

**TALLGRASS PRAIRIE REMNANTS AS “LIVING MUSEUMS”:  
LANDSCAPE CONTEXT, METACOMMUNITY DYNAMICS, AND PRIVATE  
MANAGEMENT PRACTICES OF NATIVE PRAIRIE HAY MEADOWS**

**by**

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## CHAPTER ONE

# **Tallgrass Prairie Remnants as Living Museums: Historical and Environmental Factors Overpower Metacommunity Dynamics**

### ABSTRACT

Metacommunity theory is currently the dominant framework for understanding processes of species assembly. It predicts that local communities are linked in a network by dispersal of species, the establishment of which is filtered through local environmental conditions. In a fragmented landscape, such as native tallgrass prairie remnants within eastern Kansas, smaller patch area, greater isolation, and poorer matrix quality are predicted to result in (1) decreased species richness, (2) decreased site ‘quality’ (measured with a Floristic Quality Index), and (3) decreased presence of specialist species. Within five counties in eastern Kansas, 301 native tallgrass remnants were surveyed and mapped. Sites were grouped by cluster analysis to test these hypotheses of metacommunity theory. The total area of a site, as well as habitat heterogeneity of sites, contributed to richness and quality of sites, but isolation and matrix quality did not. In fact, isolation and matrix quality appeared to contribute to decreasing richness and quality of sites. The relationship of isolation with richness and quality can be explained by regional environmental variation having a greater influence on community composition than habitat connectivity in this system. The relationship of landscape matrix quality with richness and floristic quality may be explained by the historic non-random process of human landscape changes at the time of settlement, with soil quality rather than connectivity affecting composition. These results suggest that the hypothesis

that dispersal plays a role in the community assembly of tallgrass prairie remnants is probably false. Although species-rich communities may persist for decades and possibly centuries after fragmentation, in the long term the effects of random local extinction and the burden of “extinction debt” have been underestimated for native tallgrass prairies, and the forecast for long-term persistence of these communities is grim.

**KEY WORDS:** fragmentation, island biogeography, coexistence, competition-colonization trade-off, dispersal, extinction

## INTRODUCTION

Metacommunity theory has become the dominant framework for understanding species assembly on the regional level. Metacommunity theory draws on the principles of island biogeography theory (MacArthur and Wilson 1967), e.g., relationships between colonization, extinction, site area, and site isolation; as well as community ecology principles of species interactions, including competition and trophic interactions (Leibold et al. 2004, He et al. 2005). A *metacommunity* is defined as “a set of local communities that are linked by dispersal of potentially interacting species” (Leibold et al. 2004). Unlike island biogeography theory, metacommunity theory allows for the consideration of variable matrix quality, which can affect rates of dispersal between sites (Laurance 2008, Prugh et al. 2008). The metacommunity concept also allows ecologists to consider factors driving species assembly across multiple spatial scales, and particular emphasis has been given to distinguishing between local (e.g., environmental conditions or species interactions) and regional (e.g., immigration of organisms from the regional species pool)

effects, and the interactions between the two scales (Mouquet and Loreau 2002, Leibold et al. 2004, Urban 2004, Chase 2005).

Fragmented landscapes, such as wetlands (Matthews et al. 2005), tropical forest fragments (Galanes and Thomlinson 2009), and grassland fragments (Krauss et al. 2004) are considered ideal systems for testing metacommunity theory *in situ* because they consist of ‘islands’ of suitable habitat within landscapes of variable degrees of unsuitable habitat. The limitation of studying metacommunity dynamics *in situ*, rather than using a model, is that generally the data obtained is a snapshot of species composition at one point in time. Therefore, the dynamics of migration and extinction inferred to take place are tested indirectly.

Numerous studies have shown that species richness increases with increasing patch area and perimeter and decreasing isolation (Krauss et al. 2005, Matthews et al. 2005, Galanes and Thomlinson 2009). Floristic quality, an estimate of the conservation value of a vascular plant community, and the proportion of native or specialist species also increase with increasing area and perimeter and decreasing isolation (Matthews et al. 2005). However, in many cases the suitability of the intervening landscape, or matrix, may be more important than area or isolation (Debinski and Holt 2000, Prugh et al. 2008), because the presence of corridors or suitable intermediary habitat increase the ability for species to disperse and colonize, and in these cases, the degree of dispersal limitation drives metacommunity dynamics.

### *Extinction Debt*

Tilman et al. (1994) demonstrated quantitatively that random destruction of habitat results in time-delayed but deterministic extinction of dominant competitors. This

time lag between sufficient habitat destruction and the eventual extinction of the best competitors (or weakest colonizers) is known as “extinction debt.”

Although extinction debt is difficult to detect, empirical studies have shown evidence for extinction debt for a wide variety of communities, and a strong focus has been placed on vascular plant communities (reviewed in Kuussaari et al. 2009). One of the most succinct examples of evidence for extinction debt was given by Vellend et al. (2006), who show that European forests which have undergone fragmentation since 1775 owe an extinction debt, and given time, will lose “slow” species (poor colonizers, better competitors) until their community composition more closely resembles that of the forests which have been fragmented for the past thousand years. European grassland communities which have been fragmented for 70 years may owe an extinction debt of 40% of specialist plant species (Helm et al. 2006).

#### *Eastern Kansas Tallgrass Prairie*

Historically, North American prairie vegetation extended from Manitoba and Saskatchewan, Canada, south to Texas and Mexico, with a wide peninsula extending into Illinois, Indiana, and Ohio. The eastern portion of this range was dominated by tallgrass prairie prior to Euro-American settlement in this region. However, within the past 200 years, human landscape changes resulted in the destruction of over 99% of the original tallgrass prairie (Howe 1994). Most native habitat loss occurred quickly at the end of the 19<sup>th</sup> century, and the rate of habitat loss and fragmentation has slowed since. Since Euro-American settlement of eastern Kansas following the Homestead Act of 1862, the landscape has been dominated by agriculture and introduced cool-season grasses (Foster et al. 2009). Native tallgrass prairie persists in two forms: as grazed pasture, which is



often severely degraded, and as prairie hay meadows, which have been managed mainly for hay production since the time of settlement.

Eastern Kansas hay meadows are typically hayed annually in the summer, usually in July. Although native pastures or mixed-use hay meadows/pastures are common throughout the area, “high-quality” native prairie hay meadows, or those most closely resembling species composition prior to Euro-American settlement, are only rarely used for cattle grazing. Continued grazing degrades the plant community, resulting in losses of specialist species and increases of exotic species. Prescribed burning of hay meadows is common; about one-third of owners burn every 1-4 years (Chapter 2). Although resting (not haying or burning) of native prairie hay meadows is recommended to serve as refuges for wildlife and to allow seed production of native plants (Kindscher and Byczynski 2009), private landowners very rarely rest their hay meadows; nearly 80% of owners have never rested their prairie (Chapter 2). When resting does occur, it is likely accidental; a farmer may fail to hay a meadow in a year due to adverse weather conditions or the low priority of obtaining hay compared to harvesting grain crops.

Due to their high level of fragmentation, native prairie hay meadows of eastern Kansas provide an ideal system for testing hypotheses of metacommunity theory and especially for looking for evidence of differential local extinction of specialist species in areas which have suffered greater degrees of habitat loss and fragmentation. A central hypothesis of this study is that smaller patch area, greater isolation, and poorer matrix quality would result in (1) decreased species richness, (2) decreased site ‘quality’ (measured with a Floristic Quality Index), and (3) decreased presence of specialist species.

## METHODS

### *Study site*

Native tallgrass prairies were surveyed in Anderson, Douglas, Leavenworth, Linn, and Miami counties in eastern Kansas. Most of the study area is within the Osage Cuestas physiographic province, though the northern portion of this area falls within the Glaciated Region, which had glacial cover during a portion of the Pleistocene. Major underlying strata are limestone and shale, with scattered sandstone deposits. Most of the five-county region is primarily agricultural, with crops of corn, soybeans, wheat, grain sorghum, and alfalfa, as well as pastures of native warm-season grasses or cultivated cool-season grasses (Table 1.1). Douglas, Leavenworth, and Miami counties also have suburban development west of Kansas City. High-quality native prairie fragments are scattered throughout all counties, making up less than 1% of the landscape, and are mostly used as hay meadows (Kindscher et al. 2005, Kindscher et al. 2009).

### *Field methods*

In 2004 and 2005, the Kansas Natural Heritage Inventory (KNHI) of the Kansas Biological Survey identified and surveyed high-quality natural areas of tallgrass prairie in Douglas, Leavenworth, and Miami counties (Kindscher et al. 2005). In 2008 and 2009, a similar survey was conducted for Anderson and Linn counties (Kindscher et al. 2009). High-quality natural areas support plant communities that closely approximate the vegetation that existed prior to Euro-American settlement. Field crews conducted surveys along county roads and examined topographic maps and satellite imagery to identify high-quality native prairie hay meadows. Once a landowner granted permission,

field crews assessed site condition and compiled a list of all plant species present (nomenclature was from the USDA Plants Database [USDA, NRCS 2010]) during a walk-through of the site. Methodology of site identification and censusing are described in Kindscher et al. (2005) and Kindscher et al. (2009).

Particular attention was given to identifying previously unknown populations, and verifying known populations, of Mead's milkweed (*Asclepias meadii* Torr ex A. Gray), a species listed as threatened under the Endangered Species Act (USFWS 1988). *A. meadii* is a long-lived clonal perennial native to tallgrass prairie, whose range has been severely restricted due to habitat loss and fragmentation. As of 2003, only 171 populations of *A. meadii* were known about (USFWS 2003); yet populations were present on 123 of 349 sites surveyed within the five counties.

#### *Testing effects of area and isolation on species richness and floristic quality*

All sites were digitally mapped in ArcMap 9.2 (ESRI 2008). Digitization of sites was done by field crews after visiting sites, with reference to aerial photography (USDA 2003). Area and perimeter of sites, and two measures of isolation (distance to nearest site and average distance to nearest three sites) were calculated using Hawth's Tools extension for ArcMap (Beyer 2004). Landscape composition immediately surrounding sites (using a 0.5 km buffer) was summarized by calculating the percent of seven land use/land cover categories, with the use of a Kansas land cover map (Kansas Applied Remote Sensing Program 2005). The distance of 0.5 km was chosen in order to incorporate effects of adjacent fields (e.g., nutrient run-off from cropfields), while also providing effects of the larger landscape that sites were situated in. While a larger distance class could have been used to include all of the potential sites that a new

colonizer could disperse to from the site, the dispersal distance of most prairie plants is both highly variable and largely unknown. Calculations were done in ArcMap, and the land cover categories included were: urban, cropland, grassland, Conservation Reserve Program (CRP) fields, woodland, water, and other. Soil data was downloaded from the USDA National Resources Conservation Service (NRCS), and the average depth to bedrock and average water availability to 150 cm were calculated for each site (Soil Survey Staff, NRCS, USDA n.d.).

The following measures were taken to ensure only one plant community type, upland tallgrass prairie, was included in the analysis. The NRCS soil data was used to exclude sites with soil types that were occasionally or frequently flooded, as these floodplain soils would contain “wet prairie” plant communities. All species were classified by their wetland indicator status (USFWS 1988; USFWS 1993; USDA, NRCS 2010), and all obligate wetland species (which occur exclusively in wetlands, USFWS 1988) were removed from analysis. Tree and large shrub species, which were only present as seedlings and annually mowed resprouting individuals (which never become reproductive), were also removed from analysis.

Floristic quality was calculated for each site in the following manner. All species were either assigned a Kansas coefficient of conservatism or classified as an exotic species (Freeman and Morse 2002). Coefficients of conservatism are values which are assigned to native species and range from 0 to 10, indicating the relative fidelity of species' occurrence to habitats relatively unaltered from pre-settlement times. For each site, a floristic quality index (FQI) was calculated according to the formula from Swink and Wilhelm (1994):  $FQI=R/\sqrt{N}$ , where R equals the sum of coefficients of

conservatism for all plants recorded at the site and N equals the total number of native species recorded. Plants with high coefficients of conservatism tend to be habitat specialists, which are more sensitive to habitat disturbances than plants with low coefficients of conservatism, which tend to be generalists that are more tolerant of a variety of conditions. Non-native species are not assigned a coefficient of conservatism, and so are not considered in the equation. Total species richness, native species richness, and richness of native upland species (according to the wetland indicator status) were calculated for each site.

