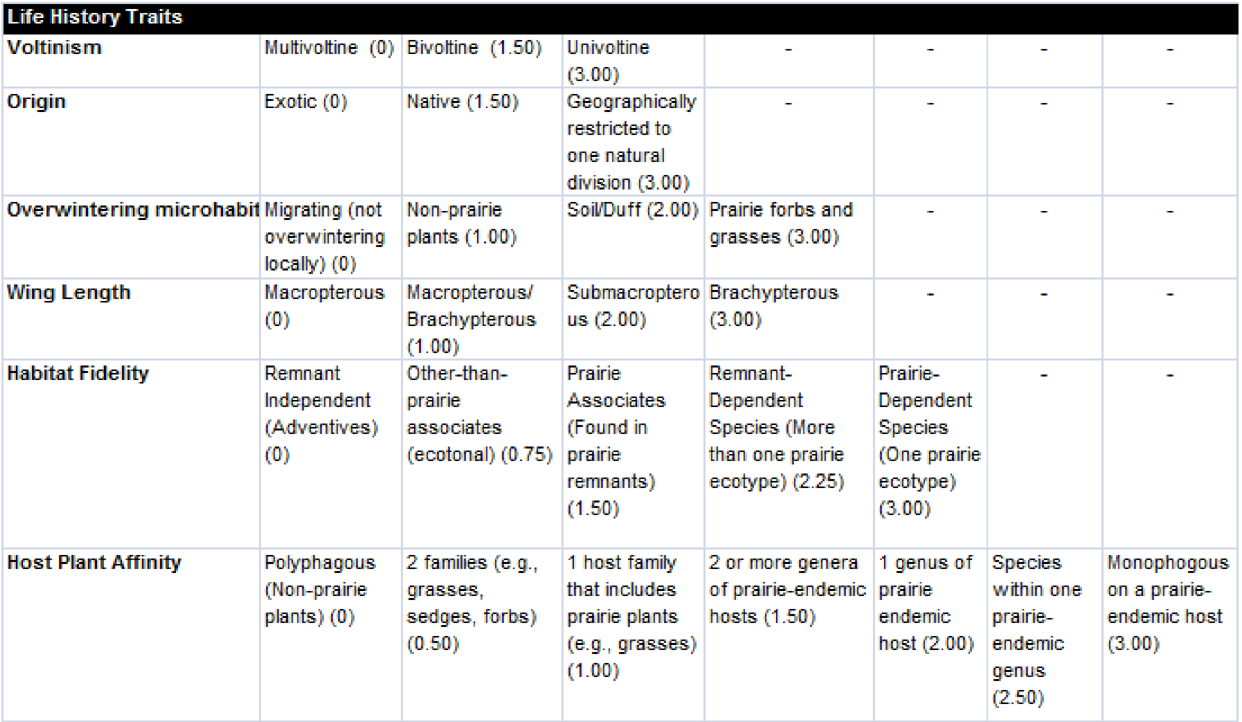
TABLES AND FIGURES

Table 2.1 Life history traits of Auchenorryncha used in calculating coefficient of conservatism values



Life History Traits							
Voltinism	Multivoltine (0)	Bivoltine (1.50)	Univoltine (3.00)	-	-	-	-
Origin	Exotic (0)	Native (1.50)	Geographically restricted to one natural division (3.00)	-	-	-	-
Overwintering microhabit	Migrating (not overwintering locally) (0)	Non-prairie plants (1.00)	Soil/Duff (2.00)	Prairie forbs and grasses (3.00)	-	-	-
Wing Length	Macropterous (0)	Macropterous/ Brachypterous (1.00)	Submacropterous (2.00)	Brachypterous (3.00)	-	-	-
Habitat Fidelity	Remnant Independent (Adventives) (0)	Other-than-prairie associates (ecotonal) (0.75)	Prairie Associates (Found in prairie remnants) (1.50)	Remnant-Dependent Species (More than one prairie ecotype) (2.25)	Prairie-Dependent Species (One prairie ecotype) (3.00)	-	-
Host Plant Affinity	Polyphagous (Non-prairie plants) (0)	2 families (e.g., grasses, sedges, forbs) (0.50)	1 host family that includes prairie plants (e.g., grasses) (1.00)	2 or more genera of prairie-endemic hosts (1.50)	1 genus of prairie endemic host (2.00)	Species within one prairie-endemic genus (2.50)	Monophogous on a prairie-endemic host (3.00)

Table 2.2 Distribution of 35 sites visited in the Midwestern USA. Research Questions 1 = Sampling Design x Sampling Methods; 2 = Sampling Design x Prairie Community, 3 = Sampling Method x Prairie Community, 4 = Seasonality x Sampling Method, 5 = Sampling Efficiency, 6 = Sampling Intensity

	Sites	County	State	Acres	Years Visited	Prairie Community	Research Questions
1	12mile	Marion	IL	94	2004	Wet/Mesic	3. 5, 6
2	Fults	Monroe	IL	10	2004, 2005	Loess	1, 2, 3, 5, 6
3	Chloe Marsh	Mercer	MO	15	2004	Wet/Mesic	3, 5, 6
4	Little Tarkio	Holt	MO	129	2004	Loess	3, 5, 6
5	Sylvan-Runkel	Monona	IA	330	2004	Loess	3
6	Overlook Prairie	Monona	IA	220	2004	Loess	3
7	Cayler	Dickinson	IA	160	2004	Wet/Mesic	3, 5, 6
8	Hayden	Howard	IA	240	2004	Wet/mesic	3, 5, 6
9	Kettle Moraine	Waukesha	WI	250	2004	Wet/mesic	3
10	Oquawka	Henderson	IL	200	2004	Sand	3, 5, 6
11	Snyder Prairie	Cass	IL	5	2004	Loess	3, 5, 6
12	Harlem Hills	Winnebago	IL	20	2004	Loess	3, 5, 6
13	Midewin	Will	IL	15,000	2004, 2005	Wet/Mesic	1, 2, 3
14	Gooselake Prairie	Grundy	IL	600	2005	Wet/Mesic	1, 2, 3
15	Illinois Beach	Lake	IL	300	2004	Sand	3, 5, 6
16	Fountain Creek	Green Lake	WI	252	2004	Wet/Mesic	3
17	Faville Prairie	Jefferson	WI	60	2005	Wet/Mesic	1, 2, 3
18	Scuppernong Prairie	Waukesha	WI	2013	2005	Wet/Mesic	2
19	Rock River Prairie	Rock	WI	37	2005	Loess	2
20	Hardscrabble Prairie	Lafayette	WI	160	2005	Loess	2
21	Muralt Prairie	Green	WI	10	2005	Loess	2
22	Oliver Prairie	Green	WI	4	2005	Loess	2
23	Avoca Prairie	Iowa	WI	5,743	2005	Wet/Mesic	2
24	Ayers Prairie	Carroll	IL	109	2005	Sand	2
25	Lost Mound Prairie A	Jo Daviess	IL	NA	2005	Sand	4
26	Lost Mound Prairie B	Jo Daviess	IL	NA	2005	Sand	4
27	Lost Mound Prairie	Jo Daviess	IL	NA	2005	Sand	3, 4
28	Longbranch Prairie	Mason	IL	93	2005	Sand	2
29	Sand Prairie Scrub Oak	Mason	IL	1460	2005	Sand	2
30	Matanzas Prairie	Mason	IL	28	2005	Wet/Mesic	2
31	Revis Hill Prairie	Mason	IL	417	2005, 2008	Loess	2, 4
32	Leeville Prairie	Kankakee	IL	15	2005	Sand	2

33	Des Plaines A	Will	IL	NA	2005	Wet/Mesic	1,3
34	Des Plaines B	Will	IL	NA	2008	Wet/Mesic	4
35	Lost Mound C	Jo Daviess	IL	NA	2008	Sand	4

Table 2.3 Means and standard errors of all research questions addressed in examining the robustness and sensitivity of both versions of the AQI (i.e., AQI$_{w/N}$ and AQI$_{w/outN}$). Variables in bold with different letters indicate significant differences among sampling methods, sampling designs, prairie communities, and seasonality or a combination of these variables (e.g., sampling method and seasonality), $p < 0.01$ (Holm-Sidak test). Abbreviations include vac (i.e. vacuum) and NA (not applicable).

Research Questions	**AQI$_{w/N}$**	**F value**	**P value**	**AQI$_{w/out}$**	**F value**	**P value**
Research Question 1: (Sampling Design vs. Sampling Method)	**Vac 38.67 ± 3.70^{a} > Sweep 18.24 ± 3.70^{b}**	**15.24**	**P <0.001**	**Vac 33.58 ± 3.09^{a} > Sweep 18.94 ± 3.00^{b}**	**11.25**	**P < 0.001**
Research Question 2: (Sampling Method vs. Prairie Community)	**Vac 36.15 ± 3.38^{a} > Sweep 26.26 ± 3.38^{b}**	**4.29**	**P = 0.05**	**Vac 35.68 ± 2.68^{a} > Sweep 24.63 ± 2.09^{b}**	**8.48**	**P < 0.001**
Research Question 3: (Sampling Design vs. Prairie Community)	Hill 40.69 ± 1.87 : Wet-Mesic 39.24 ± 6.15 : Sand 35.77 ± 1.41	0.25	P = 0.78	Hill 38.67 ± 47 2.60 : Wet-Mesic 33.40 ± 2.36 : Sand 29.41 ± 2.60	3.17	P = 0.06
Research Question 4: (Seasonality vs. Sampling Method - Landscape Level)	**Vac 44.85 ± 3.25^{a} > Sweep 26.63 ± 3.25^{b}**	**15.73**	**P < 0.001**	**Vac 38.92 ± 2.76^{a} > Sweep 26.57 ± 2.85^{b}**	**10.01**	**P = 0.01**
Research Question 4: (Individual Sites):						
Des Plaines (Sampling Method)	**Vac 33.07 ± 2.27^{a} > Sweep 15.45 ± 2.27^{b}**	**30.08**	**P < 0.001**	**Vac 23.46 ± 1.79^{a} > Sweep 11.86 ± 1.79^{b}**	**21.17**	**P < 0.001**
Lost Mound (Sampling Method)	**Vac 33.33 ± 1.67^{a} > Sweep 16.33 ± 1.08^{b}**	**51.83**	**P < 0.001**	**Vac 33.66 ± 1.16^{a} > Sweep 20.33 ± 1.22^{b}**	**65.54**	**P < 0.001**
Des Planes (Seasonality)	**June 45.34 ± 3.94^{a} > July 26.26 ± 4.69^{b} > August 27.59 ± 2.27^{a}**	**10.14**	**P < 0.001**	June 29.32 ± 3.66: July 21.88 ± 0.69 : August 19.26 ± 0.94	1.416	P = 0.280
Lost Mound (Seasonality)	**August 36.96 ± 2.89^{a} > July 35.61 ± 5.15^{a} > June 27.38 ± 2.46^{b}**	**6.19**	**P = 0.01**	**June 37.00 ± 1.07^{a} > July 30.44 ± 3.59^{b} > August 33.55 ± 2.64^{a}**	**5.62**	**P = 0.02**

Figure 2.1 A total of 274 Auchenorrhyncha species, represented by 9 families and 1 superfamily that were collected on 35 tallgrass prairie remnants visited in 2004, 2005, 2006, and 2008 from Illinois, Missouri, Iowa, and Wisconsin

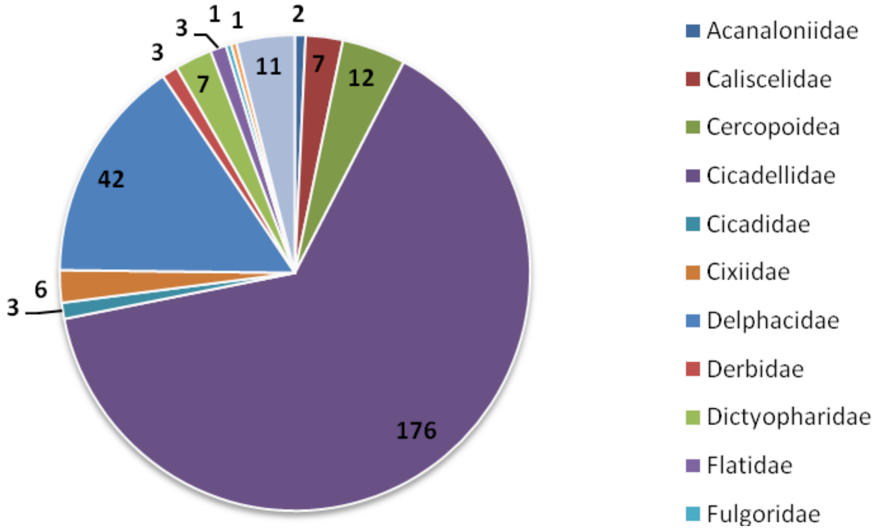


Figure 2.2 Total number of auchenorrhynchan species for each coefficient of conservatism value, with values ranging from 0 – 18.

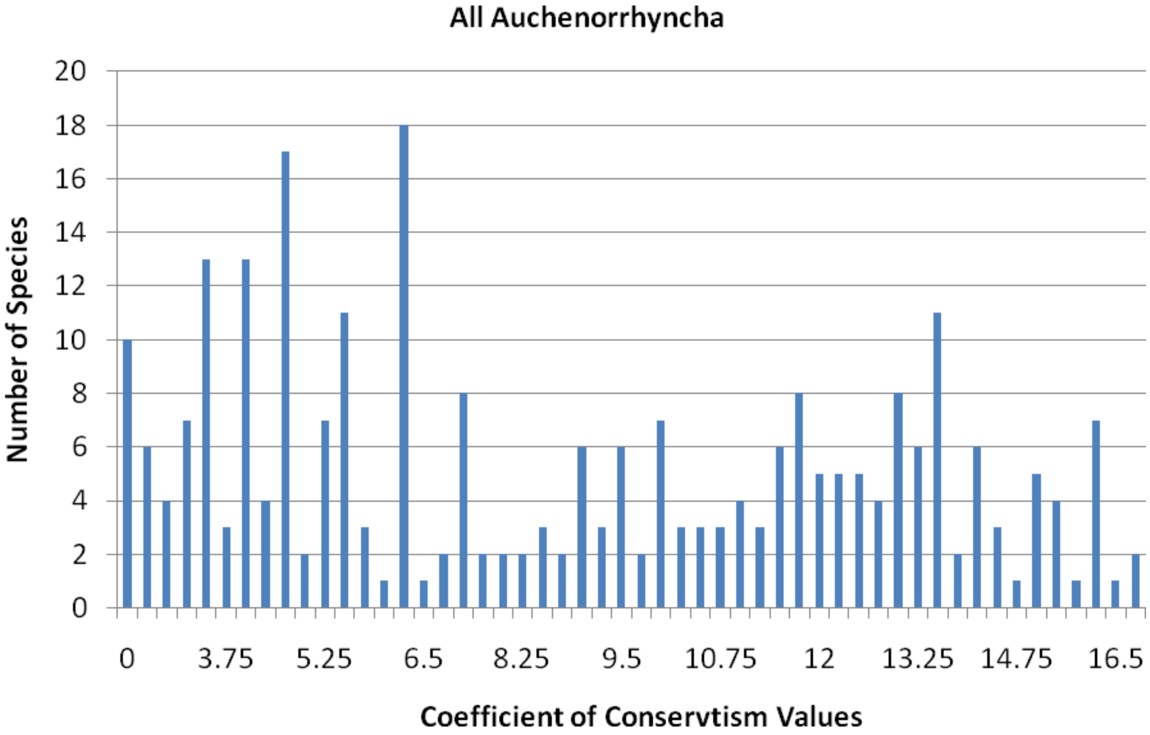


Figure 2.3 Total number of A) Cicadellidae (leafhoppers), B) Delphacidae (Planthoppers), C) Membracidae (Treehoppers), D) Cercopoidea (Spittlebugs), and E) Fulgoromorpha (other small groups of planthoppers) species for each coefficient of conservatism value, with values ranging from 0 – 18

A

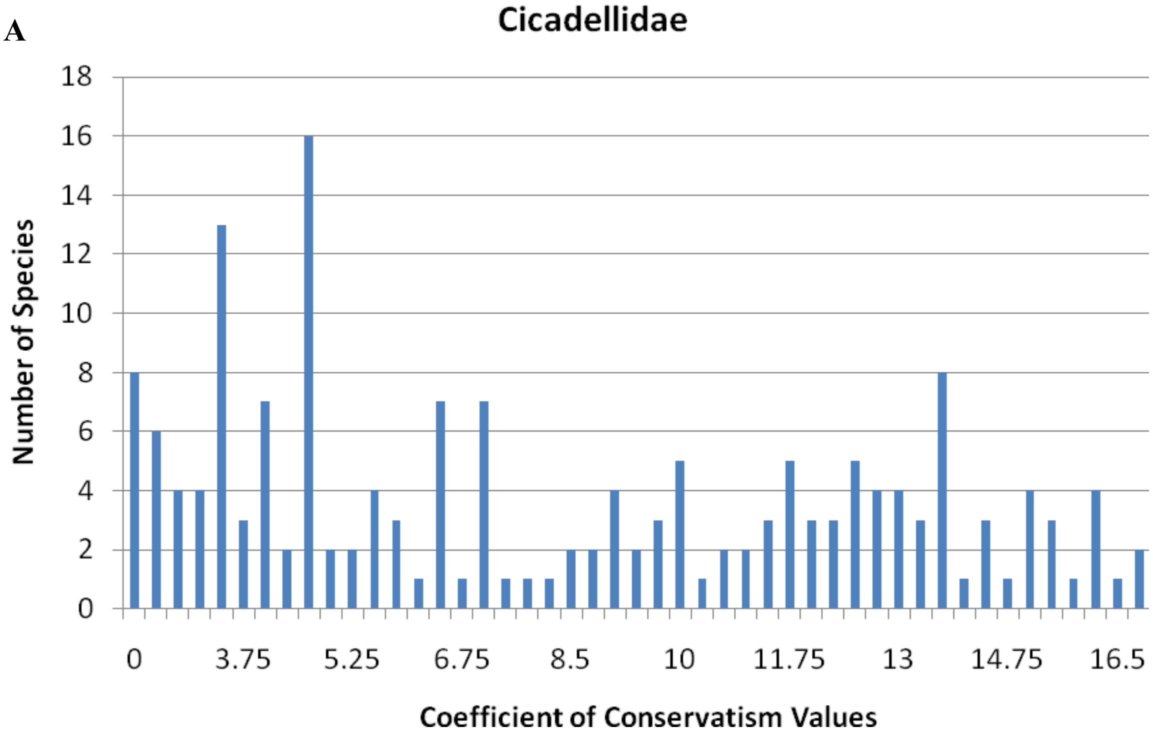


Figure 2.3 (cont.)

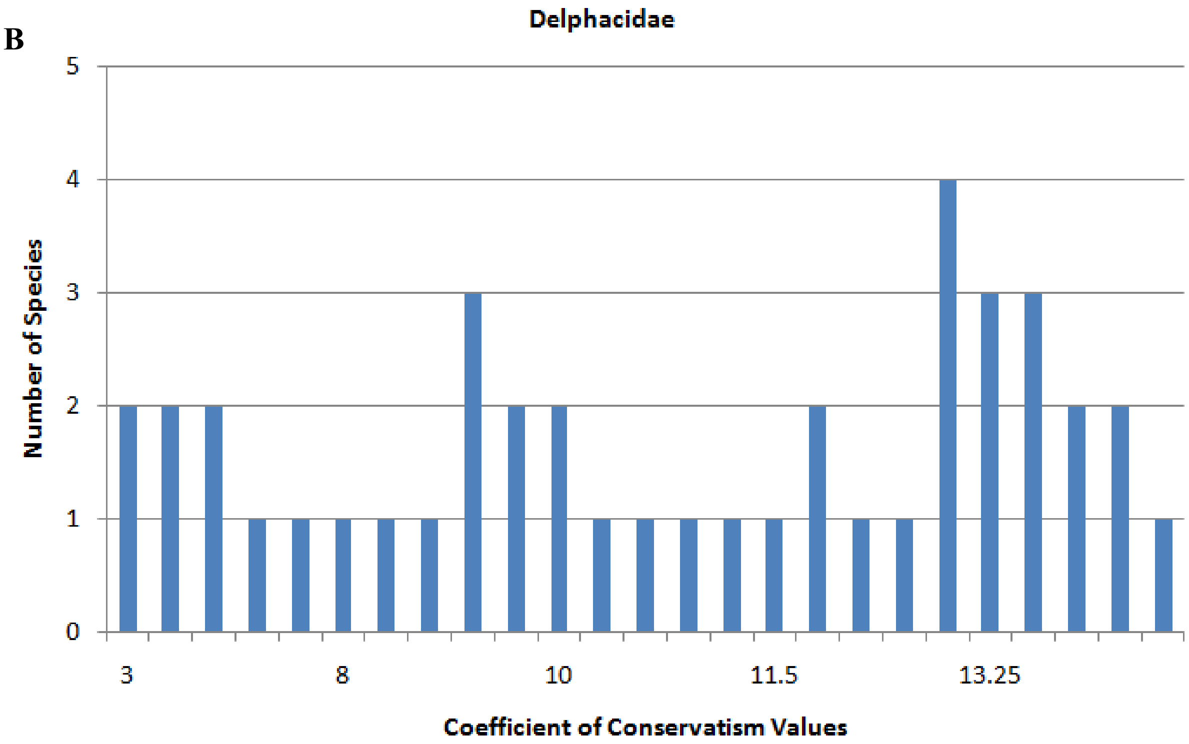


Figure 2.3 (cont.)

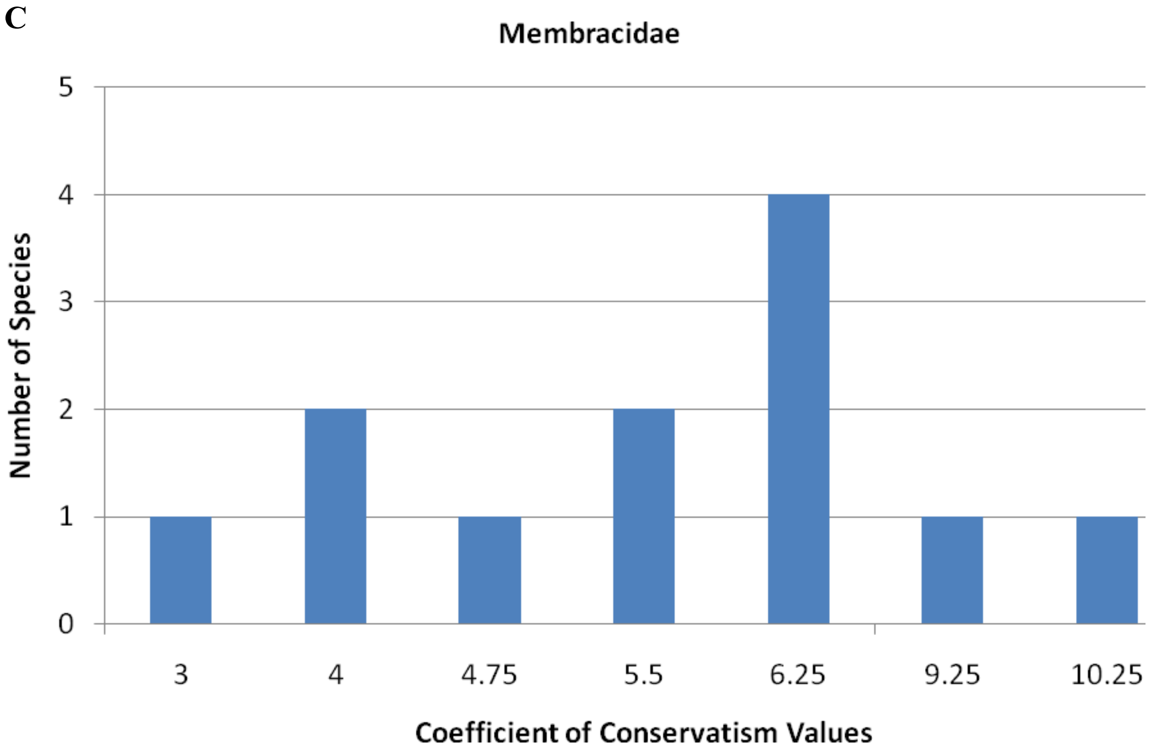


Figure 2.3 (cont.)

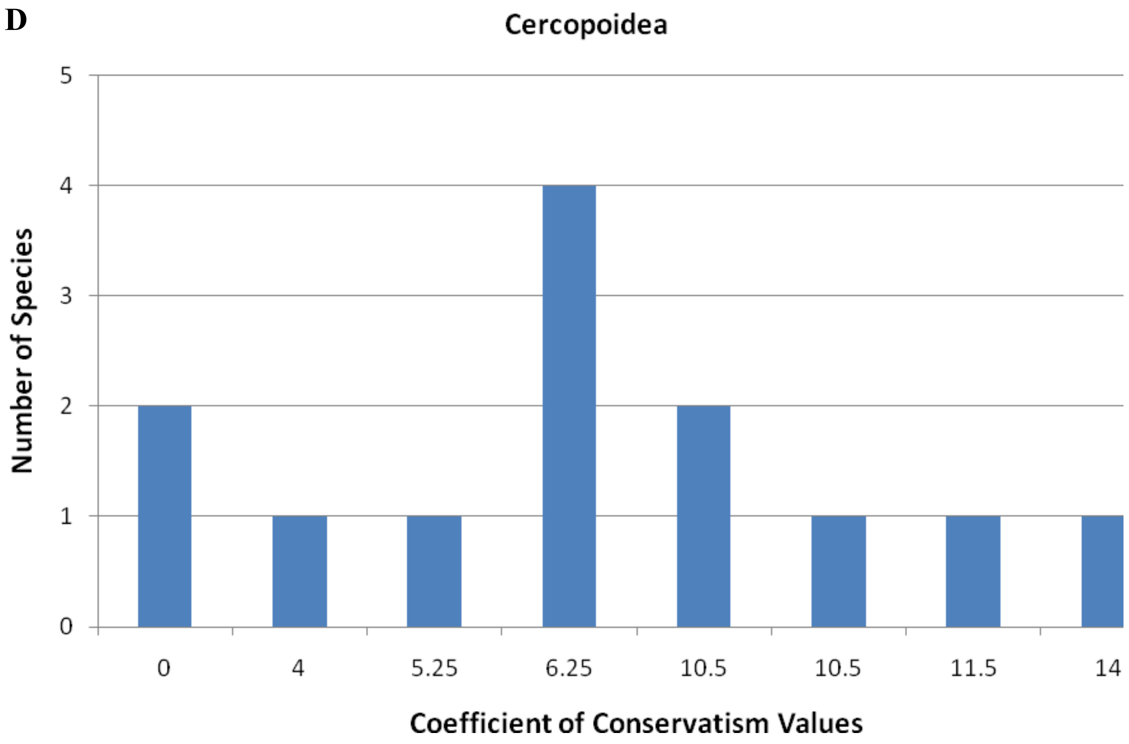


Figure 2.3 (cont.)

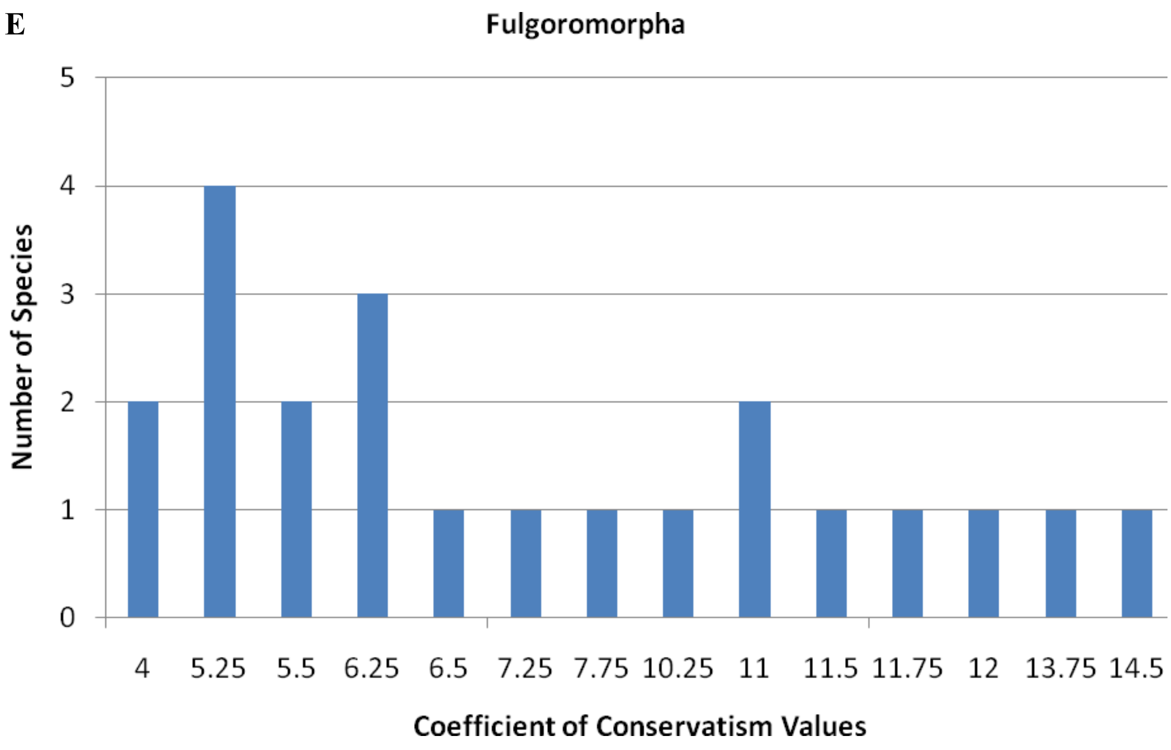


Figure 2.4. Species accumulation curves wilth standard deviation for all sites sampled when using a vacuum (A) and sweep net (B). Chao2 is species richness estimator based on the proportion of singletons (species represented by one individual)/doubletons (species represented by two individuals) plus species observed in one sample (i.e. transect); and SOBs are species observed in one sample (i.e. transect)

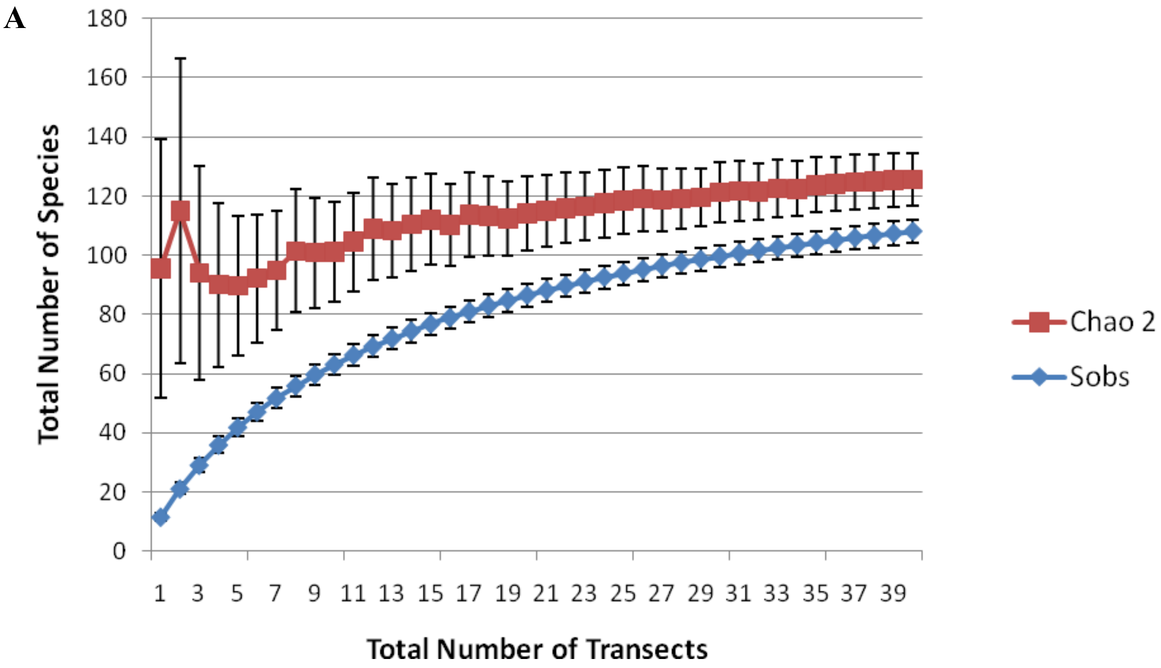


Figure 2.4 (cont.)

