MODELING THE POTENTIAL FOR GREATER PRAIRIE-CHICKEN AND FRANKLIN'S GROUND SQUIRREL REINTRODUCTION TO AN INDIANA TALLGRASS PRAIRIE

by

Zachary Travis Finn

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THE PURDUE UNIVERSITY GRADUATE SCHOOL
STATEMENT OF COMMITTEE APPROVAL

Dr. John Dunning, Jr., Chair

Department of Forestry and Natural Resources

Dr. Patrick Zollner

Department of Forestry and Natural Resources

Dr. Elizabeth Flaherty

Department of Forestry and Natural Resources

Dr. Jennifer Duggan

Department of Applied Environmental Science
California State University at Monterey Bay

Approved by:

Dr. Robert Wagner

Dedicated to my wife, Sarah, and my son, Ezra. You keep me going.

Also dedicated to nature. Your restoration is my passion.

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ABSTRACT

Greater prairie-chickens (*Tympanuchus cupido pinnatus*; GPC) have declined throughout large areas in the eastern portion of their range. I used species distribution modeling to predict most appropriate areas of translocation of GPC in and around Kankakee Sands, a tallgrass prairie in northwest Indiana, USA. I used MaxEnt for modelling the predictions based on relevant environmental predictors along with occurrence points of 54 known lek sites. I created four models inspired by Hovick et al. (2015): Universal, Environmental, Anthropogenic-Landcover, and Anthropogenic-MODIS. The Universal, Environmental, and Anthropogenic-MODIS models possessed passable AUC scores with low omission error rates. However, only the Universal model performed better than the null model according to binomial testing. I created maps of all models with passing AUC scores along with an overlay map displaying the highest predictions across all passing models. MaxEnt predicted high relative likelihoods of occurrence for the entirety of Kankakee Sands and many areas in the nearby landscape, including the surrounding agricultural matrix. With implementation of some management suggestions and potential cooperation with local farmers, GPC translocation to the area appears plausible.

Franklin's ground squirrels (*Poliocitellus franklinii*; FGS) have declined throughout a large portion of the eastern periphery of their range. Because of this, The Nature Conservancy is interested in establishing a new population of these animals via translocation. The area of interest is tallgrass prairie in northwest Indiana, USA: Kankakee Sands and the surrounding landscape. Species distribution modelling can help identify areas that are suitable for translocation. I used MaxEnt, relevant environmental variables, and 44 known occurrence points to model the potential for translocation of FGS to Kankakee Sands and the surrounding area. I created four models inspired by Hovick et al. (2015): Universal, Environmental, Anthropogenic-Landcover, and Anthropogenic-MODIS. I created maps of models with passing AUC scores. The final map was an overlay map displaying the highest relative likelihood of occurrence predictions for the area in all passing models. Only the Universal and Anthropogenic-MODIS models had passable AUC scores. Both had acceptable omission error rates. However, none of the models performed better than the null model ($p < 0.05$). MaxEnt predicted that a few areas in and outside of Kankakee Sands possess high relative likelihoods of occurrence of FGS in both the Universal and Anthropogenic-MODIS models. However, MaxEnt predicted high relative likelihoods in the

surrounding agricultural matrix in the Universal Model. FGS prefer to cross through agricultural areas via unmowed roadside instead of open fields (Duggan et al. 2011). Because of this, high predictions in agricultural matrices in the Universal model are irrelevant. High relative likelihood predictions for linear sections that are obviously roads are disregardable in the context of my modeling efforts. Because of my low sample size, none of the models are really reliable in predicting relative likelihoods of occurrence for this area. Despite high relative likelihood predictions, the appropriateness of a translocation effort to the area is inconclusive.

CHAPTER 1. THE IMPORTANCE OF ANIMAL REINTRODUCTIONS IN THE CONTEXT OF THE GLOBAL BIODIVERSITY CRISIS: AN INTRODUCTION

1.1 Global biodiversity crisis

The global biodiversity crisis is the term for the ongoing loss of species at a rate faster than biologists can describe the species being lost (Singh 2002). The main drivers of this rapid species decline are residential and commercial development, agriculture and aquaculture, energy production and mining, transportation and service corridors, biological resource use, human intrusion and disturbance, natural systems modification, invasive species, loss of genetic diversity, diseases, pollution, and climate change (Driscoll et al. 2018). These causes are largely driven by an increasing human population and corresponding resource use (Driscoll et al. 2018). To slow or even stop and reverse the trend of species loss, especially related to habitat and genetic diversity loss, biologists now have opportunities to restore species to areas where they have once lived but no longer do. The opportunity is known as species reintroduction.

1.2 Regional biodiversity loss

For this thesis, the area of interest is the northeast Illinois and northwest Indiana. This area is part of the eastern periphery of what was once a large expanse of the historical range of the tallgrass prairie. The tallgrass prairie habitat type has largely been converted to row crop agriculture. As a result, numerous species have declined due to the wide scale loss of habitat. Due to this historical and recent habitat restoration efforts, there is potential to reintroduce species to portions of their historic ranges.

1.3 Animal reintroductions

The International Union for the Conservation of Nature (IUCN) defines reintroduction, in their “Guidelines for Reintroductions and Other Conservation Translocations,” as “the intentional movement and release of an organism inside its indigenous range from which it has disappeared” (IUCN 2016). According to these guidelines, reintroduction becomes an acceptable conservation method when the conservation benefits outweigh the ecological, social, and economic risks (IUCN

2016). Furthermore, reintroductions are one method to reverse the global biodiversity crisis by "undoing" extirpations. However, when the planning stage of a reintroduction effort has been reached, after the definition of reintroduction goals, feasibility assessments must be performed (IUCN 2016).

1.4 Modeling approach

To model habitat suitability, I developed maximum entropy (MaxEnt) models. MaxEnt modeling is a presence-only modeling approach that uses known location sample data for the species of interest without requiring on difficult- and expensive-to-gather species absence data against a background of biologically-relevant environmental data variables (Hovick et al. 2015, Philips and Dudík 2008). Instead, MaxEnt generates its own 10,000 "pseudo-absence" points to test against the location data input for a given species (Philips and Dudík 2008).

Because MaxEnt does not rely on true absence data, and instead generates pseudo-absence points, one cannot interpret the results as an estimation of true probability of occupancy relative to relevant environmental variables (i.e. "habitat suitability") for a given modeled species (Guillera-Arroita et al. 2014, Merow and Silander Jr. 2013, Perkins-Taylor and Fray 2020). Instead, MaxEnt generates results of "relative likelihoods of occupancy" for modeled species, which are still useful due to direct proportionality to true occupancy probabilities. The MaxEnt program generates model results that include area-under-the-curve (AUC) scores, omission error rates with regards to presence data, one-tailed binomial testing to evaluate model performance, and graphical outputs that display predicted relative likelihoods of occupancy over the desired study area. Finally, these graphical outputs display predictions on a 0 – 1 scale, where zero means least likely for the species to occupy a given area.

CHAPTER 2. GREATER PRAIRIE-CHICKEN

2.1 Introduction

The International Union for the Conservation of Nature (IUCN) defines reintroduction, in their “Guidelines for Reintroductions and Other Conservation Translocations,” as “the intentional movement and release of an organism inside its indigenous range from which it has disappeared” (IUCN 2016). According to these guidelines, reintroduction becomes an acceptable conservation method when the conservation benefits outweigh the ecological, social, and economic risks. When biologists and land managers reach the planning stage of a reintroduction effort, after defining specific goals, biologists must perform feasibility assessments (IUCN 2016). I performed the habitat suitability inspection portion of a feasibility assessment relative to the greater prairie-chicken (*Tympanuchus cupido pinnatus*; GPC) within 3,400 ha restored tallgrass prairie property in northwestern Indiana, USA, known as Kankakee Sands.

Information imperative to any reintroduction effort, and important for population managers includes: species status, cause(s) of decline, habitat characteristics (among the strongest factors in determining reintroduction success), diet, foraging behavior, morphology, behavior (defensive, social, and reproductive), and reproduction itself (Gedir et al. 2004). Moreover, assessors should understand what threatens the species (including threats of anthropogenic or natural origination), species mobility (especially in regard to individuals abandoning the release group/area), and the impact of exotic species on the target species (Short 2009; Appendix A). Anthropogenic factors are also significant determinants of habitat quality. GPC are sensitive to power lines and poles as they provide perching sites for predatory raptors (Robb and Schroeder 2005). Additionally, wind turbine presence is thought to negatively affect lek site selection (Hovick et al. 2015). Habitat information is also important in identifying predictor variables in Species Distribution Modeling (SDM).

2.1.1 Species Distribution Modeling

SDM is a method of modeling potential distributions based on ecological niche models (Hovick et al. 2015). Others used SDMs for predicting the spread of invasive species, climatic effects on species distribution, and niche-based presence, management of threatened species, phylogeographic patterns, and management of landscapes (Guisan and Thuiller 2005, Guillera-Arroita et al. 2015, Mutascio et al. 2018). SDMs are useful for identifying areas of interest for species-specific conservation action, including areas for reintroductions of endangered species (Guillera-Arroita et al. 2015, Maes et al. 2019, Westwood et al. 2020). These models are based on known location data with natural and anthropogenic land use data (Rodríguez et al. 2007, Hovick et al. 2015). Moreover, these models predict species occurrence based on habitat suitability related to relative likelihoods comparable to true occurrence probabilities (Guillera-Arroita et al. 2015). I developed several SDM models using maximum-entropy methods based on environmental variables that are associated with suitable GPC habitat and other anthropogenic variables associated with avoidance (Philips et al. 2004, Hovick et al. 2015).

Maximum-entropy techniques use sample location data and environmental variables to estimate the distribution closest to uniform (maximum entropy) under the constraint of the empirical averages of said environmental variables (Philips et al. 2004). Moreover, between 2006 and 2013, MaxEnt was used in over 1,000 publications (Merow et al. 2013). Merow et al. (2013) offers 2 reasons for the volume of use. The first is that MaxEnt often outperforms other programs/methods based on predictive accuracy and its ease-of-use. The second is that others used and validated MaxEnt over a wide range of species and habitats (Philips and Dudík 2008). An additional reason is that MaxEnt works well even with small datasets (Elith et al. 2006).

2.1.2 Logistical and Operational Constraints of Species Reintroductions via Translocation

Multiple logistical and operational constraints exist relative to species translocations, including: locations of source populations, required federal and state permits, facilities that could and are willing to house a captive source, release strategies, and genetic concerns. John Shuey (Director of Conservation Science with The Nature Conservancy [TNC]), recommended I collect this information in order to facilitate a potential future translocation for this species (J. Shuey, TNC, personal communication). This information is located in Appendix B.1.

2.1.3 Objectives and Hypothesis

I chose to pursue this project at the landscape scale (multiple states) because the rarity and decline of this species highlights the need to obtain location data from more than one state. I shared state-specific results with the DNRs or similar organizations of the state that have provided the observation data. However, the purpose of this study was to model results specific to Kankakee Sands and the surrounding landscape of northwest Indiana and northeast Illinois to identify areas suitable for GPC leks for a potential future reintroduction effort. Specific objectives were to:

1. Predict GPC habitat suitability values for Kankakee Sands and the surrounding landscape using spatial data with pixelated, binary (presence or absence) output format in a map.
2. Provide logistical and operational information for TNC in regard to a GPC reintroduction to the area regardless of the results.

Our prediction and hypothesis is that the majority of the Kankakee Sands property and surrounding landscape will be suitable (suitability values $\geq$ 0.5; arbitrary cutoff) to reintroduce GPC because of the availability of thousands of hectares of potential habitat relative to the recommended minimum habitat area for a population of GPC in Illinois being 610 ha (Svedarsky et al. 2003).