Stepwise linear regressions were conducted in PASW 17.0 (2009) to determine the effects of area, perimeter, isolation, percent of land use/land cover categories within 0.5 km, and soil qualities on overall species richness, native species richness, native upland species richness, and FQI of sites. (See Appendix 2 for a correlation matrix of variables entered into all models.) The parameters were  $p < 0.05$  to enter model and  $p < 0.10$  to be removed from model. In order to assess whether species which differ in their competition-colonization trade-offs are differentially affected by isolation, regressions were also conducted on species with CoCs of 0-2 (assumed to be generalist species, or better colonizers) and species with CoCs of 7-10 (assumed to be specialist species, or better competitors).

Additionally, sites were analyzed for gradients of species composition across the landscape. A Mantel test between Euclidean distance of sites and Jaccard's community dissimilarity showed significant spatial autocorrelation of community composition across the study area. In PC-ORD (McCune and Mefford 2006), a cluster analysis was conducted to group communities (with Ward's linkage method and Jaccard's distance

measure). Community groups separated more or less geographically, with some overlap. The groups of the cluster analysis were corroborated by a principal components analysis in PC-ORD of species presence, which resulted in an axis that grouped sites similarly to the cluster analysis.

Community patterns across spatial scales were analyzed in two ways. A Mantel correlogram was conducted using the Ecodist package in R (Goslee and Urban 2007) to show at what spatial scale communities were positively correlated, and at what threshold communities became negatively correlated with distance (Legendre and Fortin 1989). Community groups determined by the cluster analysis were analyzed separately for relationships among area, isolation, and landscape matrix composition on quality and richness. The same parameters were used for these models as were used for the entire study area, but if no significance was found, the parameters were relaxed to allow  $p < 0.10$  to enter the model and  $p < 0.15$  to be removed from the model (Legendre and Fortin 1989).

## RESULTS

### *Cluster Analysis*

A total of 301 sites were used in the analysis, 72 of which were excluded from analyses involving isolation metrics because they fell within 5 km of the edge of the study area. Overall, sites averaged 93.3 ha, with an FQI of 38.0, and had 86 species, 76 of which were native and 15 of which had a CoC of 7 or above (Table 1.2).

The first split in the cluster analysis divides northern sites (almost all of the sites within Leavenworth, Douglas, and Miami counties) from southern sites (all of the sites within Anderson and Linn counties, plus six sites from the northern counties; see Figure

1.1). The northern sites further split into two clusters, one which includes almost every site in the Leavenworth and Miami counties, and one which includes most sites in Douglas County. These clusters are designated N1 and N2. The southern sites further divide into four main clusters, designated S1, S2, S3, and S4, with a rough gradient from east to west, though with substantial overlap, particularly between S1, S2, and S3 (Figure 1.1; see tables of representative species of each cluster in Appendix 1).

Species which are most overrepresented in Cluster N1, relative to their average presence throughout the study area, are *Comandra umbellata*, *Lithospermum canescens*, *Coreopsis palmata*, *Hesperostipa spartea*, and *Phlox pilosa*. These sites have disproportionately low presences of *Callirhoe alcaeoides*, *Baptisia australis*, *Lespedeza virginica*, *Ambrosia psilostachya* and *Asclepias sullivantii*. N1 sites have higher average total species richness, higher average FQI, and have a greater average depth to bedrock than other clusters (Table 1.3, Table 1.4). This cluster is distinguished from the others by being mainly in the glaciated region; deeper soil substrate and glacial till likely affect the vegetation community of this area. In their vegetation descriptions of Kansas, Lauver et al. (1999) differentiate between Central Tallgrass Prairie, in the glaciated, northern portion of the study area, and Southeast Tallgrass Prairie in the south.

Cluster N2 is characterized by having disproportionately high presences of the native species *Dalea multiflora*, *Comandra umbellata*, *Hesperostipa spartea*, *Physalis pumila*, and *Oenothera speciosa*. It also has the highest presence of the exotic *Bromus inermis*. These sites have disproportionately low presences of *Penstemon tubiflorus*, *Allium canadense*, *Carex bushii*, *Callirhoe alcaeoides*, and *Baptisia australis*. The vegetation of N2 and N1 are quite similar, but N2 contains a much higher presence of

species found in dry, rocky areas, such as *Dalea multiflora*, *Oenothera speciosa*, *Desmanthus illinoense*, and *Asclepias stenophylla*; whereas N1 contains a higher presence of species associated with rich, sandy, or moist areas, such as *Penstemon tubifloris* and *Pedicularis canadensis* (McGregor et al. 1986). N2 sites have the highest average percent cropland in the areas surrounding them (Table 1.4).

Species which are overrepresented in Cluster S1 are mainly annual and weedy species: *Valerianella radiata*, *Myosotis verna*, *Carex brevior*, *Desmodium paniculatum*, and *Chaerophyllum tainturieri*, and exotic winter annual *Barbarea vulgaris*. These sites have disproportionately low presences of *Linum sulcatum*, *Elymus virginicus*, *Dalea purpurea*, *Desmodium illinoense*, and *Solidago nemoralis*. Cluster S1 has the highest richness and FQI of the southern clusters (Table 1.3), and also has the highest presence of the federally-threatened *Asclepias meadii* (Figure 1.2).

Cluster S2 is characterized by having disproportionately high presences of the native species *Elymus virginicus*, *Cyperus echinatus*, *Setaria parviflora*, *Juncus interior*, and *Desmanthus illinoensis*. These sites have disproportionately low presences of *Viola pedatifida*, *Lithospermum canescens*, *Phlox pilosa*, *Dicanthelium oligosanthes*, and *Symphotrichum oolentagiense*. These sites have a high presence of exotic and native ‘weedy’ species, and have the lowest average FQI. S2 has the lowest average presence of *Asclepias meadii* (Figure 1.2). Clusters S1 and S2 overlap geographically, but differ significantly in site quality. Of all the clusters, S2 sites have the highest average percent grassland in the areas surrounding them, while S1 has the lowest (Table 1.4).

Cluster S3 is characterized by having disproportionately high presences of the native species *Lespedeza virginica*, *Baptisia australis*, *Asclepias viridiflora*, *Pediomelum*



*esculentum*, and *Brickellia eupatorioides*. These sites have disproportionately low presences of *Helianthus grosseserratus*, *Zizia aurea*, *Liatris pycnostachya*, *Spartina pectinata*, and *Solidago canadensis*. Sites in S3 have the lowest average depth to bedrock, and the highest proportion of upland sites (Table 1.4). This cluster is made up of the driest and rockiest sites.

Cluster S4 is characterized by having disproportionately high presences of the native species *Physostegia angustifolia*, *Coreopsis grandiflora*, *Allium canadense*, *Camassia angusta*, and *Packera plattensis*. These sites have disproportionately low presences of *Ratibida pinnata*, *Silphium laciniatum*, *Comandra umbellata*, *Silphium integrifolium*, and *Lespedeza violacea*.

### *Regression Results*

For the entire study area, perimeter and isolation were both positively correlated with FQI (Table 1.5). For half of the clusters, perimeter was positively correlated with FQI; though for S3 area and perimeter/area ratio negatively correlated with FQI. Depth to bedrock was a significant factor of FQI for N2. Percent CRP immediately surrounding sites was negatively correlated with FQI for S1.

Perimeter was also a significant predictor for total species richness for the entire study area and four of the six clusters (Table 1.6). Although isolation was positively correlated with richness for the entire study area, it was negatively correlated with richness for cluster N2 and not significant for any of the other clusters. Percent cropland immediately surrounding sites was positively correlated with species richness for the entire study area and for cluster S2. For cluster N2, depth to bedrock was positively correlated with richness.

Results for native species richness (Table 1.7) and native upland species richness (Table 1.8) were similar to the results for total richness. Greater perimeter, greater percent cropland, and less percent grassland and CRP were common predictors of greater native or native upland species richness. In cluster S4, isolation was weakly negatively correlated with native species richness. In cluster S3, depth to bedrock was positively correlated with upland species richness.

Significant predictors, according to stepwise linear regressions, of richness of conservative species (defined as species with CoC values of 7-10; Table 1.9) and native ‘weedy’ species (defined as species with CoC values of 0-2; Table 1.10) were variable across clusters. Perimeter was positively correlated with conservative species across the entire study area and for half of the clusters, and with weedy species across the entire study area, but was not a significant predictor for weedy species within individual clusters. Sites with a smaller area tended to have more weedy species across the study area. Within cluster S3, the hypothesized differential species’ response to isolation was supported: isolation was negatively correlated with conservative species but positively correlated with weedy species. However, the northern clusters showed the opposite: isolation was positively correlated with conservative species for the entire study area and for cluster N1, and was negatively correlated with weedy species in cluster N2.

### *Summary of Isolation Effects*

Isolation, either as distance to the nearest site or average distance to the three nearest sites, was positively correlated with richness and floristic quality across the entire study area, but negatively correlated with richness within a few clusters (Table 1.11). Support for the hypothesis that conservative species are differentially affected by

isolation was ambiguous, as it was supported in cluster S3, but the opposite trend was found in the northern clusters.

Cluster N1 had both the greatest isolation between sites and the greatest species richness (Table 1.3). These results indicate that the apparent positive relationship when considering the entire study area is due to differences between clusters (Figure 1.3, Figure 1.4), which differ in species composition, rather than to a real effect between isolation and richness.

A Mantel correlogram (Figure 1.5) showed there was positive spatial autocorrelation between community composition of sites in distance classes of approximately 70 km; in greater distance classes there is negative spatial autocorrelation, indicating either a vegetation gradient or step-like structure.

## **DISCUSSION**

Metacommunity theory predicts that area, isolation, and matrix quality drive species richness and floristic quality of fragmented landscapes. However, two of these predictions were not supported by the data from these native tallgrass prairie hay meadows. The area of a site, strongly correlated with perimeter, does appear to contribute to species richness and FQI. For most of the regression analyses, perimeter consistently was the strongest predictor of richness or quality. Perimeter is a combination of area plus site diversity, as boundaries were drawn to exclude streams, swales, or forested or shrubby areas (Figure 1.6). While in most studies an increase in perimeter can be detrimental to a community (due to ‘edge effect’), in this case an increase in site

edge reflects the presence of environmental heterogeneity within a site, which increases diversity.

Isolation was predicted to decrease richness and floristic quality, but for the study area as a whole, isolation appeared instead to increase richness and quality. This seemingly contradictory result was due to the variation in richness across the community gradient (Figure 1.3). The sites with the highest richness and FQI were the sites in N1, the sites which were located in the glaciated region. These sites were situated in areas that received more development due to their proximity to the Kansas City Metropolitan Area, and so were also more isolated than sites which were in the unglaciated region, leading to an apparent correlation between isolation and richness (Figure 1.4). Caution must be taken when interpreting results of data over a large geographic area, because a difference in one geographic area (such as N1) can skew a dataset so that a linear relationship appears to be present when, in fact, it may not.

If matrix quality contributed to species richness or floristic quality, then the matrix surrounding sites with the highest richness and FQI would consist of more grassland and CRP, which could support sink populations that might serve as an intermediary for dispersal between higher-quality habitats. On the other hand, sites surrounded by urban areas and cropland would have low richness and FQI, due to both the unsuitability of the matrix and to detrimental run-off that could cause degradation. However, the results of the regression analyses were exactly the opposite of the predictions. A greater percentage of cropland, and a lower percentage of grassland and CRP, contributed to higher richness and FQI. This result is not likely due to direct effects

of the matrix on site community composition, but rather is likely due to a combination of soil and environmental qualities and historical factors of fragmentation.

Habitat fragmentation is usually a non-random process. In eastern Kansas, locations in low-lying areas close to streams or rivers (which in addition to providing access to water also contain the richest soils) were preferentially plowed for agriculture upon settlement (Webb 1981 [1931], Forsythe 1977). Thin-soiled, rocky areas were more likely to be left as pasture or used as native or introduced hay meadows.