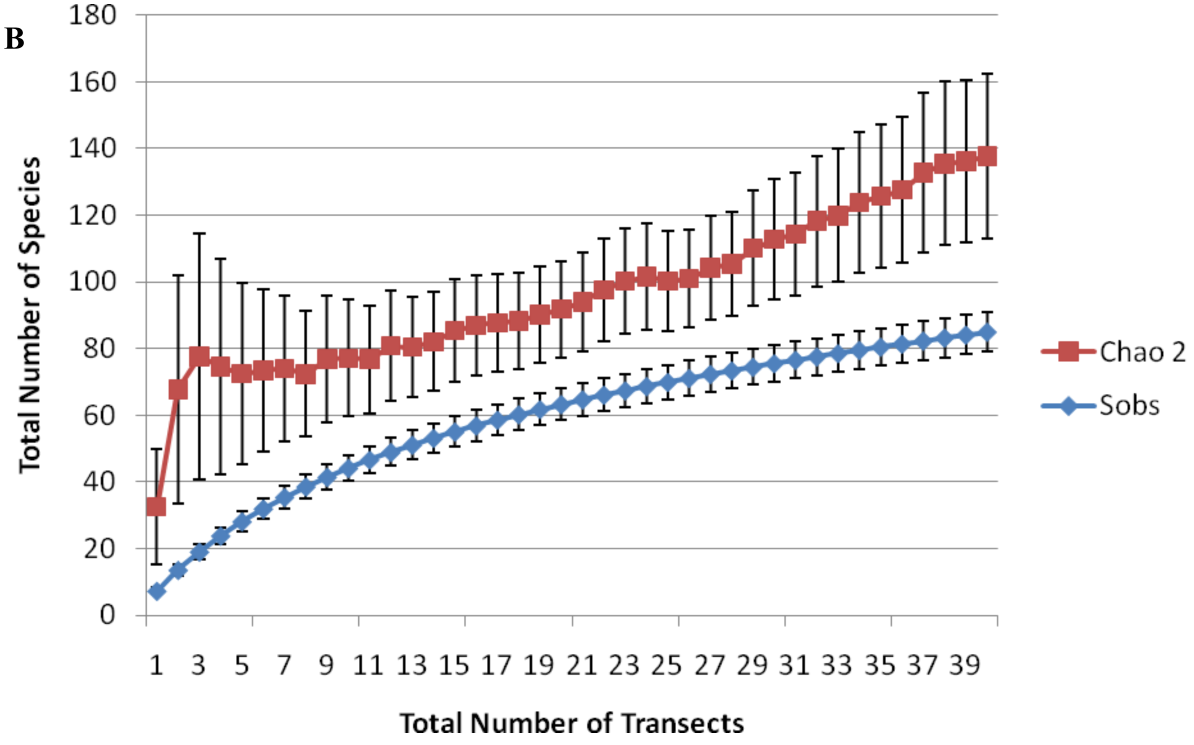


Figure 2.5. Species accumulation curves and standard deviation for (A) wet-mesic prairies (B), hill prairies (C), and sand prairies when using a vacuum. Chao2 is species richness estimator based on the proportion of singletons (species represented by one individual)/doubletons (species represented by two individuals) plus species observed in one sample (i.e. transect) and SOBs are species observed in one sample (i.e. transect)

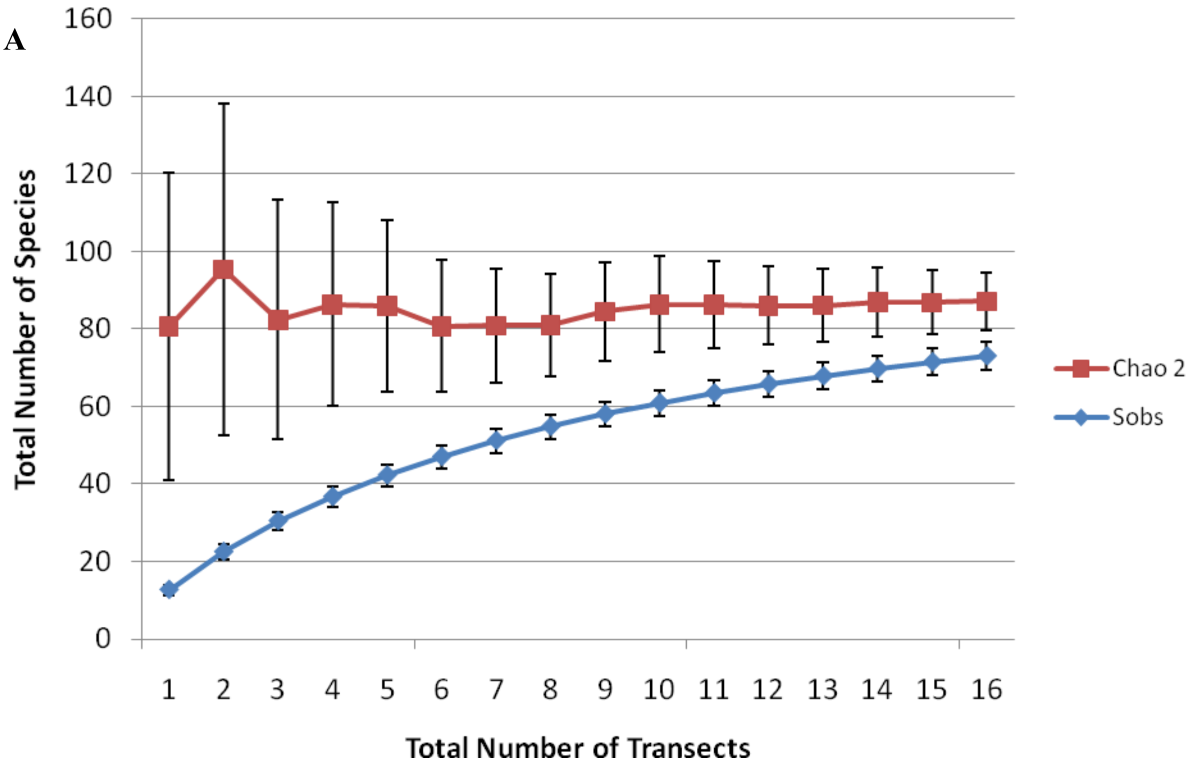


Figure 2.5 (cont.)

B

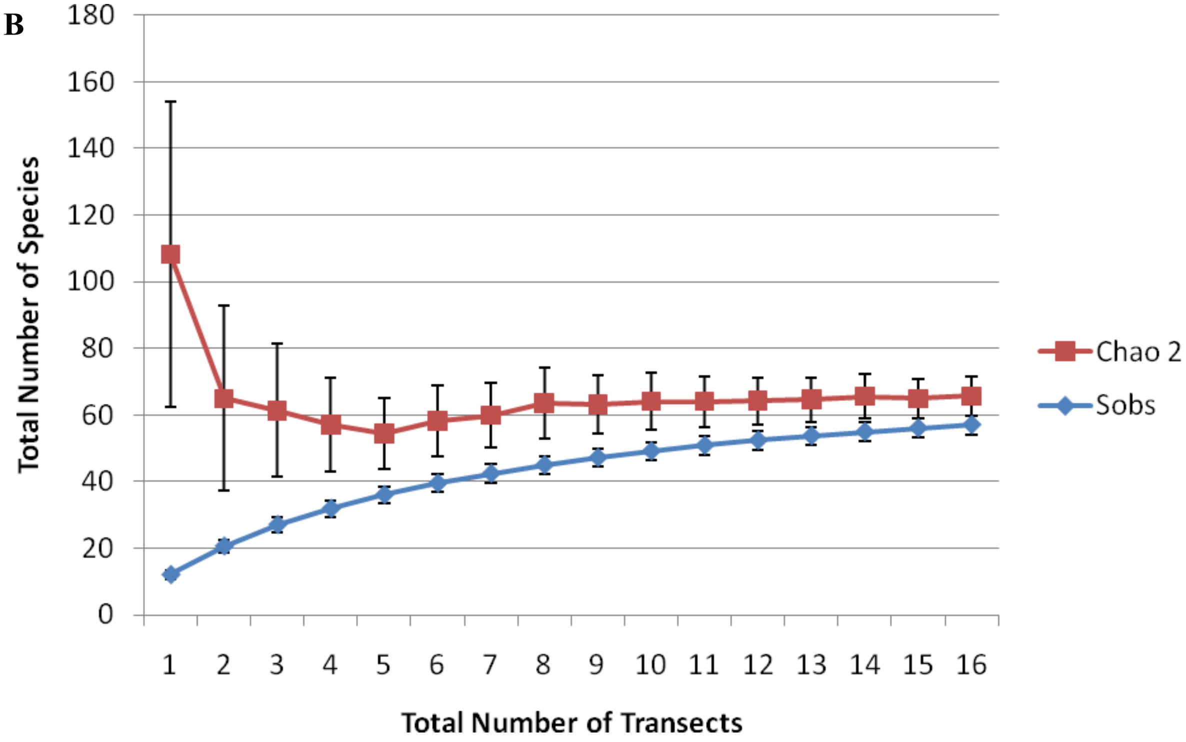


Figure 2.5 (cont.)

C

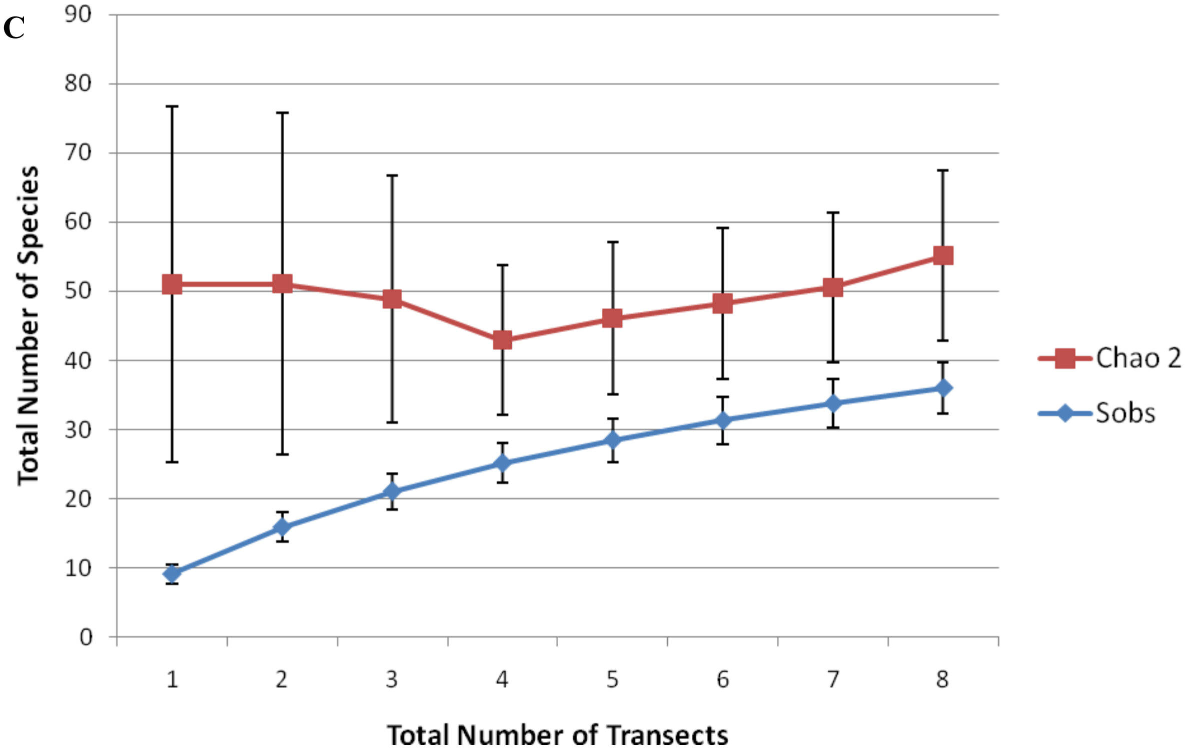
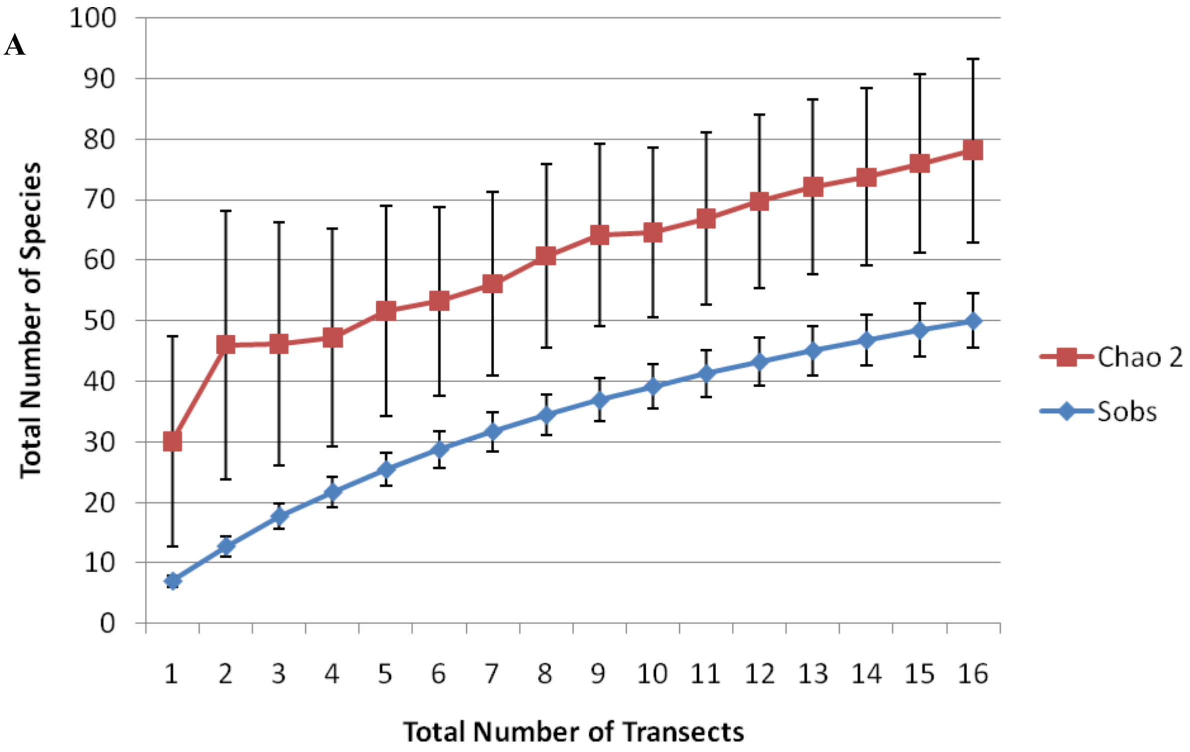


Figure 2.6. Species accumulation curves and standard deviation for (A) wet-mesic prairies (B), loess prairies (C), and sand prairie when using a sweep net. Chao2 is species richness estimator based on the proportion of singletons (species represented by one individual)/doubletons (species represented by two individuals) plus species observed in one sample (i.e. transect) and SOBs are species observed in one sample (i.e. transect)



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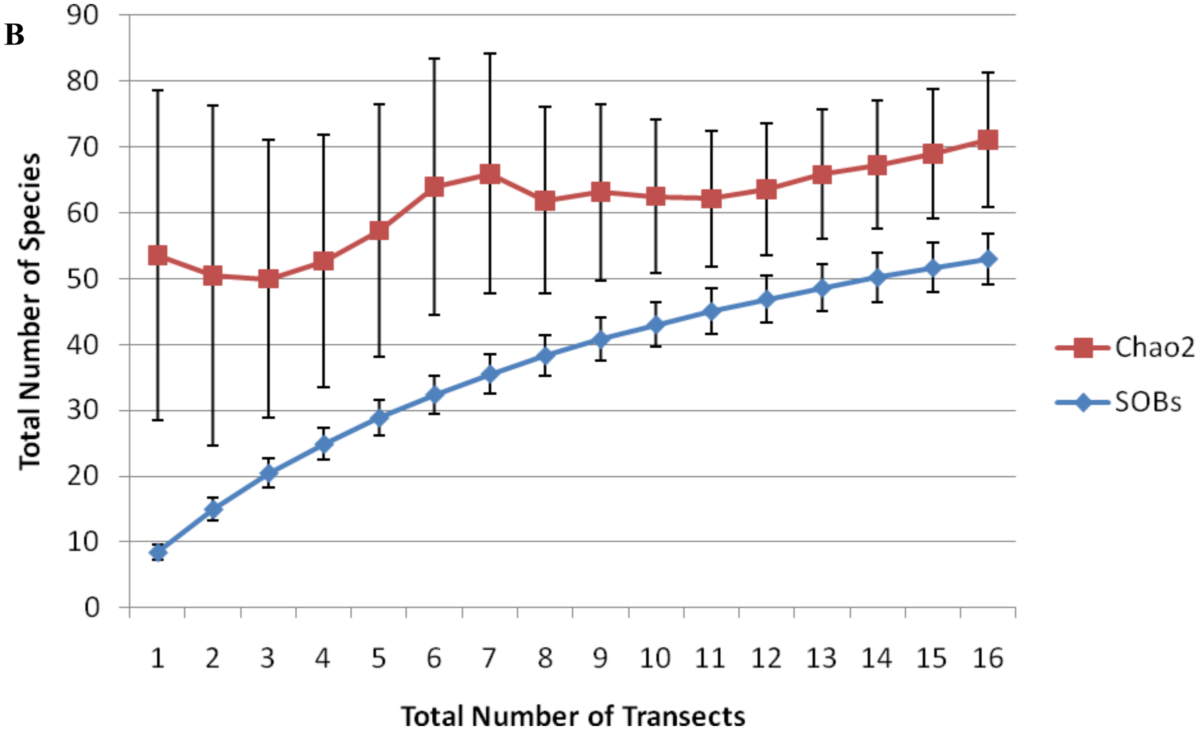


Figure 2.6 (cont.)

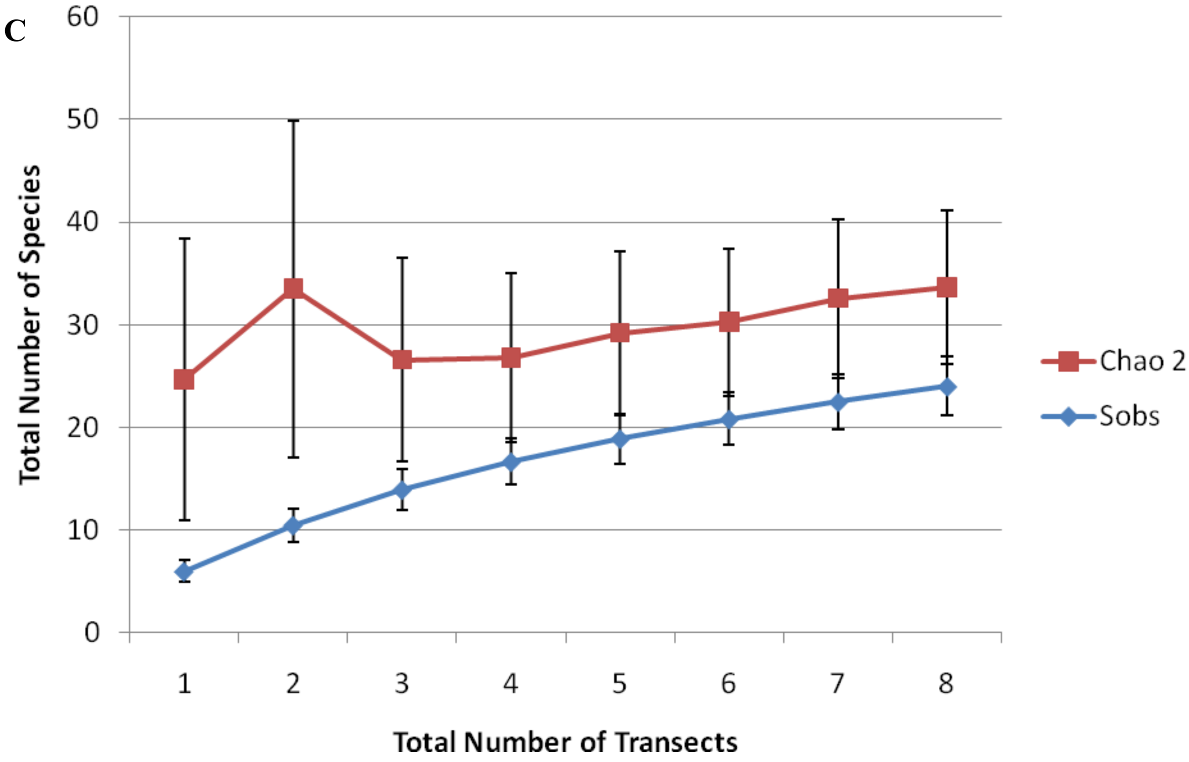


Figure 2.7. Cummulative means and standard deviations of AQI$_{w/N}$ (A) and AQI$_{w/outN}$ (B) values calculated from 10 sites using a vacuum

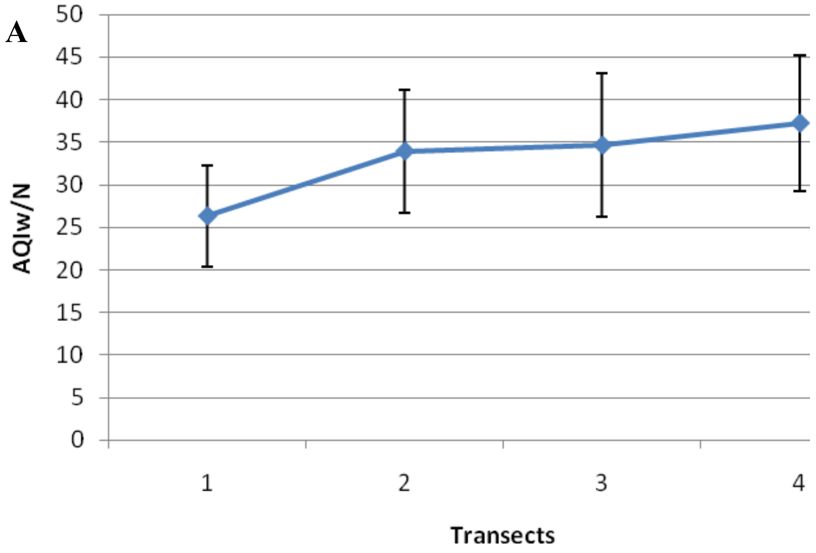


Figure 2.7 (cont.)

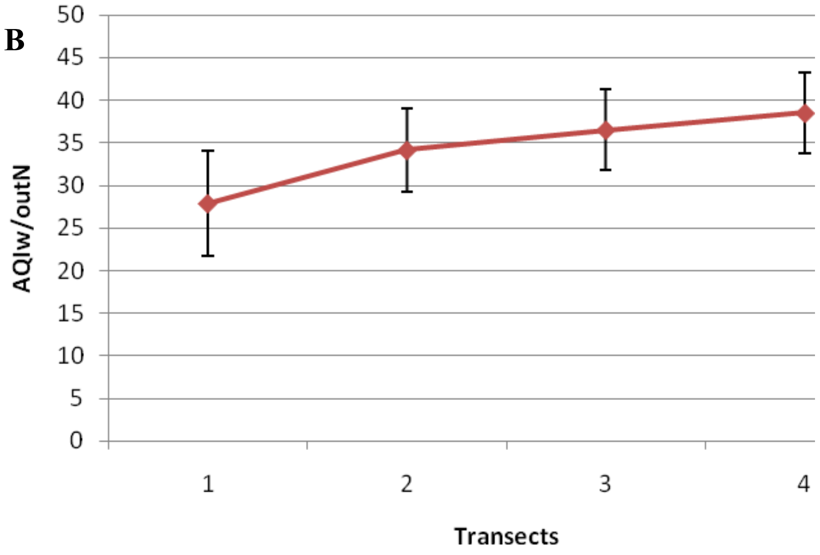


Figure 2.8. Cummulative means and standard deviations of $AQI_{w/N}$ (A) and $AQI_{w/outN}$ (B) values calculated from 10 sites using a sweep net.

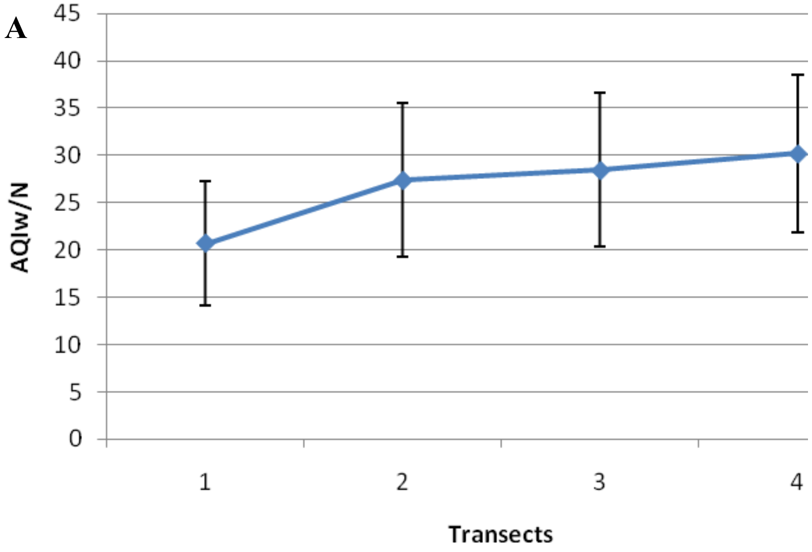


Figure 2.8 (cont.)

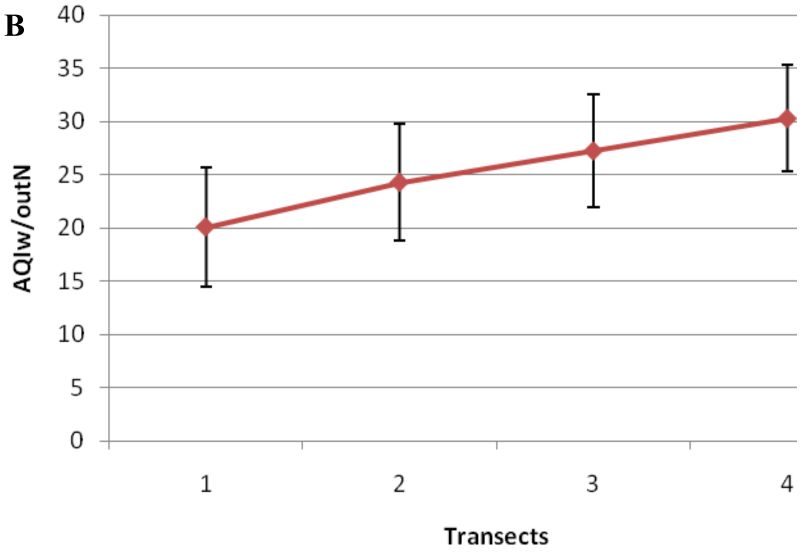


Figure 2.9. Cummulative means and standard deviations of $AQI_{w/N}$ (A) and $AQI_{w/outN}$ (B) values calculated from four wet prairies using a vacuum

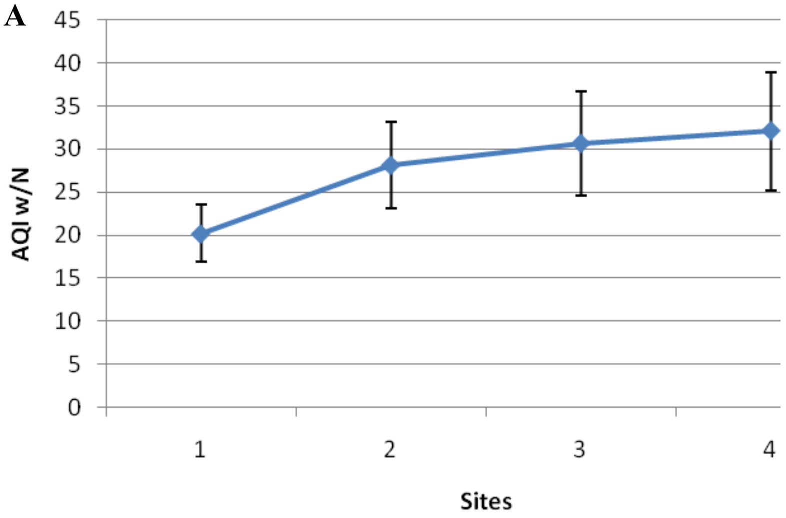


Figure 2.9 (cont.)

