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**OCCUPANCY OF SMALL MAMMALS IN MISSOURI'S
FRAGMENTED PRAIRIES**

A Master's Thesis

Presented to

The Graduate College of
Missouri State University

In Partial Fulfillment

Of the Requirements for the Degree

Master of Science, Biology

By

Morgan E. Rodery

May 2021

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OCCUPANCY OF SMALL MAMMALS IN MISSOURI'S FRAGMENTED PRAIRIES

Biology

Missouri State University, May 2021

Master of Science

Morgan E. Rodery

ABSTRACT

Prairies once extended across much of Missouri, but now are dwindling and often surrounded by agriculture or other developed land. This has resulted in a mosaic-like landscape made up of small patches that are often isolated and distant from other patches. The theory of Island Biogeography posits that species richness of habitat patches exists as a dynamic equilibrium between colonization and extinction rates and that larger islands should contain more species than smaller islands. The purpose of this study is to examine how occupancy of a small mammal prairie specialist, the prairie vole, *Microtus ochrogaster*, and other small mammal species, such as the deer mouse, *Peromyscus maniculatus*, and the hispid cotton rat, *Sigmodon hispidus*, are affected by patch size, fragmentation, and isolation of prairie patches from other similar habitats. This study will be conducted by trapping small mammals using transects surveys with Sherman live traps (a standard in the field) and individuals will be marked with a unique ear tag. Small prairie fragments throughout Missouri were used as a model system, to analyze community characteristics of these patches, I 1) created single season multi-species occupancy models to determine effects of habitat characteristics such as area, isolation, and perimeter, on small mammal occupancy, and 2) compared observed species richness to area, perimeter, and isolation. Six species of small mammals were found over the course of this study. Prairie area, perimeter, or isolation were not found to have an impact on occupancy or species richness. An unknown mechanism seems to be driving small mammal community composition and further data collection is needed to find the mechanism causing these variations.

KEYWORDS: small mammals, prairie, occupancy, Island Biogeography Theory, fragmentation, edge effect, metacommunities

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Sean P. Maher, Ph.D., Thesis Committee Chair

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Julie Masterson, Ph.D., Dean of the Graduate College

In the interest of academic freedom and the principle of free speech, approval of this thesis indicates the format is acceptable and meets the academic criteria for the discipline as determined by the faculty that constitute the thesis committee. The content and views expressed in this thesis are those of the student-scholar and are not endorsed by Missouri State University, its Graduate College, or its employees.

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INTRODUCTION

Tallgrass prairie is an imperiled ecosystem, with an 85% decline across North America (Newbold et al. 2020). Missouri has experienced an even more alarming rate of prairie loss, with less than 1% of its original tallgrass prairie remaining of the nearly 6 million ha that used to cover the landscape (Newbold et al. 2020). The reduction of prairie in Missouri, and across all North America, has created a patchily distributed landscape of habitats (Figure 1). Determining the importance of these habitat fragments to plant and animal populations is important due to the rapid vanishing of prairie patches in Missouri. For instance, Jaeger (2000) found remnants and restored areas have many impacts on the local ecosystems in which they are found such as decreased erosion and carbon sequestration. Schulte et al. (2017) indicated biodiversity and

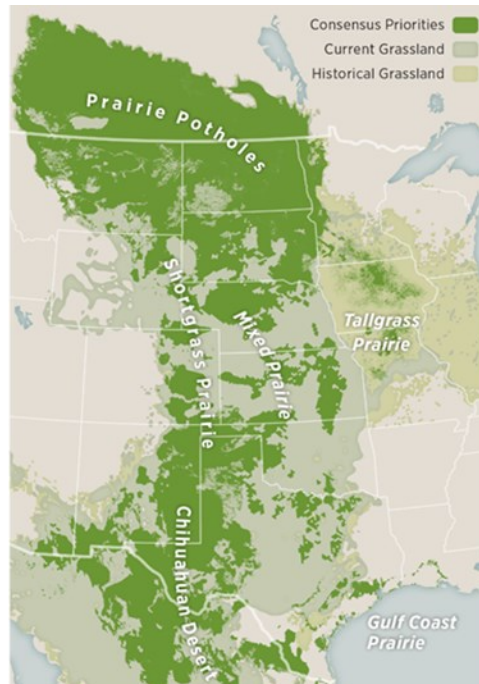


Figure 1. Map showing historical, current, and projected areas that are strongholds of prairie habitat (Huffman 2019).

ecosystem services are positively impacted by prairie vegetation and that small patches of maintained prairie can have large ecosystem impacts as well as improve conservation efforts of the species that reside in prairie areas.

Multiple studies comparing oceanic islands and habitat fragments, or “continental islands” have found that these isolated habitats share many similarities. This phenomenon is increasingly observed in areas where land has been highly developed by agriculture and other anthropomorphic involvement, and fragmentation occurs as a result of multiple uses of land surrounding habitats (Fox et al. 2000). Remnant habitat such as maintained and restored prairies throughout Missouri is becoming increasingly isolated as a result. Predictions of species abundance in these isolated fragments are so similar to that of oceanic islands that the theory of island biogeography can be applied to such habitats (Fox et al. 2000). The extinction rate of larger islands is lower than that of smaller islands. Therefore, larger islands retain more species. Blue points represent the expected number of species on each island (Rosenzweig 1995) as seen in Figure 2.

In Missouri small mammal species *Microtus ochrogaster*, *Peromyscus maniculatus*, *Peromyscus leucopus*, *Sigmodon hispidus*, *Neotoma floridana*, *Blarina hylophaga*, and *Reithrodontomys megalotis* are some species commonly found in prairie habitats. In a 15-year study done in similar prairie habitat in Kansas, heterogeneity and successional stage of habitat patches was a large determinant for presence, abundance, and distribution of small mammal species (Swihart and Slade 1990). In this study *Microtus ochrogaster* was the most commonly trapped species of all eleven species captured over the fifteen years. All species were prone to temporal variations in community composition which suggests a non-equilibrial small mammal community in Swihart and Slade’s study area.

Six species of small mammals were observed over the course of my study. These species were *Peromyscus maniculatus*, *Microtus ochrogaster*, *Peromyscus leucopus*, *Sigmodon hispidus*, *Reithrodontomys megalotis*, and *Sylvilagus floridana*. *Peromyscus maniculatus* was the most commonly captured species in this study.

Notably, *Blarina hylophaga* (Elliot's short-tailed shrew), *Blarina carolinensis* (Southern short-tailed shrew), and *Cryptotis parva* (North American least shrew) were missing from my study. These three species are commonly found in grasslands throughout Southwestern Missouri, and I expected to detect them in my study. In a study done on shrews in Wyoming, researchers found that sunken cans, or pitfall traps, were much more effective than snap traps or Sherman traps for capturing shrews (Brown 1967). It is likely that no shrew species were captured due to the use of Sherman traps rather than pitfall traps that are more successful for trapping shrew species.

Theoretical Spatial Population Dynamics

The theory of island biogeography (MacArthur and Wilson 1967) predicts the number of species on island as functions of colonization and extinction, such that the expected number of species will be a dynamic equilibrium without designating species identity. For instance, under the island area effect, the number of expected species increases with patch size due to lower extinction rates, although species will be replaced regularly. Isolation of habitat patches also will impact species richness by reducing colonization rates.

For example, Simberloff and Wilson (1969) empirically tested the model using arthropods on islands in mangrove swamps. They measured richness before and after fumigation and found similar values, albeit with different species occurring on islands. Simberloff and

Wilson concluded that colonization, with little impact of interaction of species with the environment, produce the original species richness curves. A slight decline can be observed as interactions between species and the environment become more important, these interactions will eventually lead to a dynamic equilibrium consistent with MacArthur and Wilsons IBT (Simberloff and Wilson 1969).

Another example of IBT, involving small mammals, comes from a study done on re-introduced small mammals in New Zealand. This study found that area had the most significant effect on small mammal species richness (Russell 2004). Russell found that a slight change in the magnitude (increase of one) of area increases the small mammal species richness 2.53 times over (Russell 2004). A similar result was also found in a study of island-dwelling mammals in Japan (Parra and Jaeger 2001). In this study, the assumption of IBT that island area has a positive effect on species richness was upheld (Parra and Jaeger 2001).

Observationally, Brown (1971) used the model to examine small mammal communities in boreal montane islands. Here, the number of small mammal species on “islands” was highly correlated with the area of the island being examined (Brown 1971). Brown (1971) also found that islands were so isolated that colonization did not have an impact on richness. He found that populations on montane islands were formed exclusively by extinctions. Brown concluded that after isolation of mountaintops occurred, extinctions have been the only reduction of diversity of fauna in his study areas. Brown also notes that there has been a low rate of extinction since these groups of species were isolated.

Lawlor (1998) drew different conclusions about the relationship between island area and species richness for these montane islands. He found a weak relationship between species richness and area of the habitat patch (Lawlor 1998). Lawlor posits that species composition on

montane islands in the Great Basin are made up of extinction-resistant species that were able to move from habitat patch to habitat patch. It is important to note that Lawlor has a more complete dataset than Brown and trapping effort was greater in Lawlor's study of small mammals.

Swihart et al. (2003) identified physical qualities of habitat in a fragmented landscape, such as patch area and patch isolation, as predictors for which subsets of species would make up small mammal communities. Species inability to migrate over large bodies of water in the case of Simberloff and Wilson, is similar to prairie species attempting to immigrate over large agricultural fields with little food or water resources.

Furthermore, according to MacArthur and Wilson's equilibrium theory regarding species numbers on islands, larger islands should have greater species richness, due to low extinction rates compared to extinction rates of smaller islands (Lawlor 1986). The same is true of species numbers for islands more proximal to the mainland. These islands should have greater species richness than islands more distal to the mainland due to higher rates of colonization (Lawlor 1986).

Multiple studies comparing oceanic islands and habitat fragments, or "continental islands" have found that these isolated habitats share many similarities. This phenomenon is increasingly observed in areas where land has been highly developed, and fragmentation occurs as a result of multiple uses of land surrounding habitats (Fox et al. 2000). Remnant habitat such as maintained and restored prairies throughout Missouri is becoming increasingly isolated as a result. Predictions of species abundance in these isolated fragments are so similar to that of oceanic islands that the theory of island biogeography can be applied to such habitats (Fox et al. 2000).