2.2 Methods

2.2.1 Study Area

This study incorporated data from Illinois, Wisconsin, Iowa, and Missouri, USA, to create a habitat suitability prediction for the GPC at Kankakee Sands, a 3,400 ha restored tallgrass prairie and wetland site, owned by TNC (TNC; Fig. 2.1). Kankakee Sands is located approximately 8 km north of Morocco, Indiana in Newton County. As designated by TNC, Units E and I are currently/were recently under agricultural leasing (Fig. 2.1). TNC returned Unit E to tallgrass prairie in fall 2020 and will restore Unit I sometime in 2021 (Nyberg, TNC, personal communication). All other units are tallgrass prairie with some wetlands and creeks in Units D and G. Kankakee Sands is also a part of a larger collection of tallgrass prairie properties in northwest Indiana and northeast Illinois (Fig. 2.2).

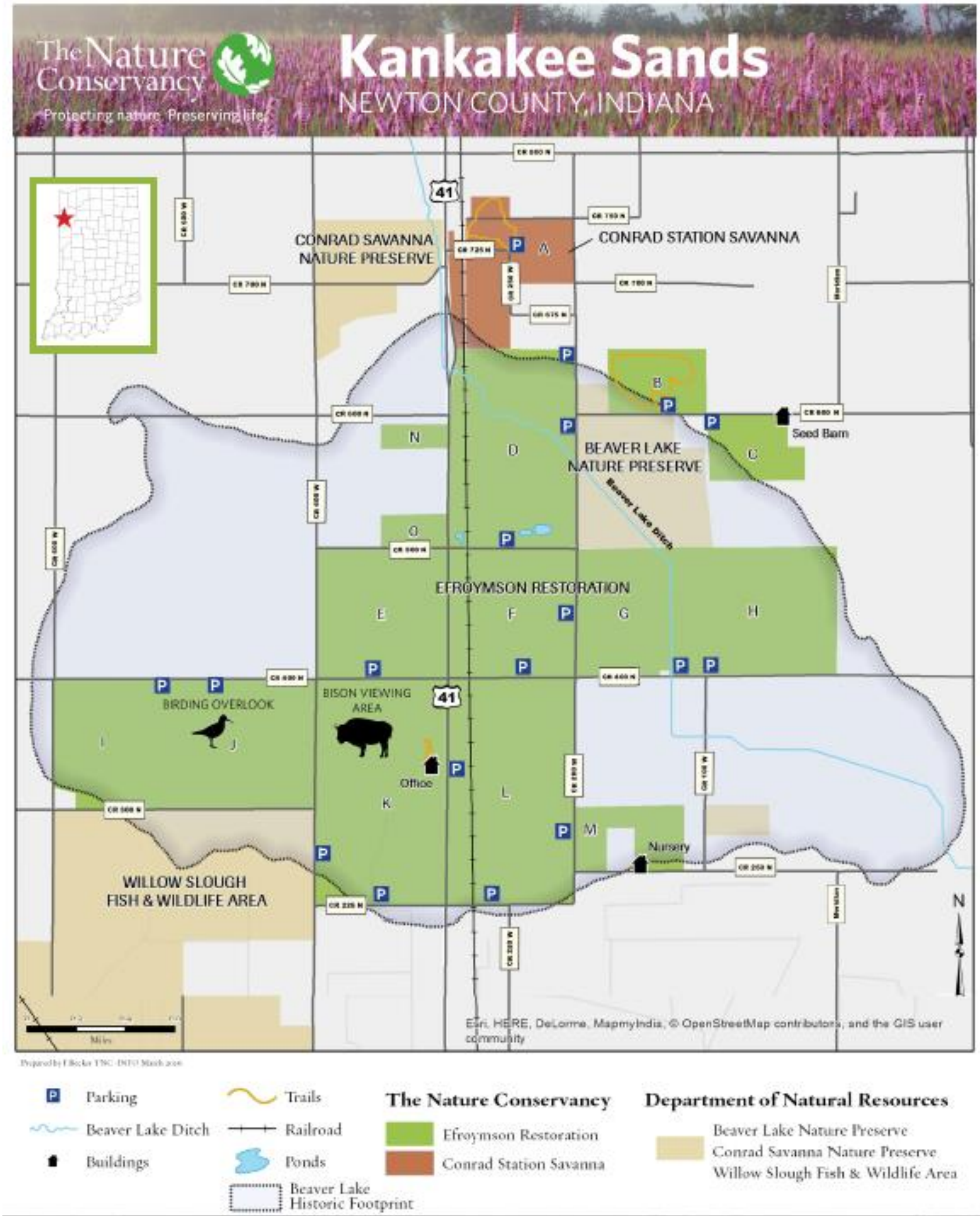

Figure 2.1. Map of Kankakee Sands in northwest Indiana, USA created by Becker (2016), owned by The Nature Conservancy. This property could become a translocation area for an Indiana extirpated species, the Greater Prairie-Chickens (*Tympanuchus cupido*). Noteworthy are the roads dividing each section, the railroad line in the middle, and the bison grazing pasture. To the right of the bison grazing pasture is a major highway (US-41). The letters are the unit labels.

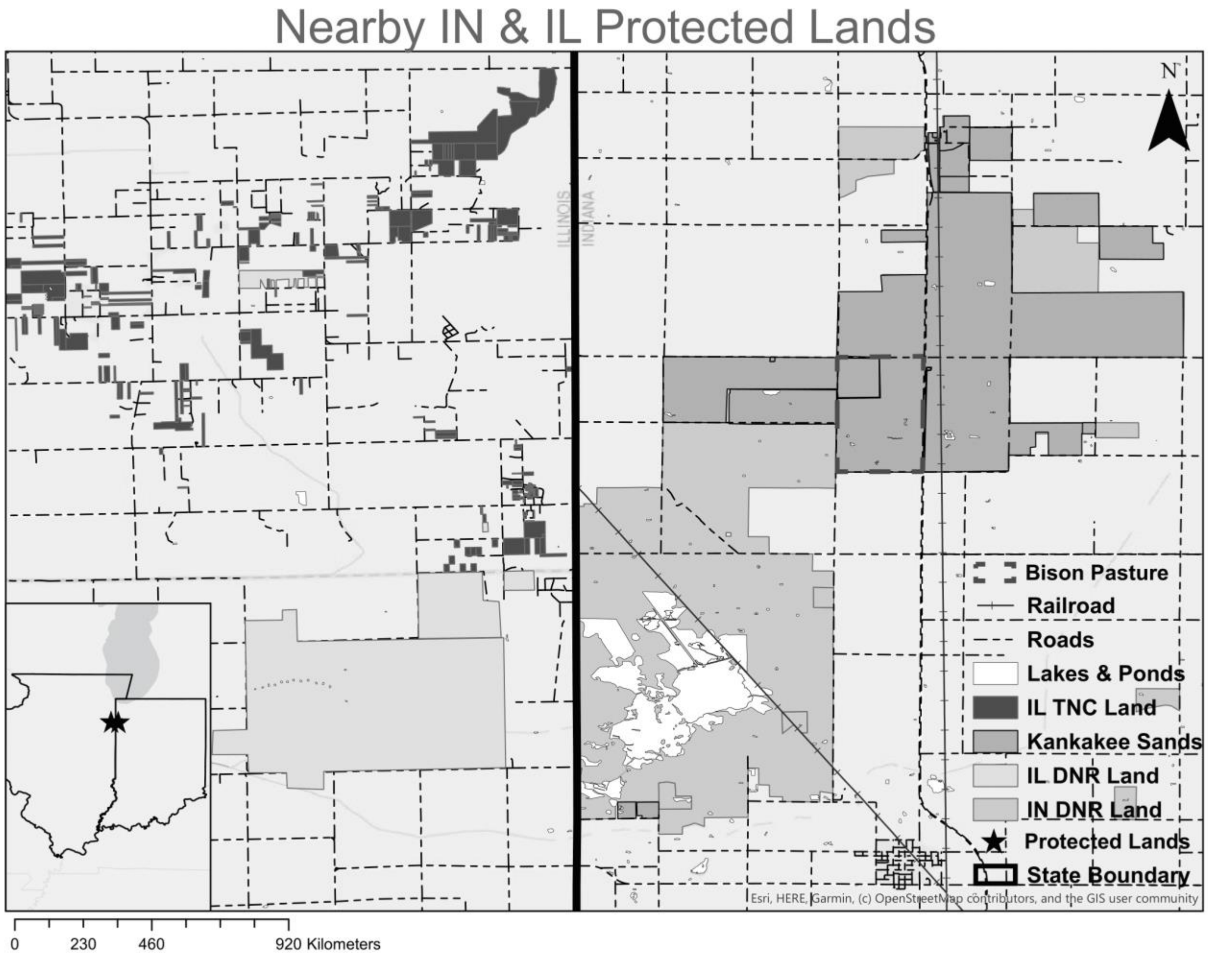

Figure 2.2. Map of the tallgrass prairie properties near and including Kankakee Sands. This landscape could become the core of a translocation effort for the Greater Prairie-Chicken (*Tympanuchus cupido*). Notice the state border between Indiana and Illinois, USA. All DNR and The Nature Conservancy boundaries are from Becker (2018).

Kankakee Sands, located in northwest Indiana, was flattened during the Wisconsin glaciation event. It was also part of the Grand Kankakee Marsh, located in the footprint of what was once Beaver Lake (TNC 2020). Farmers drained the site and converted it to agricultural fields in 1873 (Schmal 2005). TNC acquired the property in 1997, when they began their ongoing work to restore the area to tallgrass prairie and wetland habitats. There is an abandoned railroad track and parallel Indiana highway going through the property from north to south. There are also multiple graveled county roads arrayed east-west that divide the property into its multiple sections. TNC administers prescribed burns and invasive species removals across the property (TNC 2020).

The prairie portion of Kankakee Sands hosts flora including prairie dropseed (*Sporobolus clandestinus*), junegrass (*Koeleria cristata*), and sedges (Carex spp.; Homoya et al. 1985). The wetland portion features bluejoint grass (*Calamagrostis canadensis*), common reed (*Phragmites communis),* arrowheads (*Sagittaria* spp.), and sedges (both *Scirpus* spp. and *Carex* spp.; Homoya et al. 1985). Invasive horsetail reeds (*Equisetum* spp.) are present in large areas as well (ZF, personal observation). The property also provides habitat for mammalian fauna such as American bison (*Bison bison*), coyote (*Canis latrans*), plains pocket gopher (*Geomys bursarius*), and eastern cottontail (*Sylvilagus floridanus*). Avian fauna includes introduced ring-necked pheasant and 227 other species including waterfowl and raptors (Barnes 1985, eBird 2020, TNC 2020). Reptile species include ornate box turtle (*Terrapene ornata*), eastern hognose snake (*Heterodon platirhinos*), and bull snake (*Pituophis melanoleucus*; Homoya et al. 1985).

Nearby Renssalaer, Indiana (28 km southeast of Kankakee Sands) experiences an average annual temperature of 10.2° C (U.S. Climate Data 2007-2019). Annual snowfall ranges 0.25-22.6 cm/month (Western Regional Climate Center 2020). Annual rainfall ranges from 2.5-10.2 cm/month (World Climate 2020). The soils of Kankakee Sands are primarily Mollisols, Entisols, and Histosols (Soil Survey Staff 2017).