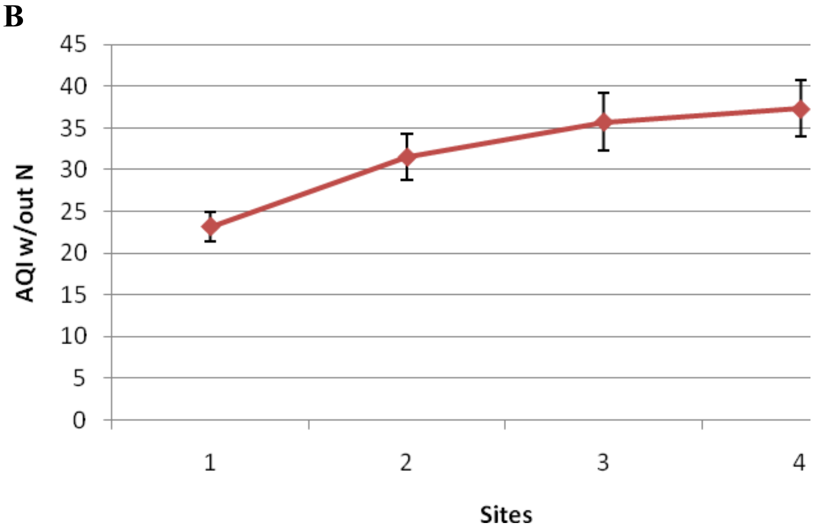


Figure 2.10. Cummulative means and standard deviations of AQI$_{w/N}$ (A) and AQI$_{w/outN}$ (B) values calculated from four hill prairies using a vacuum

A

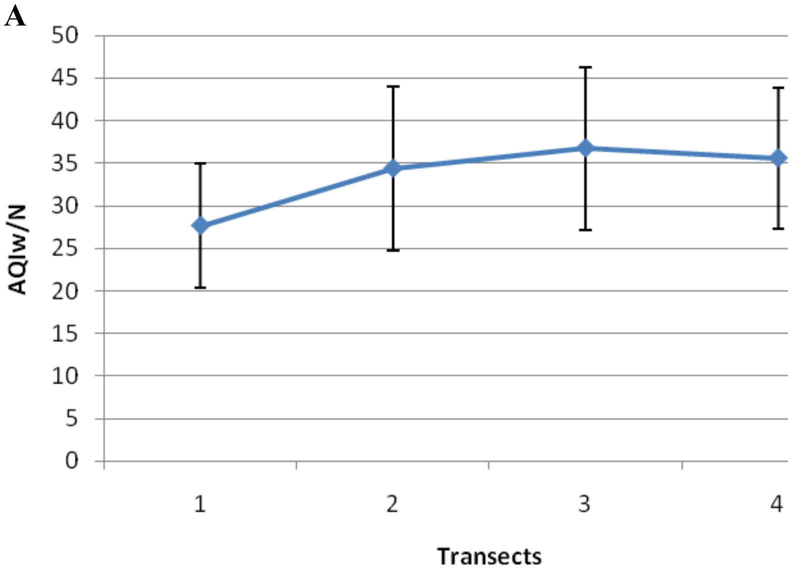


Figure 2.10 (cont.)

B

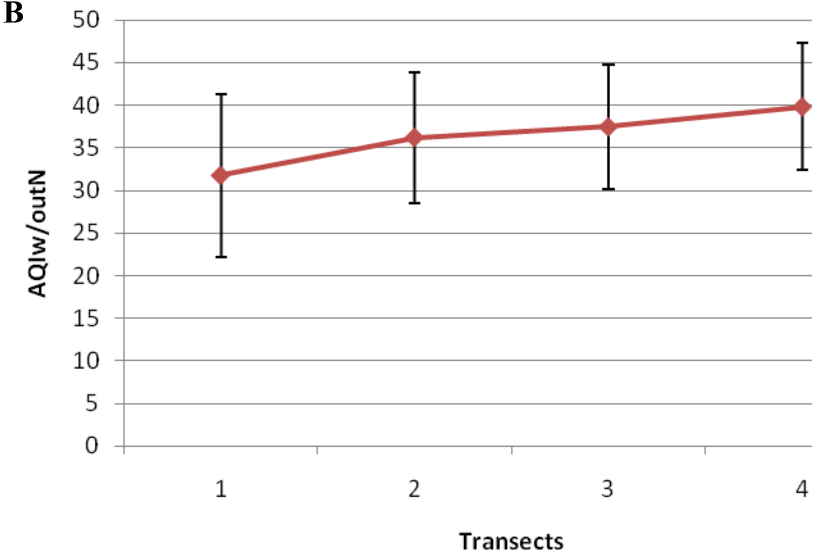


Figure 2.11. Cummulative means and standard deviations of $AQI_{w/N}$ (A) and $AQI_{w/outN}$ (B) values calculated four wet-mesic prairies using a sweep net

A

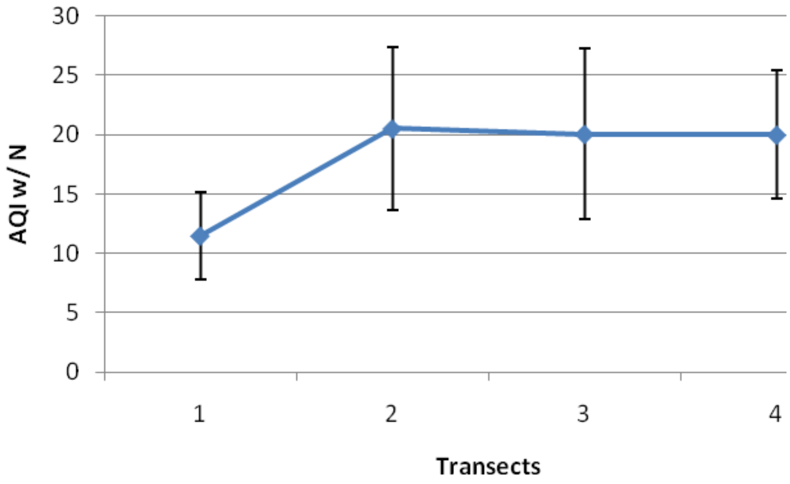


Figure 2.11 (cont.)

B

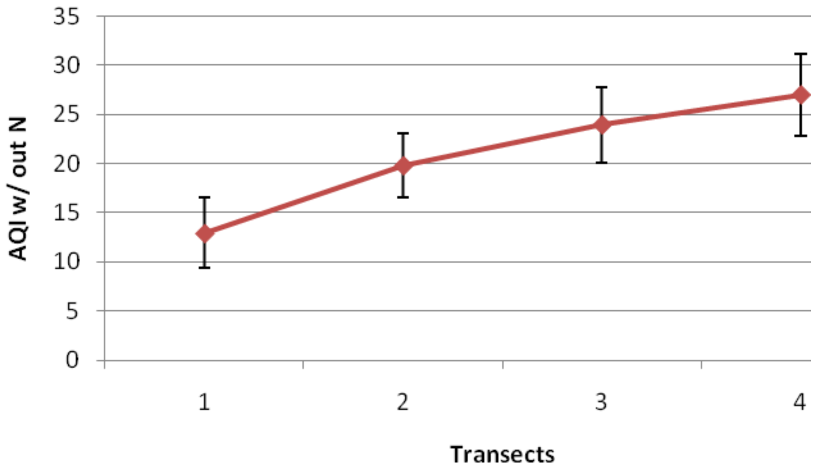


Figure 2.12. Cummulative means and standard deviations of $AQI_{w/N}$ (A) and $AQI_{w/outN}$ (B) values calculated four hill prairies using a sweep net

A

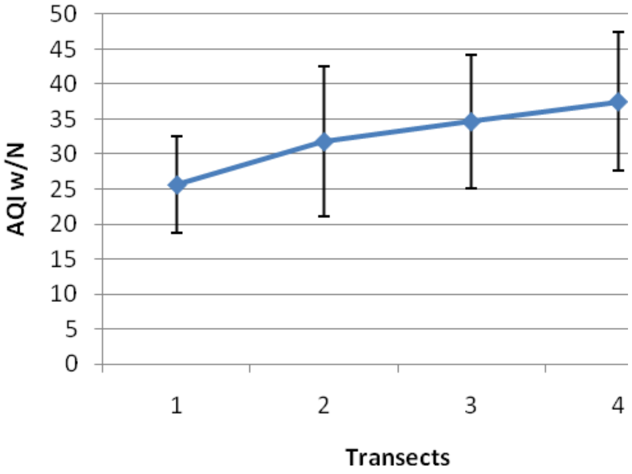


Figure 2.12 (cont.)

B

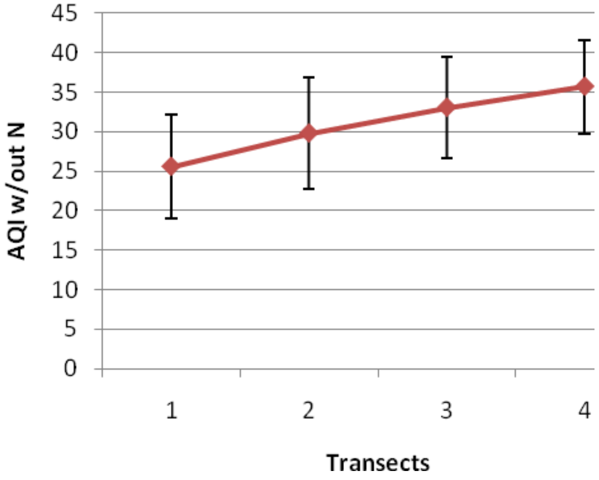


Figure 2.13 Cummulative AQI$_{w/N}$ (A) and AQI$_{w/outN}$ (B) values calculated from hill prairies using a vacuum

A

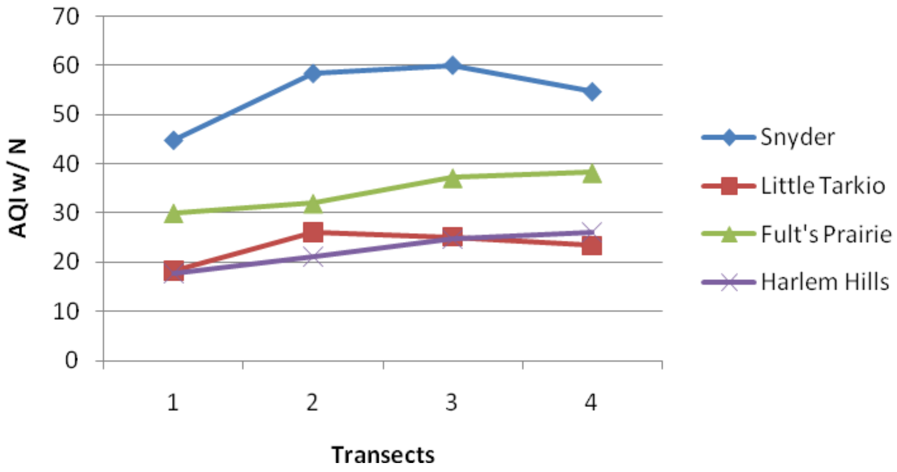


Figure 2.13 (cont.)

B

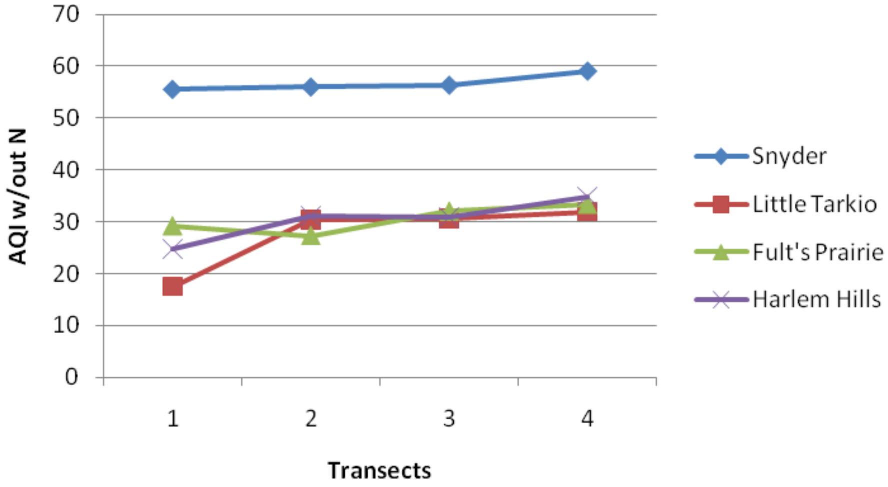


Figure 2.14 Cummulative $AQI_{w/N}$ (A) and $AQI_{w/outN}$ (B) values calculated from sand prairies using a vacuum

A

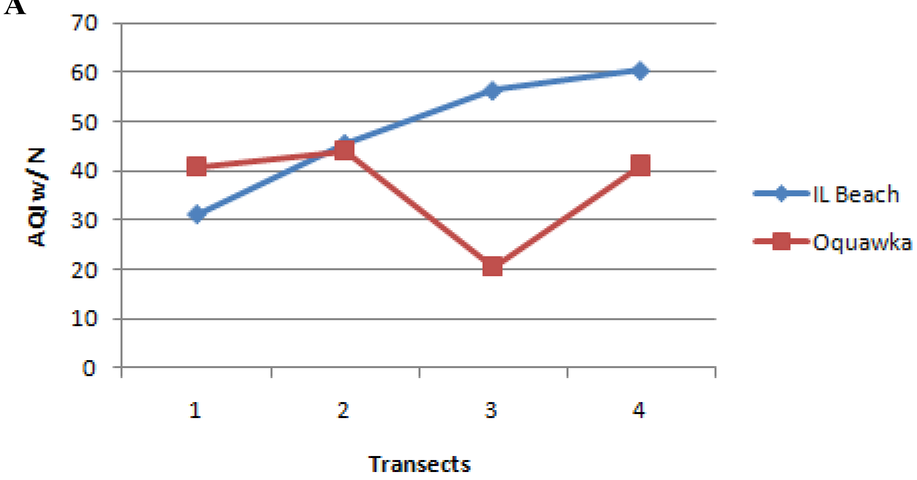


Figure 2.14 (cont.)

B

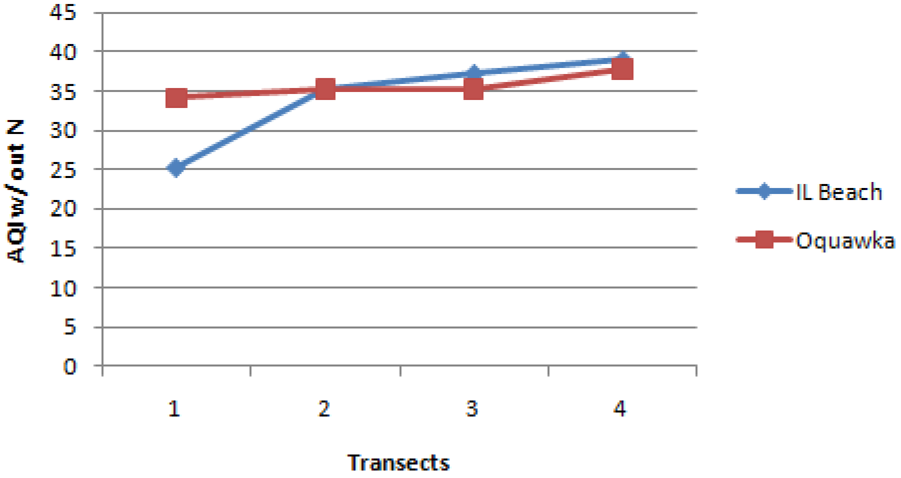


Figure 2.15 Cummulative AQI$_{w/N}$ (A) and AQI$_{w/outN}$ (B) values calculated from four wet-mesic prairies using a vacuum

A

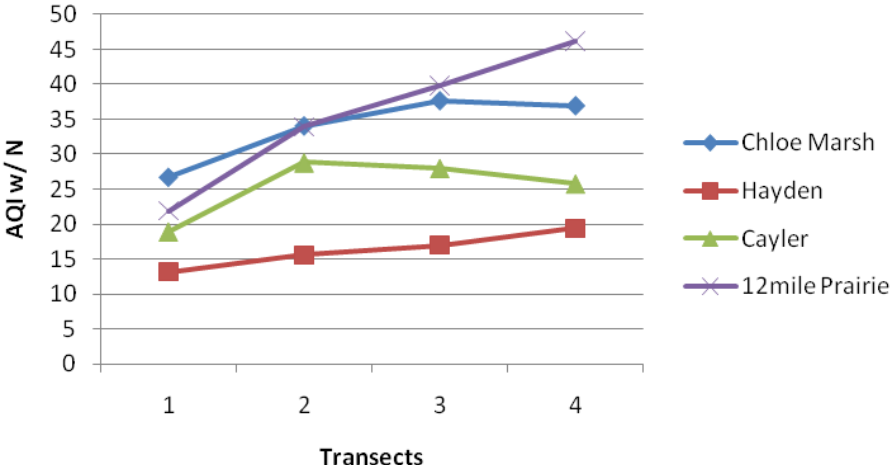


Figure 2.15 (cont.)

B

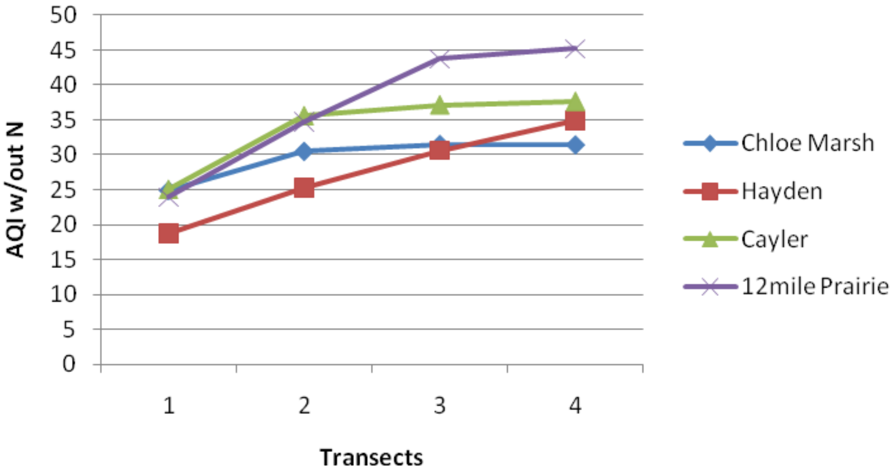


Figure 2.16 Cummulative AQI$_{w/N}$ (A) and AQI$_{w/outN}$ (B) values calculated from four hill prairies using a sweep net

A

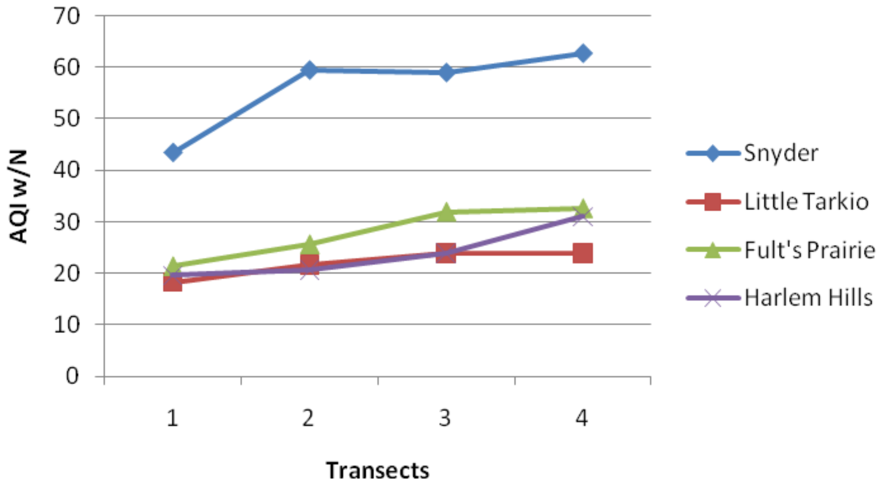


Figure 2.16 (cont.)

B

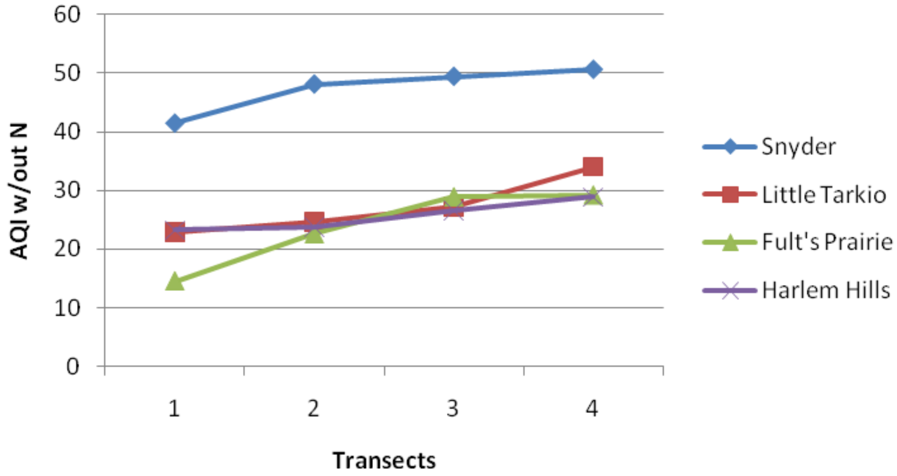
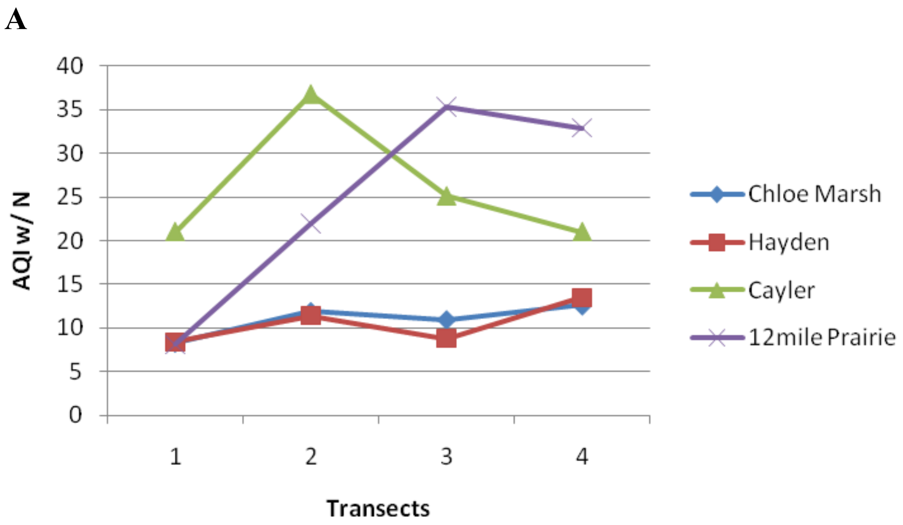


Figure 2.17 Cummulative $\mathrm{AQI}_{\mathrm{w/N}}$ (A) and $\mathrm{AQI}_{\mathrm{w/outN}}$ (B) values calculated from four-mesic wet prairies using a sweep net



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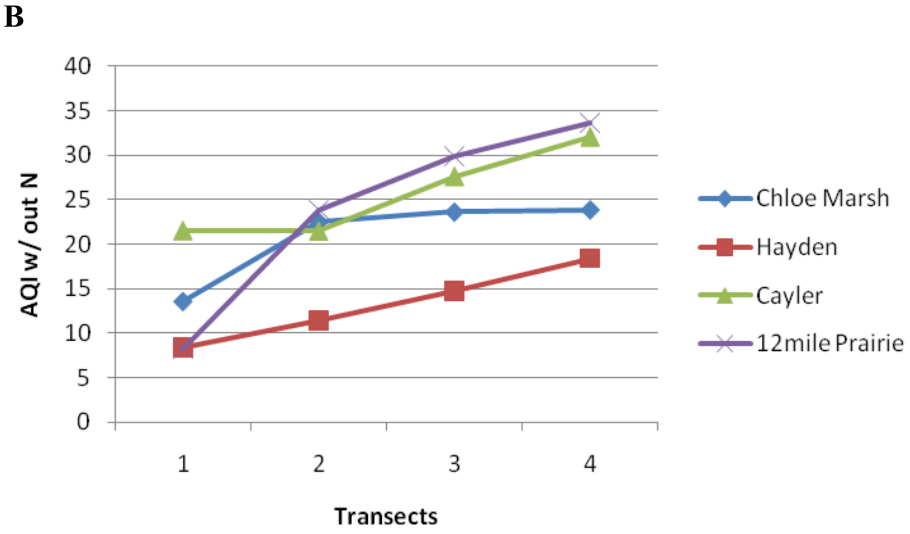


Figure 2.18 Cummulative $AQI_{w/N}$ (A) and $AQI_{w/outN}$ (B) values calculated from sand prairies using a sweep net

A

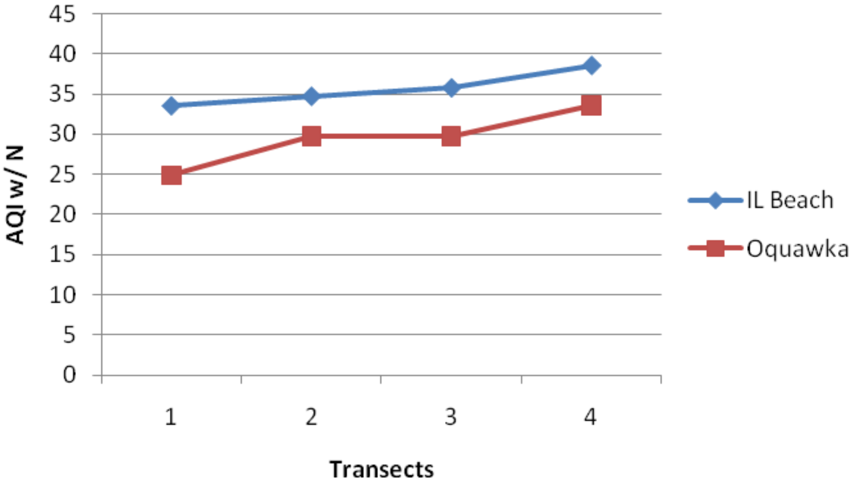
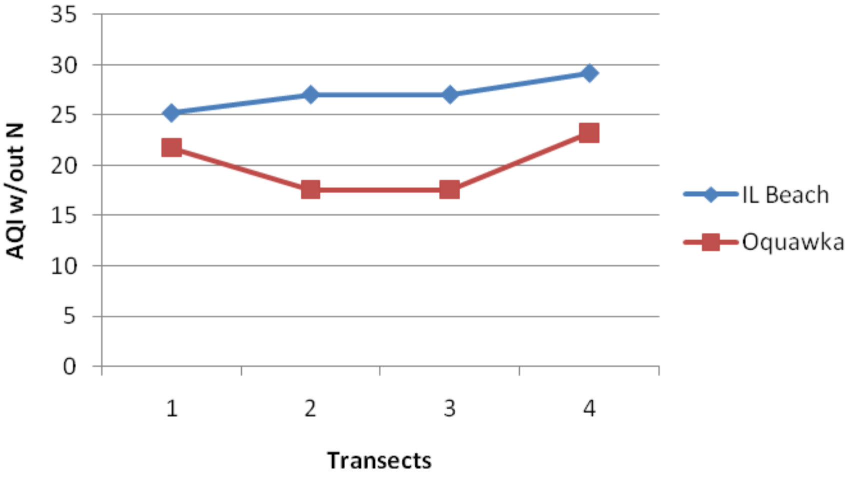


Figure 2.18 (cont.)

B



Chapter 3: Evaluating Glacial Drift Hill Prairie Quality in Illinois Using Auchenorrhyncha (Insecta: Hemiptera) and Vascular Plants

ABSTRACT

In this study a habitat quality index based on Auchenorrhyncha (Insecta: Hemiptera) species composition was used to investigate the indexs' ability in discriminating hill prairie quality along a gradient of disturbance, based on the Illinois Natural Areas Inventory grading criteria, and whether this index differs from other vegetation-based measures of quality; and examine the relationships between Auchenorrhyncha and vegetation integrity and diversity. Auchenorrhyncha and vascular plants were sampled from 14 Illinois glacial-drift hill prairies representing a range in quality. Insects and plants were sampled from late July through August, 2007. Results from this study showed that Floristic Quality Index, Auchenorrhyncha Quality Index, as well as other Auchenorrhyncha and plant diversity and integrity index values are greater in high, followed by mid, then low quality hill prairie remnants. Also, perennial C4 grasses are strongly associated with prairie Auchenorrhyncha. These data suggest that prescribed burning or brush removal are needed to prevent woody-encroachment from eliminating prairie vegetation and Auchenorrhyncha on low quality sites; and restoration of perennial C4 grasses on low quality sites are needed to support more prairie Auchenorrhyncha fauna. Reintroductions of conservative (i.e., prairie-dependent and fire-sensitive) Auchenorrhyncha may also be needed to improve Auchenorrhyncha integrity on mid and low quality sites but these reintroductions need to be used in combination with reduced burn management or the inclusion of unburned refugia to reduce the extirpation of these fire-sensitive insects.

INTRODUCTION

Hill prairies are island-like openings of prairie that occur on steep slopes surrounded by forests that are adjacent to major rivers in the Midwestern United States (Evers 1955, Robertson *et al*. 1995). These prairies are found on south to southwest-facing slopes and on well drained soils. In Illinois, a total of 453 moderate to high quality acres of hill prairies have been identified (Taft *et al*. 2009), representing four main types: loess, glacial drift, gravel, and sand (White 1978). Presently, loess is still the most common hill prairie (463 acres), followed by glacial drift (51.5 acres), gravel (14.7 acres), and then sand (5.2 acres) (personal communication John Taft - Illinois Natural History Survey). Because of their steep slopes and inaccessibility, hill prairies have not been plowed or converted to agriculture, but have been grazed (Robertson *et al*. 1995). However, they remain some the most numerous of all high quality prairie communities in Illinois, with 93 hill prairie remnants falling in this category (Taft *et al*. 2009). Hill prairies are dominated by bunch-grass species, such as little bluestem (*Schizachyrium scoparium*), sideoats gramma (*Bouteloua curtipendula*), and several conservative perennial forb species (Robertson *et al*. 1995, Taft *et al*. 2009).