The abundance and maintenance of individual species within a set of patches can be related through the concept of metapopulations, in which populations consist of many local populations (Hanski 1999). Metapopulation models are helpful for interpretation of abundance rates and distribution of species when the spatial extent is large (Gotelli 1991). Models of metapopulation link local rates of immigration and extinction with events happening between sites.

Common models associated with metacommunities, termed “The Big 4” by Brown et al. (2017), are the species sorting model, neutrality, mass effects, and patch dynamics. Island biogeography theory is a neutral model, it assumes that all species are equal, all species are expected to have the same dispersal potential, same competitive ability, same habitat preferences, and be equal in all other regards. The mass effects model says that immigration and emigration (Leibold et al. 2004). Mass effects are likely influencing metacommunity structure in these grasslands. In habitats like grasslands, divisions between habitat patches are less distinct, mass effects influence community composition (Leibold et al. 2004).

The species-sorting model says that patch resources cause differences in species composition and interactions. In this perspective, quality of patches and dispersal ability of species most strongly influence species composition (Leibold et al. 2004). The patch dynamics model assumes that patches are equal in their ability to support populations and species composition is limited by dispersal, extinctions, and colonization (Leibold et al. 2004).

For instance, less isolated patches are more likely to exhibit the rescue effect, wherein local extinction followed by colonization occur between observations. This is due to the increased chance of emigration of individuals from surrounding habitat patches (Gotelli 1991). Here, patches of better-quality habitat should maintain populations of sensitive species, and

patches of poor quality would experience extirpations followed by recolonization (Pulliam 1988).

Though the concept of metapopulation and the theory of Island Biogeography share similarities, they are not the exact same. The theory of island biogeography uses extinction and emigration rates to predict species richness of islands or habitat patches. A metapopulation is a group of populations that have been separated but are of the same species and have some level of interaction. Thus, island biogeography is useful to examine communities at equilibrium, while metapopulations are inherently at disequilibrium.

Another characteristic sometimes associated with metacommunities is nestedness. Many fragmented habitats are made up of varied sizes of the same subsets of small mammal communities. These subsets are often composed of nested communities where islands that are smaller or more isolated have a smaller nested subset of species than that of larger islands with higher levels of connectivity (Lomolino 1996). Differences in species composition in local communities typically point toward differences in

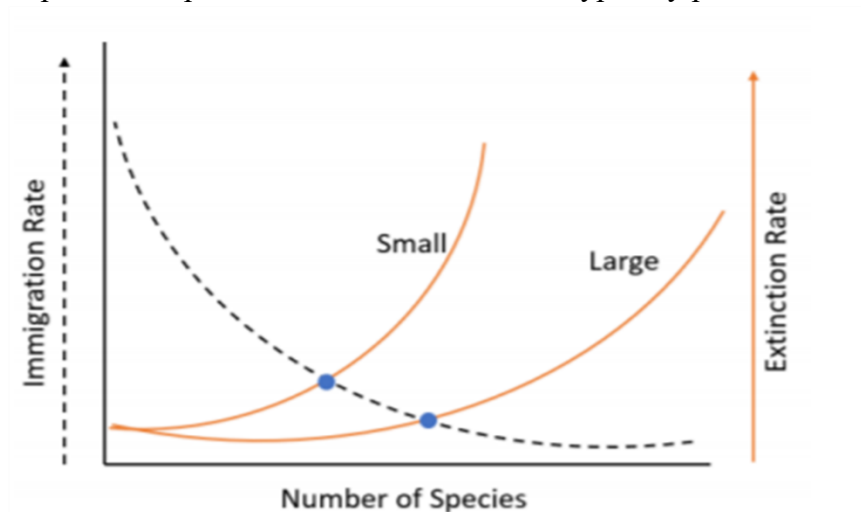


Figure 2. The extinction rate of larger islands is lower than that of smaller islands. Therefore, larger islands retain more species. Blue points represent the expected number of species on each island (Rosenzweig 1995).

species' sensitivity to patchiness or fragmentation (Swihart et al. 2003).

This relates to the idea of the single large or several small (SLOSS) argument. This is the debate between single large or several small fragments being more beneficial to species within fragments. In many fragmented landscapes, conservation of one large reserve of habitat is often not a feasible option (Vieira et al. 2009). A greater number of nested subsets of species within several small fragments can be supported than in one larger single preserve (Patterson and Atmar 1986). Larger areas of suitable habitat are able to support more species, however, when the area of a larger patch is reduced, the number of species is not reduced proportionally to reduction of area (Patterson and Atmar 1986). As the amount of area decreases, more species are expected to blink off.

Imperfect Detection. For field studies of species that may exhibit rarity or are difficult to capture, failure to observe an individual during a survey would not necessarily indicate absence as can be seen in Figure 3. Occupancy estimation is an effective method to study the change in the distribution of species in cases where detection is not always certain (Nichols et al. 2008). Mackenzie et al. (2002) describes the modeling method that implements a hierarchical approach to detection and occupancy, wherein these processes are examined simultaneously. The necessary sampling method uses multiple visits to individual study sites within a window of time to assess detection, and multiple sites to assess occupancy. As such, it is ideal for sites that are representative of habitat patches within a metapopulation (Mackenzie et al. 2002).

Detection is often imperfect, so estimators of community-level characteristics made from model-based estimators of occurrence provide a measure of uncertainty (Dorazio and Royle 2014). Ikanyan et al. (2012) detail how to model a full community to leverage information for

abundant species and better assess rare species. Further, these models can yield estimates of species richness while accounting for unsampled species. As can be seen in Figure 3, species 1 is detected in three of the habitat patches in (A) and only detected in one habitat patch in (B). Number and timing of visits can affect species detection, and occupancy modeling is useful in such cases where imperfect detection is inferred.

Eaton et al. (2011) demonstrated occupancy of lower keys marsh rabbits (*Sylvilagus palustris hefneri*) is influenced by patch size, wherein larger habitat patches were found to have higher probability of occupancy. This study also found that detection is not influenced by patch size (Eaton et al. 2011).

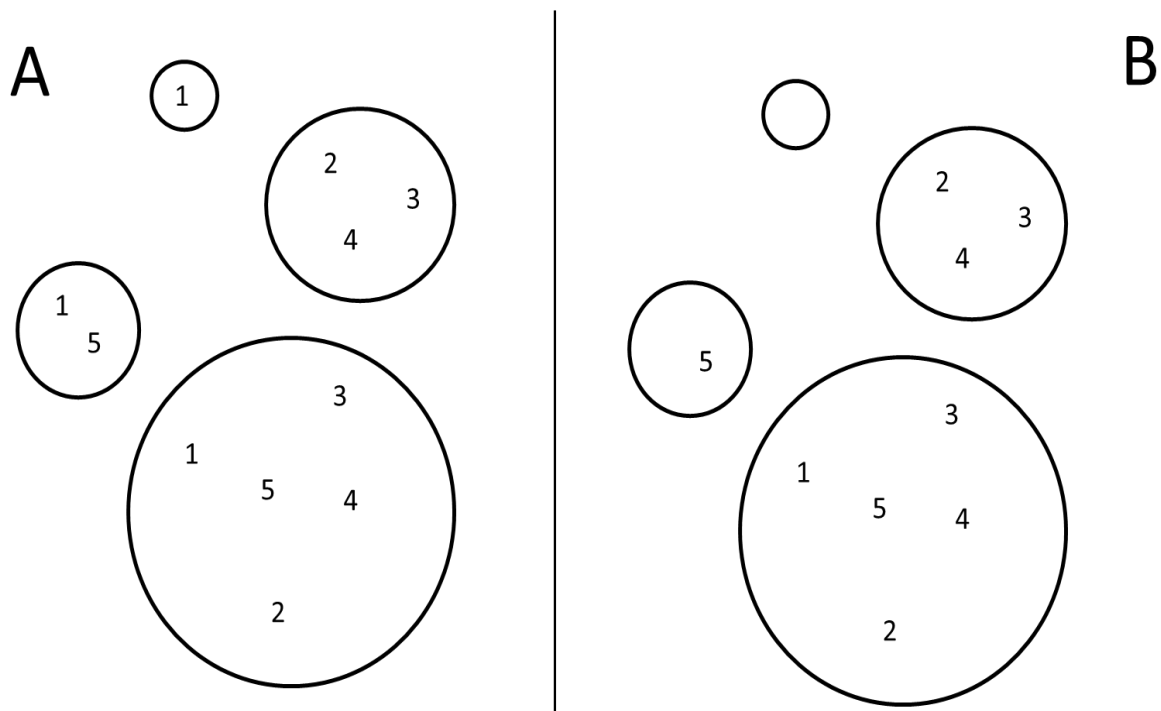


Figure 3. A network of patches containing species 1-5. Imperfect detection can lead to apparent absences of species such as species 1 in network B. Effects of habitat characteristics on species composition and richness can remain ambiguous due to imperfect detection.

Beasley and Maher (2019) also examined occupancy of small mammals in Missouri. This study focused on mammals in glades throughout Missouri and tested whether glade patches were following the trends predicted by the theory of island biogeography proposed by MacArthur and Wilson. This study found that increases in small mammal species richness was related to patch area but not isolation and effects of patch shape were different amongst different species (Beasley and Maher 2019).

Grasslands as Important Ecosystems. Studies of grasslands have shown climate change likely will negatively affect plant species composition and net primary production in grasslands (Liu et al. 2018). Liu (2018) also found that changes in net primary production as well as allocation of belowground resources was caused by a change in community composition. Global climate change has caused need and timely execution of biological monitoring of biological communities (Colwell and Coddington 1994). This is exacerbated by the increasing threats of species extinctions and baseline data for biological communities is necessary for future monitoring efforts to determine the effects of climate change on global biodiversity (Colwell and Coddington 1994).