2.2.2 Data Collection and Analysis

GPC Lek Locations

I obtained GPC lek location data from the natural heritage database programs of the natural resource departments in the states of Illinois, Iowa, Missouri, and Wisconsin. These natural heritage programs are all part of the NatureServe network of natural heritage databases (NatureServe 2014). I requested GPC lek location data from the Illinois Department of Natural Resources (DNR; Illinois Department of Natural Resources 2020), Wisconsin DNR (Wisconsin Department of Natural Resources, Wisconsin Natural Heritage Program 2020), Iowa DNR (Iowa Department of Natural Resources 2021), and the Missouri Department of Conservation (Missouri Department of Conservation 2021) in November, 2020. I received the final data in February 2021. All data were provided as-is (i.e. not updatable) based on the time of request according to the natural heritage databases' data license agreements. The states collected these data through annual breeding bird surveys, scientific literature, and incidental, biologist-sourced or biologist-approved observations. While Illinois, as Indiana's closest neighbor, has a few GPC leks, that state did not have enough GPC lek sites for this project. Therefore, I incorporated data from additional states to increase my sample size. GPC lek sites are a useful habitat use metric because lek sites are associated with other GPC habitats year-round including for nest sites and brood rearing (Hamerstrom, Jr. and Hamerstrom 1973, Schroeder and White 1993, Hovick et al. 2015).

To reduce errors from land use change over time, I only used points that were from 2010–2021. Modeling errors (i.e., translating and applying results to the real-world) would result from using lek points considered "too old" and located in non-habitat due to land-use conversion over time. I considered polygon and line data I received, in addition to point data, as non-point representations of the lek location data. Using ArcGIS, I converted these polygons and lines to point values. I also removed two points positioned in the center of a road because of assumed GPS error.

Spatial Autocorrelation Reduction

To assess the role of spatial autocorrelation, I used Program R (v. 4.1.0, R Core Team, Vienna, Austria) along with the packages "raster" (Hijmans 2021), "elsa" (Naimi et al. 2019), and "usdm" (Naimi 2017). I used packages "raster" combined with "elsa" to create correlograms to determine the distance at which Moran's I was equal to zero for several of the data layers. Moran's I coefficients are a measure of relatedness of neighboring cells at various distances (Diniz-Filho et al. 2003). Their derivative correlograms are useful for testing potential spatial filtering distances at which to consider spatial autocorrelation negligible when the Moran's I value is zero for a given distance (Diniz-Filho and Bini 2005). I used preliminary model runs to identify the most influential, and likely to be spatially-autocorrelated, layers from respective models. These layers were MODIS and landcover. Both had a Moran's I of zero at 325 km, which left two GPC lek points in the five-state area after filtering.

Semivariograms are another useful way to identify spatial autocorrelation-reducing spatial filtering distances (Griffith 2005). Semivariograms are the inverse of correlograms (Getis 2010). I used the "raster" and "usdm" packages in R to create semivariograms for the MODIS and landcover layers. I tested these layers out to a cutoff distance of 400 km in an attempt to find the distance at which a sill occurs. Sills occur at large distances where the semivariance value equals the variance of the data layer (Griffith 2005). The resulting plotted semivariogram did not have a sill out to 400 km. Because these filtering distances were as large as they were, I was unable to account for spatial autocorrelation in my models. Instead, I used another distance value for spatial filtering to preserve data volume.

Spatial Filtering

To help ensure independence of data points, I used 1.52 km as a spatial filter for minimum distance between distinct leks. This distance is the median value of a range of mean observed distances between leks in Illinois (range: 1.33–1.71 km; Gillespie 2016). To filter out data points within this minimum distance, I used the Buffer tool in ArcGIS Pro to create circles around point values based on a specified input radius. I manually removed data points until no buffer circles overlapped while trying to preserve the greatest amount of data points.

Landcover and Anthropogenic GIS Layers

For the states of Indiana, Illinois, Wisconsin, Iowa, and Missouri, I collected digital landcover and anthropogenic geographic information system (GIS) data layers from publicly-available internet sources. I retrieved most of the data layers for the above states from the Geospatial Data Gateway provided by the Natural Resources Conservation Service (NRCS 2020). These data layers include: NRCS States by State, Cropland Data Layer by State, NRCS Conservation Easement Areas by State, National Elevation Dataset 30 Meter, TIGER Primary and Secondary Roads, and TIGER Streets (NRCS 2020).

NRCS States by State provided state borders. Cropland Data Layer by State provided specific landcover information to identify potential habitat and non-habitat, in addition to cropland location information. This layer, updated annually since 1997, consisted of 134 land cover, crop cover, and cropland classes in 30 m pixels, and were current as of 2019 (USDA National Agricultural Statistics Service Cropland Data Layer 2020). The accuracy of the landcover is 85-95% for major crop categories and was based on Advanced Wide Field Sensor (AWiFS), Landsat Thematic Mapper (TM), Enhanced Thematic Mapper (ETM+), and Moderate Resolution Imaging Spectroradiometer (MODIS) satellite data (Boryan et al. 2011).

NRCS Conservation Easement Areas by State identified locations and types of conservation easements in the state that could provide additional habitat. I included the National

Elevation Dataset 30 Meter GIS layer because lek sites are typically located on elevated areas relative to the surrounding landscape (Robb and Schroeder 2005). I included TIGER Primary and Secondary Roads, along with TIGER Streets, because roads fragment habitat.

For an additional general landcover layer, I used the Application for Extracting and Exploring Analysis Ready Samples (AρρEEARS, Ver. 2.51, NASA EOSDIS Land Processes Distributed Active Archive Center, Sioux Falls, SD) to download a MODIS layer MOD13Q1 v006 (hereon "MODIS"; Didan 2015, Hovick et al. 2015). These satellite data included MODIS Normalized Difference Vegetation Index (NDVI), which provided spatial and temporal comparisons of vegetative cover in the blue, red, and near-infrared reflectance (Hovick et al. 2015).

I downloaded power transmission line data from Homeland Infrastructure Foundation-Level Data (HIFLD, 2020). I also included wind turbine location data from the U.S. Wind Turbine Database (USWTD 2020). These landscape characteristics were the covariates I identified as potentially significantly influencing model results. In order to highlight my areas of interest among model results, I requested boundary polygon layers from Becker (2018). These layers included boundaries for Kankakee Sands, along with TNC lands in and other protected areas in the surrounding landscape.

Data Layer Conversions

After downloading all GIS data, I used ArcGIS Pro (v. 2.6.0, Esri, Redlands, CA) to clip the power transmission line, railroad, and wind turbine layers to only include data from the states of interest. I then combined TIGER Primary and Secondary Roads and TIGER Streets into one Feature layer (hereon "Roads"). I ensured all GIS layers were in the same projection coordinate system (WGS 1984), except for roads, railroads, power transmission lines, and wind turbines (Albers Equal Area Conic; for more accurate density calculations, Hovick et al. 2015). I converted the Cropland Data Layer by State (hereon "Landcover"), NRCS Conservation Easement Areas by State (hereon "Conservation Easements") and National Elevation Dataset 30 Meter (hereon "Elevation"; NRCS 2020) feature layers into raster format using the Feature to Raster (Conversion) tool.

To ensure more accurate measurements, I changed the projection coordinate system of the new "Roads" layer, railroads, and power transmission lines to Albers equal area. I used the Near tool in ArcGIS Pro to calculate distance to the nearest power transmission line and road (respectively) relative to lek locations since power transmission lines and roads are potential sources of mortality and may affect lek site location selection. Using the Line Density tool, I created density raster layers from the linear polyline layers. I repeated this process for wind turbines, then used the Point Density tool to create a density raster layer for the wind turbines.

The Slope tool in ArcGIS Pro created a slope raster layer from the resampled elevation layer with degree values from 0–90 to measure maximum change between a certain cell and all 8 neighbor cells. I performed all of the above in accordance with the methods used in Hovick et al. (2015).

Instead of using the default 10,000 random points from MaxEnt, I created my own 10,000 pseudo-absence points using the command prompt and a bias file containing GPC habitat only (Philips 2017). I used custom points instead of the default because custom background points generated only in sampled habitat types focuses MaxEnt on differentiating between presence and background distributions instead of inherent sampling bias (Mutascio et al. 2018). To obtain a layer of only sampled habitat types for the bias file, I used the USDA CropScape tool (v. 2020, National Agricultural Statistical Service, Washington, D.C.) to extract only the Grass/Pasture landcover type. MaxEnt assumed that the data collectors for GPC lek site data did not sample every possible habitat patch in the five-state area (Fourcade et al. 2014). My bias file served to assess the sampling bias that resulted from the MaxEnt assumption of surveyors surveying only the known presence areas (Stolar and Nielsen 2015, Young et al. 2011).

To finish the use of ArcGIS Pro, I used the Raster to ASCII tool to ensure all data layers matched in extent (combined state boundaries) and resolution (same as MODIS, the variable with the coarsest resolution), which is a requirement for MaxEnt (Young et al. 2011). I removed all data columns from the attributes table of the observation data point before creating two new fields in the attribute table for longitude and latitude. Then I calculated the longitude and latitude geometry, in that order, for every animal observation point. I proceeded by converting the attribute table of the points layers to a .csv input file (Excel v. 2016, Microsoft, Redmond, WA).

MaxEnt for Data Analysis

To analyze the data, I used maximum entropy modeling (MaxEnt; v. 3.4.1, Philips et al. 2020) to create SDMs of lek site suitability across the landscape sampled based on the GIS layers above. An advantage of MaxEnt is that this program does not require known (and expensive) absence data. This is because MaxEnt uses 10,000 pseudo-absence points from individual background pixels (Hovick et al. 2015, Philips and Dudík 2008).

To determine model covariates and assess different parsimonious options, I used four models inspired by or also used in Hovick et al. (2015) to determine the influence of the above environmental variables on predicting lek site suitability. The first model included all of the data layers (Landcover, MODIS, Conservation Easements, Elevation, Slope, Roads, power transmission lines, and wind turbines). I referred to it as the "Universal model". The second model included only environmental factors (Table 2.1), called the "Environmental model". I called the third model "Anthropogenic-Landcover". It included anthropogenic layers (Table 2.1) along with the Landcover layer. I called the final model "Anthropogenic-MODIS". It featured the same anthropogenic layers and the MODIS layer. I created the additional models that did not include all variables based on known GPC behavior to develop more simplistic models, easily-interpreted by a wide variety of stakeholders (i.e. DNRs, farmers, land stewards, local politicians, etc.; Hovick et al. 2015).

Table 2.1. Data from IN, IL, IA, WI, and MO, USA involved in species distribution modeling to predict likelihood of presence (suitability) for Greater Prairie-Chicken (*Tympanuchus cupido*, GPC) at Kankakee Sands and the surrounding landscape in northwest IN and northeast IL.

Data	Explanation	Source(s)
GPC lek site/booming ground location	Animal data	IN, IL, IA, WI, and MO Natural History Inventories/Databases
NRCS State by State	State boundaries	Natural Resources Conservation Service (NRCS) Geospatial Data Gateway (GDS)
Cropland Data Layer by State	Detailed landcover	NRCS GDS
NRCS Conservation Easement Areas by State	Additional habitat	NRCS GDS
National Elevation Dataset 30 m	Influences lek site selection	NRCS GDS
Slope	Influences lek site selection	Created in ArcGIS Pro (Slope tool)
TIGER Primary and Secondary Roads	Fragments habitat; Inhibits metapopulation movements	NRCS GDS
TIGER Streets	Fragments habitat; Inhibits metapopulation movements	NRCS GDS
MOD13Q1 v006	Additional general landcover	Application for Extracting and Exploring Analysis Ready Samples (AppEEARS)
Power transmission lines	Provide raptor perch sites	Homeland Infrastructure Foundation-Level Data
Wind turbines	Negatively-associated with GPC presence	U.S. Wind Turbine Database

The modelling format I used is known as samples with data (SWD). SWD format creates models based on species sample locations and background points that contain environmental data values for the respective sets of data files. Users can then project these models in MaxEnt with

GIS environmental data to obtain pictures of model results across the geographic area of interest. This method is useful for reducing time running models of large environmental datasets (Philips 2017). Furthermore, Mutascio et al. (2018) found that SWD format performed better for bias correction than the regular MaxEnt method of using a bias file.