Within the past 50 years hill prairies throughout Illinois have declined in total area by over 50% due to woody encroachment, invasion from exotic species (non-native grasses, forbs, shrubs, and trees), limestone quarrying, urban development, and livestock grazing (McClain and Anderson 1990, Robertson *et al*. 1995, Schwartz *et al*. 1997, Taft *et al*. 2009). As a consequence many populations of native plant species have declined dramatically or have become extirpated (Robertson and Schwartz 1994, Leach and Givinish 1996), resulting in the deterioration of hill prairie integrity (i.e., native biotic and abiotic components). In an attempt to prevent further loss of this integrity a variety of vegetation-based measures (e.g., species richness and diversity/quality indices) have been used to identify and conserve hill prairies (White 1978, McClain *et al*. 2002, Owens *et al*. 2008), as well as to provide tools for long-term monitoring of hill prairie quality

Despite these advantages, some plants do not always show a strong correlation in patterns of species richness to those of other groups of organisms, and thus any assessments based solely on vascular plants may be incomplete (Kushlan 1979, Landres *et al*. 1988, Kremen 1992). By using a variety of organisms, a more complete picture of hill prairie integrity might be obtained than assessments based on plants alone. One such group that is numerically dominant in abundance and species richness in native grasslands, in particular on hill prairies, is Auchenorrhyncha (Insecta: Hemiptera [DeLong 1948, Waloff 1980, Hamilton 1995, Nickel 2003]). These insects include leafhoppers (Cicadellidae), planthoppers (Fulgoroidea), treehoppers (Membracidae), spittlebugs (Cercopoidea), and cicadas (Cicadoidea) (Hamilton 2005). Many of these species display strong affinities to a few species of prairie plants (Whitcomb *et al*. 1987, Hamilton and Whitcomb 1993) and respond in predictable ways to grassland degradation (Harper *et al*. 2000, Hilderbrandt and Nickel 2003).

Although Auchenorrhyncha and vascular plants represent significant and important components of hill prairie biodiversity and may be integral to ecosystem function, quantitative studies that have explored the relationships between hill prairie Auchenorrhyncha and plant integrity along a gradient of quality are lacking. In addition to the scarcity of these studies is the continued reliance of measuring prairie integrity based on vascular plants. Previous studies (Hamilton 1995, and Panzer *et al*. 1995) have suggested that some groups of prairie insects (including Auchenorrhyncha) respond differently to management (particularly burning) than many prairie plants (Harper *et al*. 2000). Therefore, measures of prairie integrity based only on plants (or, for example, any single group of insects) may not facilitate an adequate assessment of integrity of hill prairies. Thus, by examining the relationships of Auchenorrhyncha and vascular plants and their response to disturbance, land managers may be provided with a more complete understanding of hill prairie integrity. Also, this information may be invaluable in managing and conserving Illinois prairies, as well as determining the redundancy, if any, in using both Auchenorrhyncha and plants in discriminating hill prairie quality. Ultimately, these new studies could aid land managers in developing a more comprehensive approach in the conservation and management of native grassland biota then those based on plants alone.

The aim of this study was to determine whether a habitat quality index based on Auchenorrhyncha species composition can discriminate hill prairie quality using the Illinois Natural Areas Inventory grading criteria (1978) from 14 Illinois glacial-drift hill prairies that reflect a gradient of degradation; whether this index ranks sites differently in terms of quality than other vegetation-based biotic indices and diversity measures; and to compare the integrity and diversity of plants and Auchenorrhyncha. The variables to be compared include Auchenorrhyncha species composition, species richness, Shannon-Wiener and Simpson's Diversity indices, and an Auchenorrhyncha Quality Index (with and without abundance) and their corresponding mean Coefficient of Conservatism or meanCC values. Quality rankings of sites based on these variables will be compared with those based on plant species composition, plant cover of functional groups, species richness of functional groups, exotic species, Shannon-Wiener and Simpson's Diversity indices, and the Floristic Quality Index and its corresponding meanCC (Swink and Wilhelm 1994, Taft *et al*. 1997). Since Auchenorrhyncha respond differently to land management, such as burning, shrub removal, and grazing (Reed 1996, Harper *et al*. 2000, Nickel and Hildebrandt 2003), I hypothesize that habitat quality indices based on

Auchenorrhyncha will discriminate hill prairie quality differently than vegetation based indices. Because many of these conservative (i.e., prairie-dependent and fire-sensitive) Auchenorrhyncha are strongly associated with native perennial grasses (Whitcomb *et al.* 1987, Hamilton and Whitcomb 1993, Nickel 2003, Hamilton 2005), I hypothesize that perennial C4 grass abundance and species richness will be more strongly correlated with Auchenorrhyncha integrity than other plant functional groups.

MATERIALS AND METHODS

Study Sites

I sampled 14 glacial drift hill prairies visited in Peoria County and Marshall County (Central, IL) near the Illinois River (Table 3.1). These sites covered a wide range in quality that was based on the grading criteria of the Illinois Natural Areas Inventory (INAI, White 1978), with grades ranging from A (undegraded) to D (degraded) (Table 3.1): three sites were of high quality, with an INAI grade of A to B+ and were dominated by native prairie plants (i.e., high quality); seven sites were of intermediate quality, ranging from B to C+, with a majority of their vegetation being prairie plants as well as some trees and shrubs (i.e., mid quality); and four sites were of poor quality, ranging from C to D, often dominated by trees, shrubs, and some prairie plants (i.e., low quality). These INAI grades were assigned by land managers in the summer of 2007 based on the combination of native prairie vegetation that was present, presence of woody invasion, and any degradation (e.g., quarry, and abandoned roads) present on these sites. All sites were surrounded by forests and have different management history. Nine sites were under a management of controlled burning (Table 3.1). High quality sites were not burned in 2 to 3 years. Mid quality sites displayed a range of management with four sites burned 2 to 3 years ago, one site burned 10 years ago, and one site that was only managed with shrub removal. Low quality sites also displayed a range of management with two sites unmanaged, one site was burned 10 years prior to collecting, and one site had not been burned in 16 years prior to collecting. Management history was provided by land managers.

Vegetation Sampling

At each site, botanical surveys were conducted by placing four 10m-linear transects perpendicular to a 22m baseline. Each transect was 5m apart from the neighboring transect. All transects were placed in the center of the prairie to capture the native prairie biota and minimize edge effects. However, Detweiller South and Robinson Park North Prairie B (Table 3.1) had narrow strips of prairie vegetation, thus transects were placed in a single-file orientation. Along each transect 10 1/4 m^2 quadrats were used to assess vegetation for a total of 40 quadrats per site. Successive quadrats were placed every meter on alternate sides of the transect. All plant species rooted within the quadrats were identified and recorded. Plant cover (individual species and functional groups [e.g., forbs, grasses]), bare ground, and litter were recorded using a modified Daubenmire cover class system (Daubenmire 1959, Baily and Paulton 1968). All tree saplings and shrubs 1m tall within the quadrats were also identified and recorded. Sampling took place between the third week of July and second week of August of 2007. Botanical nomenclature followed Mohlenbrock (2002).

Auchenorrhyncha Sampling

On the same vegetation sampling transects Auchenorrhyncha were sampled with a modified leaf blower vacuum (STIHL™) for approximately 5 minutes. Specimens were transferred into Photo Tactic Optimal Insect Extractors (PTOIE) for approximately 45min – 1hour. All Auchenorrhyncha were stored in 95% ethanol. Adult Auchenorrhyncha were identified to species, total numbers of individuals were tallied, and specimens were deposited at the Illinois Natural History Survey Insect Collection. Sampling took place between the third week of July and second week of August, 2007, when Auchenorrhyncha abundance and species richness is at its peak (Blocker *et al.* 1972, Blocker and Reed 1976). Auchenorrhyncha and plants were sampled during the same day and time to avoid confounding effects, such as changes due to seasonality and climate (Waloff 1980).

Vascular Plant Indices Terms and Definitions

The following plant indices were calculated per site:

Exotic Species = total number of exotic species

Shannon-Wiener Index of Diversity (H') = $- \sum [p_i \ln (p_i)]$ (Weaver and Shannon 1949); pi is proportion of each species in the sample. The value of H' ranges from 0 to 5. Values were calculated with and without exotic species.

Simpson's Index of Diversity (D) = $\sum (n / N)^2$ (Whittaker 1975); n = total number of individuals (total mean cover of each taxa) of a particular species, N = total number of individuals (total mean cover of all taxa) of all species. The value of D ranges from 0 to 1. Values were calculated with and without exotic species.

Species Richness = total number of native species.

Vascular Plant Functional Group Variables

Total cover and total species richness were determined for each of the following functional groups:

Perennial C4 Grasses (PC4); Perennial C3 Grasses (PC3); Annual C3 Grass (AC3); Annual C4 Grass (AC4); Annual Forbs (AF); Biennial Forbs (BiF); Perennial Forbs (PF); Ferns; Sedges; tree saplings; Shrubs; and Vines (herbaceous and woody species were combined)

Bare Ground and Litter Variables

Total cover of bare ground (BG) and litter were determined for each site.

Weighted Indices of Floristic Quality Variables

Mean Coefficient of Conservatism (Mean C) = $\sum$ CC/S. CC = Coefficient of Conservatism; S = total species richness (including exotic species). Mean C was also calculated using just native species or Mean C_n.

Floristic Quality Index (FQI) = Mean C ($\sqrt{Sn}$); Mean C = $\sum$ CC/S. Sn = native species richness, and S = native and exotic species richness.

The CC is an integer from 0 to 10 assigned to each species in the Illinois flora (Taft *et al.* 1997). All exotic species were assigned values of 0. Species displaying a ruderal ecology were assigned CC values from 0 to 3. These species are adapted to frequent disturbances. Matrix and competitive species were assigned CC values from 4 to 6. Many of these species can persist in intermediate levels of disturbance, but may decline with an increase in intensity, frequency, or duration of disturbance (Grime *et al.* 1988, Taft *et al.* 1997). Species that are restricted to natural areas (i.e. remnant-dependent) were assigned CC values from 7 to 10.

Auchenorrhyncha Indices Terms and Definitions

The following Auchenorrhyncha indices were calculated for site:

Shannon-Wiener Index of Diversity (H') = $-\sum [p_i \ln(p_i)]$ (Weaver and Shannon 1949), pi is proportion of each species in the sample. The value of H' ranges from 0 to 5.

Simpson's Index of Diversity (D) = $\sum (n/N)^2$ (Whittaker 1975); n = total number of individuals (total mean cover of each taxa) of a particular species, N = total number of individuals (total mean cover of all taxa) of all species. The value of D ranges from 0 to 1.

Species Richness = total number of species.

Weighted Indices of Auchenorrhyncha Quality Variables

AQI_t = meanCC*√S; MeanCC = Mean coefficient of conservatism value for all species encountered per sampling effort; √S = Square root transformation of the total number of species encountered at the site sampled. AQI were calculated using all species and just native grassland species (AQI_p).

The AQI was also computed using abundance (i.e., AQIwN).

$AQI_{w/N} = \sum [(n_i/N) * CC_i] * \sqrt{S}$, in which n_i = The total number of individuals for species i N = The total number of individuals for all species; CC_i = Coefficient of Conservatism for species i √S = Square-root transformation of the total number of species encountered per sampling effort. $AQI_{w/N}$ were computed using all species and grassland species prairie species ($AQIwN_p$).

The CC is a value from 0 to 18 assigned to each auchenorrhynchan species encountered. Species that are adapted to frequent disturbance and feed and overwinter on a variety of host plants received values ranging from 0 to 5. Species that can tolerate moderate levels of disturbance, are found in edge habitat and in native grassland, and feed and overwinter on prairie and some non-prairie plants received values ranging from 6 – 10. Species that are sensitive to disturbance, restricted to native grassland, and feed and overwinter on native grassland vegetation received values ranging from 11 – 18 (Appendix B.1).

Data Analysis

To determine if weighted biotic indices (AQI and FQI), diversity measures, and plant functional groups could discriminate between differences in glacial drift hill prairie quality, One-way ANOVAs followed by Tukey's *post hoc* tests were conducted. Kruskal-Wallis followed by

Dunn's *post hoc* tests were conducted when data were not normally distributed. In the case of AQI, two versions were calculated: 1) based on all 71 auchenorrhynchan species encountered, 2) and with the exclusion of 20 forest-associated auchenorrhynchan species. I removed forest-associated Auchenorrhyncha species to determine if the discriminating ability of these indices would improve. Data were checked for normality (Kolmogorov-Smirnov test) and homogeneity of variance (Levene's test [Levene 1960]). All statistical tests were conducted using SigmaStat Statistical Software package version 3.1 (2004).

Relationships among Auchenorrhyncha and vegetation variables were examined with Pearson's product-moment correlation analysis. Two separate analyses were conducted: 1) correlations between plant and Auchenorrhyncha variables calculated from all 71 auchenorrhynchan species encountered; and 2) correlations between plant and Auchenorrhyncha variables calculated from grassland Auchenorrhyncha. Tests of normality for these variables were conducted with the one-sample Kolmogorov-Smirnov test. All statistical tests were conducted using SigmaStat Statistical Software package version 3.1 (2004). Bonferroni-corrections and other related tests (e.g., Bernouilli equation) used to control for Type I error were not conducted for the ANOVA and Pearson's product-moment correlation multiple comparisons because more Type II errors (biologically meaningful patterns lacking statistical significance) may be created than Type I errors being eliminated (Moran 2003).

Relationships between Auchenorrhyncha and plants were explained with Canonical Correspondence Analysis (CCA) using CANOCO 4 was used (ter Braak and Smilauer 1998). This multivariate analysis generates ordinations of species and samples that are constrained along axes determined by the environmental variables (ter Braak and Prentice 1988). Scaling was set to inter-species distance, which displays dissimilarities between the occurrence patterns of different species and correlations among the environmental variables. Auchenorrhyncha abundance data were log-transformed to reduce the chance of highly abundant species (e.g., *Oecleus borealis* and *Erythroneura obliqua*) influencing the ordinations (ter Braak and Smilauer 1998). Statistical significance of fitting CCA axes to the relationship depicted between Auchenorrhyncha and vegetation variables (e.g., plant function groups, litter, and bare ground data) was tested using a global permutation test (Monte-Carlo test) of the species data at 1,000 iterations. Forward selection of the environmental variables tested with Monte Carlo permutations also was used in determining statistical significance for each variable singly (marginal effects) and in order of additionally explained variance (conditional effects).

CANOCDRAW (ter Braak and Smilauer 1998) was used to graph CCA ordinations, whereby significant variables (indicated by asterisks) are plotted with non-significant variables. Vectors represent vegetation and bare ground and litter variables, in which the length of the vector indicates the amount of variance explained in the ordination. The direction of the vector indicates the vector's relationship, either positive or negative, with other environmental variables.

RESULTS

Discriminating Hill Prairie Quality based on Vegetation

Out of the 17 vegetation variables, only exotic plant species richness ($F = 16.83$, $P < 0.001$), total plant meanCC ($F = 16.873$, $P < 0.0001$), shrub species richness ($F = 11.30$, $P = 0.0012$), native plant species meanCC ($F = 11.24$, $P = 0.001$), vine cover ($H = 7.93$, $P = 0.02$), vine species richness ($H = 7.28$, $P = 0.03$), tree cover ($F = 6.82$, $P = 0.02$), and perennial C4 grass species richness ($F = 6.11$, $P = 0.02$) were able to significantly discriminate between differences in glacial-drift hill prairie quality. These vegetation variables reflected the INAI grading classification, with higher values observed in high quality sites, followed marginally lower values in mid quality sites, and lowest values observed in low quality sites (Table 3.2). Although, native plant species, total plant FQI, perennial forb cover, perennial forbs species, perennial C3 grass cover, perennial C3 grass species richness, perennial C4 grass cover, shrub cover, and tree sapling species richness could not significantly discriminate between differences in glacial-drift hill prairie quality, these plant functional group, diversity, and integrity variables did reflect INAI grading classification (Table 3.2; High > Mid > Low quality).

Discriminating Hill Prairie Quality based on Auchenorrhyncha

Out of the eight Auchenorrhyncha variables, only total grassland auchenorrhynchan species meanCCw/N ($F = 17.99$, $P = 0.001$) and grassland auchenorrhynchan species meanCCw/outN ($F = 3.956$, $P = 0.05$) were able to significantly discriminate between differences in glacial-drift hill prairie quality. These total auchenorrhynchan species $\text{meanCC}_{\text{w/N}}$ values were consistently higher on high quality sites, marginally lower values on mid quality sites, and exhibited the lowest values on low quality sites (Table 3.3). Although, other Auchenorrhyncha-integrity and diversity indices with and without forest Auchenorrhyncha, such as $\text{AQI}_{\text{w/N}}$, $\text{AQI}_{\text{w/outN}}$ with, $\text{meanCC}_{\text{w/N}}$, and $\text{meanCC}_{\text{w/outN}}$ could not significantly discriminate between differences in glacial-drift hill prairie quality these indices were reflecting INAI classification, with higher values on high quality sites, followed by marginally lower values on mid quality sites, and lower values on low quality sites (Table 3.3).

Pearson’s Product Correlations

Total auchenorrhynchan species meanCC was positively correlated with all plant species meanCC ($r = 0.65$, $P = 0.01$), native plant species meanCC ($r = 0.65$, $P = 0.01$), perennial C4 grass cover ($r = 0.57$, $P = 0.03$), and perennial C4 grass species ($r = 0.70$, $P = 0.01$). But, all auchenorrhynchan species meanCC was negatively correlated with native plant species richness (r = -0.56, P = 0.04), plant species Shannon-Wiener index (r = -0.53, P = 0.05), shrub species richness (r = -0.80, P = 0.001), tree cover (r = -0.57, P = 0.03), and tree species richness (r = -0.61, P = 0.02).

The native grassland Auchenorrhyncha meanCC was positively correlated with native plant species meanCC (r = 0.67, P = 0.01), perennial C4 grass cover (r = 0.58, P = 0.03), and perennial C4 grass species (r = 0.55, P = 0.04), but was negatively correlated with tree cover (r = -0.70, P = 0.01).

The grassland auchenorrhynchan species $AQI_{w/N}$ was positively correlated with perennial C4 grass species, but negatively correlated with perennial forbs species (r = -0.57, P = 0.03) and tree cover (r = -0.65, P = 0.01). Grassland auchenorrhynchan species $AQI_{w/outN}$ was also positively correlated with perennial C4 grass species (r = 0.58, P = 0.03) and negatively correlated with tree cover (r = -0.67, P = 0.01). Lastly, total auchenorrhynchan species $AQI_{w/N}$ was negatively correlated with tree cover (r = -0.55, P = 0.04).

The all Auchenorrhyncha Shannon-Wiener diversity index was positively correlated with bare ground cover (r = 0.64, P = 0.01), but negatively correlated with litter cover (r = -0.81, 0.001), and the plant species Shannon-Wiener diversity index (r = -0.61, P = 0.02). Also, the all Auchenorrhyncha Simpsons's diversity index was positively correlated with exotic plant species (r = 0.61, P = 0.02), perennial C3 grass species (r = 0.52, P = 0.06), but negatively correlated with the all plant species meanCC (r = -0.53, P = 0.05) and tree cover (r = 0.54, P = 0.05).

Grassland Auchenorrhyncha abundance was positively correlated with shrub cover (r = 0.62, P = 0.02), but was negatively correlated with all species FQI (r = -0.59, P =0.03).

CCA for Auchenorrhyncha and Plant Cover Variables

Out of the 12 plant variables, only shrub cover significantly explained variation along the first CCA axis (P = 0.0014; Figure 3.1). This first axis explained 34.6% of the variation. Tree cover significantly explained variation along the second axis (P = 0.025; Figure 3.1). This second axis explained 25.5% of the variation.

This analysis showed that most conservative auchenorrhynchan species, such as *Bruchomorpha dorsata*, *B. tristis*, *Flexamia pectinata*, and *Laevicephalus unicoloratus* were either negatively associated with shrub and tree cover or were not responding to these variables (Figures 3.1). Non-conservative species, such as *Exitianus exitiosus*, *Endria inimica*, and typhlocybine leafhoppers (e.g., *Dikraneura* spp., *Forcipata loca*) were positively associated with shrub, and tree cover. Also the ordination diagram (Figure 3.1) clearly demonstrated that many conservative auchenorrhynchan species were positively associated with perennial C4 grass cover.

CCA for Auchenorrhyncha and Plant Species Richness Variables

Out of the 12 plant variables, perennial forbs species richness significantly explained variation along the first axis (P = 0.021; Figure 3.2). This first axis explained 36.0% of the variation. Perennial C4 grass species richness significantly explained variance along the second axis (P = 0.01; Figure 3.2). The second axis explained 26.7% of the variation in the ordination.

This analysis showed that most conservative auchenorrhynchan species such as *Bruchomorpha dorsata*, *Flexamia pectinata*, *F. prairiana*, and *L. unicoloratus* are positively associated with the perennial grass C4 species and negatively associated with perennial forbs species (Figure 3.2). A few conservative species, such as *Polyamia apicata*, *L. melsheimeri* and *L. minimus* were slightly negatively associated with perennial C4 grasses and positively associated with biannual forbs (Figure 3.2). Most non-conservative species were strongly associated with forbs, shrubs, sedges, and low quality sites and negatively associated with perennial C4 grass species (Figure 3.2).

CCA for Auchenorrhyncha and Plant Diversity Variables

Out of the eight plant diversity variables, native species FQI significantly explained variance along the first axis (P = 0.021; Figure 3.3). This first axis explained 31.8% of the variance in the ordination. The all plant species meanCC significantly explained variance along the second axis (P = 0.041; Figure 3.3). This second axis explained 25.0% of the variance in the ordination.

This analysis showed that most conservative auchenorrhynchan species are positively associated with total plant species meanCC rather than native species FQI and other measures of diversity (Figure 3.3). Some moderately conservative species, such as *Mesamia nigridorsum*, *Memnonia flavida*, and *L. unicoloratus* are positively associated with diversity measures, such as the all plant species Shannon-Wiener diversity index, and native and all species FQIs (Figure 3.3). On the opposite end of the spectrum, most non-conservative species, such as *Forcipata loca*, *Balclutha neglecta*, and *Endria inimica* are positively associated with exotic plant species richness and negatively associated with biotic indices and plant diversity measures (Figure 3.3). However, a few conservative species, such as *P. apicata* and *L. melsheimeri* are positively associated with exotic plant species and low quality sites and negatively associated with total plant species meanCC, native FQI, and other plant diversity measures (Figure 3.3).

CCA for Auchenorrhyncha and Bare Ground and Litter Variables

Out of the bare ground and litter variables, litter significantly explained variance along the first axis (P = 0.021; Figure 3.4). The first axis explained 25.6% of the variance in the ordination. Bare ground explained 15.6% of variance along the second axis (Figure 3.4) in the ordination.

This analysis showed that most Auchenorrhyncha are not affected by litter or bare ground (Figure 3.4). However, a few conservative species, such as *Scaphytopius dorsalis* and *Laevicephalus unicoloratus* are negatively responding to both variables. Other conservative species, such as *Bruchomorpha tristis*, *Delphacodes caerulata*, *Pendarus punctiscriptus*, and *Texananus decorus* are positively associated with litter and negatively associated with bare ground. Some conservative species, such as *Memnonia flavida*, and *Polyamia apicata* are positively associated with bare ground and negatively associated with litter (Figure 3.4). Other non-conservative species, such as *Ceratagallia agricola*, *Forcipata loca*, *Limotettix anthracinus*, *Graphocephala hieroglyphica*, *Scaphytopius frontalis*, and *Draeculacephala antica* are positively associated with bare ground and negatively associated with litter (Figure 3.4).

DISCUSSION

One of the objectives of this study was to determine if a habitat quality index based on Auchenorrhyncha or vegetation could discriminate hill prairie quality. Although results showed that few vegetation and Auchenorrhyncha variables were able to statistically discriminate hill prairie quality, both groups of organisms reflected the INAI classification (Table 3.2 and Table 3.3). In the case of Auchenorrhyncha, data showed that high quality sites harbor more conservative Auchenorrhyncha on average, followed by mid quality sites, then low quality sites, thus partly supporting my hypothesis (H_o: Auchenorrhyncha integrity/diversity discriminates hill prairie quality).

Several explanations can be provided for the lack of statistical significance among site quality for most of the vegetation and Auchenorrhyncha variables: 1) INAI habitat classification is subjective, whereby land managers made individual assessments of the overall condition of natural remnants based on the successional stage of the vegetation present on these sites and presence of disturbance (White 1978); 2) additional sites within each INAI classification may be needed to increase statistical power to discriminate glacial-drift hill prairie quality; 3) there was too much similarity in perennial C4 grass cover and plant species composition between high and mid quality sites resulting in similar conservative auchenorrhynchan species composition at these sites, thus suggesting that not enough differentiation in plant and Auchenorrhyncha species exist to discriminate between high and mid quality sites (Whitcomb *et al*. 1987, Hamilton 1995, and Hamilton and Whitcomb 2010); and 4) the possible homogenizing effects of burning on high and mid quality sites. Since most of these sites were burned 2 to 3 years prior to sampling (Table 3.1), Auchenorrhyncha fauna may not have had time to recover post-fire, and thus more highly vagile, host- and habitat-generalist species such as typhlocybine leafhoppers and a few conservative species, such as *Bruchomorpha dorsata*, *Flexamia pectinata*, and *Laevicephalus minimus* were able to re-colonize these sites, producing a similar species composition. This trend was observed in Harper *et al*. (2000) study, in which habitat-generalist Auchenorrhyncha species and a few conservative species, such as *Laevicephalus minimus* were able to re-colonize a previously burned prairie.

Conversely, exotic plant species, shrub species richness, vine cover, vine species richness, tree cover, and perennial C4 grass species were richness were able to discriminate high and mid quality hill prairies from low quality hill prairie. These findings are similar to several studies, which have found pronounced decreases in hill prairie vegetation as tree and shrub encroachment intensifies (McClain and Anderson 1990, Robertson *et al*. 1995, and Taft *et al*. 2009). There is also evidence that elimination of management, for example prescribed fire, brush and tree removal, and mowing, reduces species richness of prairie vegetation (Leach and Givinish 1996).

In addition to these plant functional groups, three meanCC values were also able to discriminate glacial-drift hill prairie quality: total plant meanCC, native plant meanCC and total grassland Auchenorrhyncha meanCC$_{w/N}$. These Auchenorrhyncha and vegetation meanCC integrity indices were the most sensitive in differentiating levels of habitat quality compared with other measures of Auchenorrhyncha and vegetation integrity (AQI, FQI). One explanation may be site area. These glacial-drift hill prairies were relatively similar in area supporting similar numbers of species. Consequently, habitat quality indices, such as the AQI and FQI that take into account species richness, may yield similar AQI and FQI values among these high, mid, and low quality sites. On the other hand, Auchenorrhyncha and vegetation meanCC indices ignore species richness, and thus the discriminating ability of these measures may not be diminished when site area is similar. Therefore, Auchenorrhyncha and vegetation meanCC may be more appropriate in assessing habitat quality when area is similar, but equalizing sampling effort (e.g. number of transects) is needed.

The second objective of this study was to examine which plant variables are correlated with Auchenorrhyncha integrity and diversity. My results show that Auchenorrhyncha integrity and diversity were strongly correlated with perennial C4 grass cover and species richness. These data support my hypothesis that perennial C4 grasses are indicators of native grassland Auchenorrhyncha integrity and diversity. These results agree with data from other studies that show many native grassland Auchenorrhyncha species displaying some degree of specificity to perennial C4 grasses (Whitcomb *et al*. 1987, 1988, Whitcomb and Hicks 1988, Hamilton 1995, 2004, 2005).

Total plant species meanCC was also identified as a predictor of Auchenorrhyncha integrity, particularly conservative species. The total plant species meanCC exhibited a strong positive association with conservative Auchenorrhyncha because this index is partly computed from perennial C4 grasses, such as *Bouteloua curtipendula*, *Sporobolus heterolepis*, *Schizachyrium scoparium*, and *Andropogon gerardii*, which are hosts for many conservative Auchenorrhyncha (Whitcomb *et al*. 1988).

A negative correlation was found between shrub cover, tree cover, and perennial forb species richness and Auchenorrhyncha integrity, suggesting that these plant variables may be appropriate surrogates of low Auchenorrhyncha integrity. These functional groups may be out-competing native perennial grasses for resources, and thus reduce host plant availability of native grassland Auchenorrhyncha. Ultimately, as more ruderal and pioneer plant species invade these prairies, it may allow for more niches to become available to pioneer and generalist auchenorrhynchan species, which prefer these, host plants. Biederman (2005) found a similar trend in European grasslands.

Changes in vegetation and their effect on Auchenorrhyncha community assemblages have been shown in other studies. For example, Hollier *et al*. (1994) found that univoltine and sedentary species replace mobile bi- or multi-voltine species as fallow fields succeed to grassland. There are parallel changes in other life history traits as succession advances, such as the shift from brachyptery to macroptery (Denno *et al*. 1991) and a change from host plant specialists to generalists (Novonty 1994, Biederman 2005).

Other variables that were negatively associated with Auchenorrhyncha integrity (e.g., $\text{AQImeanCC}_{\text{w/outN}}$) and diversity (e.g., all auchenorrhynchan species Shannon-Wiener) include the native vegetation Shannon-Wiener diversity index. Unlike shrub cover, tree cover, and perennial forbs species richness, these plant variables may be inappropriate surrogates of Auchenorrhyncha integrity because they ignore differences in species composition (Shannon-Weaver 1949). If land managers were to use these variables in predicting Auchenorrhyncha integrity they may conserve prairie remnants with high Auchenorrhyncha diversity, but depauperate in conservative auchenorrhynchan species.