Above ground net primary production has an impact on most ecosystem processes and studies of changes in productivity are essential for grassland habitats (Knapp and Smith 2001). As climate change, nitrogen deposition, and varied land use are altering natural landscapes more and more, these impacts will negatively alter the flow of carbon and energy within the ecosystem (Knapp and Smith 2001). This study found that assessing variability in primary production of grasslands temporally is essential to be able to quantify energetic constraints on population dynamics and community structure. Knapp and Smith's study also found evidence that the long-

term impacts of climate change will cause a major shift in the distribution of plant species throughout grasslands in North America (Knapp and Smith 2001).

Prairies are also useful for removing large amounts of CO₂ from the atmosphere by carbon sequestration (Berger et al. 2020). Dinerstein et al. (2019) suggest that climate stabilization could be possible through restoration and maintenance of 50% of earth's original grasslands. Restoration and maintenance of grasslands and conservation-conscious agricultural practices could comprise an estimated 1/5th of the natural climate solution mitigation to keep temperatures from increasing more than 2°C (Berger et al. 2020). This can also be done without significant impact on food production.

Landscape Dynamics. Surrounding land use has been found to have pronounced effects on species occupying habitat patches. For reintroduced fishers (*Pekania pennanti*) throughout Washington state and Canada, it was found that occupancy was strongly driven by characteristics of lands near the wilderness areas occupied by fishers (Happe et al. 2019). This study found that fisher occupancy was strongly associated with wilderness, meaning the edges of designated extensive wilderness areas. It is suggested that wildlife managers should endeavor to keep an appropriate combination of suitable habitat within animal's normal ranges as well as biotic factors in the surrounding areas and habitat edges (Happe et al. 2019).

The shape of patches and their contrast to the surrounding environment are expected to affect species. Patches with more sudden transitions to the surrounding environment are less permeable to organism movement, negatively impacting immigration and emigration rates (Collinge and Palmer 2002). Patch shape can have pronounced effects on individuals within the patch, such is the case for grassland birds (Davis 2004).

Isolation and size of fragments has been found to be an important driver of species richness in many cases. Species richness of small mammals inhabiting forested islands in the Atlantic was found to be strongly related to fragment size and isolation (Vieira et al. 2009). In Vieira's study area, habitat fragments have been isolated by human development and all remnant habitat patches are small, >100 hectares (Vieira et al. 2009). Species richness of small mammals was also found to be decreased when agriculture is the dominant surrounding land use (Vieira et al. 2009).

This Study

The main objectives of this study are to examine how landscape structure, patch isolation, and patch dynamics impact local small mammal communities. Another purpose of this study is to compile a small mammal species list for the prairie areas managed by the Missouri Prairie Foundation, as they have not been documented, but are important for management decisions as well as future comparative studies. I will focus on the mammal community structure and individual species' responses to qualities of their habitat. The sites chosen will vary in size, flora, and surrounding environment (e.g. grassland vs. agriculture).

These sites included 37 acre La Petite Gemme Prairie, 160-acre Bruns Tract and 40-acre Friendly Prairie in Pettis county, the 80-acre Welsch Tract and 80-acre Coyne Prairie in Dade county, 440-acre Denison Prairie and 320-acre Golden Prairies in Barton county, 440-acre Lattner Prairie and 180-acre Pleasant Run Creek Prairie in Vernon county, 171-acre Linden's Prairie in Lawrence county, and a newly acquisitioned by MPF prairie that is still unnamed, hereinafter referred to as Unnamed Prairie. A map of these sites can be seen in Figure 4. Some

individual sites are too close to each other to be distinguished on the map, so site areas are marked with a red dot.

I examined two hypotheses based upon principles of island biogeography and metapopulation dynamics. Hypothesis 1 predicts that patch shape and size will affect small mammal communities by influencing suitability. I expected that species occupancy will increase as patch size increases because larger patches will decrease extinction risk as seen in Figure 2. I also predicted that the ratio of perimeter to area would positively affect species occupancy for certain species such as *Sigmodon hispidus* because they are associated with disturbance and edge habitat.

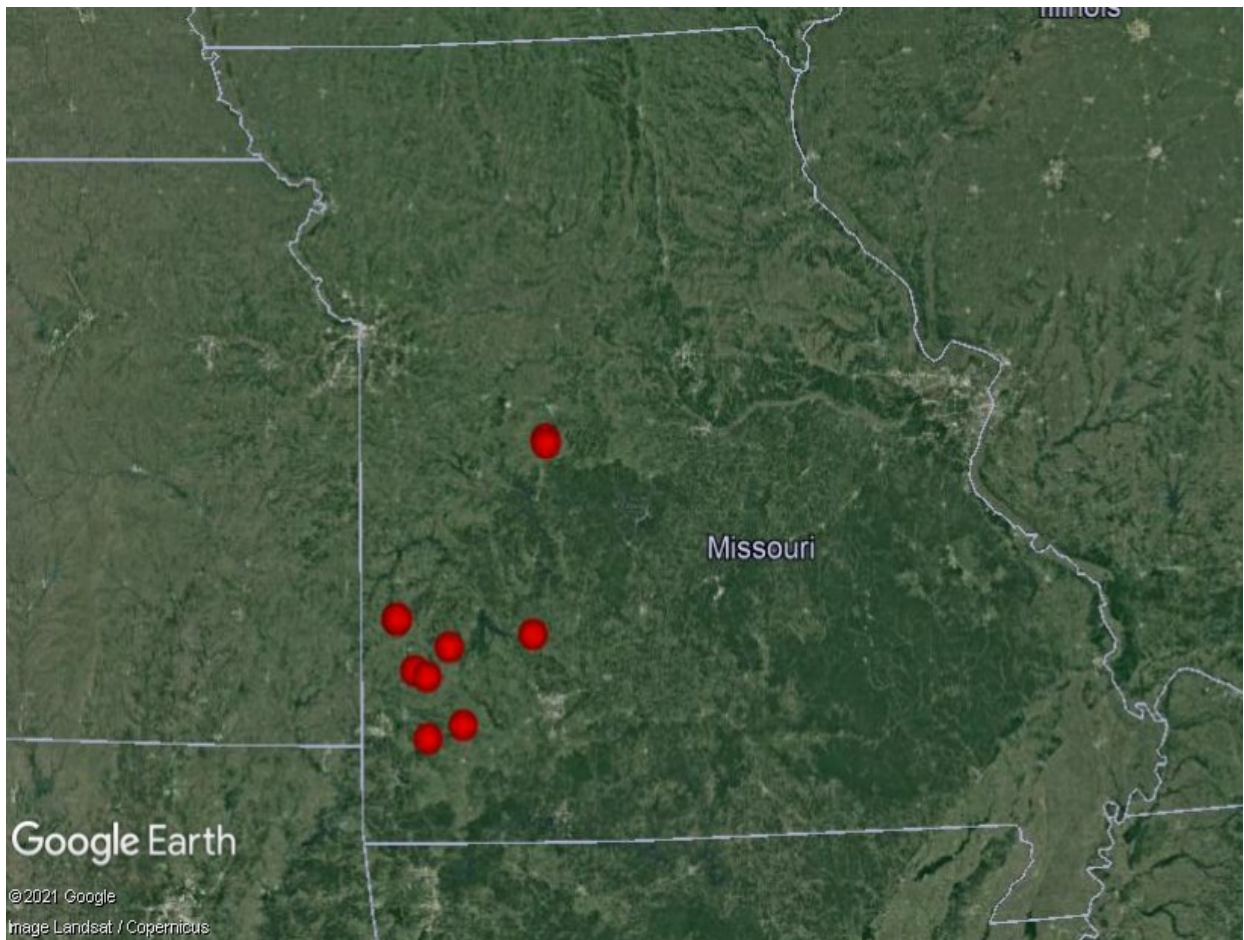


Figure 4. Map of study sites.

Hypothesis 2 states that species richness of small mammals is impacted by the amount of isolation of a prairie patch from other similar habitat types. I predict that prairie patches with a high level of isolation will have lower values of species richness than those patches that are closer to similar habitats.

Species occupancy is expected to increase as patch size increases as seen in Figure 2. Species occupancy is an important factor when related to the size of each prairie in my study. The prairies studied ranged from 40-440 acres.

I expect habitat patches surveyed during this study will follow a similar pattern of occupancy and detection. I predicted that the same can be said for sites sampled in this study. Isolation of patches from one another impacts species' ability to migrate from patch to patch. The earliest colonizers to immigrate to a patch are often driven to extinction by physical attributes of the surrounding habitat such as, in the case of the islands in Simberloff and Wilson's study area, large bodies of water or lack of sufficient breeding grounds. Species inability to immigrate over large bodies of water in the case of Simberloff and Wilson, is similar to prairie species attempting to immigrate over large agricultural fields with little food or water resources.

Rodents of MO Prairies. *Peromyscus maniculatus*, hereinafter referred to as the deer mouse, is one of the most common mammals in Missouri, and are found throughout North America. Deer mice can often be found in open habitat such as pastures and cultivated fields (Schwartz et al. 2001). Deer mice are often found in areas that are also inhabited or used by humans such as agricultural fields. They are not often found in areas with populations of a similar species captured, *Peromyscus leucopus*, due to the deer mouse's preference for open areas with little woody encroachment (Schwartz et al. 2001).

Peromyscus leucopus, hereinafter referred to as the white-footed mouse, is one of the most abundant animals in wooded areas in Missouri (Schwartz et al. 2001). White-footed mice are not often found in concurrence with the deer mouse due to their preference for wooded habitats. If these species do co-occur it is often a sign of succession in the plant community within the habitat (Schwartz et al. 2001).

Microtus ochrogaster, hereinafter referred to as the prairie vole, is a very short-lived (average of less than one year in wild populations), species and experience times of population abundance approximately every four years (Schwartz et al. 2001). Reduction of population numbers can happen quickly and are often caused by severe weather or changes in land use. Prairie voles are an important food source for many animals including owls and snakes. Prairie voles are often found in herbaceous fields and grasslands but are not typically found in wooded areas. Prairie voles are also known to inhabit small home ranges of one acre or less (Schwartz et al. 2001).