Additionally, it is possible that fields with poorer soils are more likely to be converted to CRP fields than sites with rich soils and high productivity. Weber et al. (2002) found that the surrounding landscape of CRP fields contained less row crops than the landscape surrounding non-CRP fields, and the row crops that were near CRP fields had a smaller patch size. Because the main goal of CRP is to convert lands which are highly erodible (McKenzie 1997), CRP is aggregated in areas which are less productive as cropland (Weber et al. 2002). The relationship between matrix composition and richness/FQI may well be explained by richer soils contributing both to species-rich, high-quality native prairies, and to a landscape matrix consisting of more cropland but less grassland and CRP.

### *Conservation Implications*

Superficially, species richness does not appear to be affected by increased fragmentation, nor is there evidence that more conservative species are beginning to be lost disproportionately to ‘weedier’ species that are better able to disperse. However, the fact that the hypotheses of metacommunity dynamics were rejected (decreased isolation does not correlate with greater richness or quality, and grassland and CRP fields do not

serve as corridors for dispersal) means that the risk of species loss due to fragmentation is exponentially greater than if these prairies were acting as a metacommunity with dispersal occurring between sites. Although no evidence for repayment of extinction debt was found, previous studies have found that fragmented communities may owe extinction debt hundreds of years after fragmentation (Helm et al. 2006, Vellend et al. 2006, Kuussaari et al. 2009), so it is not surprising that evidence of differential responses to fragmentation are not apparent a mere 150 years after initial fragmentation. Species with long generations and relatively stable populations are at the greatest risk for delayed future extinctions (Kuussaari et al. 2009), and many prairie species fall into this category. Sperry (1982) categorized nine prairie species as ‘documentary species,’ defined as species which had persisted for forty years after a restoration planting, but which had not spread from their original planting areas. All of Sperry’s documentary species (*Amorpha canescens*, *Baptisia bracteata* [*leucophaea*], *Ceanothus americanus*, *Helianthus pauciflorus* [*laetiflorus*], *Phlox pilosa*, *Rosa carolina*, *Silphium integrifolium*, *S. laciniatum*, and *Hesperostipa* [*Stipa*] *spartea*) were found on at least 25% of these hay meadows.

One species of conservation interest, which may also be long-persisting but poor at dispersing, is *Asclepias meadii*, which has large, apparently stable populations throughout this area (despite being extirpated from most of its original range). *A. meadii* has long generations; Betz (1989) estimated that an individual could live up to a century; recent studies that explicitly study survival rates suggest adult plants have an average longevity of 20 years (Alexander et al. 2009; H. M. Alexander, pers.comm.). However, there is evidence that *A. meadii* persists almost solely through vegetative reproduction

and individual persistence, because the cutting of hay prevents sexual reproduction from occurring (Kettle et al. 2000). Because native prairie hay meadows are cut for hay every year, it is likely that many species are not able to complete their life cycles to successfully reproduce and disperse.

Data are severely lacking on dispersal of grassland species, although Schott and Hamburg (1997) showed that seed rain of an old field adjacent to native tallgrass prairie had a reduced richness and reduced quantity of seed rain, indicating that prairie species could be dispersal-limited. In eastern Kansas, it appears that higher soil quality (indicated by location in glaciated areas and/or areas with high proportion of cropland) rather than connectivity contributes to richness and floristic quality. Each prairie is a relict of the fine-scale region in which it was situated prior to fragmentation; a “living museum.” And if each site is subject to the risks of random local extinction, without the possibility of rescue from dispersal from nearby sites, then the outlook for the conservation of the richness of species found native prairie communities is indeed dire.

## CHAPTER TWO

### **The Value of Native Prairie Hay Meadows: Management Practices of Private Landowners and Implications for Conservation**

#### **ABSTRACT**

Privately-owned prairie hay meadows make up most of the existing remnants of native tallgrass prairie in eastern Kansas. Conservationists view them as reservoirs of biodiversity and as the last remaining havens for rare species. The unique management of native hay meadows is a factor in their harboring most locations of the federally-threatened Mead's milkweed (*Asclepias meadii*). A survey of 48 private landowners in Anderson and Linn counties, Kansas, was conducted to assess the current management practices of privately-owned native prairie hay meadows. Most native prairie hay meadows are not grazed (41 of 48), but those that are have a lower quality plant community composition than those that are not (*t*-test,  $p = 0.07$ ). A variety of burning regimes are practiced by 71% of landowners, but only 21% of respondents stated that their hay meadows had ever been rested, and never more frequently than every 5 years. Although some landowners show interest in conservation, native prairie hay meadows are more often managed for maximum yield and profit of hay than for conservation interests.

**KEY WORDS:** tallgrass prairie, grassland, floristic quality assessment, conservation



## INTRODUCTION

Within the past 200 years, human landscape changes resulted in the destruction of over 99% of the original tallgrass prairie (Anderson 1990, Howe 1994). All that is left of North American tallgrass prairie are highly fragmented remnants (Steinauer and Collins 1996), which must be actively managed to be maintained (Howe 1994). Management practices that maintain native grassland communities include mowing or light grazing by ungulates and prescribed burning (Collins 1987, Howe 1994, Collins et al. 1998, Knapp et al. 1998). In eastern Kansas, native tallgrass prairie persists in two forms: as grazed pasture, which is often severely degraded, and as prairie hay meadows, which have been managed mainly for hay production since the time of settlement. These hay meadows are important reservoirs of tallgrass prairie species, and function as sources of forage (Towne and Ohlenbusch 1992, Jog et al. 2006, Foster et al. 2009). Privately-owned hay meadows also contain most of the known populations of the federally-threatened plant Mead's milkweed (*Asclepias meadii* Torr. ex A. Gray), and management of hay meadows is an integral component of this species' recovery plan (USFWS 2003). Hay meadow systems in Europe have received attention for their conservation value (Jefferson 2005), but there has been less research of native hay meadows in the tallgrass prairie region, possibly due to the lack of policy addressing conservation issues within agricultural landscapes in the United States (Foster et al. 2009).

Recommendations on managing native prairie hay meadows in eastern Kansas are provided to private landowners in publications from the Kansas State Agricultural Experiment Station and Cooperative Extension Service (Towne and Ohlenbusch 1992) and the Kansas Biological Survey (Kindscher and Buczynski 2009). The two

publications differ in perspective—the former provides recommendations to maximize financial profit of hay meadows, while the latter focuses on recommendations for conserving biodiversity and restoring degraded prairies. Both recommend prescribed burning of hay meadows, but the Kansas Biological Survey publication emphasizes the role of fire in suppressing unwanted non-native species, while the Kansas State publication emphasizes that fire can increase yield. The Kansas State publication is neutral on fertilizing, stating that fertilizers may provide a limited economic benefit but may also benefit unwanted non-native grasses, and on whether to have cattle graze a hay meadow, stating that winter grazing will not adversely affect hay yield. The Kansas Biological Survey publication explicitly recommends against fertilizing and grazing, as well as broadcast-spraying herbicides. These two publications on hay meadow management recommendations provide a contrast between agricultural and conservation paradigms.

The extent to which private landowners apply management practices that promote prairie conservation remains unknown. I surveyed private landowners with the goals of (1) obtaining an overview of management practices of privately-owned prairie hay meadows, and (2) determining to what extent the management of privately-owned prairie hay meadows aligns with the goals of conservation of tallgrass prairie species and communities.

Historically, North American prairie vegetation extended from Manitoba and Saskatchewan, Canada, south to Texas and Mexico, with a wide peninsula extending into Illinois, Indiana, and Ohio. The eastern portion of this range was dominated by tallgrass prairie prior to Euro-American settlement in this region. Since Euro-American settlement

of eastern Kansas following the Homestead Act of 1862, the landscape has been dominated by agriculture and introduced cool-season grasses (Foster et al. 2009). Existing hay meadows, which have never been plowed, have been managed mainly for hay production since the time of settlement. Shortridge (1973, p. 548) suggests that although many hay meadows could have been converted at anytime to more profitable uses, they persist due to “cultural inertia.”

## METHODS

Anderson and Linn are adjacent counties in east-central Kansas and are located within the Osage Cuestas physiographic province. They are traversed by the Marais des Cygnes River and several of its tributaries (Figure 2.1). Major underlying strata are limestone and shale, with sandstone deposits. The area is primarily agricultural, with crops of corn, soybeans, wheat, grain sorghum, and alfalfa making up 29% of the landscape, and additional land used as pastures of native warm-season grasses or cultivated cool-season grasses (Table 2.1). High-quality native prairie fragments are scattered throughout both counties, and most are used as hay meadows. When Euro-American settlement began in the mid-19<sup>th</sup> century, federal land surveyors estimated prairie to cover 94% of Anderson County and 81% of Linn County (Public Land Surveys of Kansas 1850s, Kansas State Board of Agriculture 1877). Current cover of high-quality native tallgrass prairie in the two counties is estimated at less than 1% (Kindscher et al. 2009).

In 2008 and 2009, the Kansas Natural Heritage Inventory (KNHI) of the Kansas Biological Survey identified and surveyed high-quality natural areas of tallgrass prairie

and oak-hickory forest in Anderson and Linn counties. High-quality natural areas support plant communities that closely approximate the vegetation that existed prior to Euro-American settlement. Field crews identified sites, by driving on county roads and examining satellite imagery, obtained permission of landowners, and then visited sites to assess condition and compile a list of all plant species present (nomenclature was from the USDA, NRCS 2010). Methodology of site identification and censusing are described by Kindscher et al. (2009). Of the 236 hay meadows surveyed, 6 were owned by the U.S. Fish and Wildlife Service or the Kansas Department of Wildlife and Parks; the remaining 230 were privately owned. In the autumn of each year, each landowner who had participated in the natural areas survey was mailed a thank-you letter, the plant species list, and a brochure on native prairie hay meadow management.

In 2009, a management questionnaire, with a stamped envelope for return, was included in the mailing to each landowner. A total of 103 questionnaires were sent; 2 were returned by the U.S. Postal Service as undeliverable, and 48 were completed and returned. The management questionnaire included questions on the history and recent management practices of the hay meadow. A subset of questions were concerned with grazing practices. Space was provided for respondents to provide additional comments or information on the hay meadow.

All hay meadows owned by the respondents are shown in Figure 2.1. During the KNHI survey, some adjacent hay meadows owned by different landowners were surveyed as a single site, and only one species list was compiled. Three of the respondents owned portions of multi-owner sites, and so were excluded from analyses relating management practices to species richness or floristic quality of a site. Seven

landowners owned 2 or 3 hay meadows that had been surveyed. In these cases, to ensure independence of samples, one site per landowner was selected at random to be included in the analysis.

A floristic quality assessment was conducted to assess how well each hay meadow served as a refuge for highly conservative species, i.e., species which are restricted to natural, undisturbed habitats. Floristic quality assessment is a quantitative method which uses assigned coefficients of conservatism ranging from 0 to 10 to indicate the relative fidelity of species' occurrence in habitat largely unaltered from pre-settlement times. Plants with high coefficients of conservatism tend to be habitat specialists, which are more sensitive to habitat disturbances than plants with low coefficients of conservatism, which tend to be generalists that are more tolerant of a variety of conditions. Coefficients of conservatism were assigned to Kansas plant species by Freeman and Morse (2002). For each hay meadow, a floristic quality index (FQI) was calculated according to the formula from Swink and Wilhelm (1994):  $FQI = R/\sqrt{N}$ , where R equals the sum of coefficients of conservatism for all plants recorded at the site and N equals the total number of native species recorded. Non-native species are not assigned a coefficient of conservatism, and so are not considered in the equation. One-way ANOVA and *t*-tests were conducted in PASW 17.0 (2009). Due to the nature of questionnaire data, a *p*-value of 0.10 was used as the threshold for statistical significance. When applicable, the Kolmogorov-Smirnov test for normality and Levene's test for equal variance were conducted to evaluate whether statistical assumptions were met.