Lastly, although several plant variables (e.g., shrub cover, tree cover, exotic plant species richness, and perennial C3 grass species richness) and bare ground cover were poor predictors of Auchenorrhyncha integrity they were positively correlated with Auchenorrhyncha abundance, Simpson's D, and the Shannon-Weiner index. However, these Auchenorrhyncha related indices and measures ignore ecological characteristics (host plant and habitat preferences) and tolerance to varying levels of degradation, and thus may not be adequate candidates in reflecting Auchenorrhyncha integrity. Furthermore, Auchenorrhyncha diversity may be positively associated with increases in trees, shrubs, perennial C3 grasses, bare ground, and exotic plants because these variables may increase habitat structure allowing for more niches to become available (Hollier *et al*. 1994, 2005); although, these niches may be occupied by more pioneer, adventives, and other species that are more tolerant to disturbance.

The CCA analyses also showed that least conservative auchenorrhynchan species were positively associated with trees, shrubs, forbs, and exotic plants and negatively associated with perennial C4 grasses (Figures 3.1, 3.2). Conversely, the majority of conservative Auchenorrhyncha were positively associated with perennial C4 grasses (e.g., little blue stem, side oats gramma grass, indian grass, and big blue stem) and negatively associated or were not affected by trees, shrubs, forbs, and exotic plants (Figures 3.1, 3.2, Table 3.4). These results were consistent with Pearson's product mean correlation analysis and supported the hypothesis that perennial C4 grass species richness and abundance are strongly associated with of native grassland Auchenorrhyncha integrity. These results also agree with other studies (Whitcomb *et al*. 1987, Nickel 2003, Hamilton 2004, 2005) which showed a strong association between native grassland Auchenorrhyncha and perennial C4 grasses.

Several plant diversity measures were also positively associated with conservative Auchenorrhyncha. These measures include: all plant species meanCC, and native plant species FQI (Figure 3.3). Of these measures, highly conservative Auchenorrhyncha were positively associated with native plant species meanCC, while moderately conservative species were associated with native plant species FQI (Figure 3.3). The total plant species meanCC may be a better predictor of conservative Auchenorrhyncha because it only takes into account plant species composition and not species richness (Taft *et al*. 1997, Matthews 2003, Bourdaghs *et al*. 2006).

Conversely, adventives and generalist Auchenorrhyncha species (e.g., *Delphacodes puella*, *Ceragallia agricola*, and *Balclutha neglecta*) were positively associated with exotic plant species richness and diversity measures (e.g., Shannon-Wiener) and negatively associated with both total plant meanCC and FQI (Figure 3.3). These species have the ability to switch from native to exotic host plants (Whitcomb *et al*. 1987), explaining their strong positive association with exotic plant species richness. However, their positive association with high plant diversity (Figure 3.3) may be attributed to the mid and low quality sites having a combination of forest and prairie plants. For example, Robinson North B and Forest Park (Table 3.1), plant species composition is a mix of forest and grassland plant species resulting in higher plant species richness, but fewer conservative plant species. As a consequence of the encroaching forest on these sites, more low to moderate conservative, polyphagous Auchenorrhyncha are being supported as opposed to remnant dependent Auchenorrhyncha that are restricted to one or a few prairie plants (Nickel 2003, Biedermann 2005).

Variables that were not strongly associated with either conservative or non-conservative Auchenorrhyncha were litter and bare ground, suggesting that these variables are poor predictors of Auchenorrhyncha integrity (Figure 3.4). Lack of an association between litter and Auchenorrhyncha species was unexpected since studies have documented that many conservative grassland Auchenorrhyncha overwinter in dead vegetation (Whitcomb *et al*. 1987, Hamilton 1995, Reed 1997, Panzer 2002). The lack of an association between auchenorrhynchan species and the amount of litter in this study could be explained by the surrounding forest, which deposited forest litter, providing overwintering habitat for many non-conservative Auchenorrhyncha. As a consequence, many habitat generalist-Auchenorrhyncha, such as *Liburniella ornata*, *Limotettix anthracinus*, and *Neocoelidia tumidifrons* may have obscured the association between Auchenorrhyncha and litter (DeLong 1948, 1965).

There were a few conservative species that were negatively associated with perennial C4 grass species richness (Figure 3.2) and total plants species meanCC (Figure 3.3) and positively associated with perennial forb or biennial forb species richness (Figure 3.2) and cover (Figure 3.1), and exotic plant species richness (Figure 3). These species include: *Memnonia flavida*, *Polyamia apicata*, *Laevicephalus minimus*, and *L. melsheimeri*. *L. minimus* and *M. flavida* feed on *Bouteloua curtipendula* (Hamilton 2000, 2004) and *Polyamia apicata* feeds on *Panicum villosissimum* (DeLong 1948). Although, these grasses are moderate to highly conservative species (Taft *et al*. 1997), they may not compete well with other grasses, favoring habitats that are often dominated by forbs (personal communication Paul Marcum - Illinois Natural History Survey). Since their host plants occur near forbs rather than grasses, this may explain their negative association with these grasses. On the other hand, *L. melsheimeri* feeds exclusively on *Danthonia spicata* (poverty oats grass) which is a shade-tolerant grass species and is often found along the border of forests and grasslands. Their strong association with *D. spicata* may explain their negative association with perennial C4 grass cover (Figure 3.1) and species richness (Figure 3.2), as well as meanCC derived from all plant species (Figure 3.3).

Both Pearson's product mean correlations and CCA identified several plant variables that were strongly associated with conservative Auchenorrhyncha species. These variables include perennial C4 grass cover and species richness, and total plant species meanCC. Perennial forbs were also strongly associated with several conservative Auchenorrhyncha because their host plants are poor competitors with other native perennial grasses than with forbs. Adventives auchenorrhynchan species were strongly associated with exotic species richness, shrub cover and species richness, tree cover and species richness, annual forb cover and species richness, and biennial forbs cover and species richness. These results indicate that the relative condition of native grassland Auchenorrhyncha integrity can be obtained from measuring the amount of perennial C4 grass cover and species richness. Thus, efforts to conserve and restore Auchenorrhyncha integrity should aim at the restoration or conservation of this plant functional group.

CONCLUSIONS

These results show that species composition of plants and Auchenorrhyncha were similar on high and mid quality sites, suggesting that FQI and AQI may only be able to discriminate quality between high and low quality sites in these glacial-drift prairies. Additional studies are needed to determine whether the AQI and its corresponding mean coefficient of conservatism can discriminate quality on other prairie communities (i.e., sand, mesic, other hill prairie communities) as they have considerably different fauna and flora, and thus AQI (and FQI) could discriminate quality within these prairies.

My results show a strong association between perennial C4 grasses and conservative Auchenorrhyncha, particularly in high and mid quality sites. Conversely, perennial C4 grass species richness and abundance (e.g., side oats gramma and little bluestem) was low on low quality sites compared to high and mid quality sites, and thus many conservative auchenorrhynchan species (e.g., *Laevicephalus minimus*) that are restricted to these grasses (Harper *et al*. 2000, Panzer 2002, Hamilton and Whitcomb 2010) were absent or had small population sizes on these sites (Appendix B.1). Hence, to maintain the prairie Auchenorrhyncha integrity on high and mid quality Illinois glacial-drift hill prairies, conservation and restoration efforts need to focus on maintaining trhe perennial C4 grass community. Improving Auchenorrhyncha integrity on low quality sites should also emphasize on restoring the perennial C4 grass community. Additional land management may also be needed in improving these mid and low quality sites. For example, judicious use of prescribed burning (Panzer 2002), shrub and tree removal, and grazing may be needed to prevent extensive encroachment of woody, invasive, and exotic species from displacing these prairie Auchenorrhyncha and vegetation. However, the frequency, intensity, and during of these prescribed burns on glacial-drift hill prairies requires additional testing. Reintroductions of Auchenorrhyncha may also be needed in establishing populations of conservative Auchenorrhyncha that may have been extirpated on these low quality sites and on high and mid quality sites that have undergone extensive burn management.

Lastly, evidence has shown that fire was once a natural process in the maintenance of grasslands (Collins and Gibson 1990). Since many of these prairie auchenorrhynchan species are fire-sensitive (Panzer and Schwartz 2000), suggests that in order for these fire-sensitive insects to survive a burn, glacial-drift hill prairies must have had unburned and burned prairie. Unburned prairies may have been refugia for fire-sensitive insects, allowing these insects to re-colonize recently burned prairie. Therefore, including unburned areas (i.e., refugia) into the current management strategy of glacial-drift hill prairies may be beneficial in conserving these fire-sensitive insects.

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TABLES AND FIGURES

Table 3.1 Illinois Natural Area Inventory (INAI) grades (based on plant species composition and presence of degradation to the site) and land management history of 14 nature preserves that are located in Peoria and Marshall Counties, IL.

Sites	General Quality	INAI Grade	Management
1. Robinson South	Mid	B/C+	Burned 3 years ago
2. Detwieller South	Low	C/D	Unburned
3. Robinson Park North C	Mid	B/B-	Unburned
4. Singing Woods	High	A	Burned 2 years ago
5. Gentiana 1	High	A	Burned 3 years ago
6. Camp Wokanda	Mid	B/C+	Burned 2 years ago
7. Hopewell	High	A	Burned 3 years ago
8. Robinson Park North B	Low	D	Burned 10 years ago
9. Gentiana 2	Mid	B	Burned 2 years ago
10. Wier	Mid	B	Burned 3 years ago
11. Forest Park	Low	C	Burned 10 years ago
12. Detwieller North	Mid	B	Burned 10 years ago
13. Robinson Park North	Low	C	Burned 16 years ago
14. Peoria Park Prairie	Mid	B	Shrub removal 2 years ago

Table 3.2 Mean and standard error of all plant variables used in discriminating glacial-drift hill prairie quality. Variables in bold with different letters indicate significant differences among high, mid, and low quality glacial-drift hill prairies, $p < 0.05$ (Tukey's test).

Variables	High	Mid	Low
Exotic Species	**1.33 ± 0.88^b**	**1.57 ± 0.57^b**	**6.50 ± 0.65^a**
Native mean CC	**5.12 ± 0.06^a**	**4.96 ± 0.13^a**	**4.14 ± 0.18^b**
meanCC$_{total}$	**5.00 ± 0.06^a**	**4.79 ± 0.14^a**	**3.62 ± 0.22^b**
Tree Sapling Cover	**4.16 ± 1.70^c**	**9.84 ± 2.21^b**	**32.60 ± 10.04^a**
Vine Cover	**0.03 ± 0.02^b**	**0.75 ± 0.73^b**	**12.29 ± 4.93a**
Perennial Grass C4 Species	**4.33 ± 0.33^a**	**4.57 ± 0.48^a**	**2.25 ± 0.48^b**
Shrub Species	**2.33 ± 0.33^b**	**2.43 ± 0.57^b**	**6.50 ± 0.87^a**
Vine Species	**0.67 ± 0.33^b**	**0.71 ± 0.57^b**	**4.50 ± 0.96^a**
Perennial Forb Cover	202.67 ± 22.26	185.36 ± 29.84	122.98 ± 12.23
Native Species	44.67 ± 3.29	47.43 ± 6.31	52.50 ± 7.26
FQI$_{total}$	33.42 ± 1.56	32.7 ± 2.32	26.64 ± 3.15
Perennial C3 Grass Cover	22.72 ± 5.13	12.80 ± 4.42	13.83 ± 3.50
Perennial C4 Grass Cover	176.03 ± 48.33	138.43 ± 26.28	78.36 ± 43.17
Shrub Cover	43.85 ± 9.92	16.03 ± 6.44	59.05 ± 21.74
Perennial Forb Species	31.67 ± 2.73	30.14 ± 3.80	27.50 ± 5.81
Perennial Grass C3 Species	1.33 ± 0.33	3.43 ± 1.13	6.00 ± 1.08
Tree Sapling Species	5.00 ± 1.73	5.00 ± 1.07	7.75 ± 0.48

Table 3.3 Mean and standard error of all Auchenorrhyncha (i.e. Auch) variables used in discriminating glacial-drift hill prairie quality. Variable in bold with different letters indicate significant differences among high, mid, and low quality glacial-drift hill prairies, $p < 0.05$ (Tukey's test).

Variables	High	Mid	Low
All Auch. $AQI_{w/N}$	33.49 ± 3.93	29.27 ± 1.97	24.54 ± 4.96
All Auch. AQI	37.73 ± 6.51	33.00 ± 2.70	25.75 ± 7.27
All Auch. $meanCC_{w/N}$	7.57 ± 0.28	7.28 ± 0.36	5.97 ± 0.40
All Auch. $meanCC_{w/outN}$	10.87 ± 0.75	9.68 ± 0.61	7.56 ± 1.05
Grassland $AQI_{w/outN}$	37.72 ± 6.51	33.00 ± 2.70	25.75 ± 7.27
Grassland $AQI_{w/N}$	33.49 ± 3.94	29.27 ± 1.97	24.54 ± 4.96
Grassland Auch. $meanCC_{w/outN}$	**7.57 ± 0.28^{a}**	**7.28 ± 0.36^{a}**	**5.97 ± 0.40^{b}**
Grassland Auch. $meanCC_{w/N}$	**9.80 ± 0.25^{a}**	**8.54 ± 0.34^{b}**	**7.54 ± 0.68^{c}**

Table 3.4 Complete list of significant and non-significant variables that explained variance in CCA ordinations. Variables in bold were identified by the Monte Carlo permutation test as significantly explaining most of the variance in the ordination.

Significant Variables	Non-Significant Variables
Tree Seedling Cover (P = 0.03)	Bieannual Forb Cover
Shrub Cover (P = 0.00)	Annual Forb Cover
Perennial C4 Grass Species Richness (P = 0.01)	Annual C4 Grass Cover
Perennial Forb Species Richness (P = 0.02)	Perennial C4 Grass Cover
All plant Species meanCC (P = 0.04)	Bieannual Forb Species Richness
Native Plant Species FQI (P = 0.02)	Shrub Species Richness
Litter (P = 0.02)	Perennial C3 Grass Species Richness
	Sedge Species Richness
	Annual Forb Species Richness
	Native Plant Species meanCC
	All Plant Species FQI
	All Plant Species Shannon-Wiener
	Exotic Plant Species Richness
	Bare Ground

Figure 3.1 Ordination of Auchenorrhyncha species abundance, sites, and plant cover variables on Axis 1 and Axis 2 of a Canonical Correspondence Analysis (CCA) with scaling based on inter-species distances (see Appendix 1 for list of Auchenorrhyncha species and their corresponding acronyms). Variables that significantly explained most of the variance are labeled with asterisks. Non-significant variables include biannual forb cover (biforbcover), annual forb cover (annualforbcover), annual C4 grass cover (AC4cover), and perennial C4 grass cover (PC4 cover). Low quality sites are shown with solid green symbols, mid quality sites are shown star symbols, and high quality sites are shown with red diamond symbols. Acronyms for species are first three letters of genus and species.

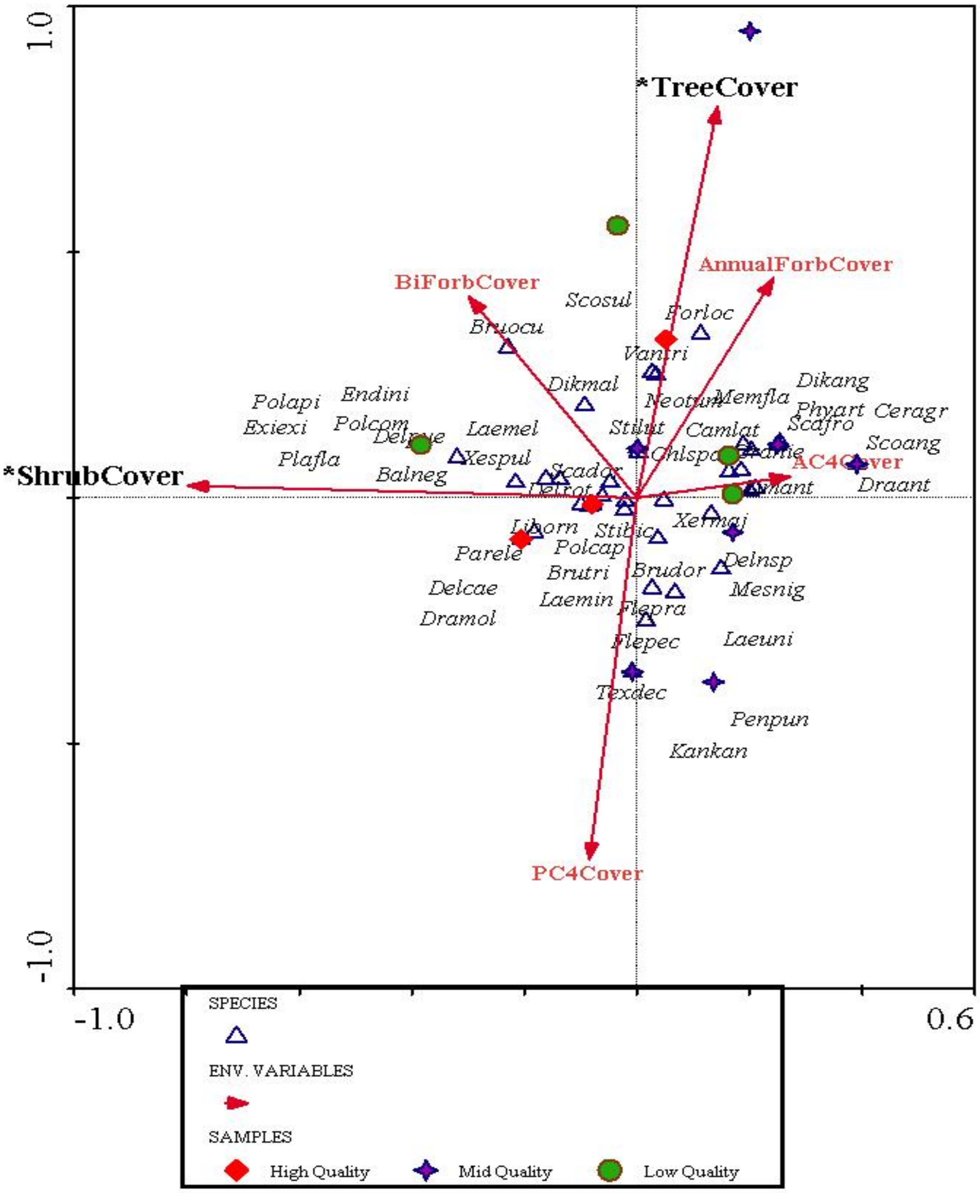


Figure 3.2 Ordination of Auchenorrhyncha species abundance, sites, and plant species richness variables on Axis 1 and Axis 2 of a Canonical Correspondence Analysis (CCA) with scaling based on inter-species distances (see Appendix 1 for list of Auchenorrhyncha species and their corresponding acronyms). Plant variables that significantly explain most of the variance in the ordination are perennial C4 grass species richness (PC4), and perennial forb species richness (PForbSp). Non-significant variables include biannual forb species richness (biforbSp), shrub species richness (ShrubSp), perennial C3 species richness (PC3Sp), sedge species richness (SedgeSp), and annual forb species richness (AForbSp).

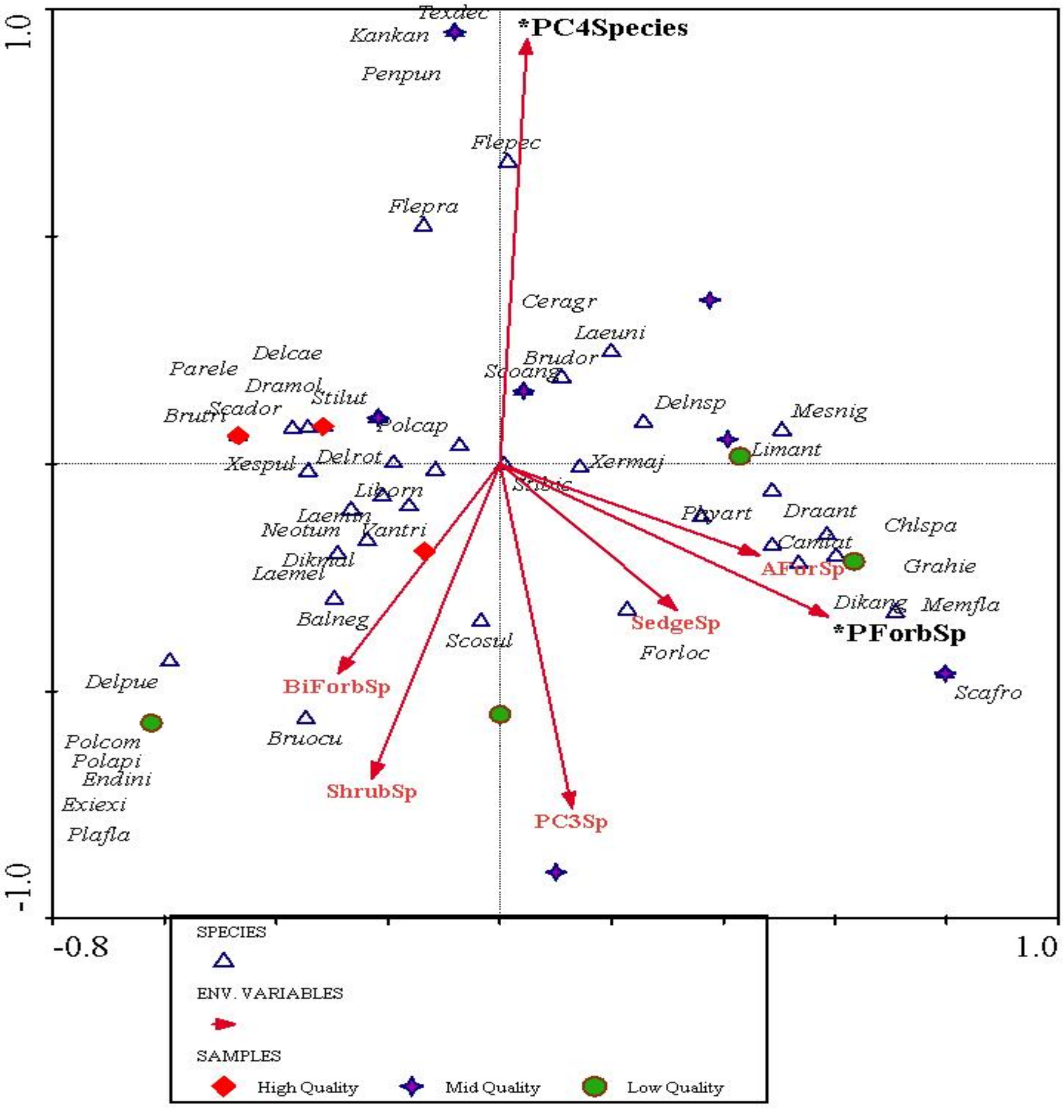


Figure 3.3 Ordination of Auchenorrhyncha species abundance, sites, and plant diversity and biotic index variables on Axis 1 and Axis 2 of a Canonical Correspondence Analysis (CCA) with scaling based on inter-species distances (see Appendix 1 for list of Auchenorrhyncha species and their corresponding acronyms). Plant variables that significantly explained most of the variance in the ordination are meanCC (all plant species) and FQI (native plant species). Non-significant variables include meanCC (native plant species), FQI (all plant species), Shannon-Wiener diversity index (Shannon), and exotic plant species richness (exotics).

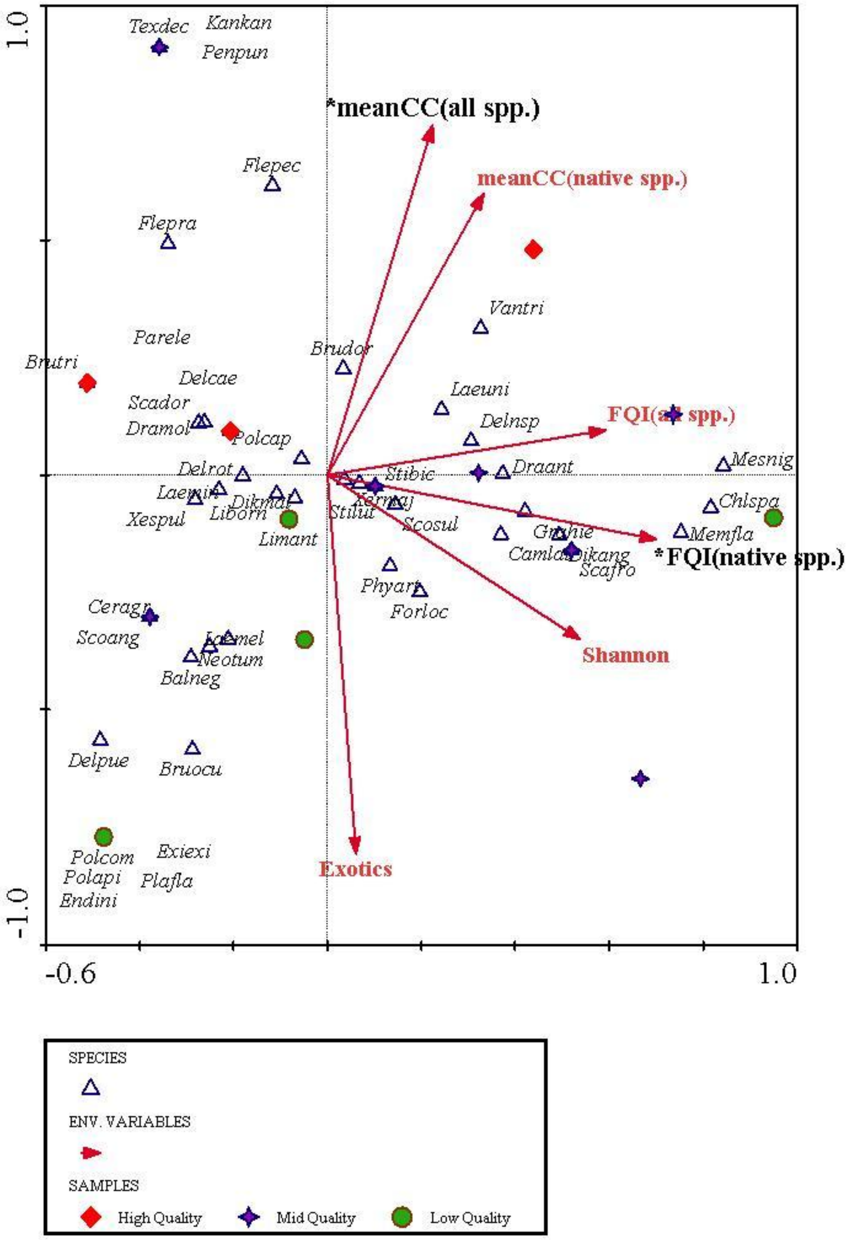
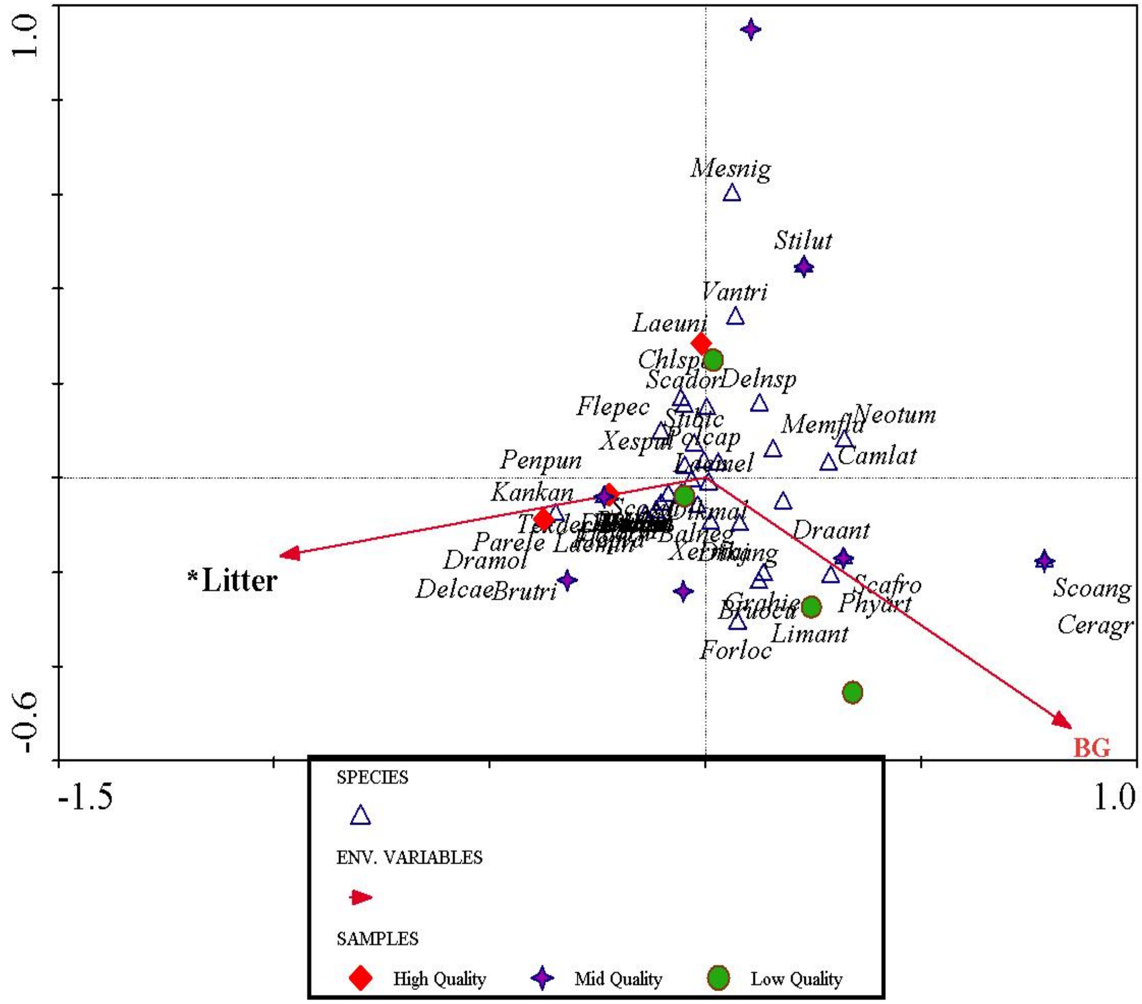


Figure 3.4 Ordination of Auchenorrhyncha species abundance, sites, and landscape variables on Axis 1 and Axis 2 of a Canonical Correspondence Analysis (CCA) with scaling based on inter-species distances (see Appendix 1 for list of Auchenorrhyncha species and their corresponding acronyms). Landscape variable that significantly explained the most variance in the ordination is litter cover. Non-significant landscape variable is bare ground cover (BG).