Sigmodon hispidus, hereinafter referred to as the hispid cotton rat, is a recently immigrated species to Missouri, the first records of its presence appearing in 1945 (Schwarz et al. 2001). The hispid cotton rat is expanding northerly from South America and the Southern US, currently, Missouri is one of the most northerly states it inhabits. *Sigmodon hispidus* inhabits dense fields and can live near and within agricultural areas. Patches with more edge might be preferable for some species of small mammals such as *Sigmodon hispidus*. Predictions of climate change indicate that there will be increases in extreme weather events which could result in the expansion of the cotton rat (Rehmeier et al. 2005). The expansion of the cotton rat to temperate climates has subjected the species to harsh winters for which it is not well-suited (Reed and Slade 2006). The hispid cotton rat could be an appropriate model organism for changes in weather patterns

in the Great Plains region. Stress at the northern reaches of its habitat may cause more distribution of the species and could indicate climate fluxes in the Great Plains (Rehmeier et al. 2005). Due to its use as a model organism, monitoring efforts of hispid cotton rats in the Midwest could be an informative indicator of climate change.

Reithrodontomys megalotis, hereinafter referred to as the western harvest mouse, was only captured at two sites. Western harvest mice inhabit fields with dense groundcover and tend to occur near water or marshy areas (Schwartz et al. 2001).

METHODS

Data Collection

I sampled 12 prairies from June–July 2020. These prairies included 37 acre La Petite Gemme Prairie, 160-acre Bruns Tract and 40-acre Friendly Prairie in Pettis county, the 80-acre Welsch Tract and 80-acre Coyne Prairie in Dade county, 440-acre Denison Prairie and 320-acre Golden Prairies in Barton county, 440-acre Lattner Prairie and 180-acre Pleasant Run Creek Prairie in Vernon county, 171-acre Linden's Prairie in Lawrence county, and a newly acquisitioned by MPF prairie that is still unnamed, hereinafter referred to as Unnamed Prairie. All prairies have similar management strategies, but differ in size, shape (amount of perimeter), and plant species composition.

Google Earth Pro (Version 7.3.3.7786) was used to delineate the extent of prairies. Sites were located on the map and polygons were drawn for use in spatial analysis of each site. Polygons were then imported to QGIS (Version 3.18.1. QGIS.org, 2021) to determine area and perimeter of polygons, as well as distances between sites. I used QGIS to determine relevant conservation areas and designated prairie habitat proximal to sampling sites.

Program R was also used to determine isolation of prairie patches. The package `rgdal` (R package version 1.5-18, Bivand, Keitt and Rowlingson 2020) and function `spDists` were used to determine distances between sampling sites and other designated prairies or conservation areas. QGIS (Version 3.18.1. QGIS.org, 2021) was used to find managed conservation areas near sampling sites. Minimum and average distances of isolation were then calculated for each site in kilometers. QGIS was used to determine areas of prairies and other characteristics used for spatial analysis. Program R package `vegan` (R package version 2.5-6) was used to perform

mantel tests using Bray-Curtis distance to test correlation between isolation metrics and species richness.

Area of each sampling site was found using QGIS and information from the Missouri Prairie Foundation. Area was compared to species richness of sites. I also quantified shape by calculating the ratio of perimeter: area. Program R package *vegan* (R package version 2.5-6) was used to perform mantel tests using Bray-Curtis distance to test correlation between prairie area and species richness.

I used standard field methods to set and capture small mammals, using large folding aluminum treadles and doors galvanized (LFATDG) Sherman live-traps (dimensions 7.62x8.89x22.86 cm) baited with sunflower seeds. Generally, mammal trapping stations were placed along 250 m transects using a 10m interval point-transect method, with two traps were set at each trap station (e.g. Beasley and Maher 2019). The Welsch Tract was not large enough for two 250 m transects and multiple smaller transects had to be used, and a third smaller transect was set at Linden's Prairie for two nights. Traps were set in the evening and checked the following morning to minimize trap fatality. The geographic coordinates of the beginning and ending points of trap lines were recorded using the ViewRanger app (Version 10.3.5 (613)) for reproducibility of the study (Appendix A). Maps of each prairie can be found in Appendix B.

The number of trap nights was recorded for each site daily. Sites that contained two trap lines could have a possible 100 trap nights as 100 total traps were set per night. Traps that were empty but closed were counted as half trap nights. Weather conditions, moon phase of trap night, and temperature were noted for each sampling day. Vapor pressure deficit, average temperature, minimum temperature, and maximum temperature was obtained for each sampling site using data obtained by PRISM Climate Group (<http://prism.oregonstate.edu>).

Mammals captured were identified to species, measured (right hind foot length, ear length, tail length), and sexed. Individuals captured were also given an ear tag with a unique number so they could be identified if recaptured later in my study. Approval for this project was obtained from the Missouri State University Institutional Animal Care and Use Committee and Missouri Department of Conservation before data collection began (IACUC, ID #18-029.0; MDC #18692). Documentation for IACUC approval can be found in Appendix C.

Data Analysis

Program R (Version 3.0 or 4.0x 1.3.1073) was used for statistical analysis of data collected. Data was converted to a useable format using the `readxl` function (Wickham 2019). Using the minimum number of individuals alive at each site, I calculated species richness (S) as the number of species encounter at each site and two diversity indices, Shannon-Wiener (H') and Simpson's (S_d) using the `vegan` package in R. The Simpson's diversity index quantifies the probability that two randomly selected individuals from the sample do not represent the same species (Tuomisto 2010). The Shannon-Wiener diversity index quantifies the uncertainty in the species identification if you were to randomly select an animal from the dataset (Tuomisto 2010).

To account for uncertainty in effort, I also calculated the Chao estimate for species richness for each site. The Chao estimator accounts for rarity in the samples and provides a value per site that represents potential richness given the number of singleton and doubleton observations. To test for effects of area and shape on species richness, I used a generalized linear model, with a Poisson error distribution. Site-level species richness was plotted against area

(km²) for each site. Site-level species richness was also plotted against perimeter: area (km²) ratio of each site.

The package unmarked (Version 1.01, Fiske and Chandler 2011) was used to build and compare occupancy models for small mammal species. I used the `occu` function `occu` for fitting the single season occupancy model developed by Mackenzie et al. (2002). This model estimates site occupancy rates when detection probabilities are less than 1 (Mackenzie et al. 2002). The trapping events have two possible outcomes, capture or no capture, and are the Bernoulli trials in this modeling approach.

Occupancy models are often used in instances where biological communities are observed using repeated measures, live trapping in the case of this study. These models estimate the composition of species and the number of species (N) within a study community. Detection is often imperfect, so estimators of community-level characteristics are made from model-based estimators of occurrence (Dorazio and Royle 2014).

Five occupancy models were created for each species. Model 1 was a dot model, testing the null hypothesis that trap effort, or number of traps set, did not influence detection of species. Model 2 tested the effect of the number of visits on the detection of species. Model 3 tested the effect of trapping effort on the detection of species. Model 4 tested the effects of number of trap nights vs trap effort on the detection of species. Model 5 tested the effects of cumulative trap effort (over 3 trap nights) on the detection of species.

I identified the most parsimonious model by comparing Akaike information criterion (AIC) values. AIC combines the likelihood of the model and a penalty for the number of parameters, which can determine which model was most informative while accounting for the model complexity. Models with the smallest AIC value were deemed the best-supported, and

models within two AIC units were statistically similar. Additionally, I calculated AIC weights to provide a comparable value across species, as comparing AIC units between species is uninformative.

The plant communities of study sites were also analyzed to find any relationship between characteristics of the plant community and small mammal species richness, if one existed. The Missouri Prairie Foundation has compiled Floristic Integrity reports for some of the prairies managed throughout Missouri (Thomas 2018). These reports were compiled for Coyne Prairie, Denison Prairie, La Petite Gemme, Linden's Prairie, and Pleasant Run Creek Prairie. The coefficient of conservatism (C-value) is rated on a 0-10 scale and is assigned to each species of plant native to Missouri. The 0-10 scale indicates the probability that each plant would be found in a habitat that is stable and in the late stages of succession (Thomas 2018). The C-value was compared with species richness of small mammals in each of the above-mentioned sites. Values used from the floristic integrity report were mean C, native mean C, %C value 0, %C value 1-3, %C value 4-6, %C value 7-10, plant species richness, native plant species richness, and non-native plant species richness.

To estimate abundance of *P. maniculatus*, I used the fishmethods package (R package version 1.11-1, Nelson 2019), which includes an implementation of the Schnabel method. Abundance estimates were not possible for the remaining species due to a small sample size or few recapture events. This approach uses the number new captures and marked individuals in more than two sampling events. A closed population within each patch was assumed due to the short time frame of sampling. From these estimates, linear models were fit to determine if abundance of *P. maniculatus* was affected by patch attributes.

To examine patterns of community similarity, I used non-metric multidimensional scaling (NMDS) ordination, using the vegan package. Raw data for the ordination were the observed individual captures (e.g. minimum number alive) for each of the 12 sample sites. Compositional dissimilarity of sites was measured using a Bray-Curtis distance and then displayed in a scatterplot. To examine further factors influencing community composition, I used a canonical correspondence analysis (CCA) in vegan to quantify the effect of perimeter, area, mean distance to similar habitat, and minimum distance to similar habitat, on presence of species in each patch.

RESULTS

A total of 172 unique small mammals were captured, and I detected six species of small mammals (Table 1). The most widespread species was the North American Deer mouse (*Peromyscus maniculatus*), which occurred at 11 sites and was only absent from the recently acquired prairie. The Prairie Vole (*Microtus ochrogaster*) occurred at six sites and the Hispid Cotton Rat (*Sigmodon hispidus*) occurred at 5 sites. Both the White-footed Deer mouse (*P. leucopus*) and the Western Harvest Mouse (*Reithrodontomys megalotis*) were each detected at two sites. Only one individual of *S. floridanus* was captured, likely due to the size of traps used. LFATDG Sherman traps were used which generally are not large enough for adult *S. floridanus*, the individual captured was a juvenile. Species captures in each site can be seen in Table 1 and Figure 5.