## RESULTS

Total species on a site ranged from 34 to 141, with a mean of 82. Altogether, 255 native and 35 non-native plant species were identified. Mead's milkweed was located on 17 sites. Respondents had owned their hay meadows an average of 36.4 years, with a range of 2 to over 100 years (due to some respondents reporting total ownership time within a family, with ownership being passed from parent to child). Hay production ranged from 0.75 to 2.5 tons per acre, with a mean of 1.57. Hay was usually cut between July 4<sup>th</sup> and July 31<sup>st</sup>, although the time of cutting ranged from June to after September 1<sup>st</sup>.

Seven respondents used their hay meadows for cattle grazing. Six of the 7 who grazed cattle answered the subset of questions dealing with grazing practices; the seventh landowner had limited grazing in the past and did not fill out this section. Four of 6 respondents allowed cattle to graze on their hay meadow once every 2-3 years. One allowed cattle to graze every year, while one allowed cattle to graze every 4 years. Time of grazing was variable, occurring in the summer (2 respondents), winter (2 respondents), spring and summer (1 respondent), or spring, summer, and fall (1 respondent). Duration of grazing ranged from 1 to 6 months.

Grazing did not have a significant effect on species richness ( $t=1.42$ ,  $df=43$ ,  $p=0.16$ ), but it did have significant effect on floristic quality ( $t=1.88$ ,  $df=43$ ,  $p = 0.07$ ). Grazed prairies had a lower mean FQI ( $\bar{x} = 33.1$ ) than non-grazed prairies ( $\bar{x} = 38.2$ ; Figure 2.2).

Twenty-eight of 38 respondents who did not graze their hay meadows said that the sites had fences or evidence of old fences, which are presumed to be indicators of past

grazing, because a fence would only have been erected to contain livestock. Only 10 hay meadows had never been fenced. Presence of a fence or evidence of an old fence did not have an effect on species richness ( $F(3,41)=1.86, p=0.15$ ) or floristic quality ( $F(3,41)=2.03, p=0.13$ ).

Thirty-four of 48 respondents had burned their prairie at some point. Frequency of burning had no significant effect on species richness ( $F(1,4)=2.05, p=0.11$ ). However, for floristic quality, burning frequency was significant ( $F(4,40) = 2.20, p = 0.09$ ). Hay meadows that were burned every 1-2 years or had never been burned had higher FQIs than hay meadows that were burned every 3 or more years (Figure 2.3).

No landowner rested all or part of his prairie (by not actively managing it or cutting hay in a year) more often than every 5 years. Ten of 48 respondents had rested their prairie once every 5-7 (or more) years. Resting did not have a significant effect on species richness ( $t=1.49, df=43, p=0.73$ ) or FQI ( $t=2.08, df=43, p=0.11$ ).

More than half (27 of 45) of respondents had, at some time, spot-sprayed herbicide to eradicate invasive species such as sericea lespedeza (*Lespedeza cuneata* (Dum. Cours.) G. Don) or musk thistle (*Carduus nutans* L.). Spot-spraying did not have a significant effect on species richness ( $F(4,40)= 0.90, p=0.47$ ) or floristic quality ( $F(4,40)= 0.73, p=0.58$ ) of a site.

Thirteen of 45 respondents used broadcast- or aerial-sprayed herbicide on their hay meadows. There was no significant effect of broadcast/aerial-sprayed herbicide on species richness ( $F(3,41)= 0.50, p=0.68$ ) or floristic quality ( $F(3,41)= 0.62, p=0.61$ ). The 13 respondents who broadcast- or aerial-sprayed herbicide most commonly did so to get rid of weeds (unwanted forb species; 11 respondents), followed by the goal of improving

hay quality (by increasing the proportion of grass to forbs; 7 respondents). Other reasons given for the use of herbicide included the control of sericea lespedeza and musk thistle (4 respondents), and to improve yield (2 respondents).

## DISCUSSION

Of the management practices assessed in the questionnaire, none had a significant effect on species richness. Grazing and burning frequency had the largest effects on the floristic quality of sites. Because grazing and burning frequency affected floristic quality but not species richness, these practices may shift species composition rather than changing the overall number of species present. Collins (1987) showed experimentally that grazing by herbivores may increase species diversity by decreasing the abundance of dominant grasses, thereby releasing forbs from competitive pressure. However, the negative impact of grazing on floristic quality (Figure 2.2) indicates that grazing may also shift forb composition away from more conservative, habitat specialists and toward weedier generalists.

The role of fire in maintenance of tallgrass prairie communities is complex and can vary with other types of disturbances, edaphic conditions, season, and climate. Experimental studies of various burning regimes on both grazed and ungrazed areas of the Konza Prairie Research National Area in Kansas found that annual burning of tallgrass prairie decreased species richness, but burning every 2 or 4 years resulted in an increase in richness the year after a fire (Collins and Gibson 1996), and that species diversity increased for 6 to 7 years after burning, and then declined (Gibson and Hulbert 1987). Gibson et al. (1993) found an effect of burning on vegetation community of



formerly-grazed areas of Konza, but only after soil type had been accounted for. They also found that no species responded to burning alone, although some responded to interactions among burning, mowing, and fertilizing. However, experimental studies have produced “frustratingly few” generalities of the effects of fire on community composition (Collins and Gibson 1996, p. 97). Here, hay meadows that were frequently burned or never burned had a higher FQI than hay meadows that were less frequently burned (Figure 2.3), which is difficult to interpret in light of the complexity of the effects of fire on community composition and the lack of a clear understanding from experimental studies.

Burning and resting have been promoted as means of conserving prairie habitat and plant species (Howe 1994, Collins and Gibson 1996, Tunnell 2004, Jog et al. 2006, Kindscher and Byzcinski 2009). Private landowners practice a wide variety of burning regimes, but they rarely rest portions of their hay meadows, with only a quarter of respondents stating their hay meadows had been rested, and never more frequently than every 5 years. One comment provided by a landowner might shed light on why burning is often implemented but resting is not: “I am very conscious of prairie hay meadows used for hay and/or pasture. It is hard to pay taxes without using the acres for something.” Burning a hay meadow when the grasses are dormant does not prevent a landowner from cutting hay later in the year, but resting requires that a landowner forego some portion of the profit that he would otherwise get from the hay meadow. Therefore, very few high-quality native hay meadows are rested. The lack of a significant effect of resting on species richness or FQI may be due to the fact that no hay meadow surveyed was rested more often than once every 5 years. Occasional resting, or rotational haying, has been

promoted as a means of providing habitat for butterflies and ground-nesting birds, and allowing late-season plants to complete seed development (Kindscher and Byczynski 2009). However, it is unknown whether regular resting would have long-term effects on species richness or floristic quality of a hay meadow.

KNHI field crews noted a wide variation in the attitudes of landowners toward their hay meadows, from owners who considered them fields of weeds to owners who appreciated the native wildflowers and wildlife habitat that their hay meadows provided. The variation in the use of herbicides reflects the variation in attitudes. Some landowners do not attempt to control invasive species, while some spot-spray herbicide, impacting only the problem species with little or no effect on neighboring, desirable species. Some landowners broadcast-spray herbicide over the entire hay meadow to eradicate invasive species, although others noted in the space allotted for additional comments that they would rather dig out each invasive plant with a hoe than use herbicide. Some users of herbicide aim to get rid of unwanted forbs whether they are native or not, in an effort to improve hay quality. Landowner opinions vary widely on whether forb content increases or decreases the quality of hay, but some consider anything but grasses to be a “weed” that deteriorates hay quality.

The management practices described here pertain only to high-quality native prairie hay meadows. Lower-quality native hay meadows, which contain fewer conservative species and more non-native invasive species, were excluded from the KNHI survey. Management practices that result in degradation of a high meadow include overgrazing, the use of fertilizers, or overuse of broadcast-sprayed herbicides. The low number of respondents in this study who grazed their hay meadows or

broadcast-sprayed herbicide is not a reflection of the total extent of the use of these practices by landowners, but a reflection of the exclusion of lower-quality or degraded sites from the KNHI survey.

Although some landowners show high interest in conserving native plant species and communities, not all management practices that promote conservation are implemented in native prairie hay meadows. Although cultural inertia deters landowners from converting hay meadows to more profitable uses, most landowners appear to manage for maximum yield and profit of hay, with minimum input costs, rather than for conservation interests.

In the United States, biodiversity issues are rarely addressed in agricultural landscapes (Foster et al. 2009). One of the first steps in incorporating biodiversity policy into an agricultural landscape is to place a monetary value on the conservation assets within the landscape (Heal 2000, Santelmann et al. 2004, Crow et al. 2006, Rouquette et al. 2009). In Eastern Kansas, conserving prairie hay meadows will have the greatest impact on preserving native species diversity (Jog et al 2006), but private landowners most often consider the economic value of the hay they harvest from them rather than considering any ‘value’ of their biodiversity or ecosystem services. The economic value of native prairie hay is low compared to other assets managed by private farmers, and cultural inertia, which certainly can not be expected to continue in perpetuity, is the only thing maintaining these sites as native hay meadows. If the preservation of native prairie hay meadows is a goal of conservationists, than private owners must become aware of the value of conservation, and this value must be incorporated into a landscape-level perspective that takes into account a diversity of value systems and sources of benefits.

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## FIGURES

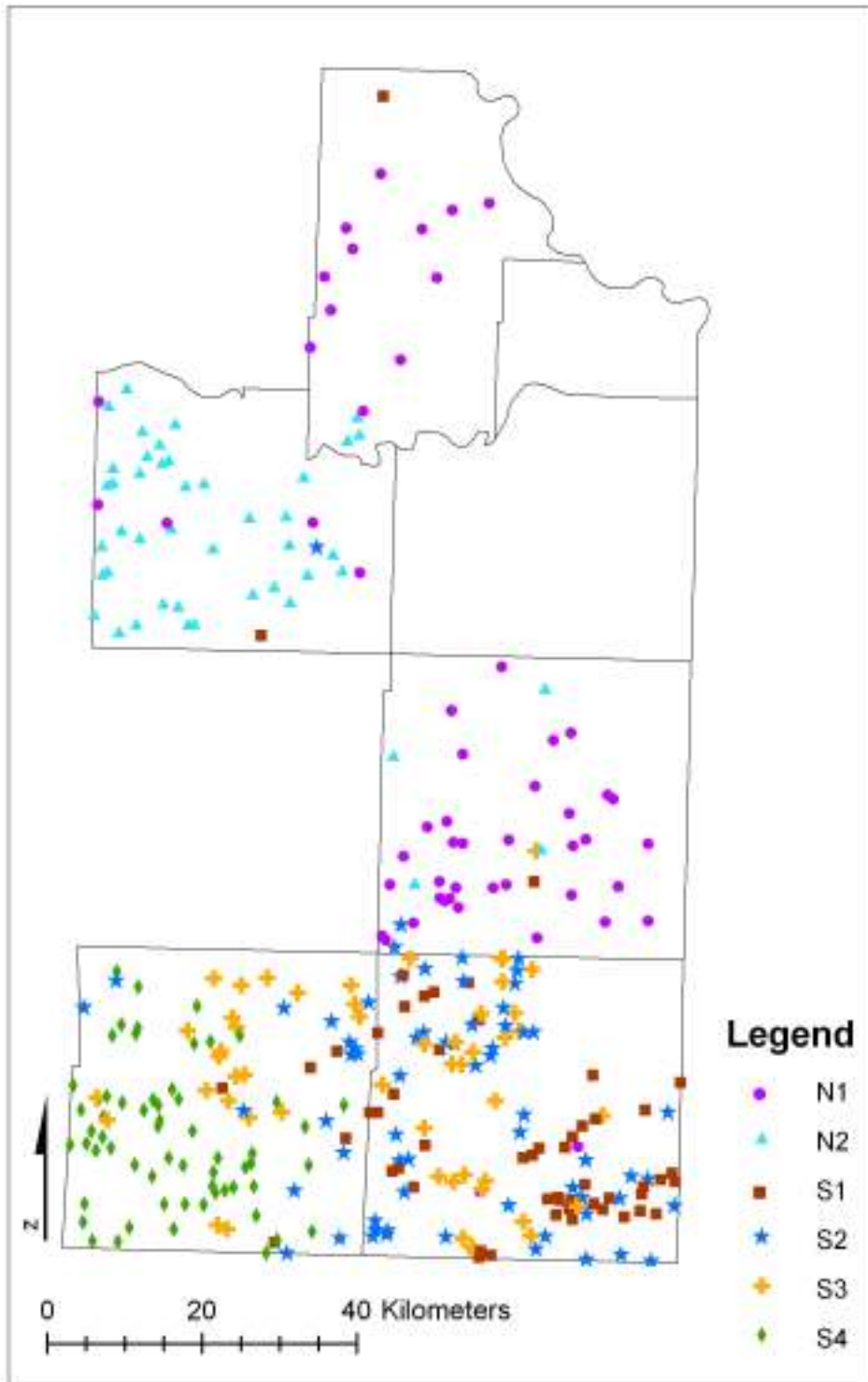


Figure 1.1. Native prairie hay meadows grouped by cluster analysis of species presence, in Leavenworth, Wyandotte, Douglas, Johnson, Miami, Anderson, and Linn counties, KS.