Setting adjustments included: 5,000 iterations, random seed, 10^-5 convergence threshold, regularization multiplier of three, and a cross-validation replicated run type (Mutascio et al. 2018). A regularization multiplier of three, instead of the default one, smoothed model output and reduced the risk of overfitting (Elith et al. 2011, Merow et al. 2013). Cross-validation is better for small data sets because it uses all data points to train and test the models (Elith et al. 2011, Merow et al. 2013, Philips 2017).

Model Evaluation

To evaluate the models, I used area-under-the-curve (AUC) test, omission error rate, and one-tailed binomial test. AUC measures the probability of a presence data point being ranked higher than a random absence point (Hovick et al. 2015). In the case of MaxEnt, the absence points are randomly-generated pseudo-absences, which I restricted the generation of using my bias file. Furthermore, an AUC score > 0.75 is considered useful for modeling purposes (Hovick et al. 2015).

MaxEnt uses suitability values from 0 to 1 and then converts these to binary predictions of present or absent to obtain omission error rates (Hovick et al. 2015). I obtained omission error rates (proportion of presences in testing data associated with areas of predicted absences) based on the 10th percentile training presence threshold from MaxEnt outputs. Using this threshold allowed for the omission of 10% of data points to account for potential errors (Hovick et al. 2015). Lastly, the one-tailed binomial tests evaluated whether the models predicted present lek locations better than random chance alone with the same area proportion predicted as "present" by the models (Hovick et al. 2015).

I concluded the MaxEnt analysis by creating maps of the "useful" model outputs in ArcGIS Pro, as well as a map showing overlapping model outputs, that show binary, present/absent (present = can occupy, values ≥ 0.5, absence = cannot occupy, values < 0.5) predictions for the raster cells in this area of interest. I also created a model overlay map. I converted the ASCII files for model replicate averages created by MaxEnt to a "fuzzy membership" raster type using the

Fuzzy Membership tool. In the Fuzzy Membership tool, I used the Linear membership type. I finished this portion by combining the fuzzy membership rasters for each model into an overlaid layer using the Fuzzy Overlay tool. In the fuzzy overlay tool I used the "Or" overlay type, which returned the maximum values out of the input raster sets.

Logistical and Operation Information Gathering

To obtain logistical and operational information for TNC as it pertains to the translocation of GPC to Kankakee Sands, I used published literature, web resources, email, and telecommunications (Appendix B). This information was requested by John Shuey, the Director of Conservation Science for TNC. He requested information on necessary federal and state-level permits, locations of source populations, facilities (i.e. zoological parks) that could and/or are willing to house a captive source, release strategies, and an overview of potential genetic concerns. This information is located in Appendix B.1.

2.3 Results

After spatial filtering, I used the remaining 54 data points from 2010–2020 in my modeling attempts. The Universal, Environmental, and Anthropogenic-MODIS models had useful (> 0.75) AUC values (Table 2.2). All passing models had an identical, low test omission error rate (0.0823; Table 2.2). MaxEnt's binomial tests indicated that the Universal model performed better than a random null model ($p = 0.0141$; Table 2.2). However, the Environmental and Anthropogenic-MODIS models did not perform better than the null ($p > 0.05$; Table 2.2). Because the Anthropogenic-Landcover model did not have a passing AUC score, I do not describe its specifics here (Table 2.3).

Table 2.2. Statistics for Greater Prairie-Chicken MaxEnt models of relative likelihood of occurrence at Kankakee Sands in northwest Indiana, USA. Omission error rates and p-values based on 10th percentile training presence training omissions as per Hovick et al. (2015).

	Mean Test AUC	Standard Deviation	Omission Error Rate	One-tailed Binomial Test P-value
Universal	0.892	0.057	0.0823	0.0141
Environmental	0.847	0.097	0.0823	0.1226
Anthropogenic-landcover	0.696	0.062	0.072	0.1848
Anthropogenic-MODIS	0.752	0.123	0.0823	0.2944

Table 2.3. Percent contributions of environmental variables for MaxEnt models predicting Greater Prairie-Chicken relative likelihood of occurrence at Kankakee Sands in northwest Indiana, USA based on state heritage program occurrence data.

	Elevation	Slope	Landcover	Cons. Ease.	MODIS	Transmission Line Density	Road Density	Wind Turbine Density
Universal	14.5	46.7	22.0	4.1	4.6	0.1	8.1	0
Environmental	19.2	65.5	0.4	9.0	5.9	-	-	-
Anthropogenic-landcover	-	-	78.4	-	-	0.1	21.6	0
Anthropogenic-MODIS	-	-	-	-	66.6	0.2	33.3	0

The best predictor for the Universal model was Slope (46.7%; Table 2.3). The second best predictor was Landcover (22.0%), followed by Elevation (14.5%; Table 2.3). The least informative predictors were Transmission Line Density (0.1%) and Wind Turbine Density (0%; Table 2.3).

For the Environmental model, the best predictor was again Slope (65.5%; Table 2.3). However, for this model the second best predictor was Elevation (19.2%), then Conservation Easements (9.0%; Table 2.3). The least informative variables were again Transmission Line Density (0.1%) and Wind Turbine Density (0%; Table 2.3).

In the Anthropogenic-MODIS model, the most informative predictor variable was MODIS (66.6%; Table 2.3). The second best predictor was Road Density (33.3%). The least informative predictor variables were again Transmission Line Density (0.2%) and Wind Turbine Density (0%; Table 2.3).

In ArcGIS Pro, I created potential distribution maps based on the respective model output rasters (Fig. 2.3, Fig. 2.4, Fig. 2.5). The results were a relative likelihood of occurrence based on a zero to one scale, with zero the least likely to support a GPC lek. The final overlay map indicates the areas of highest predictions based on all models (Fig. 2.6).

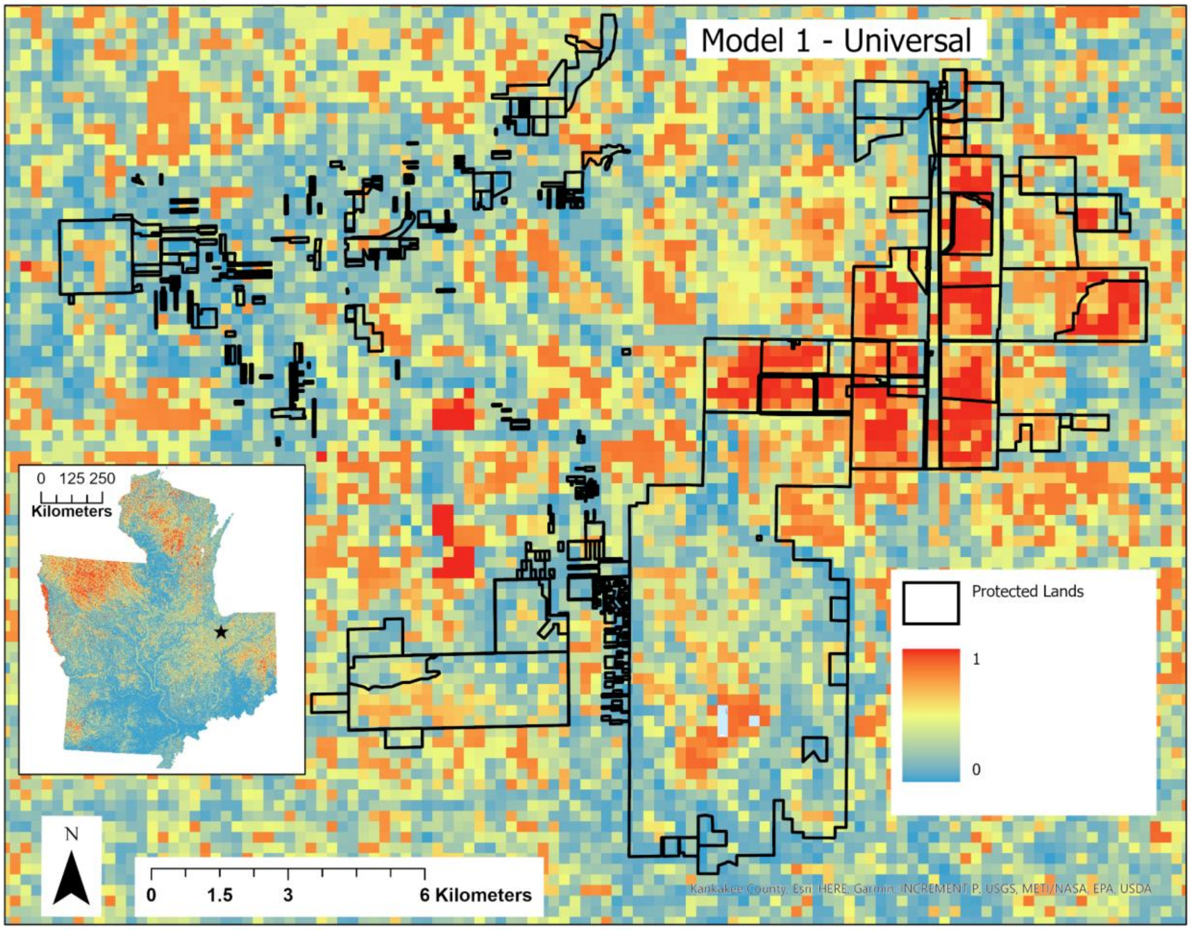

Figure 2.3. Map of the MaxEnt Universal model results for relative likelihood of Greater Prairie-Chicken occurrence at Kankakee Sands and the surrounding protected lands in northwest Indiana and northwest Illinois, USA. The Universal model incorporated all environmental variables in its predictions. Warmer colors (shades associated with the number 1) indicate a higher relative likelihood of occurrence. All boundaries are from Becker (2018).