Unnamed						
Drover's						
Friendly						
Welsch						
Coyne						
Golden						
LPG						
Lattner						
Denison						
Dorris						
Linden's						
Pleasant						
	<i>Peromyscus maniculatus</i>	<i>Microtus ochrogaster</i>	<i>Sigmodon hispidus</i>	<i>Peromyscus leucopus</i>	<i>Reithrodontomys megalotis</i>	<i>Sylvilagus floridanus</i>

Figure 5. Species detected in each site. Grey boxes indicate detection, white boxes indicate no detection.

Species richness (S) varied among sites (mode = 3, median = 2, max = 5), but this was not related to area ($P > 0.05$). Pleasant and Dorris Prairies had the greatest measure of local diversity as indicated by Simpson's (S_D) and Shannon-Weiner Indices (H'), in part because the relative evenness at Dorris and greater richness at Pleasant (Table 1).

Sites with a higher level of perimeter: area ratio are more desirable for species that prefer habitats with edge, fence lines or areas of woody encroachment such as *Sigmodon hispidus* and *Peromyscus leucopus*. The lm for the relationship was not significant ($P > 0.05$).

Detection models were fit for *P. maniculatus*, *M. ochrogaster*, *P. leucopus*, and *S. hispidus* (Table 2), but not for *R. megalotis* or *S. floridanus* due to the small sample size. For *P. maniculatus* and *M. ochrogaster*, AIC values did not differ by more than 2 for any model (Table 2), suggesting that detectability was independent of any factors tested in the models. For both species the null model has the lowest AIC value and is therefore the most informative. For *P. leucopus*, AIC values differed by more than 2 for model 2 and model 5. For *S. hispidus*, AIC value differed by more than 2 for model 3. This means that detection probability was most effected by daily trap effort.

Only one individual of *S. floridanus* was captured. Only two *R. megalotis* individuals were captured, one in Pleasant Run Creek Prairie and one in Linden's Prairie. Due to the small sample size, occupancy and detection models were not fit for *R. megalotis* or *S. floridanus*. Detection models were fit for *P. maniculatus*, *M. ochrogaster*, *P. leucopus*, and *S. hispidus* (Table 2). For *P. maniculatus* and *M. ochrogaster*, AIC values did not differ by more than 2 for any model (Table 2), suggesting that detectability was independent of any factors tested in the models. For both species, the null model has the lowest AIC value and is therefore the most informative. For *P. leucopus*, AIC values differed by more than 2 for model 2 and

model 5. This means that detection probability was most effected by the number of visits to the site as well as cumulative trap effort. For *S. hispidus*, AIC value differed by more than 2 for model 3. This means that detection probability was most effected by daily trap effort.

For *P. maniculatus*, *M. ochrogaster* and *S. hispidus* AIC values do not differ by more than 2 for any model (Table 3). This means that all models were equally supported and the best-supported model for *P. maniculatus* and *M. ochrogaster* was the null model. The best-supported model for *P. leucopus* included perimeter: area ratio+ area and area.

Abundance estimates were calculated only for *P. maniculatus* (Table 4) because there was a sufficiently large enough sample size for seven of the sites. The estimated abundance for each is as follows: La Petite Gemme (N=14.25), Linden's Prairie (N=16.875), Pleasant Run Creek Prairie (N=11.5), Dennison Prairie (N=16.67), Coyne Prairie (N=21), Welsch Tract (N=16), and Drover's Prairie (N=15). Upper confidence intervals, lower confidence intervals, and confidence interval distribution were also calculated for each site with an abundance estimate (Table 4). Abundance for this species was not related to patch attributes.

Data collected by the Missouri Prairie Foundation for floristic integrity reports were analyzed for Coyne Prairie, Denison Prairie, La Petite Gemme, Linden's Prairie, and Pleasant Run Creek Prairie The coefficient of conservatism or C-value was compared with species

richness of small mammals for each site (Table 5). Values used from the floristic integrity report were mean C, native mean C, %C value 0, %C value 1-3, %C value 4-6, %C value 7-10, plant species richness, native plant species richness, and non-native plant species richness. No measure from the floristic integrity report for these sites was correlated to species richness of small mammals. The p-values calculated for the relationship between small mammal species richness and all characteristics of the plant community were non-significant ($P>0.05$). Native mean C was also plotted against small mammal species richness and no relationship was found.

By measuring compositional dissimilarity of sites using a Bray-Curtis and then using ordination to display, I identified a cluster of seven sites that have similar small mammal communities (Figure 6). The newly acquired prairie is distinct from the others due to the presence of only *S. hispidus*, and Friendly Prairie CA is offset from others because it is represented by a single capture of *P. maniculatus*. Pleasant Run Creek and Linden's Prairies are

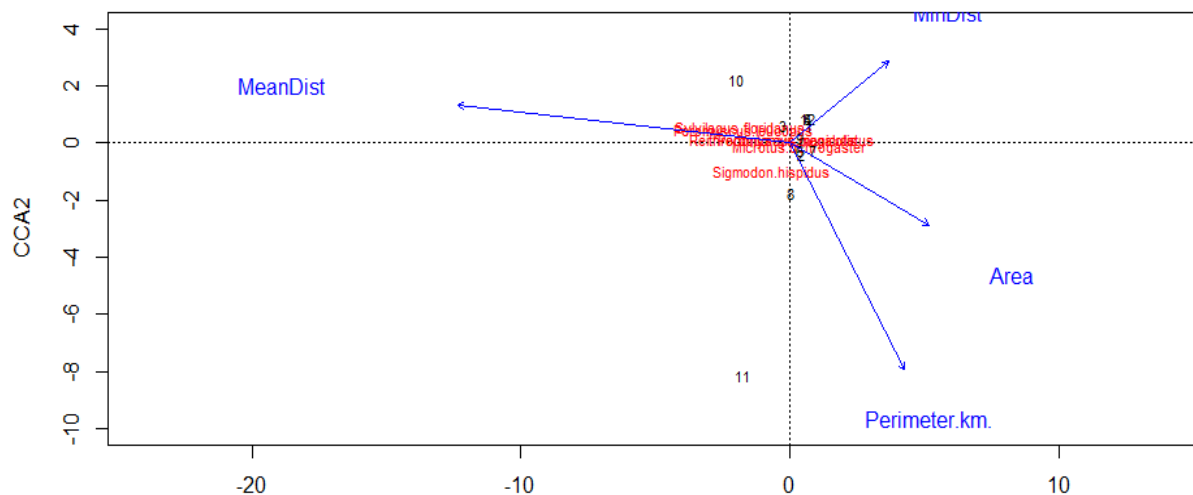


Figure 7. Canonical correspondence analysis of the effect of patch perimeter, patch area, mean distance to similar habitat, and minimum distance to similar habitat species composition of each patch.

offset because of the occurrence of *R. megalotis*, and Dorris Creek Prairie CA had few captures of *P. maniculatus*. There is no correlation between Bray-Curtis distances and geographic distance (Mantel $r = 0.225$, $P = 0.151$).

Table 1. Number of unique captures of small mammal species by site and related diversity. Species 1 is *Microtus ochrogaster*, species 2 is *Peromyscus maniculatus*, species 3 is *Peromyscus leucopus*, species 4 is *Reithrodontomys megalotis*, species 5 is *Sigmodon hispidus*, species 6 in *Sylvilagus floridana*.

Location	Species Identity						S	S_Chao	Chao SE	S _D	H'
	1	2	3	4	5	6					
Coyne	1	0	7	0	0	0	2	2	0.3535534	0.219	0.377
Denison	2	0	10	0	2	0	3	3	0	0.449	0.796
Dorris	2	1	2	0	0	0	3	3	0.1360828	0.640	1.055
Drovers'	0	0	9	0	0	0	1	1	0	0.000	0.000
Friendly	0	0	1	0	0	0	1	1	0	0.000	0.000
Golden	0	0	7	0	1	0	2	2	0.3535534	0.219	0.377
La Petite Gemme	19	0	8	0	0	0	2	2	0	0.417	0.608
Lattner	1	0	6	0	3	0	3	3	0.4082483	0.540	0.898
Linden's	0	0	13	1	2	0	3	3	0.2041241	0.320	0.602
Pleasant	1	12	8	1	0	1	5	8	4.383919	0.601	1.116
Unnamed	0	0	0	0	5	0	1	1	0	0.000	0.000
Welsch Tract	0	0	10	0	0	0	1	1	0	0.000	0.000

Table 2. Akaike Information Criteria (AIC) values and weights of detection models for each species.

Model	Species							
	<i>P. maniculatus</i>		<i>M. ochrogaster</i>		<i>P. leucopus</i>		<i>S. hispidus</i>	
	AIC	ω	AIC	ω	AIC	ω	AIC	ω
Null (1)	34.68	0.32	42.64	0.33	22.24	0.076	46.20	0.110
Number of Visits (2)	35.26	0.24	44.53	0.13	19.17	0.353	46.49	0.095
Daily Trap Effort (3)	36.47	0.13	43.22	0.25	22.95	0.053	43.35	0.454
Daily Trap Effort + Number of Visits (4)	36.50	0.13	44.19	0.15	20.55	0.177	45.02	0.197
Cumulative Trap Effort (5)	35.88	0.18	44.33	0.14	19.23	0.342	45.64	0.144

Table 3. Akaike Information Criteria (AIC) values and weights of occupancy models for each species.

	Species							
	<i>P. maniculatus</i>		<i>M. ochrogaster</i>		<i>P. leucopus</i>		<i>S. hispidus</i>	
Model	AIC	ω	AIC	ω	AIC	ω	AIC	ω
Null	34.68	0.599	42.64	0.50	22.24	0.042	46.20	0.46
Perimeter: Area +Area	42.57	0.012	45.48	0.12	16.44	0.764	48.47	0.15
Perimeter: Area	40.57	0.031	44.26	0.22	19.37	0.177	48.19	0.17
Area	35.71	0.358	44.87	0.16	23.99	0.017	47.70	0.22

Table 4. Abundance estimates and confidence intervals for *Peromyscus maniculatus*.