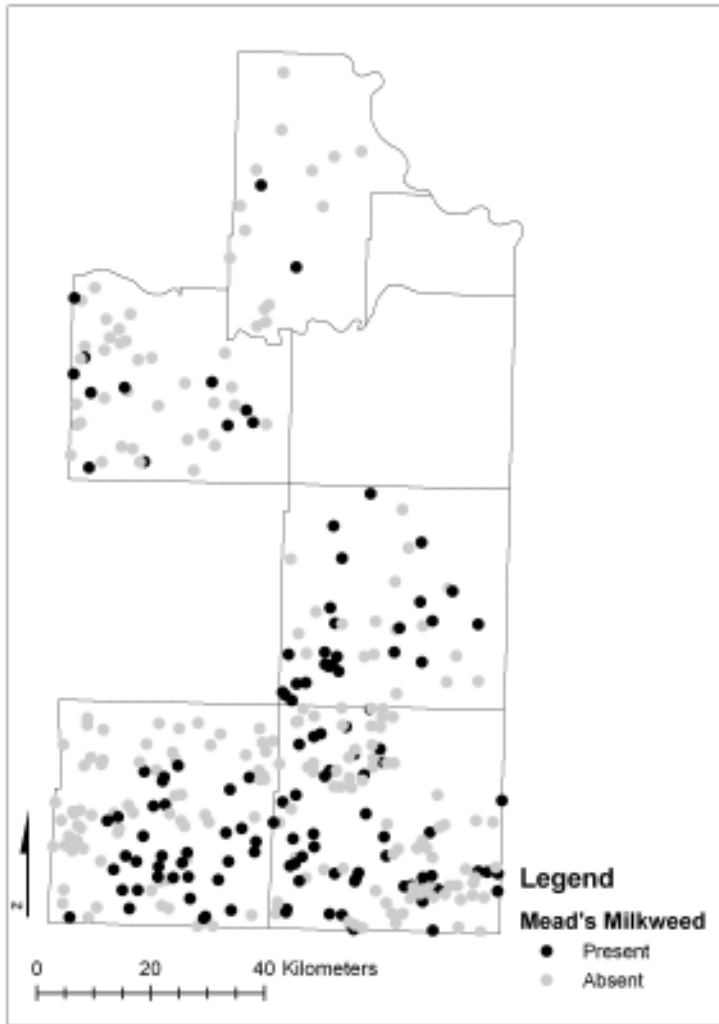


Figure 1.2. Location of Mead's milkweed (*Asclepias meadii*) populations within Leavenworth, Wyandotte, Douglas, Johnson, Miami, Anderson, and Linn counties, KS.

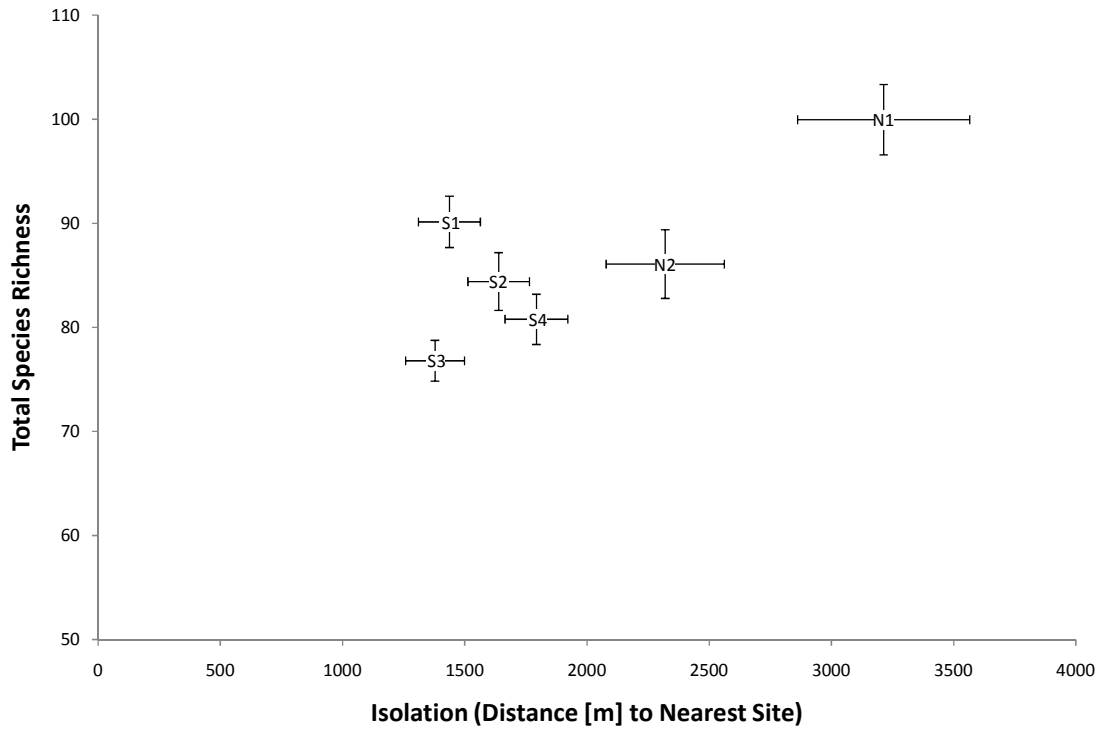


Figure 1.3. Mean isolation (distance to nearest site) and species richness of sites grouped by cluster analysis. Bars display standard error.

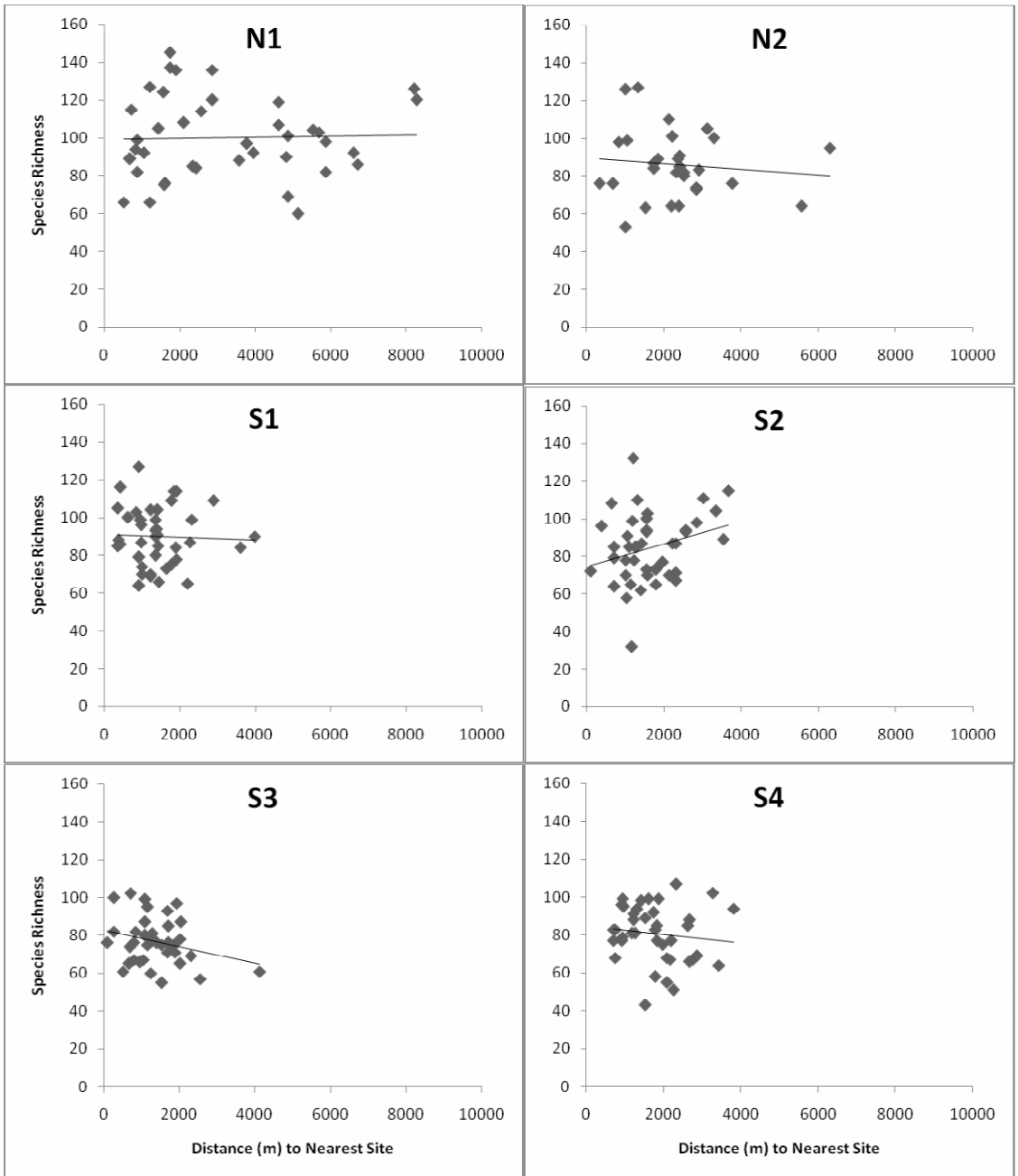


Figure 1.4. Isolation and species richness of each cluster. Note that most clusters have a neutral or negative relationship, and Cluster N1 has sites with higher values of both isolation and species richness than the other clusters.

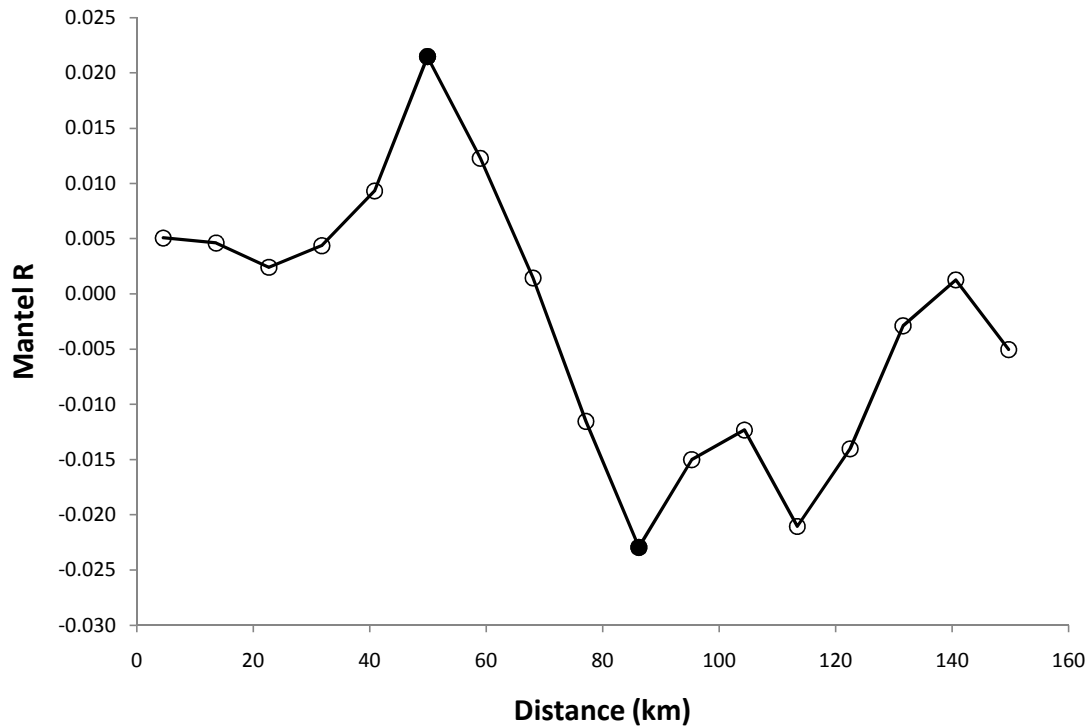


Figure 1.5. Mantel correlogram of Jaccard’s dissimilarity and geographic distance. Solid circles indicate significance of  $p < 0.05$ . There is positive spatial autocorrelation between community composition of sites in distance classes of approximately 70 km; in greater distance classes there is negative spatial autocorrelation, indicating either a vegetation gradient or step-like structure.