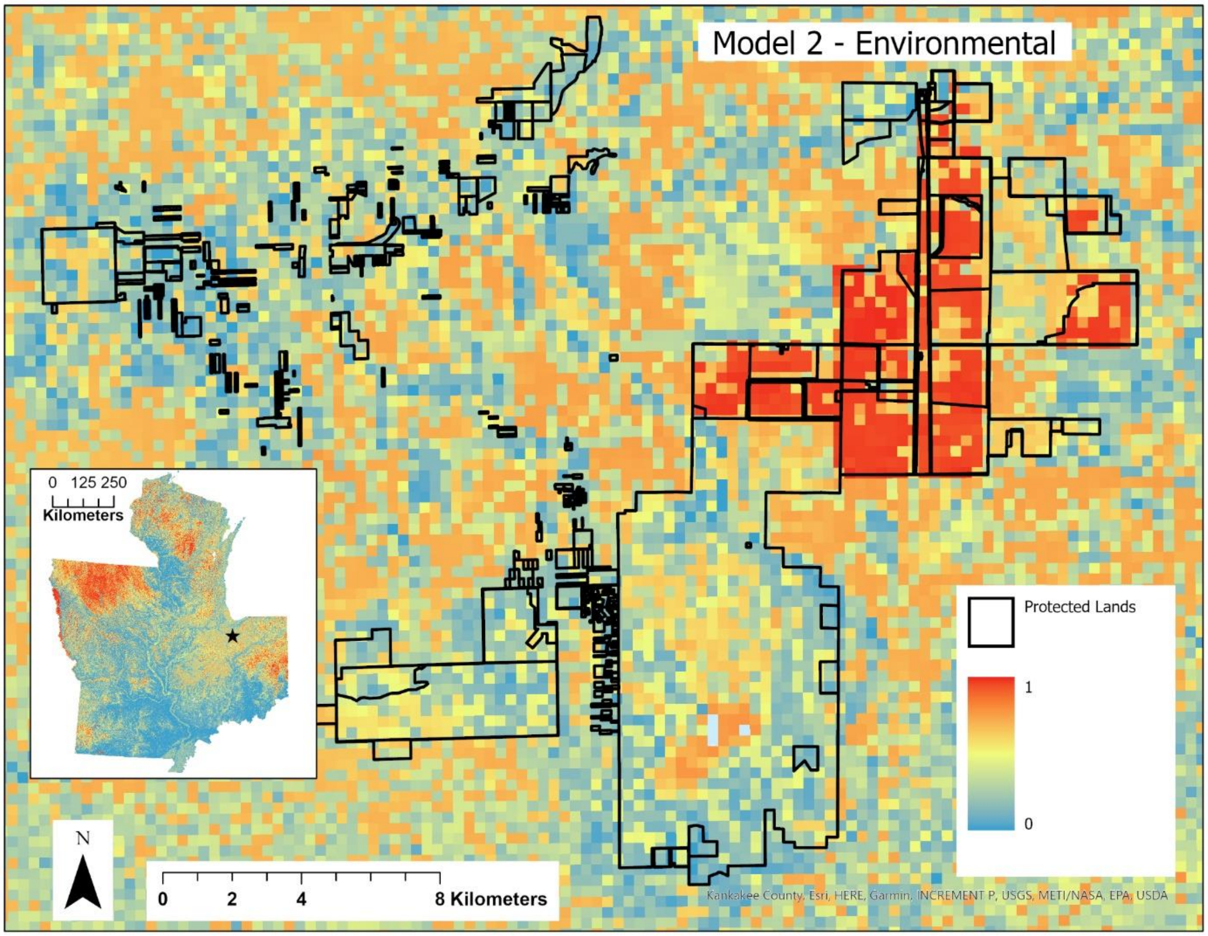

Figure 2.4. Map of the MaxEnt Environmental model results for relative likelihood of Greater Prairie-Chicken occurrence at Kankakee Sands and the surrounding protected lands in northwest Indiana and northwest Illinois, USA. The Environmental model incorporated all non-anthropogenic environmental variables in its predictions. Warmer colors (shades associated with the number 1) indicate a higher relative likelihood of occurrence. All boundaries are from Becker (2018).

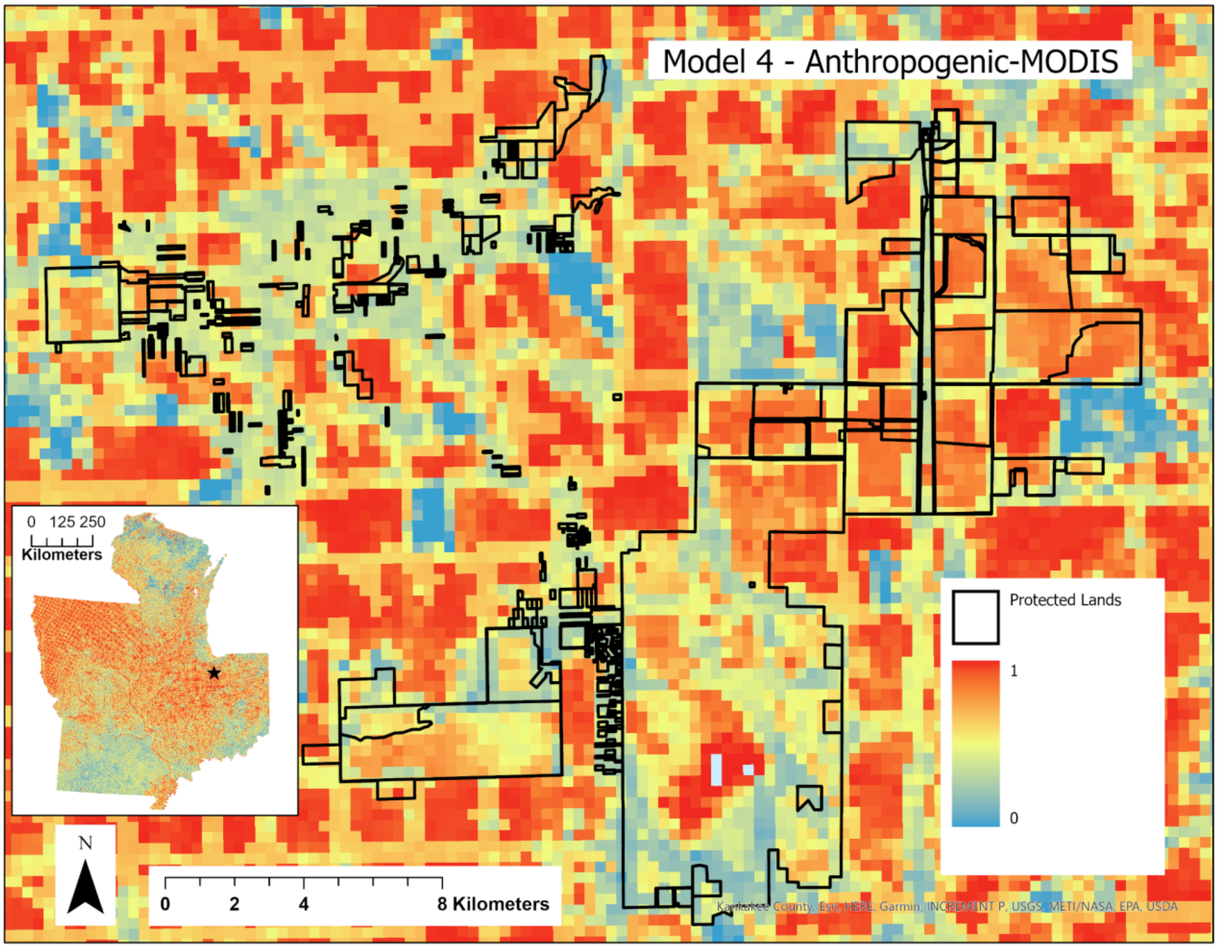

Figure 2.5. Map of the MaxEnt model results for relative likelihood of Greater Prairie-Chicken occurrence at Kankakee Sands and the surrounding protected lands in northwest Indiana and northwest Illinois, USA. This model, Anthropogenic-MODIS, incorporated all anthropogenic environmental variables along with the MODIS land cover variable in its predictions. Warmer colors (shades associated with the number 1) indicate a higher relative likelihood of occurrence. All boundaries are from Becker (2018).

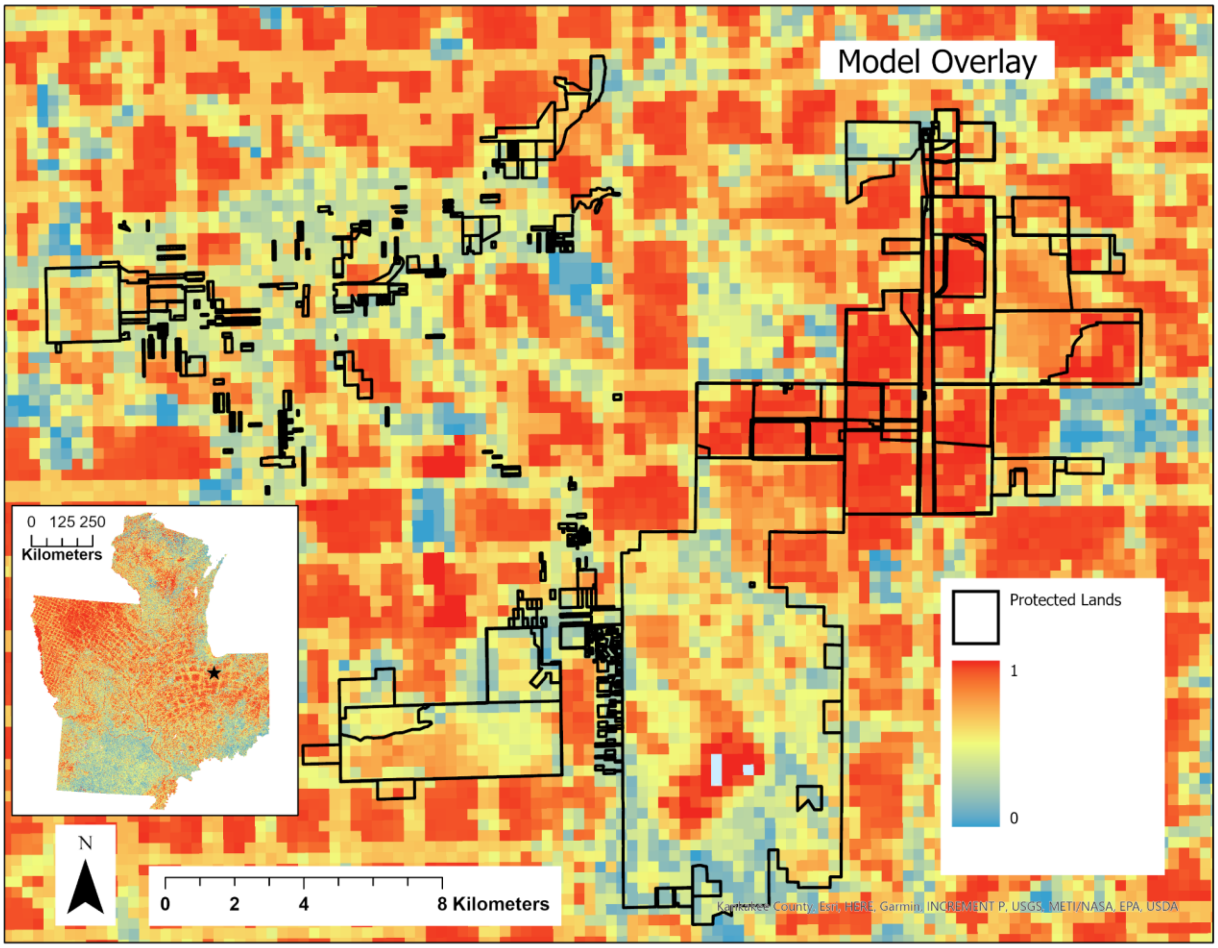

Figure 2.6. Map of the overlay of three MaxEnt models: Universal, Environmental, and Anthropogenic-MODIS. These models predict the relative likelihood of occurrence of the Greater Prairie-Chicken. Warmer colors (shades associated with the number 1) indicate a higher relative likelihood of occurrence. All boundaries are from (2018).

2.4 Discussion

My prediction that the majority of the Kankakee Sands property and surrounding landscape will be suitable (suitability values ≥ 0.5; arbitrary cutoff) to reintroduce GPC was correct given the results of the Universal model, the only model with a passing AUC score and significant binomial test (Table 2.2). The other two passing models were the Environmental and the Anthropogenic-MODIS. All models likely would have performed better with more data; I was only able to use 54 data points after processing and spatial filtering. Establishing new leks in the five-state study area or incorporating more states in a future study might be beneficial in increasing the usable sample size, as well as expanding survey efforts for states with existing leks. Another strategy to improve predictive power of future modeling attempts include expanding efforts to the range-wide scale. A final tactic is to use older data points (before 2010) while checking to ensure all data points are within suitable habitat classes or an agricultural field near suitable habitat classes that did not change based on the landcover layer from a given year corresponding to given data points.

Slope as the most influential predictor in the Universal model was surprising considering slope was the least informative predictor for all models in Hovick et al. (2015). However, GPC hens gain visibility advantages when nesting on slopes (Matthews et al. 2013). The influences of slope (and elevation) on nest site selection are relevant since proximity within 2 km of nesting habitat is important in lek site selection (Johnson et al. 2011, Svedarsky et al. 2003). The second best predictor was Landcover, as GPC rely on nearby grassland habitat for lek site selection (Hovick et al. 2015). Elevation was the third best predictor, given that GPC select more sites elevated above the surrounding landscape for visibility and acoustic advantages (Hovick et al. 2015).