Site	N	invSE	LCI	UCI	CI
La Petite Gemme	14.25	0.035	7.12	57	Poisson
Linden's Prairie	16.87	0.021	9.64	45	Poisson
Pleasant Run Creek	11.5	0.043	5.75	46	Poisson
Prairie					
Dennison Prairie	16.67	0.034	7.14	Inf	Poisson
Coyne Prairie	21	0.047	7	Inf	Poisson
Welsch Tract	16	0.036	6.85	Inf	Poisson
Drover's Prairie	15	0.047	6	Inf	Poisson

Table 5. Prairies surveyed, their area, perimeter: area ratio, perimeter, relative isolation measures, and mean C value of the plant community.

Location	Area (km ²)	Perimeter: Area Ratio	Perimeter (km)	Mean distance to other locations surveyed (km)	Minimum distance to other locations surveyed (km)	Mean distance to designated conservation area (km)	Minimum distance to designated conservation area (km)	Mean C	
Coyne	0.32	7.49	2.42	51.35	0.81	10.12	1.39	3.9	
Denison	1.29	3.72	4.82	57.79	0.91	10.08	1.87	3.93	
Dorris	0.63	4.91	3.18	55.68	8.05	10.67	0.93	NA	
Drovers'	0.32	10.59	3.42	129.45	2.23	10.42	0.61	NA	
Friendly	0.15	9.58	1.55	131.12	2.23	9.71	0.66	NA	
Golden	1.71	4.93	6.38	54.92	8.05	9.53	0.38	NA	
La Petite Gemme	0.15	10.59	1.58	79.19	51.56	10.42	2.11	4.13	
Lattner	0.48	5.79	2.81	58.28	0.9	10.44	1.75	NA	
Linden's	0.64	8.07	5.58	73.54	22.99	10.09	1.38	4.73	
Pleasant Run Creek	0.61	4.32	3.14	57.8	0.9	9.64	0.79	4.03	
Unnamed	0.27	8.35	2.26	78.87	22.99	9.95	0.83	NA	
Welsch Tract	0.31	7.32	2.37	51.18	0.81	10.15	0.45	NA	

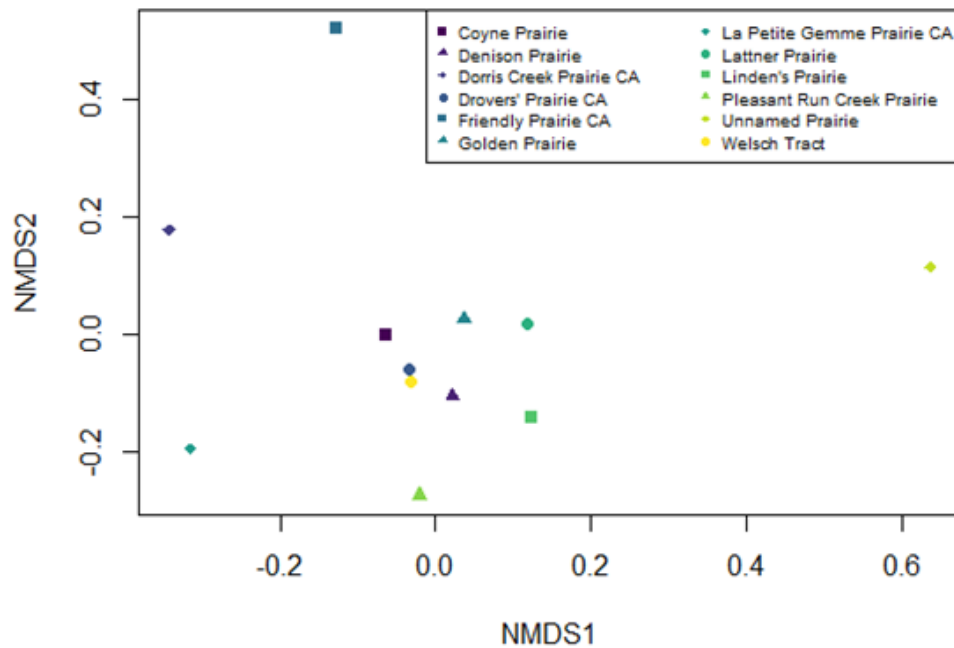


Figure 6. Scatterplot of the ordination of small mammal communities using a Bray-Curtis distance of unique individuals.

Effects of patch area, patch perimeter, mean distance to similar habitat, and minimum distance to similar habitat were also evaluated. A CCA plot was created to display the effects of these habitat characteristics on species composition within the patches (Figure 7). Many features of the habitat were measured and only perimeter, or amount of edge, influence species composition. In Figure 7, *Sigmodon hispidus* is not clustered with the other species and is somewhat affected by the amount of edge in the habitat, as is consistent with my prediction. Area or isolation of patches seemed to have no effect on small mammal species assemblage.

DISCUSSION

The results of this study can provide baseline data and the potential for repeatability of data collection in these sites by future researchers. Across the twelve study sites: La Petite Gemme Prairie, Bruns Tract, Friendly Prairie, Welsch Tract, Coyne Prairie, Denison Prairie, Golden Prairie, Lattner Prairie, Pleasant Run Creek Prairie, Linden's Prairie in, and a newly acquisitioned by MPF prairie that is still unnamed, six species of small mammals were detected. The six species detected were *Peromyscus maniculatus*, *Microtus ochrogaster*, *Peromyscus leucopus*, *Sigmodon hispidus*, *Reithrodontomys megalotis*, and *Sylvilagus floridana*.

Patches surveyed over the course of this study were equivalent in many ways regarding habitat characteristics although most patches contained different small mammal community assemblages. Neither the sites' proximity to each other nor area influenced occupancy or species richness of patches. An unknown mechanism not considered in this study is responsible for driving differences in the small mammal communities within these sites.

The results of my field observations did not support my hypotheses. Hypothesis 1 stated that patch shape and size will have an effect on the small mammal communities of sampling sites. I expected that species occupancy would increase as patch size increases. Data collected during this study showed that species richness (S) varied among sites (mode = 3, median = 2, max = 5), but this was not related to area ($P > 0.05$). For small mammal occupancy, area was not a significant covariate and larger AIC weights were associated with null models.

Hypothesis 2 states that species richness of small mammals will be impacted by the amount of isolation of a prairie patch from other similar habitat types. I predicted that prairie patches with a high level of isolation would have lower values of species richness than those

patches that are closer to similar habitats. Isolation of patches had no effect on species richness in patches observed during this study. Further, it did not influence occupancy or community composition.

I also predicted that the ratio of perimeter to area would affect occupancy for certain species such as *Sigmodon hispidus*. Some species prefer habitat with more edge, and I predicted that these species would have higher occupancy in patches with a higher ratio of perimeter to area. The CCA showed that this prediction was supported for *Sigmodon hispidus*. In patches with greater perimeter *Sigmodon hispidus* were more likely to be detected.

Only one individual of *S. floridanus* was captured, likely due to the size of traps used. LFATDG Sherman traps were used which generally are not large enough for adult *S. floridanus*, and the individual captured was a juvenile. Notably no shrews, a common species in the study areas, of any species were captured. This is likely due to the type of trap used. Shrews are not often captured in Sherman traps and pitfall traps are usually necessary to sample for these species.

Aspects of the plant community such as mean C (coefficient of conservatism), native mean C, %C value 0, %C value 1-3, %C value 4-6 %C value 7-10, plant species richness, native plant species richness, and non-native plant species richness also did not have an effect on small mammal species richness during this study. In a study done on small mammal abundance in Kansas, it was found that small mammal populations not responding to variance in vegetation structure, due to the lack of variation in vegetation across patches (Schweiger et al. 2000). Sampling sites analyzed in this study followed a similar pattern and did not vary greatly in plant community composition.

Results of the species richness-area relationship similar to this study are not unheard of. In a study of black-tailed prairie dog towns in the panhandle of Oklahoma, it was also found that species richness was not significantly correlated with the area, isolation, or characteristics of the habitats, or towns (Lomolino and Smith 2003). The results of Lomolino and Smith's study also are not consistent with island biogeography predictions. The researchers in this study found that characteristics of the surrounding landscape (within 10 km of the habitat) was most significantly correlated with species richness (Lomolino and Smith 2003).

Lomolino and Smith also came to a similar conclusion that habitat fragmentation, resulting in isolation of native habitats is one of the greatest threats to biodiversity. Grassland ecosystems have been especially impacted by fragmentation, and suffered in loss of extent, connectivity, and biological diversity (Lomolino and Smith 2003). To conserve native grassland species, a network of prairie reserves in areas similar to historic ranges of native prairie must be maintained for native species to persist (Lomolino and Smith 2003).

The results of this study demonstrate the need for long-term studies of small mammal population trends. Although patterns of abundance can be observed in short-term studies, long-term studies are necessary to establish general patterns of abundance and the factors that contribute to the variation (Rehmeier 2005). The underlying mechanism driving differences between patches remains ambiguous in this study, but long-term data could reveal the habitat characteristics responsible for variation.

The importance of long-term studies for small mammal communities has been addressed by many researchers. In a fifteen-year study done on early successional small mammal communities in Kansas, the researchers assert that if only 2-3 years of data collection had been completed, alternative conclusions could have been drawn concerning the effect on community

structure of vegetative succession, species interactions and resiliency of populations (Swihart and Slade 1990).

Similar assertions were made by researchers studying abundance patterns of Elliot's short-tailed shrew (*Blarina hylophaga*) over eighteen years. As was the case in my study, short term studies often are unable to determine possible mechanisms that affect variations in abundance. Long-term studies are necessary to root out the characteristics of the habitat or populations that impact abundance (Matlack et al. 2002). Another benefit of long-term studies is their ability include both upper and lower levels of abundance throughout multiple seasons and a multitude of extreme weather events (Matlack et al. 2002) the effects of which may not be possible to observe over a short time frame.

Populations in fragmented landscapes such as these prairie remnants in Missouri are also at greater risk of extinction due to limited gene flow (Shirk et al. 2010). Animal populations existing within fragmented landscapes are subject to reduced gene flow and loss of genetic diversity, eventually leading to heightened risk of extinction (Shirk et al. 2010). Inbreeding, fixation of deleterious alleles, and reduction of potential adaptability lead extinction of species in such landscapes (Shirk et al. 2010).