Figure 1.6. Sites with comparable areas but differing perimeter lengths. The site on the left has greater environmental heterogeneity, which leads to greater perimeter length when site is digitized.





Figure 2.1. Map of all prairie hay meadows owned by questionnaire respondents within Anderson and Linn counties, KS.

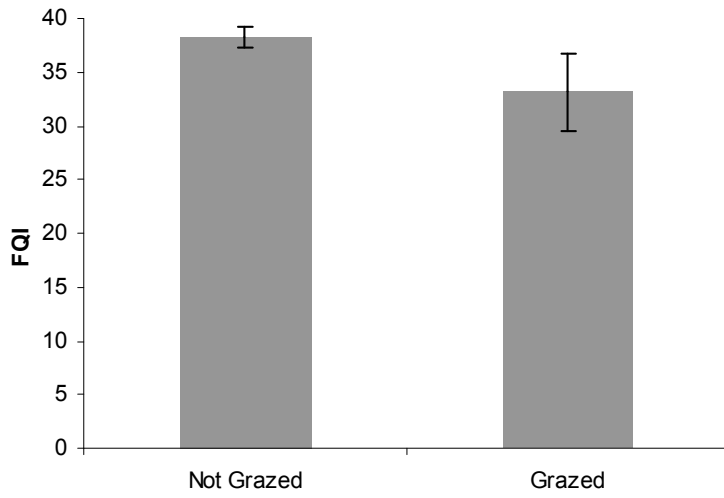


Figure 2.2. Floristic quality index of grazed (n=7) and non-grazed (n=38) prairies (*t*-test,  $p = 0.07$ ). Standard error is shown.

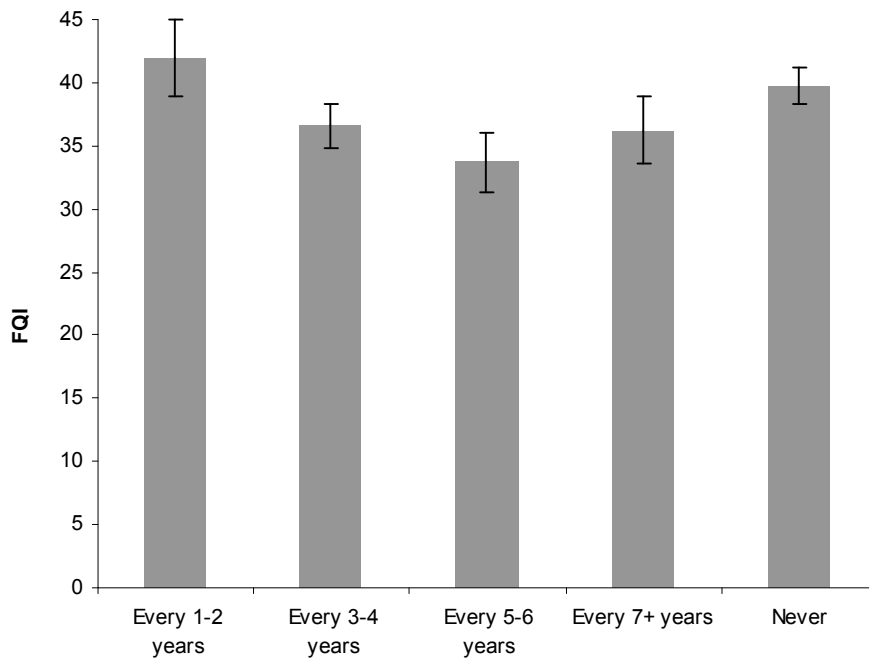


Figure 2.3. Floristic quality index of prairies by burn frequency (ANOVA  $p = 0.09$ ). Standard error is shown.

## TABLES

Table 1.1 Land Use – Land Cover of the five-county study area. Note that ‘grassland’ includes both native and cultivated non-native pastures and hay meadows. Data from Kansas Applied Remote Sensing Program 2005.

Land Use	Percent Cover
Grassland	49.4%
Cropland	23.1%
Woodland	19.4%
Urban	3.0%
CRP	2.5%
Water	2.1%
Other	0.1%

Table 1.2. Results of ANOVAs by clusters.

	F	DF (Between)	DF (Within)	p
<b>Area</b>	6.083	5	295	0.000
<b>Distance to Nearest Site</b>	12.638	5	223	0.000
<b>FQI</b>	8.478	5	295	0.000
<b>% Native Upland Species</b>	19.300	5	295	0.000
<b>Perimeter</b>	6.266	5	295	0.000
<b>Total Species</b>	13.553	5	295	0.000
<b>% High CoC Species</b>	4.246	5	295	0.001
<b>% Low CoC Species</b>	4.004	5	295	0.002
<b>Perimeter/Area Ratio</b>	4.001	5	295	0.002
<b>% Native Species</b>	2.308	5	295	0.044
<b>% Grassland</b>	1.946	5	295	0.087
<b>% CRP</b>	1.782	5	295	0.116
<b>% Woodland</b>	1.736	5	295	0.126
<b>% Water</b>	1.246	5	295	0.288
<b>Depth to Bedrock</b>	0.715	5	295	0.613
<b>% Cropland</b>	0.675	5	295	0.643
<b>Soil Available Water</b>	0.269	5	295	0.930

Table 1.3. Average properties of sites by cluster. Clusters with different superscripts differ significantly from one another (see ANOVA results, Table 1.2).

Cluster	No. of Sites	Average FQI	Average Total Species			% Native Upland Species			% Low CoC Species			% High CoC Species			Average Perimeter (m)	Average Area (ha)	Average Perimeter/Area Ratio	Distance to Nearest Site (m)*
			% Native Species	% Native Species	% Native Species	% Native Species	% Native Species	% Native Species	% Native Species	% Native Species	% Native Species	% Native Species	% Native Species	% Native Species				
N1	48	41.90 <sup>a</sup>	87.5%	30.0% <sup>a</sup>	19.8% <sup>a</sup>	19.3% <sup>a</sup>	69.6 <sup>a</sup>	1401 <sup>a</sup>	0.024 <sup>ab</sup>	3214 <sup>a</sup>								
N2	42	36.22 <sup>b</sup>	88.3%	31.0% <sup>a</sup>	22.8% <sup>b</sup>	17.1% <sup>ab</sup>	61.3 <sup>a</sup>	1240 <sup>a</sup>	0.026 <sup>a</sup>	2320 <sup>b</sup>								
S1	51	38.65 <sup>b</sup>	88.2%	26.3% <sup>b</sup>	20.5% <sup>ab</sup>	16.8% <sup>b</sup>	87.5 <sup>a</sup>	1552 <sup>a</sup>	0.025 <sup>ab</sup>	1437 <sup>c</sup>								
S2	61	36.73 <sup>b</sup>	87.6%	26.6% <sup>b</sup>	20.5% <sup>ab</sup>	15.5% <sup>b</sup>	108.2 <sup>ab</sup>	1605 <sup>a</sup>	0.024 <sup>ab</sup>	1639 <sup>bc</sup>								
S3	49	36.57 <sup>b</sup>	89.6%	34.6% <sup>c</sup>	18.9% <sup>a</sup>	17.1% <sup>ab</sup>	67.0 <sup>a</sup>	1343 <sup>a</sup>	0.028 <sup>a</sup>	1378 <sup>c</sup>								
S4	50	38.07 <sup>b</sup>	89.3%	26.1% <sup>b</sup>	19.4% <sup>a</sup>	17.5% <sup>ab</sup>	156.3 <sup>b</sup>	2142 <sup>b</sup>	0.019 <sup>b</sup>	1793 <sup>bc</sup>								

\*Sites within 5 km of the edge of the study area were excluded from this analysis.

Table 1.3. Average properties of landscape matrix and soil characteristics of sites by cluster.

Cluster	No. of Sites	% Urban Landcover		% Cropland		% Grassland		% CRP		% Woodland		% Water		Depth to Bedrock		Soil Available Water	
		% Urban Landcover	% Urban Landcover	% Cropland	% Cropland	% Grassland	% Grassland	% CRP	% CRP	% Woodland	% Woodland	% Water	% Water	Depth to Bedrock	Depth to Bedrock	Soil Available Water	Soil Available Water
N1	48	0.7%	0.7%	21%	21%	59%	59%	4%	4%	15%	15%	1.3%	1.3%	175.54	175.54	33.73	33.73
N2	42	1.1%	1.1%	25%	25%	59%	59%	3%	3%	12%	12%	0.6%	0.6%	170.32	170.32	26.18	26.18
S1	51	0.1%	0.1%	17%	17%	54%*	54%*	6%*	6%*	20%	20%	2.2%	2.2%	169.51	169.51	31.02	31.02
S2	61	0.7%	0.7%	18%	18%	63%*	63%*	4%	4%	14%	14%	0.6%	0.6%	163.22	163.22	33.84	33.84
S3	49	0.2%	0.2%	20%	20%	57%	57%	4%	4%	17%	17%	0.9%	0.9%	153.83	153.83	31.50	31.50
S4	50	0.0%	0.0%	14%	14%	61%	61%	2%*	2%*	21%	21%	1.2%	1.2%	159.40	159.40	33.66	33.66

\* Clusters were significantly different at the p<0.10 level.

Table 1.5. Significant predictors of FQI as determined by stepwise linear regression.

Cluster	Significant Predictors	Beta	p	Model R square	Model p (ANOVA)
Entire Study Area	Perimeter	0.248	0.000	0.081	0.000
	Distance to Nearest Site	0.148	0.020		
Cluster N1*	Perimeter	<i>0.296</i>	<i>0.064</i>	<i>0.088</i>	<i>0.064</i>
Cluster N2	Depth to Bedrock	0.471	0.010	0.222	0.010
Cluster S1	Perimeter	0.638	0.000	0.409	0.000
	% CRP within 0.5 km	-0.286	0.034		
Cluster S2	Perimeter	0.379	0.012	0.144	0.012
Cluster S3	Area	-0.674	0.000	0.358	0.002
	Perimeter/Area Ratio	-0.434	0.015		
	% Water within 0.5 km	-0.309	0.034		
Cluster S4*	<i>NS</i>	<i>NS</i>	<i>NS</i>	<i>NS</i>	<i>NS</i>

\*After the model with p=0.05 required for entry yielded no significance, model was adjusted to allow p=0.10 for entry. Italicized results were not significant at p=0.05.

Table 1.6. Significant predictors of total species richness as determined by stepwise linear regression.

Cluster	Significant Predictors	Beta	p	Model R square	Model p (ANOVA)
Entire Study Area	Perimeter	0.274	0.000	0.103	0.000
	Distance to Nearest Site	0.157	0.013		
	% Cropland within 0.5 km	0.142	0.023		
Cluster N1	Perimeter	0.407	0.009	0.166	0.009
Cluster N2*	Perimeter	0.445	0.010	0.409	0.011
	Depth to Bedrock	0.497	0.008		
	Soil Available Water	<i>0.328</i>	<i>0.053</i>		
	Distance to Nearest 3 Sites	<i>-0.305</i>	<i>0.079</i>		
Cluster S1	Perimeter	0.580	0.000	0.337	0.000
Cluster S2	Perimeter	0.363	0.011	0.259	0.005
	% Cropland within 0.5 km	0.336	0.018		
Cluster S3*	<i>NS</i>	<i>NS</i>	<i>NS</i>	<i>NS</i>	<i>NS</i>
Cluster S4*	% CRP within 0.5 km	<i>-0.281</i>	<i>0.083</i>	<i>0.146</i>	<i>0.083</i>

\*After the model with p=0.05 required for entry yielded no significance, model was adjusted to allow p=0.10 for entry. Italicized results were not significant at p=0.05.