The remaining predictors of Road Density, MODIS, Conservation Easements, Transmission Line Density, and Wind Turbine Density minimally influenced Universal model results. Because most lek sites in the Midwest occur in areas highly fragmented by roads, MaxEnt concluded a lower percent of importance for this variable in terms of affecting suitability since there are leks existing in this type of landscape. Landcover likely outperformed MODIS because Landcover is specific and categorical while MODIS consists of satellite imagery that measures NDVI of spatial and temporal comparisons of vegetative land cover in blue, red, and near-infrared reflectance

(Didan 2015, Hovick et al. 2015), suggesting that the Landcover layer had a clearer separation of habitat types the MODIS layer. Conservation easements are likely not prevalent enough near existing leks to be an important factor in lek site selection. The same can be said of the densities of transmission lines and wind turbines.

The Environmental model again had Slope as the best predictor. Elevation was the second most influential variable in this model, followed by Conservation Easements, then MODIS (Table 2.3). When the anthropogenic habitat features are excluded, Landcover became the least important variable for reasons unknown. Perhaps MaxEnt assumed an unfragmented matrix (from the lack of inclusion of road, transmission line, and wind turbine densities) equated to a lower significance in land cover type.

MODIS was the best predictor for the Anthropogenic-MODIS model (Table 2.3). Land cover type appears most important when topographical variation is excluded from modeling. Road Density was the second most important variable (Table 2.3). Existing leks occur in a landscape of high fragmentation due to roads. This predictor explains model results more when other relevant variables are excluded. Again, transmission line and wind turbine densities are not prevalent around existing leks in the study area to influence lek site selection.

Generally, the Universal and Environmental models provided the highest predictions of relative likelihood of occurrence for Kankakee Sands only (Fig. 2.3, Fig. 2.4). Meanwhile, the Anthropogenic-MODIS model provided higher predictions for Kankakee Sands and a large portion of the surrounding landscape, including the agricultural matrix (Fig. 2.6).

The majority of the GPC data points were in areas predicted to have relative likelihoods of occurrence ≥ 0.5, MaxEnt′s baseline prevalence. Because of this, I considered 0.5 the minimum score of interest related to suitability for translocation (Mutascio et al. 2018, Fitzpatrick et al. 2013). Given this, MaxEnt predicted that all lettered units of Kankakee Sands (Fig. 2.1, Fig. 2.6) would be suitable areas for GPC lek sites. Outside of Kankakee Sands, areas with a relative likelihood of occurrence above 0.5 included Beaver Lake Nature Preserve, the north and central portions of Willow Slough Fish and Wildlife Area (WSFWA), the southern block of Iroquois County State Wildlife Area west of WSFWA, and a few small parcels northwest of Kankakee Sands (Fig. 2.1, Fig. 2.6). However, it is important to note that in all three useful model outputs (Fig. 2.3, Fig. 2.4, Fig. 2.5), the surrounding agricultural matrix was predicted to have a relative likelihood of occurrence > 0.5.

Another mentionable note is that the environmental data did not reflect microhabitat assessments. Rather, they represent broad, macrohabitat classifications and characteristics. Therefore, I did not incorporate fine-scale habitat characteristics into my modeling and biologists should interpret results on a macrohabitat scale. Future modeling efforts that incorporate fine-scale habitat would likely have more informative results for a similar sample size.

In this area of the Midwest, GPC usually lek from March to May (Robb and Schroeder 2005). Farmers typically begin planting around 20 April – 10 May (Nielsen 2015). Before and after the planting events during the lekking period, these fields are open with open visibility. High visibility in these open fields near potential nesting habitat in nearby grasslands may serve as acceptable booming grounds outside of Kankakee Sands and other protected lands during the non-planting season. As lek site selection is related to visibility and acoustic advantages, it is also associated with proximity to nesting habitat (Hovick et al. 2015). For additional plausibility, leks in southeast Illinois are located on open ground in disturbed areas near grassland habitat (J. Dunning, personal observation).

The presence of a major highway bisecting Kankakee Sands is another concern. This busy road would present a higher risk of mortality for translocated GPC attempting to travel between the two major bisected sections of the property. Other, smaller, gravel roads fragment the smaller, individual sections of Kankakee Sands (Fig. 2.1). These roads may present mortality risk and discourage dispersal as well. Regarding the surrounding landscape, the smaller habitat patches northwest of Kankakee Sands are smaller and fragmented by roads and agricultural matrix. Increasing local habitat connectivity via corridors and future habitat restorations would be ideal to establish a metapopulation in the area and promote dispersal/gene flow.

Reintroducing populations onto isolated habitat islands can pose challenges in terms of establishing a metapopulation between nearby habitat patches. GPC can fly >11 km between habitat patches (Leopold 1931), a boon for connectivity in a fragmented landscape. I presume there is ongoing effort by TNC in both Indiana and Illinois, along with the Departments of Natural Resources of both states, to connect existing habitat patches to create larger, contiguous, or otherwise connected tallgrass prairie habitat patches in the area of proposed reintroduction. There is definitely potential for increased connectivity in the area via land acquisition by conservation organizations and landowner participation in Farm Bill programs such as the Conservation Reserve Program. Isolated habitat patches are also more likely to need supplemental translocations in order to maintain a viable population size, based on the history of GPC reintroduction in Illinois (ILDNR 2018). Additionally, establishing captive populations in nearby zoological parks could help assuage the need for frequent population input by minimizing distance traveled between the translocation site and stable, wild source populations.

I acknowledge certain limitations behind SDMs. The first limitation is that this method does not account for undetected individuals/groups in the modeling design. Second, this method also does not account for sources of mortality, such as predation or vehicle collisions. Third, this method also does not account for the effect of exotic species, such as Ring-necked Pheasant (*Phasianus colchicus*), on the species of interest. Fourth, MaxEnt users assume constant detection probability across the landscape (Merow et al. 2013). These factors would likely affect the prediction of habitat suitability if included. They had to be excluded due to the simplicity of the modeling design, which accounted for only land use as the factor that predicts habitat suitability. However, habitat destruction due to increased resource depletion and land use conversion from a growing human population is the main factor in global biodiversity loss (Singh 2002). Because of this, I was satisfied with the evaluation of habitat as the primary predictor of species occupancy. Lack of data for undetected individuals/groups, along with the inability to incorporate sources of mortality and exotic/invasive species effects into MaxEnt evaluations should not prevent modeling for conservation purposes. With respect to these concerns, other researchers have successfully used MaxEnt for similar purposes to mine (Cilliers et al. 2013, D’Elia et al. 2015).

Use of elevation and slope as predictor variables was a concern because of the relatively flat topography of northwest Indiana and northeast Illinois. The relative flatness was a concern in terms of including elevation and slope as predictors of limited value related to Kankakee Sands. However, I ultimately included these variables given that the extent of the predictive models ranged over the entirety of Indiana, Illinois, Iowa, Missouri, and Wisconsin. Over such a large area, elevation and slope contribute more as predictors because elevation varies significantly more at the larger scale than at the local.

Ideally, I would reduce potential spatial autocorrelation among environmental variables when performing this type of modeling. Although I was unable to successfully account for spatial autocorrelation using the diagnostic measures, I was confident in conducting my analyses because I incorporated a biologically-relevant spatial filter to promote independence of my observation points. I believe my results are still useful given my attempt to formally account for this potential issue. Other studies that also used MaxEnt analyses to assess species distributions for potential future reintroductions did not account for spatial autocorrelation (Adhikari et al. 2012, Hendricks et al. 2016). Moreover, regularization in MaxEnt increases its stability regarding correlated environmental variables, which is related to spatial autocorrelation (Elith et al. 2011).

An a-posteriori observation is that over 200 m of raster resolution was lost changing the landcover layer resolution from 30 m to 231.66 m to match the MODIS layer resolution, which may explain why the correlograms' and semivariograms' spatial filtering distances were > 300 km. In future modeling efforts for GPC, I recommend removing MODIS as a predictor in order to preserve as much data and resolution as possible in all environmental data layers. Because these potential spatial filtering distances were large enough to leave only two data points, I did not correct for potential spatial autocorrelation. I did, however, retain a nearest neighbor distance of 1.52 km to help ensure independence among the lek data.

MaxEnt outputs can be interpreted as estimates of absolute occupancy only if the user provides presence-absence data (Guillera-Arroita et al.2014). But when working with presence-only and background data, such as in this project, MaxEnt estimates relative likelihood of occurrence (Guillera-Arroita et al. 2014, Merow and Silander Jr. 2013, Perkins-Taylor and Fray 2020). This form of measuring occupancy probability is proportional to the actual occupancy probability and is therefore still useful (Guillera-Arroita et al. 2015, Merow and Silander Jr. 2013). Given this, biologists can interpret the relative likelihood of occurrence outputs as an index of habitat suitability on a zero to one scale (Perkins-Taylor and Fray 2020).

Overall, MaxEnt's predictions indicated high relative likelihoods of occurrence for the GPC lek sites in the area of Kankakee Sands and the surrounding landscape. Specific to the Universal model, the only one to perform better than the null and likely the only actually useful model, MaxEnt predicted all of Kankakee Sands to have a high relative likelihood of occurrence of GPC lek sites. This indicates translocation is plausible at Kankakee Sands, at least in terms of macro-habitat criteria. I believe a reintroduction effort would be worth it in terms of establishing a new population of this declining grassland bird, along with continuing the restoration of the tallgrass prairie ecosystem already begun by TNC and increased ecotourism, accompanied by local economic benefits, as a result of the restored presence of these birds.

2.5 Management Implications

I had four site-specific and one regional management suggestion for Kankakee Sands relevant to the GPC. The first suggestion is removing all fencerows and unused power lines (if possible) to reduce available perching sites for predatory birds. Because GPC requires large tracts of contiguous habitat, my second suggestion is to continue acquiring property and connect habitat patches as they become available/affordable. The third suggestion is to work with the property manager at nearby Willow Slough Fish and Wildlife Area (17.5 km), to halt, postpone, or minimize the release of captive RNP. According to M. Schoof (INDNR, personal communication), 2,500 pen-raised RNP are released each year. An alternative is to contact the INDNR about increasing pheasant harvest limitations and/or incentives in Newton County, because only 30 RNP are harvested per year at Kankakee Sands (M. Schoof, INDNR, personal communication; Appendix A). My fourth suggestion is that it may be beneficial to permanently or temporarily close the county roads that divide the property into multiple fragmented sections that would present another mortality hazard to any translocated GPC, although politics might make this difficult. For the surrounding landscape, land managers could work with the Illinois and Indiana Departments of Natural Resources to implement management similar suggestions as possible and deemed appropriate.

In the event that GPC establish leks in agricultural fields outside of Kankakee Sands, I encourage managers to continue outreach with the local farmer community to delay planting and/or provide compensation for the delay. GPC typically lek between March and May (Robb and Schroeder 2005), while farmers in Indiana typically seed their fields 20 April – 10 May (Nielsen 2015). Finally, conflicts may arise over lek disruption by farm equipment and/or the inability of a farmer to plant their fields where a new GPC occurs. Maintaining relationships with local landowners would be helpful in reaching mutually-beneficial resolutions for numerous potential conflicts, such as this.