Negative impacts of landscape fragmentation can be combated by wildlife linkages and corridors (Bier et al. 2008). These linkages support movement of wildlife from patch to patch when the areas they inhabit become inhospitable for movement and multiple barriers, such as roads or agriculture, are restricting migration. Bier et al. uses the term linkage to describe connective land intended to promote movement of multiple focal species or propagation of ecosystem processes. They define a corridor as a swath of land intended to allow passage of a

particular wildlife species between 2 or more wildland areas. The ideal wildlife linkage would be suitable for movement of multiple native focal species (Bier et al. 2008).

The designers of these linkages also keep in mind the goals of stakeholders and landscape analysts use an algorithm that tests the resistance of the surrounding area to each focal species. The importance of transparency in linkage design is also very important, and conservation biologists should be honest with stakeholders about uncertainties and assumptions that go with linkage design (Bier et al. 2008).

More studies over population dynamics in fragmented habitats are necessary to species conservation. Specifically, mammals, who often inhabit landscapes that have been fragmented and where natural processes involved in upkeep of metapopulation structure are crucial to population persistence (Olivier et al. 2009). Understanding populations within these communities and how they are affected by landscape fragmentation should be considered by in conservation planning and management of such sensitive habitats (Olivier et al. 2009).

Michael Soule defines conservation biology as the biology of species, communities, and ecosystems that are perturbed, either directly or indirectly, by human activities or other agents. The goal of conservation biology and the reason for studies such as this are carried out with the purpose of proposing and testing principles that contribute to the furtherance of biological diversity through scientific research. Soule states, “Although crisis-oriented, conservation biology is concerned with the long-term viability of whole systems (Soule 1985).”

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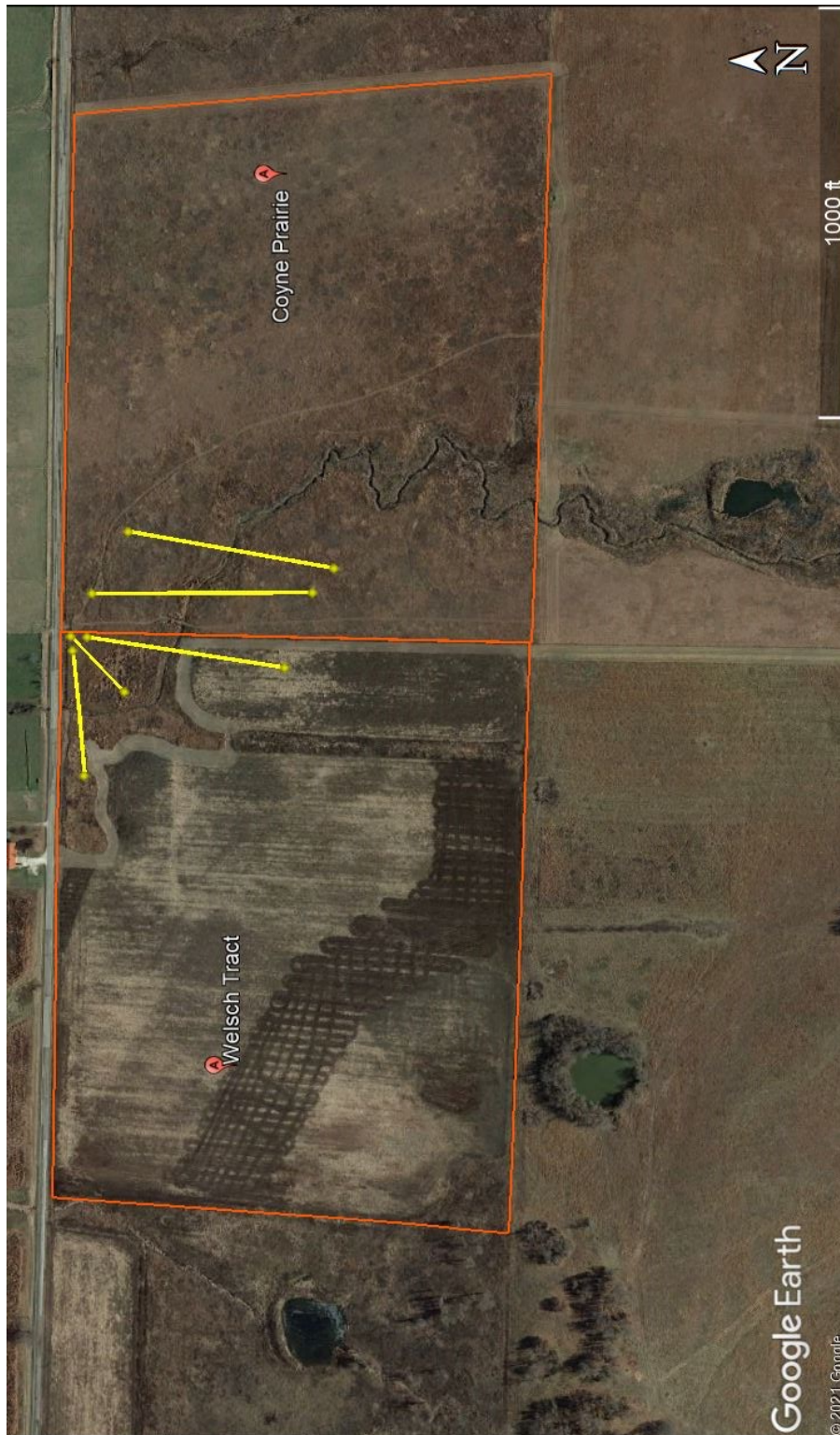
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APPENDICES

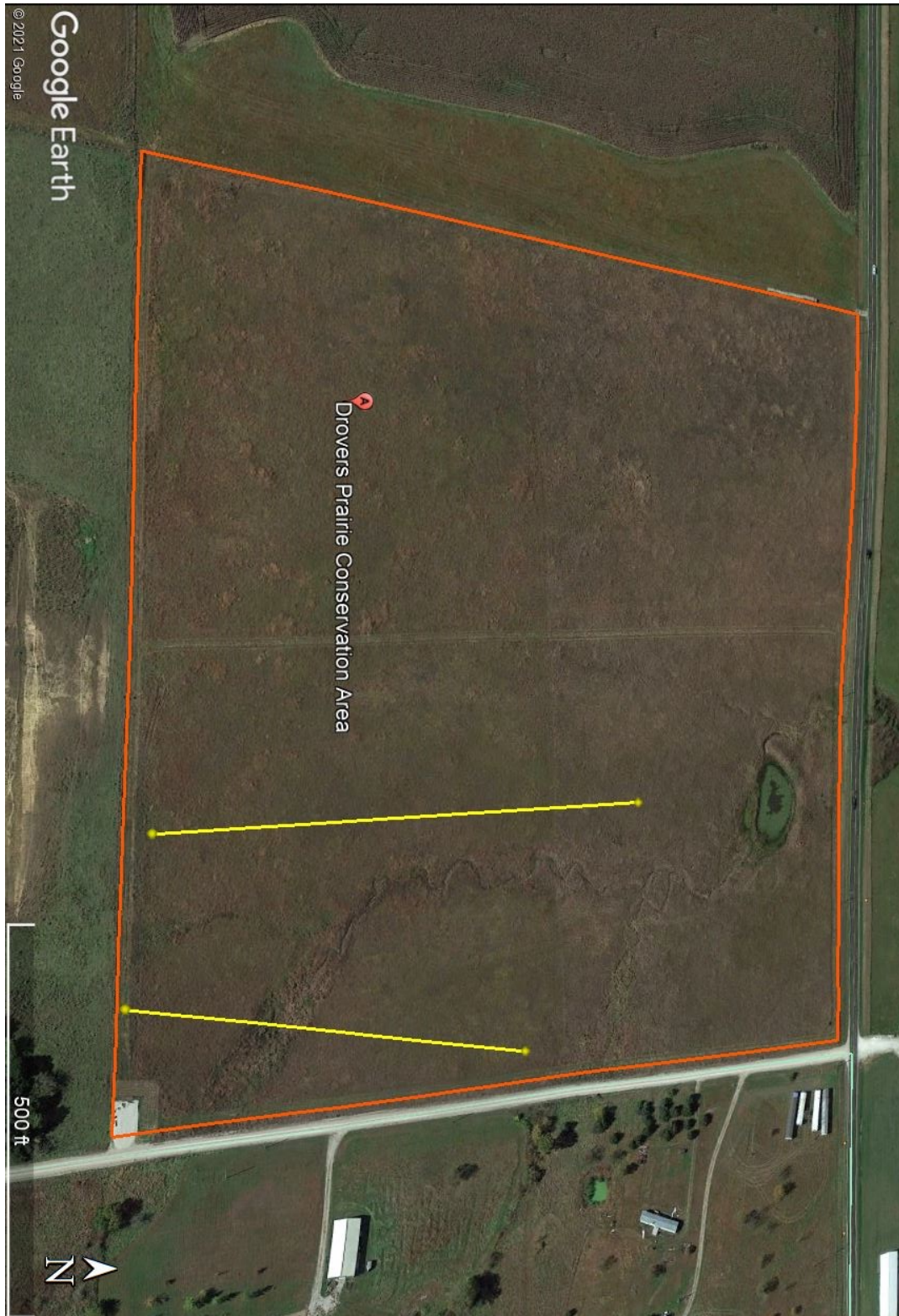
Appendix A. Geographic coordinates of the beginning and end of trap lines for each site.