Table 1.7. Significant predictors of native species richness as determined by stepwise linear regression.

Cluster	Significant Predictors	Beta	p	Model R square	Model p (ANOVA)
Entire Study Area	Perimeter	0.264	0.000	0.078	0.000
	% Cropland within 0.5 km	0.133	0.036		
Cluster N1	Perimeter	0.355	0.024	0.126	0.024
Cluster N2	% Grassland within 0.5 km	-0.377	0.044	0.142	0.044
Cluster S1	Perimeter	0.603	0.000	0.455	0.000
	% CRP within 0.5 km	-0.293	0.026		
	Distance to Nearest 3 Sites	-0.262	0.041		
Cluster S2	% Cropland within 0.5 km	0.341	0.018	0.231	0.005
	Perimeter	0.318	0.027		
Cluster S3*	<i>NS</i>	<i>NS</i>	<i>NS</i>	<i>NS</i>	<i>NS</i>
Cluster S4*	Distance to Nearest 3 Sites	<i>-0.296</i>	<i>0.068</i>	<i>0.087</i>	<i>0.068</i>

\*After the model with p=0.05 required for entry yielded no significance, model was adjusted to allow p=0.10 for entry. Italicized results were not significant at p=0.05.

Table 1.8. Significant predictors of native upland species richness as determined by stepwise linear regression.

Cluster	Significant Predictors	Beta	p	Model R square	Model p (ANOVA)
Entire Study Area	% Cropland within 0.5 km	0.190	0.003	0.107	0.000
	Perimeter	0.244	0.000		
	Perimeter/Area Ratio	0.173	0.011		
	Distance to Nearest Site	0.134	0.034		
Cluster N1	Perimeter	0.404	0.010	0.163	0.010
Cluster N2	% Grassland within 0.5 km	-0.447	0.015	0.200	0.015
Cluster S1	Perimeter	0.579	0.000	0.345	0.000
	% CRP within 0.5 km	-0.287	0.042		
Cluster S2	% Cropland within 0.5 km	0.331	0.012	0.409	0.000
	Perimeter	0.418	0.003		
	% Water within 0.5 km	-0.344	0.013		
	% CRP within 0.5 km	-0.298	0.023		
Cluster S3*	Depth to Bedrock	0.359	0.021	0.270	0.013
	% Grassland within 0.5 km	0.329	0.033		
	Perimeter/Area Ratio	<i>0.290</i>	<i>0.057</i>		
Cluster S4*	% CRP within 0.5 km	<i>-0.304</i>	<i>0.060</i>	<i>0.093</i>	<i>0.060</i>

\*After the model with p=0.05 required for entry yielded no significance, model was adjusted to allow p=0.10 for entry. Italicized results were not significant at p=0.05.

Table 1.9. Significant predictors of species with high CoC values (7-10) as determined by stepwise linear regression.

Cluster	Significant Predictors	Beta	p	Model R square	Model p (ANOVA)
Entire Study Area	Distance to Nearest Site	0.235	0.000	0.077	0.000
	Perimeter	0.181	0.005		
Cluster N1	Distance to Nearest 3 Sites	0.370	0.019	0.137	0.019
Cluster N2	Depth to Bedrock	0.634	0.001	0.385	0.002
	% Woodland within 0.5 km	0.347	0.042		
Cluster S1	Perimeter	0.548	0.000	0.301	0.000
Cluster S2	Perimeter	0.360	0.018	0.129	0.018
Cluster S3	Distance to Nearest Site	-0.528	0.001	0.438	0.001
	% Grassland within 0.5 km	0.439	0.004		
	Perimeter	-0.413	0.008		
	Perimeter/Area Ratio	-0.300	0.050		
Cluster S4*	<i>NS</i>	<i>NS</i>	<i>NS</i>	<i>NS</i>	<i>NS</i>

\*After the model with p=0.05 required for entry yielded no significance, model was adjusted to allow p=0.10 for entry. Italicized results were not significant at p=0.05.

Table 1.10. Significant predictors of species with low CoC values (0-2) as determined by stepwise linear regression.

Cluster	Significant Predictors	Beta	p	Model R square	Model p (ANOVA)
Entire Study Area*	Perimeter	0.337	0.001	0.047	0.003
	Area	-0.271	0.008		
	% Cropland within 0.5 km	<i>0.107</i>	<i>0.098</i>		
Cluster N1*	<i>NS</i>	<i>NS</i>	<i>NS</i>	<i>NS</i>	<i>NS</i>
Cluster N2	Distance to Nearest 3 Sites	-0.499	0.002	0.511	0.000
	% Grassland within 0.5 km	-0.488	0.004		
	Soil Available Water	0.415	0.007		
Cluster S1	% Cropland within 0.5 km	-0.394	0.008	0.284	0.002
	Depth to Bedrock	-0.390	0.008		
Cluster S2*	% Cropland within 0.5 km	<i>0.281</i>	<i>0.068</i>	<i>0.079</i>	<i>0.068</i>
Cluster S3	Distance to Nearest 3 Sites	0.323	0.048	0.105	0.048
Cluster S4*	% CRP within 0.5 km	<i>-0.300</i>	<i>0.063</i>	<i>0.090</i>	<i>0.063</i>

\*After the model with p=0.05 required for entry yielded no significance, model was adjusted to allow p=0.10 for entry. Italicized results were not significant at p=0.05.

Table 1.11. Correlation of isolation (distance to nearest site or distance to nearest three sites) with measures of richness and quality, as determined by stepwise linear regressions. See Tables 1-6 for statistics.

	FQI	Total Species	Native Species	Native Upland Species	Conservative Species	Weedy Species
Entire Study Area	+	+		+	+	
Cluster N1					+	
Cluster N2		-				-
Cluster S1			-			
Cluster S2						
Cluster S3					-	+
Cluster S4			-			

Table 2.1 Land Use – Land Cover of Anderson and Linn Counties. Note that ‘grassland’ includes both native and cultivated non-native pastures and hay meadows. Data from Kansas Applied Remote Sensing Program 2005.

Land Use	Percent Cover
Grassland	47.5%
Cropland	29.3%
Woodland	17.1%
Urban	1.5%
CRP	3.3%
Water	1.3%
Other	0.1%



### Appendix 1: Representative Species of Clusters

Table A1. Most abundant, overrepresented, and underrepresented species within Cluster N1, and species found only within Cluster N1.

Most Abundant Species	CoC	% Sites Occupied	Overrepresented Species	CoC	% Sites Occupied	% Sites Occupied (All Sites)
<i>Andropogon gerardii</i>	4	100%	<i>Comandra umbellata</i>	6	85%	37%
<i>Euphorbia corollata</i>	5	100%	<i>Lithospermum canescens</i>	7	83%	36%
<i>Rudbeckia hirta</i>	2	100%	<i>Coreopsis palmata</i>	7	85%	43%
<i>Schizachyrium scoparium</i>	5	100%	<i>Hesperostipa spartea</i>	8	68%	27%
<i>Amorpha canescens</i>	7	98%	<i>Phlox pilosa</i>	7	78%	38%
<i>Baptisia bracteata</i>	6	98%	<i>Zizia aurea</i>	5	75%	37%
<i>Erigeron strigosus</i>	4	98%	<i>Linum sulcatum</i>	6	70%	33%
<i>Tripsacum dactyloides</i>	3	98%	<i>Gaura longiflora</i>	2	45%	10%
<i>Oligoneuron rigidum</i>	3	95%	<i>Bromus inermis</i>	*	63%	29%
<i>Salvia azurea</i>	4	95%	<i>Pedicularis canadensis</i>	7	60%	31%
<i>Silphium laciniatum</i>	4	95%				
<i>Solidago canadensis</i>	2	95%				
<i>Tradescantia ohioensis</i>	5	95%				
Species Exclusive to Cluster	CoC	% Sites Occupied	Underrepresented Species	CoC	% Sites Occupied	% Sites Occupied (All Sites)
<i>Galium virgatum</i>	5	10%	<i>Callirhoe alcaeoides</i>	6	10%	54%
<i>Geum canadense</i>	1	8%	<i>Baptisia australis</i>	6	15%	53%
<i>Heuchera richardsonii</i>	7	8%	<i>Lespedeza virginica</i>	5	5%	36%
<i>Triosteum perfoliatum</i>	4	8%	<i>Ambrosia psilostachya</i>	3	3%	29%
<i>Ruellia strepens</i>	4	5%	<i>Asclepias sullivantii</i>	5	13%	36%
<i>Veronica arvensis</i>	*	5%	<i>Coreopsis grandiflora</i>	8	0%	23%
			<i>Polytaenia nuttallii</i>	6	60%	82%
			<i>Lepidium densiflorum</i>	0	3%	23%
			<i>Liatris punctata</i>	5	3%	20%
			<i>Carex bushii</i>	4	55%	72%

Table A2. Most abundant, overrepresented, and underrepresented species within Cluster N2, and species found only within Cluster N2.

Most Abundant Species	CoC	% Sites Occupied	Overrepresented Species	CoC	% Sites Occupied	% Sites Occupied (All Sites)
<i>Andropogon gerardii</i>	4	100%	<i>Dalea multiflora</i>	7	55%	8%
<i>Schizachyrium scoparium</i>	5	100%	<i>Comandra umbellata</i>	6	83%	37%
<i>Achillea millefolium</i>	1	97%	<i>Hesperostipa spartea</i>	8	66%	27%
<i>Amorpha canescens</i>	7	97%	<i>Bromus inermis</i>	*	66%	29%
<i>Dichanthelium oligosanthes</i>	4	97%	<i>Physalis pumila</i>	4	83%	48%
<i>Tripsacum dactyloides</i>	3	97%	<i>Oenothera speciosa</i>	2	59%	24%
<i>Asclepias viridis</i>	1	93%	<i>Medicago lupulina</i>	*	52%	48%
<i>Ruellia humilis</i>	3	93%	<i>Tragopogon dubius</i>	*	86%	24%
<i>Salvia azurea</i>	4	93%	<i>Erigeron annuus</i>	0	62%	17%
<i>Apocynum cannabinum</i>	0	90%	<i>Phlox pilosa</i>	7	69%	53%
<i>Dalea candida</i>	7	90%				
<i>Desmodium illinoense</i>	5	90%				
<i>Euphorbia corollata</i>	5	90%				
<i>Rudbeckia hirta</i>	2	90%				
<i>Solidago canadensis</i>	2	90%				

Species Exclusive to Cluster	CoC	% Sites Occupied	Underrepresented Species	CoC	% Sites Occupied	% Sites Occupied (All Sites)
<i>Stenaria nigricans</i>	5	7%	<i>Penstemon tubiflorus</i>	3	21%	80%
			<i>Allium canadense</i>	2	3%	57%
			<i>Carex bushii</i>	4	21%	72%
			<i>Callirhoe alcaeoides</i>	6	7%	54%
			<i>Baptisia australis</i>	6	7%	53%
			<i>Helianthus mollis</i>	7	28%	69%
			<i>Trifolium campestre</i>	*	3%	44%
			<i>Antennaria neglecta</i>	2	34%	68%
			<i>Lespedeza virginica</i>	5	3%	36%
			<i>Tradescantia ohioensis</i>	5	66%	92%

Table A3. Most abundant, overrepresented, and underrepresented species within Cluster S1, and species found only within Cluster S1.