Chapter 4: The Influence of Fire on Illinois Hill Prairie Auchenorrhyncha (Insecta: Hemiptera) Diversity and Integrity

ABSTRACT

Prescribed burning has been important in maintaining the structure of plant communities in the tallgrass prairie. However, implementation of these burn regimes often overlooks responses of other taxa, particularly arthropods. In this study, the timing and frequency of burns were examined on one of the most diverse and abundant groups of herbivorous insects, Auchenorrhyncha. These insects are ideal candidates in understanding the effects of fire on prairie arthropods because they are among the most numerous invertebrate herbivores in the prairie, including many species restricted to prairie habitats and they have ecological characteristics that confer a wide range of responses to prescribed burning. A total of 22 Illinois hill prairies were sampled along the Mississippi and Sangamon Rivers in the summer of 2006 using a modified leaf-blower vacuum. These sites exhibited a wide range of burn management, from unburned to recently burned, and having been burned multiple times. Species richness, Auchenorrhyncha Quality Index (with and without abundance data) and the mean coefficient of conservatism (with and without abundance data) were calculated for each site. Results suggest that unburned sites supported the greatest number of species and had higher Auchenorrhyncha Quality Index and mean coefficient of conservatism values than sites undergoing burn management. Although these data strongly suggest that recent burning and frequent burning on loess hill prairies can negatively affect the prairie Auchenorrhyncha community, the elimination of prescribed fire may erode prairie vegetation integrity. It is recommended that rotational burns of 3 years are needed to conserve the vascular plant and Auchenorrhynchya community assemblages.

INTRODUCTION

Historically, fire has played a significant role in the maintenance of North American tallgrass prairies (Gleason 1913, Anderson 1990, Collins 1990, Robertson *et al*. 1997). Burning can prevent invasion of trees and shrubs (Gibson and Hulbert 1987) and reduce the spread of invasive non-prairie species (Anderson 1972, Wilson and Stubbendieck 1997). Burning also can stimulate plant productivity, increase native perennial grass abundance, and aid in the production of seedlings (Anderson 1965, Glenn-Lewin *et al*. 1990, Seastedt and Ramundo 1990); although these increases in plant productivity is dependent on the timing of fire (Bragg 1995, Jog *et al*. 2006). Elimination of fire on prairie remnants has lead to an annual loss of 0.45 – 1.03% of plant species (Leach and Givnish 1996), potentially reducing ecological integrity of these remnants. In an attempt to reverse these changes fire has been re-introduced or increased on these remnants under highly prescribed conditions (Johnson *et al*. 2008). Controlled burning has now become a frequently used management tool for improving prairie remnant quality (Wright and Bailey 1980, Collins and Gibson 1990).

Despite these advantages prescribed burning has had a wide range of effects on other diverse and abundant groups of prairie organisms, in particular terrestrial arthropods (Rice 1932, Canceldo and Yonke 1970, Reichert and Reeder 1970, Nagel 1973, McCabe 1981, Opler 1981, Hansen 1986, Anderson et al. 1989, Bock and Bock 1991, Dana 1991, Orwig 1992, Fay and Samenus Jr. 1993, Hamilton 1995, Swengel 1996, Reed 1997, Harper *et al*. 2000, Panzer and Schwartz 2000, Swengel 2001, Panzer 2003). Swengel (1996) observed that most conservative butterflies inhabit sites that have not been burned for 4 or more years, whereas highly vagile butterflies, such as orange sulfurs, monarchs, and cabbage whites were common on burned sites.

Evans (1984, 1988) found that sites unburned for 4 years or more supported more fire sensitive, forbs-feeding grasshoppers whereas grasshoppers specializing on grasses were more abundant on sites burned annually or biennially. On the other hand, Panzer (2002) found that most prairie-restricted insects, from seven insect orders, re-colonized recently burned sites after 2 years or less. Carabid beetles (Larsen and Williams 1999), prairie ants (Trager 1990), and spiders (Halvorsen and Anderson 1980) were unaffected by burning.

Because terrestrial arthropods display a great degree of variation in response to controlled burning, no single burn management program to protect all arthropods has been successful. For instance, some studies show that 3-year rotational burns (Reed 1997) are adequate, other studies have found that 2-year rotational burns are successful (Panzer 2002), and other studies have found that a combination of rotational burning combined with an unburned refuge were effective in conserving many prairie-inhabiting butterflies (Swengel and Swengel 2006). Additional studies based on other diverse groups of prairie arthropods are required to provide land managers with a more complete understanding of the role of fire on terrestrial arthropod communities. This information would aid land managers in developing burn management practices that are less harmful to terrestrial arthropods, and in so doing conserve a greater proportion of the prairie biota.

The purpose of this study is to determine the effects of prescribed burning on Auchenorrhyncha (i.e., leafhoppers, planthoppers, spittle bugs, treehoppers, and cicadas), with the intent of providing new insights in the design and implementation of burn management practices in prairies. Auchenorrhyncha are ideal candidates in understanding the effects of fire on prairie arthropods because are they among the most numerous invertebrate herbivores in the prairie, including many species restricted to prairie habitats and they have ecological characteristics that confer a wide range of responses to prescribed burning (Hamilton 1995, Harper *et al*. 2000, Panzer 1988, Panzer and Schwartz 2000, 2002, 2003, Hamilton and Whitcomb 2010). More specifically, these insects will be used to examine the following questions: 1) what is the response of prairie Auchenorrhyncha integrity (i.e., the native species that should be present given that their host plants occur on Illinois loess hill prairies) and diversity after prescribed burns; and 2) how does burn frequency influence prairie Auchenorrhyncha integrity and diversity. Since many prairie Auchenorrhyncha exhibit life history traits (e.g., low mobility, overwinter in dead vegetation or duff, and have low numbers of generations) that make them vulnerable to fire (Hamilton 1995, Reed 1997, Nickel 2003, Hamilton and Whitcomb 2010), I hypothesize that prairie Auchenorrhyncha integrity and diversity will respond negatively to recent and frequent burning.

METHODS

Study Sites

I sampled 22 Illinois hill prairies along the Mississippi and Sangamon Rivers in the summer of 2006 (Figure 4.1). Study sites extended from Monroe County to Jo Daviess County. These sites ranged in size from 0.1 to 7 acres (Table 4.1). Many of these sites were maintained by controlled burn management: 7 sites had not been burned in 30 years (i.e. unburned), 3 sites were burned in the spring of 2006 (i.e. 0 years), 5 sites were burned in the spring of 2005 (i.e. 1 year), 3 sites were burned in the spring of 2004 (i.e., 2 year), and 4 sites were burned between the spring of 2001 and 2003 (i.e., 3 to 5 years) (Table 4.1). Also, these sites displayed a range in burn frequency: 5 sites were burned once, 4 sites were burned twice, and 6 sites were burned more than twice (Table 4.1). Besides timing and frequency of prescribed burns at the study sites, limited information was available regarding the implementation of these prescribed burns (e.g., rotational burns vs. entire unit) or other management practices (e.g., grazing). Land management history was provided by land managers and land owners.

Most of these sites were surrounded by secondary growth forest, with the exception of Snyder Hill Prairie (Table 4.1), which was surrounded by agriculture. Sites were dominated by prairie grasses, such as *Boueloua curtipendula* (Side Oats Grama), *Schizachyrium scoparium* (Little Bluestem), and *Andropogon gerardii* (Big Bluestem), with the exception of Delford Hill Prairie, which was dominated by forbs, such as *Aster pilosus* and *Liatris spicata*. Furthermore, the majority of these sites, such as hill prairies in Monroe and Randolph counties (Figure 1), was once part of a large complex of sites with similar plant species composition (Evers 1955, Robertson *et al*. 1995), and thus should harbor a similar Auchenorrhyncha fauna that are comparable between sites.

Sampling Protocol

Three 40m-linear transects were placed perpendicular to a 50m baseline. Each transect was 5m apart from the neighboring transect. All transects were placed within the prairie away from edges, which increased the likelihood of collecting most of the prairie Auchenorrhyncha. Auchenorrhyncha were collected with a modified leaf blower vacuum for approximately 5 minutes and stored in 95% ethanol. Sampling took place between 1200 and 1800 hours, between the third week of July and second week of August, 2006, when Auchenorrhyncha abundance and species richness are at their peak (Blocker *et al*. 1972, Blocker and Reed 1976). All adult Auchenorrhyncha were identified to species, individuals were tallied, and specimens were deposited at the Illinois Natural History Survey Insect Collection. Auchenorrhyncha nomenclature followed DeLong (1948), Wilson and McPherson (1980), Whitcomb and Hicks (1988), Bartlett and Deitz (2000).

Weighted Indices of Auchenorrhyncha Integrity Variables

Auchenorrhyncha integrity was measured by computing the following variables:

S = total number of species encountered at a site.

AQI = meanCC*$\sqrt{S}$; MeanCC = Mean coefficient of conservatism value for all species encountered per sampling effort (site); $\sqrt{S}$ = Square root transformation of the total number of species encountered at the site sampled.

The AQI was also computed using abundance (i.e., AQIwN).

AQIwN = $\sum [(n_i/N) * CC_i] * \sqrt{S}$, in which n_i = total number of individuals for species i; N = total number of individuals for all species; CC_i = Coefficient of Conservatism for species I; $\sqrt{S}$ = square-root transformation of the total number of species encountered per sampling effort (site).

The CC values are calculated scores from 0 to 18 assigned to each auchenorrhynchous species encountered. Species that are adapted to frequent disturbance and feed and overwinter on a variety of host plants received values ranging from 0 to 5. Species that can tolerate moderate levels of disturbance, are found in edge habitat and in native grassland, and feed and overwinter on prairie and some non-prairie plants received values ranging from 6 – 10. Species that are sensitive to disturbance, restricted to native grassland, and feed and overwinter on native grassland vegetation received values ranging from 11 – l8 (Appendix C.1).

Data Analysis

One-way ANCOVAs using site area as covariant, followed by Tukey's *post-hoc* test were used to determine the impact of prescribed burns (i.e., timing and frequency) on Auchenorrhyncha integrity and diversity. Area was included in the analysis because studies have shown a direct relationship between species diversity and area (MacArthur and Wilson

1967, Simberloff and Wilson 1969), and thus area may influence Auchenorrhyncha integrity and diversity. Kolmogorov-Smirnov test was used to test for normality and the Levene's test was used to examine if the data had equal variance. Both tests were performed with Sigma Stat 3.1 (2004). Delford and Snyder prairies were identified as outliers using the statistical software package SYSTAT 11.0. Also, Delford was dominated by forb species and Snyder was not surrounded by secondary growth forest, which can influence the auchenorrhynchan species composition (Hamilton 1995). Based on the statistical and ecological evidence Delford and Snyder prairies were removed from the analysis. Windfall prairie, east-central Illinois, was also excluded from the analysis because it is not considered part of the Mississippi/Sangamon hill prairie complex and therefore it is less likely that its original Auchenorrhyncha fauna was comparable to those of the more western hill prairies. Changes in Auchenorrhyncha species composition among different prescribed burn management treatments were also documented.

RESULTS

I collected 76 auchenorrhynchan species on these 22 sites, from 53 genera and 11 families (Appendix C.1). Of these, 73 (96%) were native, while 3 (4%) were non-native. Of the native taxa, approximately 30 species (39%) are considered remnant dependent and sensitive to disturbance; 19 species (25%) can tolerate a moderate amount of disturbance and are associated with prairies, ecotonal habitat, and non-prairie habitat; and 27 species (36%) are adapted to frequent disturbance and are not denizens of prairies (Appendix C.1). Numbers of species per family were: Cicadellidae (48), Delphacidae (6), Dictyopharidae (6), Caliscelidae (4), Cercopidae (2), Membracidae (2), Acanalinoiidae (1), Achilidae (1), Cicadidae (1), and Fulgoridae (1). The most speciose genus was *Delphacodes* with 5 species, followed by *Polyamia* with 4 species.

Unburned sites had significantly more species than recently burned sites (Figure 4.2) and frequently burned sites (Figure 4.3). Species richness differed based on time since last burn ($F = 4.87$, $P = 0.01$), but was not affected by site area ($F = 0.43$, $P = 0.52$). Species richness was significantly higher on unburned remnants (15.20 ± 1.90; Tukey's test, $P < 0.05$) compared to remnants burned most recently (0 years; 6.33 ± 0.33) and 1 year ago (7.50 ± 0.90), but not 2 years and (9.00 ± 1.15) and 3 to 5 years ago (15.67 ± 4.70; Figure 1). Species richness differed based on burn frequency ($F = 3.23$, $P = 0.06$), but was not affected by site area ($F = 0.50$, $P = 0.49$). Unburned sites (15.33 ± 1.54; Tukey's test, $P < 0.05$) had significantly more species than sites burned more than twice (6.60 ± 0.68), but not sites burned once (11.60 ± 3.43) and twice (10.00 ± 1.16).

Unburned sites yielded significantly higher $\text{meanCC}_{w/N}$ values than recently burned (Figure 4.2) and frequently burned sites (Figure 4.3). $\text{MeanCC}_{w/N}$ differed based on time since last burn ($F = 15.27$, $P = 0.00$), but was not affected by site area ($F = 0.07$, $P = 0.95$). $\text{MeanCC}_{w/N}$ was significantly higher in unburned prairie remnants (11.93 ± 0.27; Tukey's test, $P < 0.05$; Table 4.2) than sites burned most recently (0 years; 5.98 ± 0.52), but was not significantly greater than sites burned 1 year (10.54 ± 0.86), 2 years (11.13 ± 0.45), and 3 to 5 years ago (11.62 ± 0.58). $\text{MeanCC}_{w/N}$ differed based on burn frequency ($F = 7.21$, $P = 0.00$), but was not affected by site area ($F = 0.23$, $P = 0.88$). $\text{MeanCC}_{w/N}$ was significantly higher in unburned sites (11.93 ± 0.27) compared to sites burned more than twice (7.62 ± 1.20; Tukey's test, $P < 0.05$), but was not significantly greater than sites burned once (11.13 ± 0.34) or twice (11.52 ± 0.61).

Unburned sites showed significantly higher $\text{AQI}_{w/N}$ values than recently burned (Figure 4.2) and frequently burned sites (Figure 4.3). $\text{AQI}_{w/N}$ differed based on time since last burn (F =

16.13, P = 0.00), but was not affected by site area (F = 0.56, P = 0.47). $AQI_{w/N}$ values were significantly higher on unburned sites (46.20 ± 1.61; Tukey's test, P < 0.05; Figure 4.2) than sites burned 0 years (15.01 ± 1.22), 1 year (28.02 ± 3.27), 2 years (33.31 ± 2.86), and marginally greater than sites burned 3 to 5 years ago (44.83 ± 6.20; P = 0.06). $AQI_{w/N}$ values differed based on burn frequency (F = 8.66, P = 0.00), but were not affected by site area (F = 0.44, P = 0.52). Unburned sites (46.20 ± 1.61; Tukey's test, P < 0.05) yielded significantly higher $AQI_{w/N}$ values than sites burned more than twice (19.86 ± 4.12), but did not significantly differ from sites burned once (36.68 ± 5.32) or twice (36.28 ± 2.56).

Unburned sites had higher $meanCC_{w/outN}$ values compared to recently burned (Figure 4.2) and frequently burned sites (Figure 4.3). However, $meanCC_{w/outN}$ values did not differ based on time since last burn (F = 0.97, P = 0.46) and were not affected by site area (F = 0.16, P = 0.70). Also, $meanCC_{w/outN}$ values did not differ based on burn frequency (F = 2.22, P = 0.13) and site area (F = 0.17, P = 0.68).

Unburned sites yielded significantly higher $AQI_{w/outN}$ values compared to recently burned sites (Figure 4.2) and frequently burned sites (Figure 4.3). $AQI_{w/outN}$ values differed based on time since last burn (F = 15.02, P = 0.00) and were not affected by site area (F = 2.31, P = 0.15). $AQI_{w/outN}$ values were significantly higher on unburned sites (38.22 ± 1.72; Tukey's test, P < 0.05; Table 4.2) than sites burned at 0 years (20.41 ± 2.03), 1 year (26.49 ± 2.96), and 2 years ago (27.74 ± 1.38), but were not significantly different from sites burned 3 to 5 years ago (37.35 ± 0.50). $AQI_{w/outN}$ values were significantly higher on prairies burned 3 to 5 years ago compared to prairies burned recently (0 years) and 1 year ago, but not significantly greater than sites burned 2 years ago. $AQI_{w/outN}$ values differed based on burn frequency (F = 12.13, P = 0.00), but were not affected by site area (F = 1.58, P = 0.23). Unburned sites (38.22 ± 1.72; Tukey's test, P

< 0.05) yielded significantly higher $AQI_{w/outN}$ values than sites burned more than twice (21.77 ± 2.36) and once (30.19 ±2.17), but were not significantly greater than sites burned twice (34.23 ± 2.82).

I also detected declines in conservative (i.e., remnant-dependent and fire-sensitive) Auchenorrhyncha species and abundance as burn management changed from unburned to recently and frequently burned prairie management practices (Appendix C.1). For example, conservative auchenorrhynchan species, such as *Flexamia* spp., *Laevicephalus* spp., *Bruchomorpha* spp., and *Polyamia* spp.declined on recently burned sites compared to unburned sites and sites burned 3 to 5 years ago. I also observed increases in species and abundance of moderately-conservative species, such as *Draeculacephala* spp.and *Kansendria kansiensis*, as well in adventives species such as Typhlocybinae leafhopper species (*Empoasca* spp., *Erythroneura* spp.) in recently burned and frequently burned sites (Appendix 1). In contrast, these moderately-conservative and adventive species were absent or were represented by a few individuals on unburned sites, sites burned 3 to 5 years ago, and infrequently burned sites (Appendix C.1).

DISCUSSION

Overall Effects

The objective of this study was to determine the effects of prescribed burning on prairie Auchenorryncha integrity and diversity. The results from this study showed that unburned sites supported the greatest number of species, had the highest Auchenorrhyncha integrity, and area did not influence hill prairie Auchenorrhyncha integrity and diversity. These data support my hypothesis that unburned sites will have higher Auchenorrhyncha diversity and integrity than sites under constant fire management. Furthermore, these results are consistent with other studies that demonstrate the importance of fire in influencing the dynamics of prairie insect community assemblages and that prairie Auchenorrhyncha respond negatively to fire (Whitcomb *et al*. 1988, Hamilton 1995, Reed 1997, Harper *et al*. 2000, Panzer 2002, 2003, Hamilton and Whitcomb 2010).

Time since Burn Treatments

Unburned sites supported more conservative species as time since a burn increased (Figure 4.2, Appendix C.1). For instance, unburned sites harbored 41 (57%) out of the total 72 species encountered from all 19 sites used in the statistical analysis and 18 of these 41 species (44%) were dependent on prairies and intolerant to fire. Some of these species include *Bruchomorpha jocosa*, *B. tristis*, *Flexamia pectinata*, *F. prairiana*, *Laevicephalus minimus*, and *Polyamia dilata*. Similar compositions of conservative auchenorrhynchan species were observed on other unburned prairies in Kansas (Blocker *et al*. 1972), Illinois (Harper *et a*l. 2000, Panzer 2002, Hamilton 1995), and in the upper Midwest (Hamilton 1995, 2005). Other studies based on different insect taxa have found a similar pattern. For example, Swengel (1996) found a disproportionate number of prairie-remnant dependent butterflies on unburned prairies. The greater number of prairie-associated Auchenorrhyncha found on unburned sites could be the result of more structural complexity (i.e., more niches). Haysom and Coulson (1998) observed species richness of Lepidoptera associated with heather (*Calluna*) increased with more years since last fire, as a consequence of greater height and structural diversity of this plant.

Twelve out of the 41 (29%) species were moderately conservative, including *Acinopterus acuminatus*, *Cuerna costalis*, *Delphacodes rotundata*, and *Stirellus bicolor*. These species are common on native grasslands (DeLong 1948) and can occur on undisturbed and moderately disturbed prairies (Panzer 2002). These sites also harbored 11 adventive species (27%), which were represented by typhlocybine species, treehoppers, and spittlebugs. These species are common on relatively disturbed grasslands (Whitcomb *et al*. 1988, Hamilton 1995, Harper *et al*. 2000).

As time since a burn changed from unburned to 3 to 5 years the number of species decreased from 41 to 35, with 12 (34%) of these species being highly conservative, 13 (37%) moderately conservative, and 10 (29%) adventives. Conservative species absent from these sites but present on unburned sites were *B. jocosa*, *Delphacodes trimaculata*, *Flexamia sandersi*, *Laevicephalus unicoloratus*, *Phylloscelis pallescens*, and *Scolops perdix*. Not only were these conservative species absent, but declines in abundance of many conservative species were also found (Appendix C.1). These declines in abundance and species richness may be attributed to ecological characteristics of conservative Auchenorrhyncha, which have been reported to be highly sensitive to fire. Some of these characteristics include poor dispersal ability (i.e., short wings), and host-plant and habitat specificity. For example, Panzer (2002) found that fires dramatically reduced population sizes of immobile, stem boring *Papaipema* larvae; Dana (1991) observed that host and habitat specific prairie skippers were negatively affected by frequently burned prairies; and Harper *et al*. (2000), Panzer (2002), and Hamilton and Whitcomb (2010) found that host specific leafhoppers and their relatives exhibited substantial population declines immediately after a prescribed burn. Grubb Hollow prairie, a site burned 3 to 5 years ago, also yielded *Philaenus spumarius*, a spittlebug introduced from Europe (Hamilton 1983), which was absent on unburned sites.

As time since burning progressed from 3 to 5 years to 2 years, the number of species continued to decrease to 18 species, with 7 (39%) conservative species, 5 (28%) moderately conservative, and 6 (33%) adventives species. Of these conservative species, *F. pectinata* and *F.*

prairiana were the most abundant, but not as abundant as on unburned sites and sites burned 3 to 5 years ago. Other conservative species, such as *B. dorsata*, *P. dilata*, and *Fitchiella robertsonii* were found on only one site. Moderately conservative species include *Cuerna costalis*, *Gyponana ortha*, and *Stirellus bicolor*. Some adventives species include *Draeculacephala* species, *Neocoelidia tumidifro*ns, and *Empoasca* species. These results are in sharp contrast to other studies (Swengel 1996, Panzer 2002) were most prairie insects showed rapid recovery within 2 years of a burn.

Sites burned one year previously yielded slightly fewer species (15) than those burned two years ago (18), but this difference was not statistically significant. Moreover, the number of conservative species found on sites burned one year previously did not change, moderately conservative species stayed the same with 5 species, and adventives species decreased from 6 to 3 species. Abundance also declined for many conservative auchenorrhynchan species with the exception of *F. pectinata* and *F. prairiana* (Appendix C.1). However, population size of these species only increased at one site. These results are consistent with other studies that documented a steady decline in conservative prairie skippers (Swengel 1996), leaf beetles (Reed 1997), and leafhoppers following a burn conducted the previous growing season (Harper *et al*. 2000, Johnson *et al*. 2008). However, Tooker and Hanks (2004) observed that populations of *Silphium* spp. (Asteraceae) stem-boring Hymenoptera and Coleoptera dramatically rebounded the year following prairie burns, suggesting that prairie Auchenorrhyncha may be more sensitive to fire than these endophytic prairie insects.

Sites sampled immediately after a burn (0 years) yielded a similar species composition of Auchenorrhyncha compared to sites burned 1 year ago. On these sites, the number of species decreased to 13, with 5 (38%) conservative species, 5 (38%) moderately conservative species, and 3 (23%) adventives species. Many of the conservative species present on these sites, such as *B. dorsata*, *Flexamia pectinata*, and *L. unicoloratus* were represented by only 1 individual (Appendix C.1), suggesting that these insects are fire-intolerant. Other studies have reported similar declines of these species in Illinois (Harper et al. 2000, Panzer 2002) and in Kansas (Cancelado and Yonke 1970).

Burn Frequency Treatments

Unburned sites harbored more auchenorrhynchan species compared to frequently burned sites, with 41 species, 18 (44%) of which are conservative, 12 (29%) are moderately conservative, and 11 (27%) are adventives. Other studies have reported reduced species richness as prescribed burning increased in frequency. Evans (1984, 1988) and Wright and Samways (1999) demonstrate that unburned grasslands harbored more remnant-dependent species than frequently burned grasslands, and Morris (1975) showed that increased burning can reduce grassland insect species richness.

Sites burned once yielded fewer species than unburned sites, with 33 species, 10 (30%) of which were conservative, 11 (33%) were moderately conservative, and 12 (36%) were adventives. Sites burned twice supported fewer species than sites burned once, 18 species, 10 of which were conservative (55%), 6 (33%) species were moderately conservative, and two species (11%) were adventives. Although, species richness decreased from 33 on sites burned once to 18 species on sites burned twice, the number of conservative species on sites burned twice remained the same. Similar numbers of conservative species on sites burned twice may be the result of wide periods of time between burns (i.e., 3 to 5 years), thus allowing enough time for conservative prairie Auchenorrhyncha to re-colonize these prairies. A similar trend was observed by Harper *et al*. (2000), which showed that conservative leafhoppers required more than two years to recover from a prairie burn. As prescribed burning increased in frequency from twice to more than twice, the number of species increased to 21 species. Nine (43%) of these species were conservative, 5 (24%) species were moderately conservative species, and 7 (33%) species were adventives. These data are consistent with other studies that have reported more conservative Auchenorrhyncha on prairie remnants that have been managed by fire at wide intervals, between 3 to 5 years (Hamilton 1995, Hamilton and Whitcomb 2010). This pattern has also been well documented for butterflies that favor open prairie (Swengel 1998).

Additional Considerations

Although the original Auchenorrhyncha fauna of most of these sites was unknown, these sites are comparable because they supported similar plant species composition, such as grasses and forbs (personal observations 2006), and thus may support similar grass- and forb-feeding Auchenorrhyncha. Given this similarity in plant species composition, changes in prairie Auchenorrhyncha fauna can be attributed to differences in burn management rather than plants. In addition, this study showed that Auchenorrhyncha integrity and diversity seemed to recover after 3 years following a prescribed burn. However, land managers need to be aware that several conservative auchenorrhynchan species, such as *Bruchomorpha jocosa*, *Delphacodes trimaculata*, *Flexamia sandersi*, *Laevicephalus minimus*, *L. unicolo*ratus, *Phylloscelis pallescens*, *Poblicia fuliginosa*, and *Scolops perdix* (Appendix C.1) were absent from sites that had been burned in 3 years or less, suggesting that additional time between burns is needed in preserving most fire sensitive auchenorrhynchan species.

CONCLUSIONS

Fire is an important natural component in the maintenance of grassland diversity and integrity. However, frequent burning by land managers in an effort to reduce the invasion of exotic plant species, trees, and shrubs may have dire consequences on the invertebrate community. In order for land managers to maintain the prairie Auchenorrhyncha community and conserve vascular plants, this study recommends infrequent rotational burning with a minimum of 3 to 5 years; although additional studies are needed to determine the appropriate number of years between each burn.

Overall, tallgrass hill prairie preserves should be managed with a minimum of 3 to 5 year rotational burns. This management practice may be adequate in conserving most of the prairie Auchenorrhyncha, reduce the number of adventives Auchenorrhyncha that may become numerically dominant following a burn (Harper *et al*. 2000), and conserve prairie vegetation. However, monitoring with the Auchenorrhyncha Quality Index and the Floristic Quality Index (Taft *et al*. 1997) is needed to evaluate the success of these management practices. Additional studies should be conducted within other tallgrass prairies (e.g., wet-mesic, sand, glacial-drift hill prairies) to test the generality of the results obtained here.

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TABLES AND FIGURES

Figure 4.1 Distribution of 22 hill prairie remnants sampled within the Illinois tallgrass prairie, USA. Stars in selected Illinois counties represent the general location of each hill prairie sampled. Map was obtained from http://www.nationalatlas.gov/.

Figure 4.2 Summary of responses to time since a prescribed burned from 19 loess hill prairies in Illinois for Auchenorrhyncha integrity and diversity measures. Each bar depicts the mean (± SE) Auchenorrhyncha integrity and diversity measures. Bars with different letters indicate significant differences between burn treatments (*post hoc* Tukey's test, $p < 0.05$).

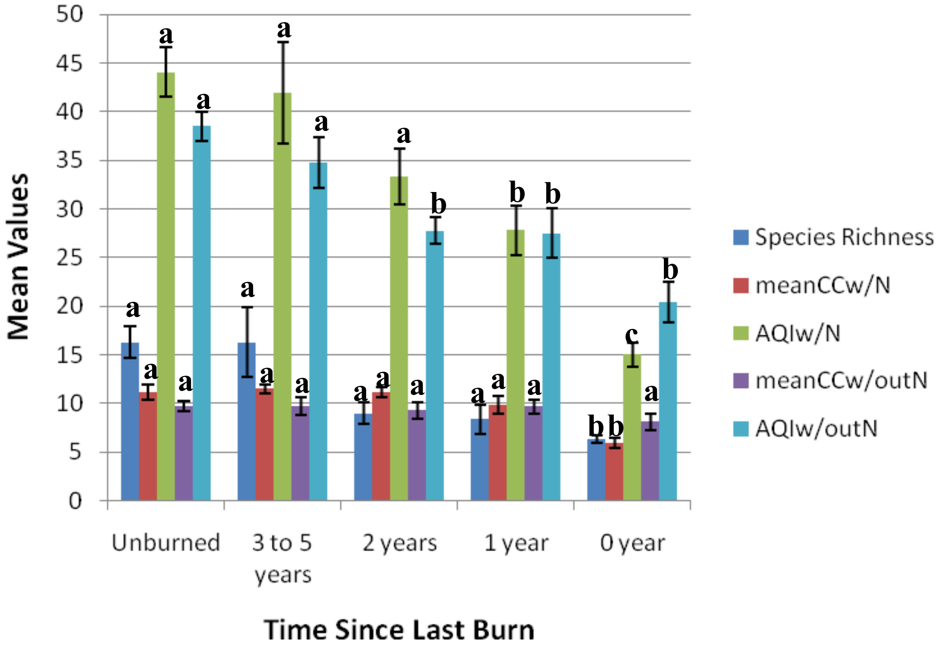

Figure 4.3 Summary of responses to frequency of prescribed burns from 19 loess hill prairies in Illinois for Auchenorrhyncha integrity and diversity measures. Each bar depicts the mean (± SE) of Auchenorrhyncha integrity and diversity measures. Bars with the different letters indicate significant differences between burn treatments (*post hoc* Tukey's test, $p < 0.05$).