	Start A	End A	Start B	End B
Linden's	37.1048, -93.9078	37.1037, -93.9078	37.1065, -93.907	37.1077, -93.9048
Golden	37.365, -94.1459	37.3646, -94.2275	37.3617, -94.1465	37.361, -94.149
Dorris	37.3821, -94.2286	37.3821, -94.2285	37.3834, -94.2286	37.3851, -94.2275
Pleasant	37.6485, -94.3461	37.6503, -94.3462	37.6486, -94.3469	37.6506, -94.347
Dennison	37.6473, -94.3459	37.6454, -94.3469	37.6483, -94.3591	37.6465, -94.3587
Lattner	37.6486, -94.3592	37.6499, -94.3604	37.6486, -94.3591	37.6504, -94.3583
Coyne	37.507, -93.9949	37.5053, -93.9952	37.5067, -93.9943	37.5051, -93.9947
Welsch	37.5071, -93.9953	37.5055, -93.9956	37.5072, -93.9954	37.5068, -93.9958
Drovers	38.5318, -93.2913	38.5335, -93.2908	38.5319, -93.2921	38.5341, -93.2921
Friendly	38.5506, -93.2989	38.5505, -93.2964	38.5512, -93.2988	38.5509, -93.2962
Unnamed	37.0412, -94.1502	37.0409, -94.148	37.041, -94.1501	37.0389, -94.1496

Appendix B. Map of Coyne Prairie and Welsch Tract. Orange lines denote property lines. Yellow lines denote trap lines for each site. Names of prairies are stated in each map.



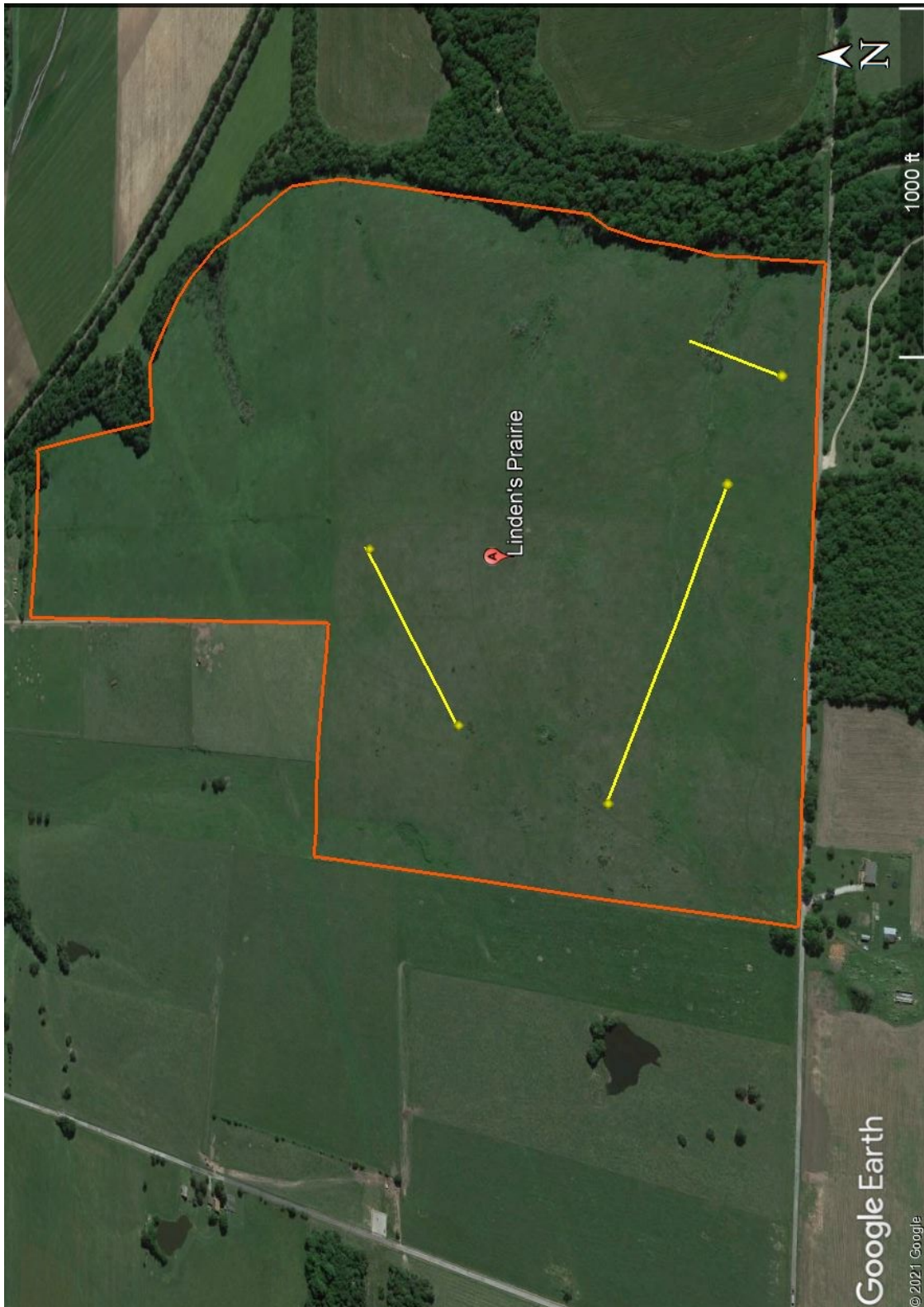
Appendix B-1. Map of Drover's Prairie Conservation Area. Orange lines denote property lines. Yellow lines denote trap lines.



Appendix B-2. Map of Dorris Creek Prairie Conservation Area. Orange lines denote property lines. Yellow lines denote trap lines.



Appendix B-3. Map of Linden's Prairie. Orange lines denote property lines. Yellow lines denote trap lines.



Appendix B-4. Map of Golden Prairie. Orange lines denote property lines. Yellow lines denote trap lines.



Appendix B-5. Map of Lattner Prairie, Denison Prairie, and Pleasant Run Creek Prairie. Orange lines denote property lines. Yellow lines denote trap lines.



Appendix B-6. Map of Unnamed Prairie. Orange lines denote property lines. Yellow lines denote trap lines.



Appendix B-7. Map of Friendly Prairie. Orange lines denote property lines. Yellow lines denote trap lines.



Appendix C. IACUC approval form.

Application to Use Live Vertebrate Animals	PI:	Sean P Maher	Page: 1 of 9
	Dept:	Biology	
	IACUC ID:	18-029.0	Web ID: 423

Title: Comparing host and ectoparasite communities across environmental gradients		Office Use Only	
Species:	Rodents, Soricomorphs, Lagomorphs	IACUC ID:	18-029.0-A
Application Type:	Continuation	Renewal Date:	05/2021
Multiple Species	Yes - I will be quantifying mammal communities and their ectoparasite communities, and captures may include a number of rodents, shrews, mole, rabbits, and hares.		
Total Animal Number: 1200 (Non-ORC - Caught in Wild)			

Yes 4.1 REQUIRED - Check this box in order to access Section 4.1, Alternatives to Proposed Procedures. Failure to check this box may result in protocol review delays.

Submission History for Continuation:
04/30/2018 - Submitted 04/30/2018 - Under Review 05/08/2018 - Reopened 05/14/2018 - Revised 05/18/2018 - Under Review 05/31/2018 - Approved 05/31/2018 - Complete 05/30/2021 - Renewal Date

Approval Date: 5/31/2018

ADDENDUM 2

PROTOCOL AMENDMENT REQUEST

This form is intended to be filled out if changes to ongoing, previously approved research or teaching activities involving animals are needed. Examples of changes include: title, funding, expiration date, animal species and numbers, procedures, personnel, etc. Protocol Amendment Requests should be submitted electronically to IACUC@missouristate.edu.

Office Use Only	Protocol Number:	Date of IACUC/DMR Review:	
	Date Filed:		

A. Investigator Information			
Principal Investigator: Sean Maher	Department: Biology	Office Address: 269 Temple	
Office Phone: 6-6916	Emergency Phone: 785-393-8272	Email: spmaher@missouristate.edu	
B. Project Information			
Original Approval Date: 05/31/2018		Current Expiration Date: 05/30/2021	
Project Title: Comparing host and ectoparasite communities across environmental gradients			
C. Amendment(s)			
Check the appropriate box(es) indicating the requested amendment(s) and provide information as needed.			
TITLE			
<input type="checkbox"/> Change Title To:			
FUNDING			
<input type="checkbox"/> Add Funding <input type="checkbox"/> Remove Funding			
Funding Source:			
EXPIRATION DATE			
<input type="checkbox"/> Change Project Expiration Date To:			
PERSONNEL			
<input checked="" type="checkbox"/> Add personnel			
Name	Title/Position	Degrees	Training/Experience
Morgan Rodery	Graduate Student	BS	None
<input checked="" type="checkbox"/> Remove personnel			
Name		Title/Position	
Casey Adkins		Graduate Student	
ANIMAL INFORMATION			
<input type="checkbox"/> Remove animal species previously approved:			
<input type="checkbox"/> Add animal species not previously approved <i>(Fill in the rest of ANIMAL INFORMATION section)</i>			
<input type="checkbox"/> Request additional animals previously approved <i>(Fill in the rest of ANIMAL INFORMATION section)</i>			
Species	Common Name	Approximate Number	Source
Justification for additional animals or species:			

Pain or Distress				
Species	Category B	Category C	Category D	Category E*

***Justification for Pain Category E:**

PROCEDURE(S)

☐ Add Procedure ☐ Remove Procedure ☐ Modify Procedure

Please describe, in detail, the procedural change(s) requested. The following topics should be included, as applicable: necessity of change, species involved, number of animals involved, treatments, drugs, euthanasia, etc.

USE OF HAZARDOUS MATERIAL(S)

List hazardous materials to be used on this project. Materials can be harmful to humans, animals, or both. It is the PI's responsibility to have, up-to-date Material Safety Data Sheets (MSDS) for materials included on this form. Some materials (e.g., radioactive materials, rDNA and biohazardous materials) require additional institutional approval; contact the Office of Research Compliance for more information.

Material(s):

D. Literature Search

Searches should provide justification of the [three "R's"](#). A new search using the same keywords from the original protocol may be sufficient. **If you completed the PROCEDURE(S) or ANIMAL INFORMATION section(s), please complete a literature search.**

Date of Search	Keywords	Resources Used	Years Covered

Results: Summarize how the search indicates necessity of an animal model, lack of duplication or the need to repeat previous studies.

Sean Maher
Principal Investigator

4 September 2019
Date