CHAPTER 3. FRANKLIN'S GROUND SQUIRREL

3.1 Introduction

Reintroductions are an acceptable conservation tool only when the conservation benefits are greater than the ecological, economical, and social risks (International Union for the Conservation of Nature [IUCN] 2016). According to IUCN (2016), "reintroduction" is defined as "the intentional movement and release of an organism inside its indigenous range from which it has disappeared" (IUCN 2016). If conservation benefits are greater than the risks, then the next step is to define the goals of a potential reintroduction project. Subsequently, a feasibility assessment must be performed (IUCN 2016). My project involved the habitat suitability assessment pertaining to the potential translocation of the Franklin's ground squirrel (*Policitellus franklinii*; FGS) to a 3,400 ha tallgrass prairie property in northwest Indiana, USA.

Species distribution modeling (SDM) is one way to assess the habitat suitability of an area for a reintroduction attempt. SDM analyses use various habitat-related ecological variables relative to the species of interest. Important variables, according to Gedir et al. (2004), are: species status, cause(s) of decline, habitat attributes (among the most important to investigate in regard to the success of a reintroduction program), behavior (foraging, reproductive, and social), diet, and reproduction itself. Furthermore, assessors must also consider management of exotic species (deemed most important) and understanding anthropogenic, predatory, and competitive threats to species longevity in an area (Short 2009; Appendix A). Some anthropogenic factors also affect the species' distribution. FGS frequently occupy unmowed strips next to roads and railroad rights-of-way (Martin et al. 2003, Huebschman 2007). Additionally, power lines and poles provide perching sites for predatory raptors (Robb and Schroeder 2005).

3.1.1 Species Distribution Modeling

SDM is a method of modeling potential distributions based on Ecological Niche Modeling (Hovick et al. 2015). Others used SDM for predicting climatic effects on species distribution, niche-based habitation, as well as the spread of invasive species, management of threatened species, phylogeographic patterns, and management of landscapes (Guisan and Thuiller 2005, Guillera-Arroita et al. 2015, Mutascio et al. 2018). SDMs are useful for identifying areas of interest for species-specific conservation action, including areas for reintroductions of endangered species (Guillera-Arroita et al. 2015, Maes et al. 2019, Westwood et al. 2020). These models are based on known location data with natural and anthropogenic land use data (Rodríguez et al. 2007, Hovick et al. 2015). Moreover, these models predict species occurrence based on habitat suitability related to relative likelihoods comparable to true occurrence probabilities (Guillera-Arroita et al. 2015).

One SDM method, maximum-entropy, uses environmental variables and sample location data to estimate the distribution closest to uniform (maximum entropy) under the constraint of the empirical averages of said environmental variables (Philips et al. 2004). Stohlgren et al. (2015), Morris et al. (2020), and (Hernandez 2015) performed maximum-entropy analyses to predict species distributions for the eastern diamondback rattlesnake (*Crotalus adamanteus*), yellow rail (*Coturnicops noveboracensis*), and bluehead shiner (*Pteronotropis hubbsi*), respectively. Between 2006 and 2013, other researchers used MaxEnt in over 1,000 scientific publications (Merow et al. 2013). Merow et al. (2013) provides 2 reasons for the widespread use. The first is that MaxEnt often outperforms other programs/methods in terms of predictive accuracy and ease-of-use. The second is that others used and validated MaxEnt over a wide range of species and habitats (Philips and Dudík 2008). An additional reason to use MaxEnt is that it works well even with small datasets (Elith et al. 2006).

3.1.2 Logistical and Operational Constraints for Species Translocations

There are multiple logistical and operational constraints pertaining to species translocations, including: locations of source populations, required federal and state permits, facilities that could and are willing to house a captive source, release strategies, and genetic concerns. John Shuey (Director of Conservation Science with The Nature Conservancy [TNC]) suggested I collect this information to aid them in a potential future translocation effort for this species (J. Shuey, TNC, personal communication). This information is located in Appendix B.2.

3.1.3 Objectives and Hypothesis

The purpose of this study was to model habitat suitability at a restored tallgrass prairie property owned by TNC as well as the surrounding landscape to identify areas for FGS translocation. The property (Kankakee Sands) is located in northwest Indiana and the surrounding area of interest extends west to northeastern Illinois, USA. I chose this species because of continuing decline in the eastern periphery of the tallgrass prairie historical range (Martin et al. 2003, Martin and Heske 2005, B. Westrich, Indiana Department of Natural Resources [INDNR], personal communication). I developed an SDM application using maximum-entropy techniques on environmental variables associated with suitable FGS habitat and other anthropogenic factors associated with avoidance. I shared state-specific results with the respective Departments of Natural Resources of the states that provided the FGS capture location data, while I focused on the model results for the Kankakee Sands area in northwest IN and northeast IL.

Specific objectives were to:

1. Predict FGS habitat suitability at Kankakee Sands and the surrounding landscape in binary (presence or absence) output in a map.
2. Provide TNC with logistical and operational information to help guide their planning of a FGS translocation to the area, with respect to the results.

Our prediction and hypothesis was that the majority of the Kankakee Sands property and surrounding landscape would be suitable (suitability values < 0.5; arbitrary cutoff) for a FGS translocation attempt because there are thousands of hectares of habitat available in the landscape relative to observed densities of FGS being 3–12 individuals/hectare (Olson 2002) combined with the dispersal capabilities of this animal of ≤ 5 km (Duggan et al. 2011).

3.2 Methods

3.2.1 Study Area

This study incorporated data from Indiana, Illinois, Wisconsin, Iowa, and Missouri, USA, to create a prediction for suitable habitat of the FGS at Kankakee Sands (Fig. 3.1), a property owned by TNC located 8 km north of Morocco, Indiana, in northwest Indiana. As designated by TNC, Units E & I are currently/were recently under agricultural leasing (Fig. 3.1). TNC returned Unit E to tallgrass prairie in fall 2020 and will restore Unit I in 2021 (A. Nyberg, TNC, personal communication). Kankakee Sands is also a part of a larger group of tallgrass prairie fragments in northwest Indiana and northeast Illinois (Fig. 3.2).

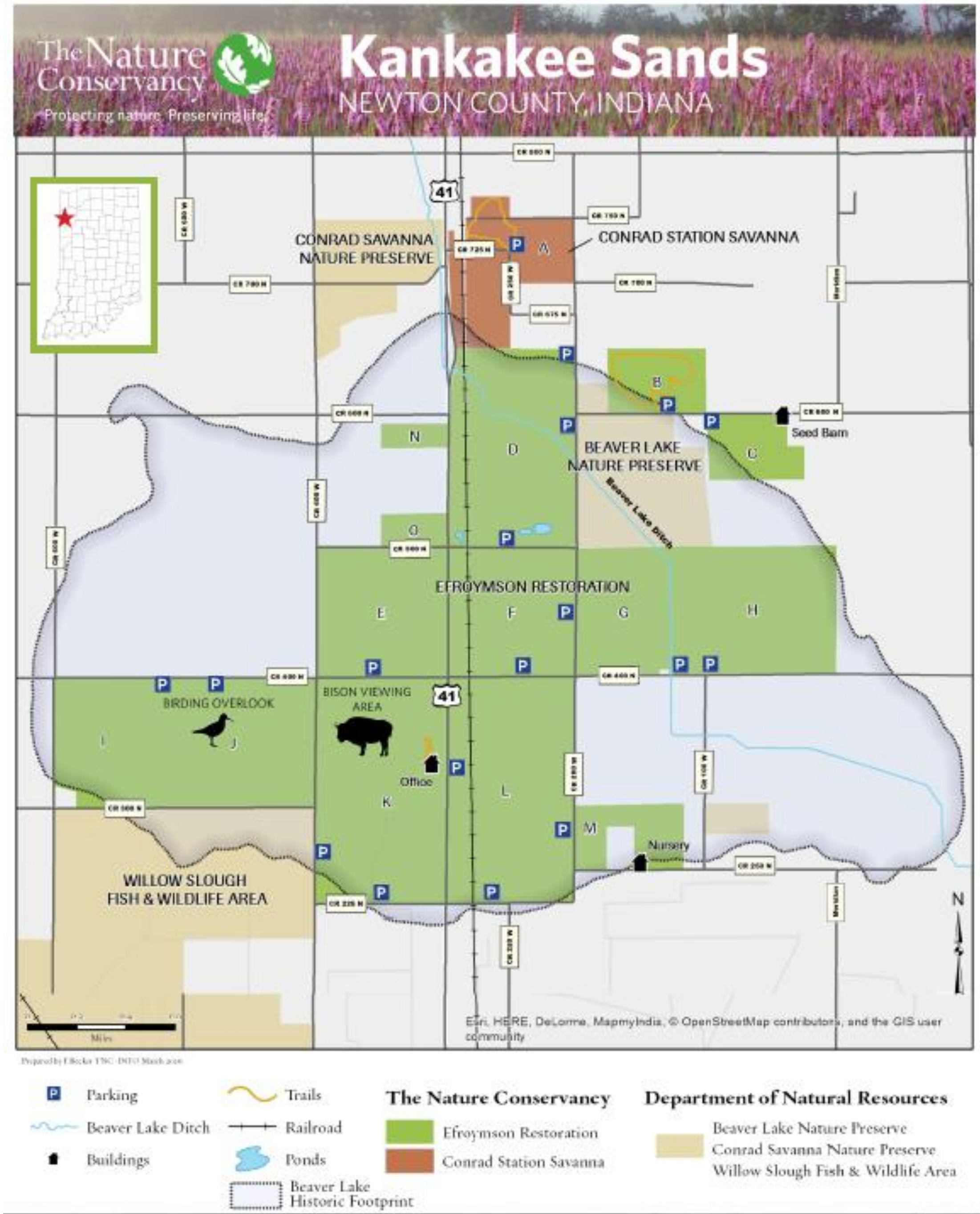

Figure 3.1. Map of Kankakee Sands property in northwest Indiana, USA created by Becker (2016), owned by The Nature Conservancy. This location could be the site of a translocation event for the Franklin's ground squirrel (*Poliocitellus franklinii*). Notable features are the roads, bison grazing pasture, and railroad line. There is a major highway (US-41) to the right of the bison pasture. Individual units are labeled with letter designations.