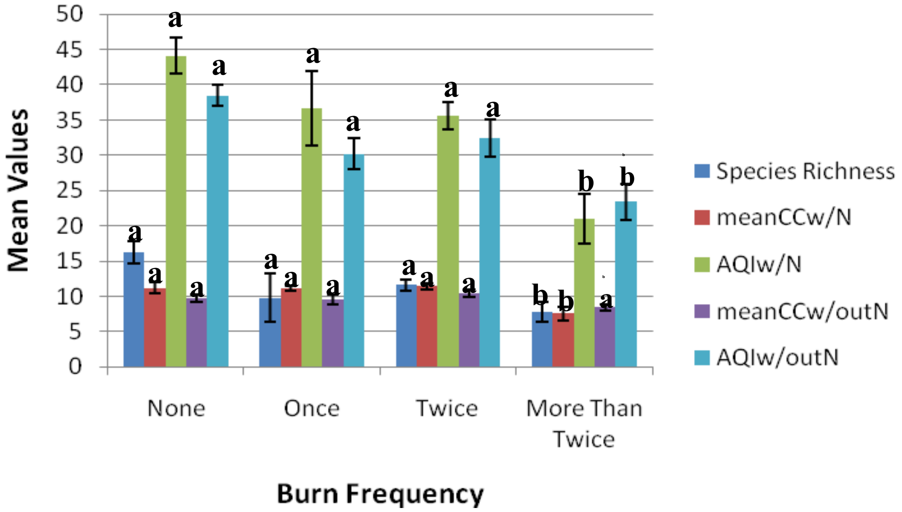

Table 4.1 Distribution of 22 sites, time since sites were burned last (TSLB), burn frequency, acres, total species richness (spp.), total abundance of all auchenorrhynchan species found on each site (N), mean coefficient of conservatism values with and without abundant for each site, and AQI values with and without abundance for each site. Sites with asterisks were removed from the statistical analyses. Mississippi Sanctuary Prairie (MSP), Brickey-Gonterman A (Brickey-GontA), Brickey-Gonterman B (Brickey-GontB)

Site	County	TSLB	Frequency	Acres	Spp.	N	meanCC$_{w/N}$	AQI$_{w/N}$	meanCC$_{w/outN}$	AQI$_{w/outN}$
MSP	Madison	1 year	More than once twice	0.5	5	7	8.21	18.37	7.95	17.78
Brickey-GontB	Monroe	0 years	More than once twice	1.1	6	28	5.26	12.88	6.88	16.84
Olin	Madison	0 years	More than once twice	1.8	6	15	6.98	17.11	9.75	23.88
Brickey-GontA	Monroe	0 years	More than once twice	1.9	7	16	5.69	15.05	7.75	20.50
Jennings	Calhoun	2 years	More than once twice	2	9	89	11.96	35.89	9.94	29.83
***Snyder**	Cass	1 year	More than once twice	6	14	132	7.20	26.92	8.43	31.54
Gonterman	Randolph	1 year	Once	0.1	7	215	12.30	32.53	11.71	30.99
Grubb Hollow	Pike	3 years	Once	2	25	203	11.42	57.11	7.60	38.02
Demint	Randolph	2 years	Once	2.4	11	57	10.99	36.44	8.52	28.27
ChalfinA	Monroe	2 years	Once	3	7	21	10.43	27.59	9.50	25.13
ChalfinB	Monroe	1 year	Once	5	8	45	10.51	29.73	10.09	28.55
Hanover	Jo Daviess	1 year	Twice	0.1	8	120	11.12	31.44	10.13	28.64
Principia	Jersey	3 years	Twice	0.6	10	121	12.72	40.23	11.50	36.37
***Windfall**	Vermilion	5 years	Twice	0.6	9	26	11.13	33.38	9.00	27.00
Fults	Monroe	5 Years	Twice	3.1	12	101	10.73	37.16	10.88	37.67
Gunterman	Monroe	Unburned	None	0.16	16	102	12.28	49.14	10.44	41.75
***Delford**	Pike	Unburned	None	0.25	22	350	6.69	31.38	8.55	40.08
Bland	Greene	Unburned	None	0.8	14	208	11.89	44.50	11.57	43.30
Walnut Grove	Pike	Unburned	None	3	20	196	10.80	48.31	8.64	38.63
Housen	Pike	Unburned	None	3	19	280	11.60	50.57	8.91	38.83
OstermanB	Calhoun	Unburned	None	6	13	179	12.44	44.84	8.91	32.12
OstermanA	Calhoun	Unburned	None	7	10	61	12.59	39.83	10.98	34.71

APPENDICES

Appendix A.1 A total of 274 auchenorrhynchan species collected from 35 sites in Illinois, Wisconsin, Missouri, and Iowa.

Species	Voltinism	Host Plant	Overwinter	Winglength	Origin	PRD	CC
Acanalonia conica	3	0	1	0	1.5	0	5.5
Acanalonia bivittata	3	0	1	0	1.5	0	5.5
Acutalis tartarea	1.5	0	1	0	1.5	0	3
Aflexia rubranura	1.5	3	3	3	3	3	16.5
Agalliopsis novella	1.5	0	1	0	1.5	0.75	4.75
Agalliopsis peneoculata	1.5	0	1	0	1.5	0.75	4.75
Agalliota constricta	1.5	0	1	0	1.5	0	3
Amblysellus curtisii	1.5	0	1	0	1.5	0	4
Amblysellus acuerus	1.5	3	3	2	3	3	15.5
Amplicephalus osborni	3	2.5	3	0	1.5	3	13
Acinopterus acuminatus	1.5	0.5	3	0	1.5	0.75	7.25
Amphigonalia gothica	3	0	1	0	1.5	0.75	6.25
Anormenis septentrionalis	3	0	0	0	1.5	0	4.5
Anoscopus serratulae	0	0	0	0	0	0	0
Aphelonema simplex	1.5	2.5	3	3	3	3	16
Aphrodes bicincta	0	0	0	0	0	0	0
Aphrophora quadrinotata	3	0	1	0	1.5	0.75	6.25
Athysanella balli	3	3	3	3	3	3	18
Athysanella incongrua	3	3	3	3	3	3	18
Athysanus argentarius	0	0	0	0	0	0	0
Attenuipyga platyrhyncus	3	2.5	2	3	1.5	1.5	13.5
Bakerella cinerea	1.5	0.5	3	1	1.5	2.25	9.75
Bakerella muscotana	1.5	1	3	1	1.5	3	11
Bakerella rotundifrons	1.5	1	3	1	1.5	2.25	10.25
Balclutha impicta	0	0	1	0	1.5	0	2.5
Balclutha punctata	0	0	0	0	0	0	0
Balclutha abdominalis	0	0	1	0	1.5	0.5	3
Balclutha neglecta	0	0	0	0	1.5	0	1.5
Bruchomorpha jocose	1.5	2.5	3	3	3	3	16
Bruchomorpha dorsata	1.5	1	3	3	1.5	2.25	12.25
Bruchomorpha oculata	1.5	0.5	3	3	1.5	1.5	11
Bruchomorpha pallidipes	1.5	2	3	3	3	3	15.5
Bruchomorpha tristis	1.5	2.5	3	3	3	3	16
Caenodelphax nigriscutelleta	1.5	2	3	3	1.5	3	14
Delphacodes nigripennata	1.5	1.5	3	3	1.5	3	13.5

Campylenchia latipes	3	0	1	0	1.5	0.75	6.25
Cedusa incise	3	0	0	0	1.5	0.75	5.25
Cedusa obscura	3	0.5	2	0	1.5	0.75	7.75
Ceratagallia uhleri	1.5	0	1	0	1.5	0.75	4.75
Ceratagallia Agricola	1.5	0	1	0	1.5	0.75	4.75
Ceratagallia viator	1.5	0	0	0	1.5	1.5	4.5
Chlorotettix fallax	3	3	3	0	1.5	2.25	12.75
Chlorotettix spatulatus	3	1	3	0	1.5	1.5	10
Chlorottetix tergatus	3	0.5	3	0	1.5	0.75	8.75
Chlorottetix tunicatus	3	1	3	0	1.5	1.5	10
Chlorottetix galbanatus	3	1	1	0	1.5	0.75	7.25
Chlorotettix unicolor	3	0.5	3	0	1.5	1.5	9.5
Beameria venosa	3	1	2	0	1.5	1.5	9
Cicadula melanogaster	3	0.5	3	0	1.5	1.5	9.5
Cicadetta calliope	3	1	2	0	1.5	0.75	8.25
Cixius basalis	3	0	0	0	1.5	0.75	5.25
Clastoptera proteus	3	0	0	0	1.5	0.75	5.25
Commellus comma	1.5	1.5	3	2	1.5	2.25	11.75
Cosmotettix delector	3	0	0	2	1.5	0.75	7.25
Cuerna alpine	3	0	2	0	1.5	0.75	7.25
Cuerna costalis	3	0	2	0	1.5	0.75	7.25
Cuerna fenestella	3	0	2	0	1.5	0.75	7.25
Delphacodes balli	1.5	1	3	3	1.5	3	13
Delphacodes puella	1.5	0.5	1	1	1.5	0	5.5
Delphacodes sagae	1.5	1	3	3	1.5	3	13
Delphacodes Andromeda	1.5	1.5	3	3	1.5	3	13.5
Delphacodes campestris	1.5	0.5	1	1	1.5	0	5.5
Delphacodes caerulata	1.5	1	3	3	1.5	2.25	12.25
Muellerianella laminalis	1.5	1	3	3	1.5	3	13
Delphacodes lutulentoides	1.5	0.5	3	1	1.5	1.5	9
Delphacodes lutulenta	0	1	3	1	1.5	1.5	8
Delphacodes mcateei	1.5	0.5	3	3	1.5	2.25	11.75
Delphacodes megadonte	1.5	0.5	3	3	1.5	2.25	11.75
Delphacodes parvula	1.5	1.5	3	1	1.5	2.25	10.75
Delphacodes rotundata	1.5	0.5	3	1	1.5	2.25	9.75
Delphacodes trimaculata	1.5	0.5	3	3	3	3	14
Delphacodes n. sp.	1.5	1	3	1	1.5	1.5	9.5
Deltacephalus balli	1.5	0.5	1	0	1.5	0.75	5.25
Deltocephalus gnarus	1.5	2.5	3	0	1.5	3	11.5
Destria fumida	1.5	2.5	3	3	3	3	16
Destria crocea	1.5	2.5	3	3	3	3	16

Dikraneura angustata	1.5	0	0	0	1.5	0	3
Dikraneura mali	1.5	0.5	1	0	1.5	0.75	5.25
Diplocolenus configuratus	1.5	1	3	2	3	1.5	12
Doratura stylata	0	0	0	0	0	0	0
Dorydiella kansana	3	1.5	3	0	3	3	13.5
Draeculacephala mollipes	1.5	0	1	0	1.5	0.75	4.75
Draeculacephala paludosa	1.5	0	0	0	1.5	0.75	3.75
Draeculacephala antica	1.5	0	0	0	1.5	0.75	3.75
Draeculacephala constricta	1.5	0	0	0	1.5	0.75	3.75
Draeculacephala robinsoni	1.5	1	1	0	1.5	0.75	5.75
Driotura gammaroides	1.5	2	3	3	1.5	1.5	12.5
Elymana acuma	1.5	1	1	0	1.5	0.75	5.75
Empoasca bifurcate	0	0	1	0	1.5	0.75	3.25
Empoasca deluda	0	0	1	0	1.5	0	2.5
Empoasca recurvata	0	0	1	0	1.5	0.75	3.25
Empoasca birdie	0	0	1	0	1.5	0.75	3.25
Empoasca erigeron	0	0	1	0	1.5	0.75	3.25
Empoasca fabae	0	0	0	0	1.5	0	1.5
Empoasca flavescens	0	0	1	0	1.5	0.75	3.25
Endria inimical	1.5	0	1	0	1.5	0	4
Entylia carinata	1.5	0	1	0	1.5	0.75	4.75
Erasmoneura vulnerata	0	0	1	0	1.5	0	2.5
Rossmoneura carbonate	1.5	2.5	2	0	1.5	3	10.5
Erasmoneura nigra	0	0	1	0	1.5	0.75	3.25
Erataneura omani	0	0	0	0	1.5	0	1.5
Rossmoneura tecta	1.5	2	2	0	1.5	3	10
Erythridula oblique	0	0	1	0	1.5	0.75	3.25
Erythroneura acuticephala	0	0	1	0	1.5	0.75	3.25
Erythroneura comes	0	0	1	0	1.5	0.75	3.25
Erythroneura prosata	0	0	1	0	1.5	0.75	3.25
Erythroneura rubrella	0	0	1	0	1.5	0.75	3.25
Erythroneura vitis	0	0	1	0	1.5	0.75	3.25
Erythroneura rosa	0	0	1	0	1.5	0.75	3.25
Exitianus exitiosus	1.5	0	1	0	1.5	0	4
Extrusanus orrysus	1.5	1	3	2	3	2.25	12.75
Fitchiella robertsonii	1.5	2.5	3	3	3	3	16
Flexamia albida	0	1.5	3	2	3	3	12.5
Flexamia areolata	0	3	3	2	1.5	2.25	11.75
Flexamia atlantica	0	3	3	2	3	2.25	13.25
Flexamia delongi	0	3	3	2	3	3	14
Flexamia grammica	0	3	3	2	3	2.25	13.25

Flexamia inflate	0	1	3	2	1.5	1.5	9
Flexamia reflexa	0	1.5	3	2	3	2.25	11.75
Flexamia sandersi	0	2	3	2	3	2.25	12.25
Flexamia clayi	0	2	3	2	3	2.25	12.25
Flexamia pectinata	0	3	3	2	3	2.25	13.25
Flexamia picta	0	2.5	3	2	3	2.25	12.75
Flexamia prairiana	0	1.5	3	2	3	2.25	11.75
Forcipata loca	0	0	0	0	1.5	0	1.5
Graminella aureovittata	1.5	3	3	0	3	3	13.5
Graminella fitchii	1.5	1	3	0	1.5	0.75	7.75
Graminella mohri	1.5	3	3	0	3	3	13.5
Graminella nigrifrons	0	0	1	0	1.5	0	2.5
Graminella oquaka	1.5	3	3	0	3	3	13.5
Graphocephala coccinea	1.5	0	1	0	1.5	0.75	4.75
Graphocephala hieroglyphica	1.5	0	1	0	1.5	0.75	4.75
Graphocephala versuta	1.5	0	1	0	1.5	0.75	4.75
Gyponana aculeate	3	1	1	0	1.5	0.75	7.25
Gyponana brevita	3	0	0	0	1.5	0	4.5
Gyponona ortha	3	0	1	0	1.5	0.75	6.25
Hecalus major	3	2.5	3	0	3	2.25	13.75
Hecalus viridis	1.5	2.5	3	0	1.5	1.5	10
Hymetta trifasciata	0	0	0	0	1.5	0	1.5
Idiocerus rotundus	3	0	1	0	1.5	0	5.5
Isodelphax basivitta	1.5	0	0	0	1.5	0	3
Japananus hyalinus	0	0	0	0	0	0	0
Jikradia olitoria	1.5	0	1	0	1.5	0.75	4.75
Kansendria kansiensis	1.5	2	3	0	1.5	1.5	9.5
Kelisia vesiculata	3	0.5	3	0	1.5	1.5	9.5
Kelisia curvata	3	2	3	0	3	2.25	13.25
Kelisia pectinata	3	2	3	0	3	2.25	13.25
Kelisia retrosa	3	2	3	0	3	2.25	13.25
Laccocera vittipennis	1.5	1	3	0	3	3	11.5
Laevicephalus acus	1.5	1.5	3	0	3	2.25	11.25
Laevicephalus melsheimeri	1.5	3	3	0	3	1.5	12
Laevicephalus minimus	1.5	3	3	0	3	2.25	12.75
Laevicephalus pravus	1.5	2	3	0	3	2.25	11.75
Laevicephalus unicoloratus	1.5	1.5	3	0	3	2.25	11.25
Latalus latidens	0	1	3	2	1.5	1.5	9
Latalus personatus	0	1	3	2	1.5	1.5	9
Latalus sayi	0	1	1	2	1	0	5
Latalus misselus	0	1	3	2	1.5	0.75	8.25

Lepyronia gibbosa	3	1	2	0	1.5	3	10.5
Lepyronia quadrangularis	3	0	1	0	1.5	0.75	6.25
Liburniella ornate	1.5	0	0	0	1.5	0	3
Limotettix anthracinus	3	1	3	0	1.5	0.75	9.25
Limotettix urnura	3	3	3	0	1.5	3	13.5
Limotettix osborni	3	1	3	0	1.5	3	11.5
Lonotura catalina	2	3	3	3	3	3	17
Macropsis rufescens	1.5	3	3	0	1.5	3	12
Macrosteles lepida	0	0.5	3	0	1.5	0.75	5.75
Macrosteles quadrilineatus	0	0	0	0	1.5	0	1.5
Macrosteles variata	1.5	0	1	0	1.5	0.75	4.75
Megamelus distinctus	1.5	2.5	3	3	1.5	3	14.5
Megamelus metazeri	1.5	2.5	3	3	1.5	3	14.5
Memnonia flavida	1.5	2.5	3	2	3	3	15
Memnonia panzer	3	3	3	2	3	3	17
Menosoma cincta	3	0	1	0	1.5	0.75	6.25
Mesamia nigridorsum	3	2.5	3	0	1.5	0.75	10.75
Mesamia straminea	3	2.5	3	0	1.5	3	13
Metcalfa pruinosa	3	0	0	0	1.5	0	4.5
Micrutalis calva	1.5	0	1	0	1.5	0	4
Myndus pictifrons	1.5	0	1	0	1.5	0	4
Neocoelidia tumidifrons	1.5	0	1	0	1.5	0.75	4.75
Neohecalus magnificus	3	2.5	3	2	3	2.25	15.75
Norvellina seminude	1.5	0	1	0	1.5	0.75	4.75
Nothodelphax hyaline	1.5	1.5	3	3	1.5	1.5	12
Nothodelphax lineatipes	1.5	1.5	3	3	3	3	15
Oecleus borealis	3	0	0	0	1.5	0.75	5.25
Oecleus chrisjohni	3	2	3	0	1.5	1.5	11
Oliarus ecologus	3	2	3	0	1.5	0.75	10.25
Oncometopia orbona	3	0	1	0	1.5	0.75	6.25
Opsius stactogalus	0	0	0	0	0	0	0
Orientus ishidae	0	0	0	0	0	0	0
Ormenoides venusta	3	0	1	0	1.5	0.75	6.25
Osbornellus auronitens	1.5	0	1	0	1.5	0	4
Osbornellus consors	1.5	0	1	0	1.5	0	4
Cosmotettix luteocephalus	3	1	1	2	1.5	0.75	9.25
Paraphlepsius electus	3	2.5	3	0	1.5	3	13
Paraphilaenus parallelus	3	2	3	0	1.5	3	12.5
Paraphilaenus solidaginis	3	2	3	0	3	3	14
Paraphlepsius irroratus	3	0	1	0	1.5	0	5.5
Paraphlepsius lobatus	3	3	3	0	1.5	3	13.5

Paraphlepsius umbellatus	3	3	3	0	3	3	15
Paraulazices irrorata	3	0	1	0	1.5	0.75	6.25
Pendarus magnus	3	3	3	0	3	3	15
Pendarus punctiscriptus	3	1	3	0	3	3	13
Penthimia Americana	3	0.5	1	0	1.5	0.75	6.75
Philaenarcys bilineata	3	1	2	0	3	1.5	10.5
Philaenarcys killa	3	3	2	0	3	3	14
Philaenus spumarius	0	0	0	0	0	0	0
Philaronia abject	1.5	0	1	0	1.5	0	4
Philaronia Canadensis	3	0	1	0	1.5	0.75	6.25
Pintalia dorsivittatus	3	0	1	0	1.5	0.75	6.25
Phylloscelis pallescens	3	3	3	1	1.5	2.25	13.75
Phylloscelis atra	3	1	3	1	1.5	2.25	11.75
Pissonotus basalis	0	0	1	3	1.5	0.75	6.25
Pissonotus divergens	0	1.5	3	3	3	3	13.5
Pissonotus niger	0	0.5	3	3	1.5	1.5	9.5
Pissonotus piceus	0	1	3	3	1.5	1.5	10
Pissonotus marginatus	0	0.5	3	3	1.5	1.5	9.5
Planicephalus flavicostus	0	0.5	1	2	1.5	0	5
Plesiommata	3	0	1	0	1.5	0.75	6.25
Poblicia fulginosa	3	2	3	0	1.5	1.5	11
Polyamia apicata	1.5	3	3	3	3	1.5	15
Polyamia caperata	1.5	1	3	3	1.5	1.5	11.5
Polyamia herbida	1.5	2.5	3	3	3	3	16
Polyamia rossi	1.5	2	3	3	3	3	15.5
Polyamia weedi	1.5	1	3	3	1.5	2.25	12.25
Polyamia compacta	1.5	2	3	3	3	2.25	14.75
Polyamia dilate	1.5	2	3	3	3	3	15.5
Ponana puncticollis	3	0	1	0	1.5	0	5.5
Ponana rubida	3	0.5	3	0	1.5	0.75	8.75
Prairiana kansana	3	2	3	0	3	3	14
Prokelisia crocea	1.5	3	3	0	1.5	2.25	11.25
Prosapia bicincta	3	0	1	0	1.5	0.75	6.25
Prosapia ignipectus	3	1	3	0	1.5	3	11.5
Psammotettix lividellus	1.5	1.5	3	0	1.5	1.5	9
Publilia concave	1.5	0	1	0	1.5	0	4
Publilia reticulate	1.5	1	3	0	1.5	2.25	9.25
Ribautodelphax puella	1.5	1	3	3	1.5	3	13
Rhynchomitra microrhina	3	0	1	0	1.5	0.75	6.25
Rosenus cruciatus	1.5	3	3	0	3	3	13.5
Sayiana sayi	3	0	0	0	1.5	0.75	5.25

Scaphoideus titanus	3	0	1	0	1.5	0	5.5
Scaphytopius abbreviates	1.5	2	3	0	3	3	12.5
Scaphytopius cinereus	1.5	0.5	3	0	1.5	1.5	8
Scaphytopius cuprescens	1.5	0	1	0	1.5	0.75	4.75
Scaphytopius frontalis	1.5	0	1	0	1.5	0.75	4.75
Scaphytopius acutus	1.5	0	1	0	1.5	0.75	4.75
Scaphytopius dorsalis	1.5	2.5	3	0	1.5	2.25	10.75
Scaphytopius hastus	1.5	0	0	0	1.5	0	3
Scolops angustatus	3	1	3	2	1.5	1.5	12
Scolops perdix	3	2	3	2	1.5	3	14.5
Scolops pungens	3	0	0	2	1.5	0.75	7.25
Scolops sulcipes	3	0.5	3	2	1.5	1.5	11.5
Sorhoanus pascuellus	0	0	0	0	0	0	0
Stenocranus delicates	1.5	0	1	0	1.5	0	4
Stenocranus lautus	1.5	1	2	0	1.5	0.75	6.75
Stenocranus pallides	1.5	1	3	0	1.5	1.5	8.5
Stictocephala lutea	3	0	1	0	1.5	0.75	6.25
Stictocephala bisonia	3	0	1	0	1.5	0	5.5
Stictocephala brevitylus	3	0	1	0	1.5	0.75	6.25
Stictocephala diceros	3	2	3	0	1.5	0.75	10.25
Stictocephala taurina	3	0	1	0	1.5	0	5.5
Stirellus bicolor	1.5	0	1	2	1.5	0	6
Stobaera tricarinata	1.5	0	1	0	1.5	0	4
Stroggylocephalus placidus	3	2	3	0	1.5	3	12.5
Texananus decorus	3	1	3	0	1.5	1.5	10
Thionia bullata	3	0	1	1	1.5	0	6.5
Tibicen canicularis	3	0	1	0	1.5	0	5.5
Tylozygus bifidus	3	0	1	0	1.5	0.75	6.25
Vanduzea triguttata	1.5	0	1	0	1.5	0.75	4.75
Xerophloea major	1.5	1	3	0	1.5	1.5	8.5
Xerophloea peltata	1.5	1	3	0	1.5	1.5	8.5
Xestocephalus brunneus	1.5	0	1	0	1.5	0	4
Xestocephalus pulicarius	1.5	0	1	0	1.5	0	4

Appendix B.1 Auchenorrhyncha collected in study area. Species shown with asterisks were considered woody-associated species. Acronyms for species are first three letters of genus and species. Study sites include Robinson South (RobS), Detweiller South (DetS), Robinson North C (RobNC), Singing Woods (SinW), Gentiana 1 (Gen1), Camp Wokanda (CamW), Hopewell (Hope), Robinson North B (RobNB), Gentiana 2 (Gen2), Wier (Wier), Forest Park (ForP), Detweiller North (DetN), Robinson North (RobN), and Peoria Park Hill Prairie (PePP).

Species	Species Code	CC	RobS	DetS	RobNC	SinW	GenA	CamW
*Acanalonia conica	Acacon	5.5	0	0	0	0	0	0
*Acanalonia bivittata	Acabiv	5.5	4	0	11	1	4	4
*Agalliopsis novella	Aganov	4.75	0	0	0	0	0	0
Balclutha neglecta	Balneg	1.5	0	0	0	0	0	0
Bruchomorpha dorsata	Brudor	12.25	1	0	1	28	0	4
Bruchomorpha oculata	Bruocu	11	0	0	0	0	0	0
Bruchomorpha tristis	Brutri	16	0	0	0	0	0	0
Campylenchia latipes	Camlat	6.25	0	0	1	0	4	1
*Cedusa incise	Cedinc	5.25	0	0	0	0	0	0
Ceratagallia Agricola	Ceragr	4.75	0	0	1	0	0	0
Chlorotettix spatulatus	Chlspa	10	0	0	0	0	0	12
*Cixius basalis	Cixbas	5.25	0	1	0	0	0	0
Delphacodes puella	Delpue	5.5	0	0	0	0	0	0
Delphacodes caerulata	Delcae	12.25	0	0	0	1	0	0
Delphacodes rotundata	Delrot	9.75	2	0	2	14	0	0
Delphacodes n.sp.	Delnsp	10	34	0	7	1	10	41
Dikraneura angustata	Dikang	3	0	1	0	0	0	8
Dikraneura mali	Dikmal	5.25	3	0	0	5	0	0
Draeculacephala mollipes	Dramol	4.75	0	0	0	2	0	0
Draeculacephala antica	Draant	3.75	0	0	1	0	0	3
*Empoasca bifurcate	Empbif	3.25	0	0	0	6	0	0
*Empoasca deluda	Empdel	*2.5*	0	0	0	0	0	0
Empoasca recurvata	Emprec	3.25	9	0	5	0	2	1
*Empoasca erigeron	Emperi	3.25	0	1	0	2	0	0
Empoasca fabae	Empfab	1.5	2	1	0	1	2	4
Endria inimical	Endini	4	0	0	0	0	0	0
*Erasmoneura vulnerata	Eravul	2.5	3	1	0	0	0	1
*Erythridula oblique	Eryobl	3.25	3	0	0	3	37	0
*Erythroneura comes	Erycom	3.25	0	0	0	0	0	0
*Erythroneura rosa	Eryros	3.25	0	0	0	0	0	0
Exitianus exitiosus	Exiexi	4	0	0	0	0	0	0
Flexamia pectinata	Flepec	13.25	0	0	0	0	2	0
Flexamia prairiana	Flepra	11.75	0	0	2	0	0	0
Forcipata loca	Forloc	1.5	0	1	0	0	0	0

Graphocephala hieroglyphica	Grahie	4.75	0	0	0	0	0	0
**Hymetta trifasciata*	Hymtri	1.5	0	0	0	0	0	0
**Jikradia olitoria*	Jikoli	4.75	0	0	0	2	0	0
Kansendria kansiensis	Kankan	9.5	0	0	0	0	0	0
Laevicephalus melsheimerii	Laemel	12	1	0	1	0	0	2
Laevicephalus minimus	Laemin	12.75	0	0	0	53	0	0
Laevicephalus unicoloratus	Laeuni	11.25	0	0	0	1	18	0
**Lepyronia quadrangularu\is*	Lepqua	6.25	1	0	0	0	0	0
Liburniella ornate	Liborn	3	0	0	0	6	0	9
Limotettix anthrancinus	Limant	9.25	0	0	0	0	0	0
Memnonia flavida	Memfla	15	0	0	0	0	0	9
Mesamia nigridosum	Mesnig	10.75	0	0	0	0	1	1
**Metcalfa pruinosa*	Metpru	4.5	0	0	0	0	0	0
Neocoelidia tumidifrons	Neotum	4.75	1	0	1	0	0	0
**Oecleus borealis*	Oecbor	5.25	9	0	10	0	12	1
**Osbornellus auronitens*	Osbaur	4	0	0	0	0	0	0
Paraphlepsius electus	Parele	13	0	0	0	21	0	0
Pendarus pendarus	Penpun	13	0	0	0	0	0	0
Penthimia Americana	Pename	6.75	1	0	0	1	0	0
**Pintalia dorsivittatus*	Pindor	6.25	0	0	0	0	0	0
Phylloscelis atra	Phyart	11.75	0	1	8	0	0	0
Planicephalus flavicostus	Plafla	5	0	0	0	0	0	0
Polyamia apicata	Polapi	15	0	0	0	0	0	0
Polyamia caperata	Polcap	11.5	107	0	11	425	2	27
Polyamia compacta	Polcom	14.75	0	0	0	0	0	0
Scaphytopius frontalis	Scafro	4.75	0	0	0	0	0	0
Scaphytopius dorsalis	Scador	10.75	1	0	0	0	0	0
**Scaphytopius hastus*	Scahas	3	0	0	0	0	0	0
Scolops angustatus	Scoang	12	0	0	5	0	0	0
Scolops sulcipes	Scosul	11.5	0	2	0	0	0	0
Stictocephala lutea	Stilut	6.25	2	0	0	0	0	0
Stictocephala bisonia	Stibis	5.5	0	0	0	0	0	0
Stirellus bicolor	Stibic	6	11	0	0	27	11	13
Texananus decorus	Texdec	10	0	0	0	0	0	0
Vanduzea triguttata	Vantri	4.75	1	0	0	0	0	0
Xerophloea major	Xermaj	8.5	3	0	0	13	0	0
Xestocephalus pulicarius	Xespul	4	4	0	0	1	0	0

Appendix B.1 (cont.)