<b>Most Abundant Species</b>	<b>CoC</b>	<b>% Sites Occupied</b>	<b>Overrepresented Species</b>	<b>CoC</b>	<b>% Sites Occupied</b>	<b>% Sites Occupied (All Sites)</b>
<i>Andropogon gerardii</i>	4	100%	<i>Valerianella radiata</i>	2	85%	40%
<i>Schizachyrium scoparium</i>	5	100%	<i>Myosotis verna</i>	2	63%	24%
<i>Amorpha canescens</i>	7	98%	<i>Carex brevior</i>	5	65%	29%
<i>Penstemon tubiflorus</i>	3	98%	<i>Desmodium paniculatum</i>	4	43%	14%
<i>Pycnanthemum tenuifolium</i>	4	98%	<i>Barbarea vulgaris</i>	*	55%	27%
<i>Tradescantia ohioensis</i>	5	98%	<i>Chaerophyllum tainturieri</i>	2	55%	31%
<i>Tripsacum dactyloides</i>	3	98%	<i>Medicago lupulina</i>	*	40%	17%
<i>Achillea millefolium</i>	1	95%	<i>Oxalis dillenii</i>	0	68%	45%
<i>Baptisia bracteata</i>	6	95%	<i>Carex meadii</i>	7	55%	33%
<i>Rudbeckia hirta</i>	2	95%	<i>Castilleja coccinea</i>	4	30%	8%
<b>Species Exclusive to Cluster</b>	<b>CoC</b>	<b>% Sites Occupied</b>	<b>Underrepresented Species</b>	<b>CoC</b>	<b>% Sites Occupied</b>	<b>% Sites Occupied (All Sites)</b>
<i>Claytonia virginica</i>	3	10%	<i>Linum sulcatum</i>	6	8%	33%
<i>Verbena bracteata</i>	0	5%	<i>Elymus virginicus</i>	3	13%	37%
			<i>Dalea purpurea</i>	7	58%	81%
			<i>Desmodium illinoense</i>	5	40%	60%
			<i>Hypericum perforatum</i>	*	25%	43%
			<i>Solidago nemoralis</i>	2	3%	19%
			<i>Asclepias viridiflora</i>	6	38%	54%
			<i>Lespedeza capitata</i>	6	50%	66%
			<i>Brickellia eupatorioides</i>	2	35%	50%
			<i>Symphotrichum pilosum</i>	0	43%	57%

Table A4. Most abundant, overrepresented, and underrepresented species within Cluster S2, and species found only within Cluster S2.

<b>Most Abundant Species</b>	<b>CoC</b>	<b>% Sites Occupied</b>	<b>Overrepresented Species</b>	<b>CoC</b>	<b>% Sites Occupied</b>	<b>% Sites Occupied (All Sites)</b>
<i>Andropogon gerardii</i>	4	100%	<i>Elymus virginicus</i>	3	70%	37%
<i>Rudbeckia hirta</i>	2	100%	<i>Cyperus echinatus</i>	3	44%	12%
<i>Schizachyrium scoparium</i>	5	100%	<i>Setaria parviflora</i>	3	51%	22%
<i>Apocynum cannabinum</i>	0	95%	<i>Juncus interior</i>	2	42%	17%
<i>Asclepias viridis</i>	1	95%	<i>Desmanthus illinoensis</i>	2	74%	53%
<i>Dalea candida</i>	7	93%	<i>Baptisia alba</i>	5	47%	26%
<i>Erigeron strigosus</i>	4	93%	<i>Melilotus officinalis</i>	*	65%	45%
<i>Tradescantia ohioensis</i>	5	93%	<i>Polygala sanguinea</i>	8	35%	15%
<i>Tripsacum dactyloides</i>	3	93%	<i>Prunella vulgaris</i>	*	40%	22%
<i>Achillea millefolium</i>	1	91%	<i>Lespedeza virginica</i>	5	53%	36%
<b>Species Exclusive to Cluster</b>	<b>CoC</b>	<b>% Sites Occupied</b>	<b>Underrepresented Species</b>	<b>CoC</b>	<b>% Sites Occupied</b>	<b>% Sites Occupied (All Sites)</b>
<i>Amhicarpaea bracteata</i>	3	5%	<i>Viola pedatifida</i>	6	42%	67%
			<i>Lithospermum canescens</i>	7	12%	36%
			<i>Phlox pilosa</i>	7	14%	38%
			<i>Dichanthelium oligosanthes</i>	4	58%	81%
			<i>Symphoricarichum oolentangiense</i>	8	30%	53%
			<i>Hesperostipa spartea</i>	8	7%	27%
			<i>Hypoxis hirsuta</i>	5	2%	22%
			<i>Asclepias viridiflora</i>	6	35%	54%
			<i>Carex meadii</i>	7	14%	33%
			<i>Erigeron annuus</i>	0	12%	30%

Table A5. Most abundant, overrepresented, and underrepresented species within Cluster S3, and species found only within Cluster S3.

Most Abundant Species	% Sites Occupied		Overrepresented Species	% Sites Occupied (All Sites)		
	CoC			CoC		
<i>Andropogon gerardii</i>	4	100%	<i>Lespedeza virginica</i>	5	76%	36%
<i>Rudbeckia hirta</i>	2	100%	<i>Baptisia australis</i>	6	84%	53%
<i>Schizachyrium scoparium</i>	5	100%	<i>Asclepias viridiflora</i>	6	82%	54%
<i>Asclepias viridis</i>	1	97%	<i>Pediemelum esculentum</i>	7	76%	49%
<i>Tradescantia ohioensis</i>	5	95%	<i>Brickellia eupatorioides</i>	2	76%	50%
<i>Amorpha canescens</i>	7	95%	<i>Cirsium undulatum</i>	4	42%	16%
<i>Dalea candida</i>	7	95%	<i>Ceanothus americanus</i>	9	68%	46%
<i>Dalea purpurea</i>	7	95%	<i>Liatris punctata</i>	5	42%	20%
<i>Erigeron strigosus</i>	4	95%	<i>Lepidium densiflorum</i>	0	45%	23%
<i>Penstemon tubiflorus</i>	3	95%	<i>Opuntia macrorhiza</i>	3	24%	8%
<i>Polytaenia nuttallii</i>	6	95%				
<i>Salvia azurea</i>	4	95%				

Species Exclusive to Cluster	% Sites Occupied		Underrepresented Species	% Sites Occupied (All Sites)		
	CoC			CoC		
<i>Brassica nigra</i>	*	3%	<i>Helianthus grosseserratus</i>	4	8%	47%
<i>Euphorbia dentata</i>	0	3%	<i>Zizia aurea</i>	5	0%	37%
<i>Hymenopappus scabiosaeus</i>	4	3%	<i>Liatris pycnostachya</i>	7	32%	68%
<i>Lespedeza stuevei</i>	4	3%	<i>Spartina pectinata</i>	4	13%	45%
<i>Stenosiphon linifolius</i>	6	3%	<i>Solidago canadensis</i>	2	45%	74%
			<i>Phlox pilosa</i>	7	11%	38%
			<i>Comandra umbellata</i>	6	13%	37%
			<i>Myosotis verna</i>	2	0%	24%
			<i>Eryngium yuccifolium</i>	7	39%	63%
			<i>Pycnanthemum tenuifolium</i>	4	61%	84%

Table A6. Most abundant, overrepresented, and underrepresented species within Cluster S4, and species found only within Cluster S4.

<b>Most Abundant Species</b>	<b>CoC</b>	<b>% Sites Occupied</b>	<b>Overrepresented Species</b>	<b>CoC</b>	<b>% Sites Occupied (All Sites)</b>	<b>% Sites Occupied</b>
<i>Allium canadense</i>	2	100%	<i>Physostegia angustifolia</i>	5	82%	16%
<i>Andropogon gerardii</i>	4	100%	<i>Coreopsis grandiflora</i>	8	79%	23%
<i>Baptisia bracteata</i>	6	100%	<i>Allium canadense</i>	2	100%	57%
<i>Schizachyrium scoparium</i>	5	100%	<i>Camassia angusta</i>	8	54%	14%
<i>Arnoglossum plantagineum</i>	6	97%	<i>Packera plattensis</i>	5	51%	13%
<i>Lobelia spicata</i>	6	97%	<i>Hypoxis hirsuta</i>	5	59%	22%
<i>Tradescantia ohioensis</i>	5	97%	<i>Prenanthes aspera</i>	8	59%	23%
<i>Achillea millefolium</i>	1	95%	<i>Callirhoe alcaeoidea</i>	6	90%	54%
<i>Amorpha canescens</i>	7	95%	<i>Ambrosia psilostachya</i>	3	62%	29%
<i>Asclepias viridis</i>	1	95%	<i>Helianthus maximiliani</i>	3	38%	8%
<i>Carex bushii</i>	4	95%				
<i>Erigeron strigosus</i>	4	95%				
<i>Penstemon tubiflorus</i>	3	95%				
<i>Polytaenia nuttallii</i>	6	95%				
<i>Psoraleidium tenuiflorum</i>	3	95%				
<i>Rudbeckia hirta</i>	2	95%				

<b>Species Exclusive to Cluster</b>	<b>CoC</b>	<b>% Sites Occupied</b>	<b>Underrepresented Species</b>	<b>CoC</b>	<b>% Sites Occupied (All Sites)</b>	<b>% Sites Occupied</b>
<i>Aristida oligantha</i>	0	3%	<i>Ratibida pinnata</i>	3	13%	55%
<i>Bouteloua gracilis</i>	5	3%	<i>Silphium laciniatum</i>	4	31%	68%
<i>Carex arkansana</i>	7	3%	<i>Comandra umbellata</i>	6	0%	37%
<i>Houstonia pusilla</i>	1	3%	<i>Silphium integrifolium</i>	4	0%	37%
<i>Tragopogon porrifolius</i>	*	3%	<i>Lespedeza violacea</i>	5	23%	56%
<i>Viola sagittata</i>	7	3%	<i>Phlox pilosa</i>	7	5%	38%
			<i>Spartina pectinata</i>	4	13%	45%
			<i>Desmodium illinoense</i>	5	28%	60%
			<i>Zizia aurea</i>	5	5%	37%
			<i>Fragaria virginiana</i>	2	33%	65%

**Appendix 2: Correlation Matrix**

Table A7. Correlation matrix of variables included in regression analyses. Pearson correlation and *p* are given in each cell.

	Perimeter	Perimeter-Area Ratio	Distance to Nearest Site	Distance to Nearest 3 Sites	% Urban	% Cropland	% Grassland	% CRP	% Woodland	% Water	Depth to Bedrock	Soil Available Water
Area	0.769	-0.580	-0.005	-0.021	0.004	-0.046	0.012	-0.035	-0.030	0.264	0.034	-0.079
Perimeter	0.000	0.000	0.940	0.755	0.951	0.486	-0.023	0.592	0.644	0.000	0.611	0.232
Perimeter-Area Ratio		-0.361	-0.040	0.080	-0.002	-0.024	-0.023	0.008	-0.024	0.237	-0.006	-0.071
		0.000	0.542	0.226	0.977	0.715	0.728	0.903	0.716	0.000	0.931	0.281
			-0.052	0.020	0.005	0.019	-0.119	0.164	0.101	-0.135	-0.076	0.072
			0.940	0.765	0.941	0.776	0.071	0.012	0.124	0.040	0.250	0.278
Distance to Nearest Site				0.237	-0.043	-0.008	0.116	-0.068	-0.101	0.006	0.050	-0.027
				0.000	0.513	0.905	0.007	0.301	0.124	0.924	0.446	0.680
Distance to Nearest 3 Sites					-0.094	-0.046	0.076	0.037	-0.030	-0.013	0.096	0.049
					0.155	0.485	0.248	0.580	0.653	0.846	0.147	0.460
% Urban						-0.100	-0.037	0.046	-0.052	-0.023	-0.003	-0.023
						0.127	0.580	0.482	0.428	0.722	0.967	0.731
% Cropland							-0.589	-0.143	-0.440	-0.096	0.146	-0.215
							0.000	0.029	0.000	0.144	0.027	0.001
% Grassland								-0.251	-0.341	-0.163	0.007	0.136
								0.000	0.000	0.013	0.910	0.038
% CRP									0.071	-0.065	0.019	0.044
									0.281	0.324	0.775	0.507
% Woodland										0.033	-0.232	0.090
										0.622	0.000	0.173
% Water											0.055	0.012
											0.405	0.857
Depth to Bedrock												0.033
												0.620