Species	**Species Code**	**CC**	**Hope**	**RPNB**	**GenB**	**Wier**	**ForP**
**Acanalonia conica*	Acacon	5.5	1	0	0	0	0
**Acanalonia bivittata*	Acabiv	5.5	0	2	0	0	1
**Agalliopsis novella*	Aganov	4.75	0	0	0	0	0
Balclutha neglecta	Balneg	1.5	0	0	0	1	15
Bruchomorpha dorsata	Brudor	12.25	1	0	0	50	0
Bruchomorpha oculata	Bruocu	11	0	2	0	0	3
Bruchomorpha tristis	Brutri	16	5	0	0	0	0
Campylenchia latipes	Camlat	6.25	0	2	0	0	0
**Cedusa incise*	Cedinc	5.25	0	0	0	0	0
Ceratagallia Agricola	Ceragr	4.75	0	0	0	0	0
Chlorotettix spatulatus	Chlspa	10	0	0	0	0	0
**Cixius basalis*	Cixbas	5.25	0	0	0	0	0
Delphacodes puella	Delpue	5.5	1	0	0	0	12
Delphacodes caerulata	Delcae	12.25	0	0	0	0	0
Delphacodes rotundata	Delrot	9.75	43	0	8	11	151
Delphacodes n.sp.	Delnsp	10	0	0	1	12	0
Dikraneura angustata	Dikang	3	0	0	0	0	0
Dikraneura mali	Dikmal	5.25	1	5	2	0	1
Draeculacephala mollipes	Dramol	4.75	0	0	0	0	0
Draeculacephala antica	Draant	3.75	0	0	0	1	0
**Empoasca bifurcate*	Empbif	3.25	0	0	0	0	0
**Empoasca deluda*	Empdel	*2.5*	0	0	0	0	15
Empoasca recurvata	Emprec	3.25	0	3	0	1	8
**Empoasca erigeron*	Emperi	3.25	0	0	0	0	0
Empoasca fabae	Empfab	1.5	14	0	1	0	1
Endria inimical	Endini	4	0	0	0	0	2
**Erasmoneura vulnerata*	Eravul	2.5	0	0	0	0	0
**Erythridula oblique*	Eryobl	3.25	0	0	0	0	1
**Erythroneura comes*	Erycom	3.25	5	0	0	0	0
**Erythroneura rosa*	Eryros	3.25	0	3	0	0	0
Exitianus exitiosus	Exiexi	4	0	0	0	0	1
Flexamia pectinata	Flepec	13.25	0	1	0	59	0
Flexamia prairiana	Flepra	11.75	28	0	0	312	0
Forcipata loca	Forloc	1.5	0	2	0	0	0
Graphocephala hieroglyphica	Grahie	4.75	0	0	0	0	0
**Hymetta trifasciata*	Hymtri	1.5	0	0	0	0	1
**Jikradia olitoria*	Jikoli	4.75	0	0	0	0	0
Kansendria kansiensis	Kankan	9.5	0	0	0	12	0
Laevicephalus melsheimerii	Laemel	12	0	0	0	2	97
Laevicephalus minimus	Laemin	12.75	365	0	0	2	34

Laevicephalus unicoloratus	Laeuni	11.25	0	0	0	4	1
**Lepyronia quadrangularu\is*	Lepqua	6.25	0	0	0	0	0
Liburniella ornate	Liborn	3	8	1	0	17	86
Limotettix anthrancinus	Limant	9.25	0	0	0	0	0
Memnonia flavida	Memfla	15	0	0	0	0	0
Mesamia nigridosum	Mesnig	10.75	0	0	0	0	0
**Metcalfa pruinosa*	Metpru	4.5	0	1	0	0	0
Neocoelidia tumidifrons	Neotum	4.75	0	0	0	0	1
**Oecleus borealis*	Oecbor	5.25	0	0	1	0	0
**Osbornellus auronitens*	Osbaur	4	1	0	0	0	0
Paraphlepsius electus	Parele	13	1	0	0	0	0
Pendarus pendarus	Penpun	13	0	0	0	0	0
Penthimia Americana	Pename	6.75	1	0	0	0	0
**Pintalia dorsivittatus*	Pindor	6.25	0	1	0	0	0
Phylloscelis atra	Phyart	11.75	0	0	0	0	0
Planicephalus flavicostus	Plafla	5	0	0	0	0	1
Polyamia apicata	Polapi	15	0	0	0	0	2
Polyamia caperata	Polcap	11.5	57	0	21	483	849
Polyamia compacta	Polcom	14.75	0	0	0	0	4
Scaphytopius frontalis	Scafro	4.75	0	0	0	0	0
Scaphytopius dorsalis	Scador	10.75	2	0	0	0	0
**Scaphytopius hastus*	Scahas	3	0	0	0	0	1
Scolops angustatus	Scoang	12	0	0	0	0	0
Scolops sulcipes	Scosul	11.5	5	0	7	0	2
Stictocephala lutea	Stilut	6.25	0	0	0	0	0
Stictocephala bisonia	Stibis	5.5	0	0	0	0	1
Stirellus bicolor	Stibic	6	2	0	5	22	292
Texananus decorus	Texdec	10	0	0	0	1	0
Vanduzea triguttata	Vantri	4.75	0	0	3	0	0
Xerophloea major	Xermaj	8.5	1	0	0	0	0
Xestocephalus pulicarius	Xespul	4	4	1	0	2	6

Appendix B.1 (cont.)

Species	**Species Code**	**CC**	**DetN**	**RobN**	**PePP**
**Acanalonia conica*	Acacon	5.5	0	0	0
**Acanalonia bivittata*	Acabiv	5.5	3	4	4
**Agalliopsis novella*	Aganov	4.75	0	1	0
Balclutha neglecta	Balneg	1.5	0	1	0
Bruchomorpha dorsata	Brudor	12.25	0	13	1
Bruchomorpha oculata	Bruocu	11	0	0	0
Bruchomorpha tristis	Brutri	16	0	0	0
Campylenchia latipes	Camlat	6.25	2	23	0
**Cedusa incise*	Cedinc	5.25	0	0	3
Ceratagallia Agricola	Ceragr	4.75	0	0	0
Chlorotettix spatulatus	Chlspa	10	0	0	1
**Cixius basalis*	Cixbas	5.25	0	0	0
Delphacodes puella	Delpue	5.5	0	0	0
Delphacodes caerulata	Delcae	12.25	0	0	0
Delphacodes rotundata	Delrot	9.75	4	0	25
Delphacodes n.sp.	Delnsp	10	0	17	15
Dikraneura angustata	Dikang	3	3	6	8
Dikraneura mali	Dikmal	5.25	0	0	0
Draeculacephala mollipes	Dramol	4.75	0	0	0
Draeculacephala antica	Draant	3.75	0	5	1
**Empoasca bifurcate*	Empbif	3.25	0	4	0
**Empoasca deluda*	Empdel	*2.5*	0	0	0
Empoasca recurvata	Emprec	3.25	0	0	3
**Empoasca erigeron*	Emperi	3.25	2	1	0
Empoasca fabae	Empfab	1.5	2	2	7
Endria inimical	Endini	4	0	0	0
**Erasmoneura vulnerata*	Eravul	2.5	0	0	1
**Erythridula oblique*	Eryobl	3.25	0	2	277
**Erythroneura comes*	Erycom	3.25	0	0	1
**Erythroneura rosa*	Eryros	3.25	0	1	0
Exitianus exitiosus	Exiexi	4	0	0	0
Flexamia pectinata	Flepec	13.25	0	0	0
Flexamia prairiana	Flepra	11.75	0	0	2
Forcipata loca	Forloc	1.5	1	0	2
Graphocephala hieroglyphica	Grahie	4.75	0	1	1
**Hymetta trifasciata*	Hymtri	1.5	0	0	0
**Jikradia olitoria*	Jikoli	4.75	0	0	0
Kansendria kansiensis	Kankan	9.5	0	0	0
Laevicephalus melsheimerii	Laemel	12	0	0	0
Laevicephalus minimus	Laemin	12.75	0	10	3

Laevicephalus unicoloratus	Laeuni	11.25	0	0	60
**Lepyronia quadrangularu\is*	Lepqua	6.25	0	0	0
Liburniella ornate	Liborn	3	1	0	8
Limotettix anthrancinus	Limant	9.25	1	0	0
Memnonia flavida	Memfla	15	0	6	0
Mesamia nigridosum	Mesnig	10.75	0	0	0
**Metcalfa pruinosa*	Metpru	4.5	0	0	1
Neocoelidia tumidifrons	Neotum	4.75	0	0	0
**Oecleus borealis*	Oecbor	5.25	35	1	2
**Osbornellus auronitens*	Osbaur	4	0	0	0
Paraphlepsius electus	Parele	13	0	0	0
Pendarus pendarus	Penpun	13	0	0	0
Penthimia Americana	Pename	6.75	0	0	0
**Pintalia dorsivittatus*	Pindor	6.25	0	0	0
Phylloscelis atra	Phyart	11.75	2	4	6
Planicephalus flavicostus	Plafla	5	0	0	0
Polyamia apicata	Polapi	15	0	0	0
Polyamia caperata	Polcap	11.5	40	10	3
Polyamia compacta	Polcom	14.75	0	0	0
Scaphytopius frontalis	Scafro	4.75	0	8	0
Scaphytopius dorsalis	Scador	10.75	0	0	0
**Scaphytopius hastus*	Scahas	3	0	0	0
Scolops angustatus	Scoang	12	0	0	0
Scolops sulcipes	Scosul	11.5	0	5	0
Stictocephala lutea	Stilut	6.25	0	0	0
Stictocephala bisonia	Stibis	5.5	0	0	1
Stirellus bicolor	Stibic	6	11	3	49
Texananus decorus	Texdec	10	0	0	0
Vanduzea triguttata	Vantri	4.75	0	0	0
Xerophloea major	Xermaj	8.5	11	3	6
Xestocephalus pulicarius	Xespul	4	0	0	0

Appendix C.1 Auchenorrhynchan species and their corresponding coefficient of conservatism values for each site visited. Sites with asterisks were removed from the analysis.

Species	**CC**	**Gunterman**	***Delford**	**Bland**	**Walnut Grove**	**Housen**	**OstermanB**
Acanalonia bivittata	5.5	0	0	0	1	1	0
Acinopterus acuminatus	7.25	0	0	6	0	0	0
Agalliota constricta	3	0	0	0	0	0	0
Aphrophora quadrinotata	6.25	0	0	1	0	0	0
Balclutha neglecta	1.5	0	11	3	0	0	0
Bruchomorpha dorsata	12.25	0	18	14	3	7	8
Bruchomorpha jocose	16	5	3	2	1	0	0
Bruchomorpha tristis	16	0	0	3	0	0	0
Campylenchia latipes	6.25	0	0	0	0	1	0
Catonia pumila	4.5	0	0	0	0	0	0
Ceratagallia agricola	4.75	1	1	217	0	2	0
Chlorotettix galbanatus	7.25	0	0	0	0	0	0
Chlorotettix spatulatus	10	1	0	0	0	0	2
Cuerna alpina	7.25	0	0	0	0	0	0
Cuerna costalis	7.25	2	1	0	0	0	0
Delphacodes caerulata	12.25	0	0	0	0	0	0
Delphacodes puella	5.5	0	0	0	0	0	0
Delphacodes n. sp.	10	0	0	0	0	0	0
Delphacodes rotundata	9.75	0	7	4	0	0	0
Delphacodes trimaculata	14	0	0	3	0	0	0
Dikraneura angustata	3	0	1	0	0	0	0
Driotura stylata	0	0	0	0	0	0	0
Draeculacephala antica	3.75	0	1	0	0	0	0
Draeculacephala constricta	3.75	0	0	1	0	0	0
Doratura gammaroides	12.5	0	0	0	0	0	0
Empoasca bifircata	3.25	0	0	3	0	0	0
Empoasca birdii	3.25	0	0	0	0	0	0
Empoasca recurvata	3.25	0	0	12	0	0	0
Endria inimica	4	0	0	0	0	0	0
Erasmanuera vulnerata	2.5	0	0	0	0	1	0
Erythroneura octonotata	2.5	0	0	0	1	0	0
Exitianus exitiosus	4	0	0	0	0	0	0
Fitchiella robertsonii	16	0	0	0	0	0	0
Flexamia pectinata	13.25	39	1	0	8	2	88
Flexamia prairiana	11.75	38	73	10	4	33	42
Flexamia sandersi	12.25	0	0	27	0	0	0
Graphocephala hieroglyphica	4.75	0	0	0	0	0	0
Gyponana ortha	6.25	0	3	0	0	0	0
Idiocerus rotundus	5.5	0	0	0	0	0	0

Kansendria kansiensis	9.5	0	0	0	0	0	0
Laevicephalus melsheimeri	12	0	0	0	0	0	0
Laevicephalus minimus	12.75	0	14	0	36	102	9
Laevicephalus unicoloratus	11.25	0	0	0	0	0	18
Latalus personatus	9	0	0	0	0	0	0
Lepyronia quadrangularis	6.25	0	0	0	0	0	1
Liburniella ornata	3	1	0	2	0	0	0
Memnonia flavida	15	2	7	0	0	0	0
Neocoelidia tumidifrons	4.75	0	0	0	0	0	0
Paraphlepsius electus	13	1	0	6	0	1	0
Paraphlepsius irroratus	5.5	0	1	0	0	0	0
Paraulazices irrorata	6.25	0	0	1	0	0	0
Pendarus punctiscriptus	13	1	0	0	0	0	0
Penthimia americana	6.75	0	0	0	0	0	0
Planicephalus flavicostus	5	0	6	0	0	0	0
Philaenus spumarius	0	0	0	0	0	0	0
Phylloscelis pallescens	13.75	0	0	1	0	0	1
Phylloscelis atra	11.75	0	0	0	0	0	0
Poblicia fulginosa	11	1	0	0	0	0	0
Polyamia caperata	11.5	5	5	21	0	7	10
Polyamia compacta	14.75	0	0	0	4	0	1
Polyamia dilata	15.5	2	24	0	2	19	5
Polyamia weedi	12.25	0	0	0	0	0	0
Ponana puncticollis	5.5	0	0	0	0	0	0
Prairiana kansana	14	0	0	0	0	0	0
Prosapia bicincta	6.25	0	0	0	0	0	0
Rhynchomitra microrhina	6.25	0	0	0	0	0	0
Scaphytopius frontalis	4.75	0	0	0	0	0	0
Scolops angustatus	12	1	0	0	0	0	0
Scolops perdix	14.5	0	0	0	0	0	1
Scolups sulcipes	11.5	0	0	0	0	0	0
Stictocephala bisonia	5.5	0	0	0	0	0	0
Stirellus bicolor	6	1	11	4	0	1	20
Texananus decorus	10	0	0	0	0	0	0
Tibicen canicularis	5.5	0	0	0	1	0	0
Xerophloea major	8.5	0	2	4	0	0	2
Xestocephalus pulicarius	4	1	6	5	0	2	0

Appendix C.1 (cont.)

Species	**CC**	**OstermanA**	**Principia**	**Grubb Hollow**	***Windfall**	**Fults**	**Jennings**
Acanalonia bivittata	5.5	6	0	1	0	0	0
Acinopterus acuminatus	7.25	0	0	0	0	0	0
Agalliota constricta	3	0	0	1	0	0	0
Aphrophora quadrinotata	6.25	0	0	0	0	0	0
Balclutha neglecta	1.5	0	0	0	1	0	0
Bruchomorpha dorsata	12.25	9	2	18	0	1	5
Bruchomorpha jocosa	16	0	0	0	0	0	0
Bruchomorpha tristis	16	0	0	0	4	0	0
Campylenchia latipes	6.25	9	0	0	0	0	0
Catonia pumila	4.5	0	0	1	0	0	0
Ceratagallia agricola	4.75	3	0	2	0	0	0
Chlorotettix galbanatus	7.25	6	0	1	0	0	0
Chlorotettix spatulatus	10	0	0	0	0	0	0
Cuerna alpine	7.25	0	0	0	0	0	0
Cuerna costalis	7.25	0	0	0	0	0	6
Delphacodes caerulata	12.25	0	0	0	0	0	0
Delphacodes puella	5.5	0	0	1	0	0	0
Delphacodes n. sp.	10	0	0	0	0	0	0
Delphacodes rotundata	9.75	3	0	2	1	0	0
Delphacodes trimaculata	14	0	0	0	0	0	0
Dikraneura angustata	3	0	0	0	0	0	0
Driotura stylata	0	0	0	0	0	0	0
Draeculacephala antica	3.75	0	0	0	0	0	0
Draeculacephala constricta	3.75	0	0	1	0	1	0
Doratura gammaroides	12.5	0	0	0	0	3	0
Empoasca bifircata	3.25	0	0	0	0	0	0
Empoasca birdie	3.25	0	0	0	0	0	0
Empoasca recurvata	3.25	0	0	0	0	0	0
Endria inimical	4	0	0	0	0	0	0
Erasmanuera vulnerata	2.5	0	0	2	0	0	0
Erythroneura octonotata	2.5	0	0	0	0	0	0
Exitianus exitiosus	4	3	0	0	0	0	0
Fitchiella robertsonii	16	0	0	0	0	0	0
Flexamia pectinata	13.25	74	63	66	0	8	39
Flexamia prairiana	11.75	129	20	14	2	41	25
Flexamia sandersi	12.25	0	0	0	0	0	0
Graphocephala hieroglyphica	4.75	0	0	4	0	0	0
Gyponana ortha	6.25	0	0	0	0	0	0
Idiocerus rotundus	5.5	0	0	0	0	0	0
Kansendria kansiensis	9.5	0	0	0	0	0	0

Laevicephalus melsheimeri	12	0	0	0	0	0	0
Laevicephalus minimus	12.75	12	0	0	9	0	0
Laevicephalus unicoloratus	11.25	0	0	0	0	0	0
Latalus personatus	9	0	0	0	0	0	0
Lepyronia quadrangularis	6.25	0	0	5	0	0	0
Liburniella ornata	3	0	0	1	0	0	0
Memnonia flavida	15	0	0	1	0	0	0
Neocoelidia tumidifrons	4.75	0	0	0	0	0	0
Paraphlepsius electus	13	0	1	0	0	0	0
Paraphlepsius irroratus	5.5	1	0	2	0	0	0
Paraulazices irrorata	6.25	0	0	0	0	0	0
Pendarus punctiscriptus	13	0	2	0	0	0	0
Penthimia americana	6.75	0	0	0	0	1	0
Planicephalus flavicostus	5	4	0	1	0	0	1
Philaenus spumarius	0	0	0	1	0	0	0
Phylloscelis pallescens	13.75	2	0	0	0	0	0
Phylloscelis atra	11.75	0	0	0	0	0	0
Poblicia fulginosa	11	0	0	0	0	0	0
Polyamia caperata	11.5	3	3	40	4	17	0
Polyamia compacta	14.75	0	8	0	0	2	0
Polyamia dilate	15.5	11	13	22	0	4	3
Polyamia weedi	12.25	0	0	2	0	0	0
Ponana puncticollis	5.5	0	0	0	0	0	0
Prairiana kansana	14	0	0	0	0	1	0
Prosapia bicincta	6.25	0	0	0	1	0	0
Rhynchomitra microrhina	6.25	0	0	0	0	0	0
Scaphytopius frontalis	4.75	0	0	0	0	0	0
Scolops angustatus	12	2	0	0	0	0	5
Scolops perdix	14.5	0	0	0	0	0	0
Scolups sulcipes	11.5	0	0	0	0	0	0
Stictocephala bisonia	5.5	0	0	0	1	0	0
Stirellus bicolor	6	1	8	9	3	21	0
Texananus decorus	10	0	0	2	0	0	0
Tibicen canicularis	5.5	0	0	0	0	0	0
Xerophloea major	8.5	1	0	0	0	1	4
Xestocephalus pulicarius	4	1	1	3	0	0	1

Appendix C.1 (cont.)

Species	**CC**	**Demint**	**ChalfinA**	**Hanover**	**Gonterman**	**MSP**	**ChalfinB**
Acanalonia bivittata	5.5	0	0	0	0	0	0
Acinopterus acuminatus	7.25	0	0	0	0	0	0
Agalliota constricta	3	0	0	0	0	1	0
Aphrophora quadrinotata	6.25	0	0	0	0	0	0
Balclutha neglecta	1.5	0	0	0	0	0	0
Bruchomorpha dorsata	12.25	0	0	2	1	0	1
Bruchomorpha jocose	16	0	0	0	0	0	0
Bruchomorpha tristis	16	0	0	0	0	0	0
Campylenchia latipes	6.25	0	0	0	0	0	0
Catonia pumila	4.5	0	0	0	0	0	0
Ceratagallia Agricola	4.75	0	0	0	0	0	0
Chlorotettix galbanatus	7.25	0	0	0	0	0	0
Chlorotettix spatulatus	10	0	0	0	0	0	0
Cuerna alpine	7.25	0	0	0	0	0	0
Cuerna costalis	7.25	0	0	0	0	0	1
Delphacodes caerulata	12.25	0	0	0	0	0	0
Delphacodes puella	5.5	0	0	0	0	0	0
Delphacodes n. sp.	10	0	0	0	0	0	0
Delphacodes rotundata	9.75	0	0	2	0	0	0
Delphacodes trimaculata	14	0	0	0	0	0	0
Dikraneura angustata	3	0	0	0	0	0	0
Driotura stylata	0	0	0	0	0	0	0
Draeculacephala antica	3.75	1	2	0	0	0	5
Draeculacephala constricta	3.75	3	0	0	0	0	0
Doratura gammaroides	12.5	0	0	0	0	0	0
Empoasca bifircata	3.25	0	0	0	0	0	0
Empoasca birdie	3.25	0	0	0	0	0	0
Empoasca recurvata	3.25	1	0	0	0	0	0
Endria inimica	4	0	0	0	0	1	0
Erasmanuera vulnerata	2.5	0	0	0	0	0	0
Erythroneura octonotata	2.5	0	0	0	0	0	0
Exitianus exitiosus	4	0	0	0	0	0	0
Fitchiella robertsonii	16	1	0	0	0	0	0
Flexamia pectinata	13.25	29	2	0	132	0	8
Flexamia prairiana	11.75	10	8	11	18	2	9
Flexamia sandersi	12.25	0	0	0	0	0	0
Graphocephala hieroglyphica	4.75	0	0	0	0	0	0
Gyponana ortha	6.25	1	0	0	0	0	0
Idiocerus rotundus	5.5	0	0	0	0	0	0

Kansendria kansiensis	9.5	0	0	1	0	0	0
Laevicephalus melsheimeri	12	0	0	0	0	0	0
Laevicephalus minimus	12.75	0	0	0	0	0	0
Laevicephalus unicoloratus	11.25	0	0	5	0	0	0
Latalus personatus	9	0	0	2	0	0	0
Lepyronia quadrangularis	6.25	0	0	0	0	0	0
Liburniella ornata	3	0	0	0	0	0	0
Memnonia flavida	15	0	0	0	0	1	1
Neocoelidia tumidifrons	4.75	0	2	0	0	0	0
Paraphlepsius electus	13	0	0	0	0	0	0
Paraphlepsius irroratus	5.5	0	0	0	0	0	0
Paraulazices irrorata	6.25	0	0	0	0	0	0
Pendarus punctiscriptus	13	0	0	0	0	0	0
Penthimia americana	6.75	0	0	0	0	0	0
Planicephalus flavicostus	5	0	0	0	0	0	0
Philaenus spumarius	0	0	0	0	0	0	0
Phylloscelis pallescens	13.75	0	0	0	0	0	0
Phylloscelis atra	11.75	0	0	0	1	0	0
Poblicia fulginosa	11	0	0	0	0	0	0
Polyamia caperata	11.5	2	2	90	1	0	16
Polyamia compacta	14.75	0	0	0	0	0	0
Polyamia dilata	15.5	0	3	0	29	0	0
Polyamia weedi	12.25	0	0	0	0	0	0
Ponana puncticollis	5.5	0	0	0	0	0	0
Prairiana kansana	14	0	0	0	0	0	0
Prosapia bicincta	6.25	0	0	0	0	0	0
Rhynchomitra microrhina	6.25	4	0	0	0	0	0
Scaphytopius frontalis	4.75	0	0	0	0	0	0
Scolops angustatus	12	1	0	0	0	0	0
Scolops perdix	14.5	0	0	0	0	0	0
Scolups sulcipes	11.5	0	0	0	0	0	0
Stictocephala bisonia	5.5	0	0	0	0	0	0
Stirellus bicolor	6	4	2	7	33	2	4
Texananus decorus	10	0	0	0	0	0	0
Tibicen canicularis	5.5	0	0	0	0	0	0
Xerophloea major	8.5	0	0	0	0	0	0
Xestocephalus pulicarius	4	0	0	0	0	0	0

Appendix C.1 (cont.)

Species	**Species Code**	**CC**	***Snyder**	**BGB**	**Olin**	**BGA**
Acanolonia bivittata	Acabiv	5.5	0	0	0	0
Acinopterous acuminatus	Aciacu	7.25	0	0	0	0
Agallia constricta	Agrcon	3	0	0	0	0
Aphrophora quadrinotata	Aphqua	6.25	0	0	0	0
Balclutha neglecta	Balneg	1.5	0	0	0	0
Bruchomorpha dorsata	Brudor	12.25	0	0	1	0
Bruchomorpha jocosa	Brujoc	16	0	0	0	0
Bruchomorpha tristis	Brutis	16	0	0	0	0
Campylenchia latipes	Camlat	6.25	0	0	0	0
Catonia pumila	Catpum	4.5	0	0	0	0
Ceratagallia agricola	Ceragr	4.75	30	6	0	4
Chlorotettix galbanatus	Chlgal	7.25	0	0	0	0
Chlorotettix spatulatus	Chlspa	10	0	0	0	0
Cuerna alpina	Cuealp	7.25	0	5	0	0
Cuerna costalis	Cuecos	7.25	0	0	5	0
Delphacodes caerulata	Delcar	12.25	0	0	0	0
Delphacodes puella	Delpue	5.5	0	0	0	0
Delphacodes n.sp	Delnsp	10	1	0	0	0
Delphacodes rotundata	Delrot	9.75	0	0	0	0
Delphacodes trimaculata	Deltri	14	0	0	0	0
Dikraneura angustata	Dikang	3	0	0	0	0
Doratura stylata	Dristy	0	0	0	0	0
Draeculacephala antica	Draant	3.75	0	12	0	7
Draeculacephala constricta	Dracon	3.75	0	0	4	0
Driotura gammaroides	Dorgam	12.5	0	0	0	0
Empoasca bifurcata	Empbif	3.25	0	0	0	0
Empoasca birdii	Empbir	3.25	1	0	0	0
Empoasca recurvata	Emprec	3.25	0	0	0	0
Endria inimica	Endini	4	0	0	0	0
Erasmoneura vulnerata	Eravul	2.5	0	0	0	0
Erythroneura octonotata	Eryoct	2.5	0	0	0	0
Exitianus exitiosus	Exiexi	4	0	0	0	0
Fitchiella robertsonii	Fitrob	16	0	0	0	0
Flexamia pectinata	Flepec	13.25	4	1	0	1
Flexamia prairiana	Flepra	11.75	12	0	0	0
Flexamia sandersi	Flesan	12.25	0	0	0	0
Graphocephala hieroglyphica	Grahie	4.75	0	0	0	0
Gyponana ortha	Gyport	6.25	0	0	0	0
Idiocerus rotundus	Idirot	5.5	0	0	0	0
Kansendria kansiensis	Kankan	9.5	1	0	0	0

Laevicephalus melshemerii	Laemel	12	0	0	1	0
Laevicephalus minimus	Laemin	12.75	0	0	0	0
Laevicephalus unicoloratus	Laeuni	11.25	9	0	1	0
Latalus personatus	Latper	9	1	0	0	0
Lepyronia quadrangularis	Lepqua	6.25	1	0	0	0
Liburniella ornate	Liborn	3	0	0	0	0
Memnonia flavida	Memfla	15	0	0	0	0
Neocoelidia tumidifrons	Neocoe	4.75	0	0	0	0
Paraphlepsius electus	Parele	13	0	0	0	0
Paraphlepsius irroratus	Parirr	5.5	0	0	0	0
Paraulacizes irrorata	Parirr1	6.25	0	0	0	0
Pendarus punctiscriptus	Penpun	13	0	0	0	0
Penthimia Americana	Pename	6.75	1	0	0	0
Planicephalus flavicostus	Plafla	5	0	0	0	0
Philaenus spumarius	Phispu	0	0	0	0	0
Phylloscelis palliscens	Phypal	13.75	0	0	0	0
Phylloscelis atra	Phyatr	11.75	0	0	0	1
Poblicia fulginosa	Pobful	11	0	0	0	0
Polyamia caperata	Polcap	11.5	7	0	0	0
Polyamia compacta	Polcom	14.75	0	0	0	0
Polyamia dilate	Poldil	15.5	0	0	0	0
Polyamia weedi	Polwee	12.25	0	0	0	0
Ponana puncticollis	Ponpun	5.5	0	0	0	0
Prairiana kansana	Prakan	14	0	0	0	0
Prosapia bicinta	Probic	6.25	0	0	0	0
Rhynchomitra microrhina	Rhymic	6.25	0	1	0	1
Scaphytopius frontalis	Scafro	4.75	1	0	0	0
Scolops angustatus	Scoang	12	0	0	0	0
Scolops perdix	Scoper	14.5	0	0	0	0
Scolops sulcipes	Scosul	11.5	0	0	0	0
Stictocephala bisonia	Stibis	5.5	0	0	0	0
Stirellus bicolor	Stibic	6	62	3	3	1
Texananus decorus	Texdec	10	1	0	0	0
Tibicen canicularis	Tibcan	5.5	0	0	0	0
Xerophloea major	Xermaj	8.5	0	0	0	1
Xestocephalus pulicarius	Xespul	4	0	0	0	0