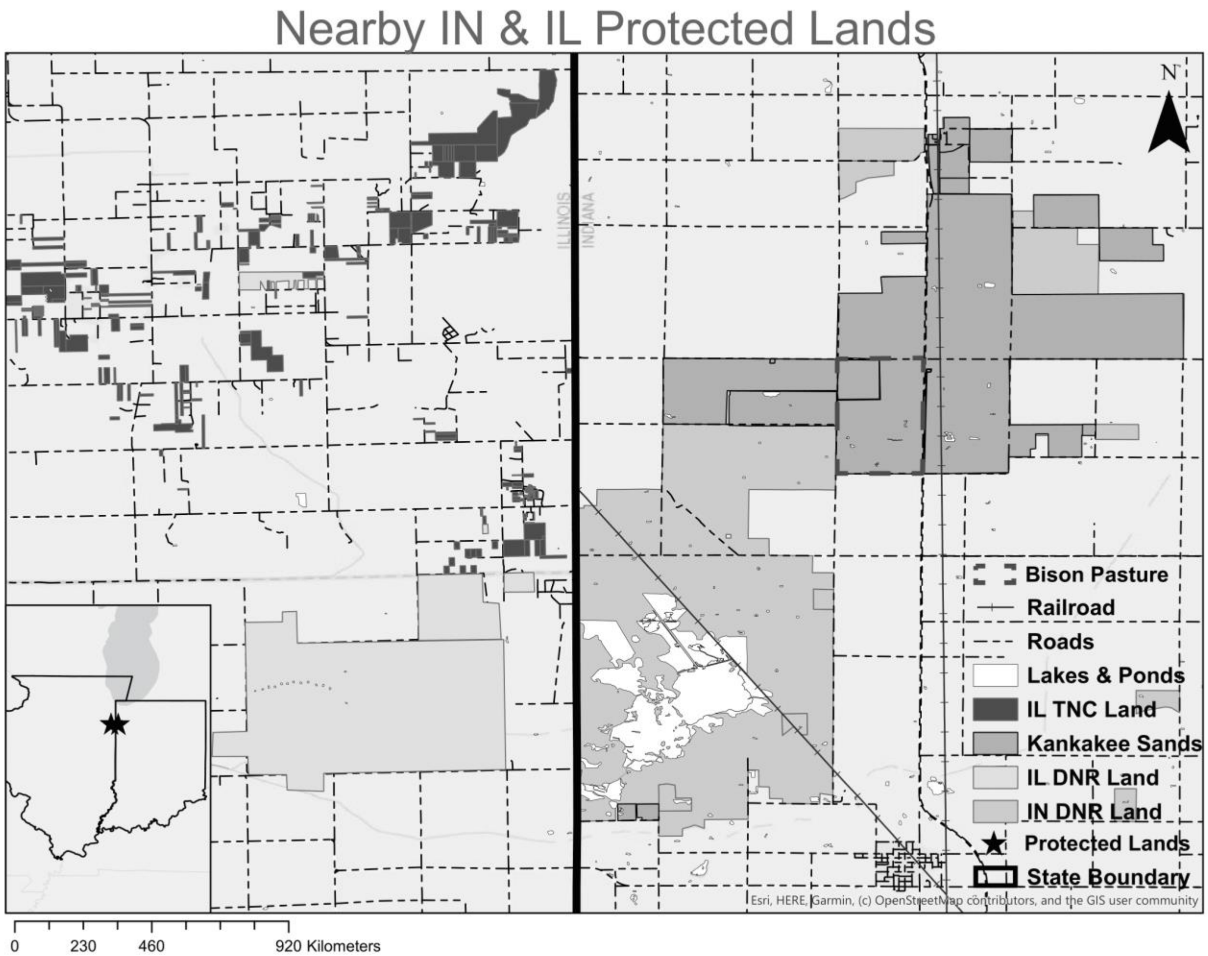

Figure 3.2. Map of the tallgrass prairie properties near and including Kankakee Sands. This landscape could host a translocated population of Franklin's ground squirrel (*Poliocitellus franklinii*). Notice the state border between Indiana and Illinois, USA. All DNR and The Nature Conservancy boundaries are from Becker (2018).

Kankakee Sands and the surrounding landscape is flat in topography due to the Wisconsin glaciation event. According to TNC (2020), the property was once part of the Grand Kankakee Marsh and tallgrass prairie ecosystems of the area. TNC stated that farmers drained and converted the property into agricultural fields in 1873 (Schmal 2005). There is an unused railroad track and active Indiana highway that run parallel to one-another, bisecting the property from north to south. TNC purchased the property in 1997, when they began to create what is now 3,400 ha of tallgrass prairie and wetland habitats (TNC 2020). There are also graveled county roads oriented east-west that divide the property into several parts.

Currently, mammals that inhabit Kankakee Sands include: coyote (*Canis latrans*), American bison (*Bison bison*), plains pocket gopher (*Geomys bursarius*), and American badger (*Taxidea taxus*; Nyberg 2019, TNC 2020). Reptiles include gopher snake (*Pituophis melanoleucus*) and prairie king snake (*Lampropeltis calligaster*; Homoya et al. 1985, TNC 2020). Avian species include exotic ring-necked pheasant (*Phasianus colchicus*) along with more than 240 bird species including waterfowl and raptors (Barnes 1952, eBird 2020, Homoya et al. 1985). Plant species of the prairie portions of this property include side-oats grama (*Bouteloua curtipendula*), little bluestem (*Andropogon scoparius*), prairie goldenrod (*Solidago rigida*), and rattlesnake master (*Erynigium yuccifolium*; Homoya et al. 1985). As for the wetland portions, plant species include spatterdock (*Nuphar advena*), bluejoint grass (*Calamagrostis canadensis*), knotweeds (*Polygonum* spp.), and sedges (*Scirpus* spp. and *Carex* spp.; Homoya et al. 1985). There are also stands of invasive horsetail reed (*Equisetum* spp.; ZF personal observation).

The nearby town of Rensselaer, Indiana (28 km southeast of Kankakee Sands) experiences an annual low temperature of 4.7° C and an annual high of 15.7° C, with an average annual temperature of 10.2° C (U.S. Climate Data 2007-2019). Annual snowfall ranges 0.25-22.6 cm/month (Western Regional Climate Center 2020). Annual rainfall ranges from 2.5-10.2 cm/month (World Climate 2020). The soil orders of the area in and around Kankakee Sands are Histosols, Mollisols, and Entisols (Soil Survey Staff 2017).

3.2.2 Data Collection and Analysis

FGS Capture Locations

I obtained FGS capture location data from the natural heritage database programs of state natural resource departments of Indiana, Illinois, Iowa, Missouri, and Wisconsin, USA, that are all part of the NatureServe network of natural heritage databases (NatureServe 2014). I requested FGS capture location data from the Indiana Natural Heritage Data Center (Indiana Natural Heritage Data Center 2020), Illinois Department of Natural Resources (DNR; Illinois Department of Natural Resources 2020), Wisconsin DNR (Wisconsin Department of Natural Resources, Wisconsin Natural Heritage Program 2020), Iowa DNR (Iowa Department of Natural Resources 2021), and the Missouri Department of Conservation (Missouri Department of Conservation 2021) in November 2020. I received the final data in February 2021. All data were as-is (not updatable) based on the time of request. These states collected FGS location information through live-trapping efforts, biologist-approved incidental observations, and road kills across the range of the animal in the respective states.

To acquire additional data points, I downloaded four FGS observation location data points from the Global Biodiversity Information Facility (GBIF 2020). GBIF is a free, open-source international and data infrastructure resource for species occurrence data with hundreds of publications each year using data downloaded from GBIF (GBIF 2020). GBIF obtains data when publishers submit datasets in machine-readable Creative Commons license formats. GBIF adheres to data standards including BioCASe/ABCD, Ecological Metadata Language, and Darwin Core (GBIF 2020).

To minimize errors of using older data points in modeling with respect to land use change over time, I only used points that were from 2010–2021. Using this date range helped minimize modeling errors resulting from capture points being located in non-habitat due to land-use conversion. I considered polygon and line data I received, in addition to point data, as non-point representations of the FGS capture location data I requested. Using ArcGIS, I converted these polygons and lines to point values.

Spatial Autocorrelation Reduction

To assess the role of spatial autocorrelation, I used Program R (v. 4.1.0, R Core Team, Vienna, Austria) along with the packages “raster” (Hijmans 2021), “elsa” (Naimi et al. 2019), and “usdm” (Naimi 2017). I used packages “raster” combined with “elsa” to create correlograms to determine the distance at which Moran’s I was equal to zero for several of the data layers. Moran’s I coefficients are a measure of relatedness of neighboring cells at various distances (Diniz-Filho et al. 2003). Their derivative correlograms are useful for testing potential spatial filtering distances at which to consider spatial autocorrelation negligible when the Moran’s I value is zero for a given distance (Diniz-Filho and Bini 2005).

I used preliminary model runs to identify the most influential, and likely to be spatially-autocorrelated, layers from respective models. These layers were soils, MODIS, and landcover. Soils had a Moran’s I value of zero at 375 km while MODIS and landcover had Moran’s I values of zero at 325 km. Both distances reduced the number of FGS data points to three after spatial filtering.

Semivariograms are another useful way to identify spatial autocorrelation-reducing spatial filtering distances (Griffith 2005). Semivariograms are the inverse of correlograms (Getis 2010). I used the “raster” and “usdm” packages in R to create semivariograms for the Soils, MODIS, and Landcover layers. I tested these layers out to a cutoff distance of 400 km to try to find a sill. The sill is where the semivariance equals the variance of the data layer (Griffith 2005). The resulting plotted semivariogram did not have a sill out to 400 km. Because these filtering distances were as large as they were, I was unable to account for spatial autocorrelation in my models. Instead, I used another distance value for spatial filtering to preserve data volume.

Spatial Filtering

To help ensure data point independence, I applied a spatial filter of 5 km, the assumed maximum dispersal distance for FGS (Martin and Heske 2005, Duggan et al. 2011). To filter out data points within this minimum distance, I used the Buffer tool in ArcGIS Pro to create circles around point values based on a specified input radius. I manually removed data points until no buffer circles overlapped.

Landcover and Anthropogenic GIS Layers

For the states of IN, IL, WI, IA, and MO, I collected digital landcover and anthropogenic geographic information system (GIS) data layers from open-source internet data platforms. I retrieved the following data layers for the above states from the Geospatial Data Gateway provided by the Natural Resources Conservation Service (NRCS 2020): NRCS States by State, Cropland Data Layer by State, NRCS Conservation Easement Areas by State, National Elevation Dataset 30 Meter, Gridded Soil Survey Geographic (gSSURGO) by State, TIGER Primary and Secondary Roads, and TIGER Streets (NRCS 2020).

The NRCS States by State layer provided state borders. Cropland Data Layer by State provided specific landcover information to identify potential habitat and non-habitat, in addition to detailed crop location information. This layer, updated annually since 1997, consisted of 134 land cover, crop cover, and cropland classes in 30 m pixels (USDA National Agricultural Statistics Service Cropland Data Layer 2020), and were current as of 2019. The accuracy of the landcover is 85-95% for major crop categories and was based on Advanced Wide Field Sensor (AWiFS), Landsat Thematic Mapper (TM), Enhanced Thematic Mapper (ETM+), and Moderate Resolution Imaging Spectoradiameter (MODIS) satellite data (Boryan et al. 2011).

NRCS Conservation Easement Areas by State identified locations and types of conservation easements, which could provide additional habitat. I incorporated the National Elevation Dataset 30 Meter layer for its own sake as a covariate as well as the creation of a slope layer. Elevation and slope are relevant because FGS burrows are often located on slopes to assist in drainage (Ostroff and Finck 2003). I used TIGER Primary and Secondary Roads, along with TIGER Streets, because roads fragment habitat.

The inclusion of the Gridded Soil Survey Geographic (gSSURGO) layer pertained to burrow site selection as well. FGS appears to select sites with sandy soils that drain well (Huebschman 2007). This layer is a derivative of the Soil Survey Geographic database (SSURGO), which includes soils data at state-wide extents as fine as county-level at a 10 m resolution (NRCS Soils 2021, NRCS Soils 2021). Stohlgren et al. (2015), Morris et al. (2020), and (Hernandez 2015) used gSSURGO soils data as part of their MaxEnt analyses to predict species distributions for the eastern diamondback rattlesnake, yellow rail, and bluehead shiner, respectively. These uses indicate applicability of gSSURGO use with MaxEnt to a wide range of fauna.

I retrieved a data layer for railroads from Homeland Infrastructure Foundation-Level Data (HIFLD 2020). I also included a power transmission line layer as a predictor of FGS presence. I retrieved these data from Homeland Infrastructure Foundation-Level Data (HIFLD 2020).

To obtain another general landcover layer, I used the Application for Extracting and Exploring Analysis Ready Samples (AρρEEARS, Ver. 2.51, NASA EOSDIS Land Processes Distributed Active Archive Center, Sioux Falls, SD) to download a MODIS layer known as MOD13Q1 v006 (hereon "MODIS"). It was created with satellite Moderate Resolution Imaging Spectroradiometer (MODIS) at a roughly 231.66 m resolution (Didan 2015). These data provide MODIS Normalized Difference Vegetation Index (NDVI), which displays spatial and temporal comparisons of vegetative landcover in blue, red, and near-infrared reflectance (Hovick et al. 2015). MODIS and the above layers were my model covariates. For the final GIS layers, I used boundary polygon layers from Becker (2018) to highlight model results for Kankakee Sands and the other tallgrass prairie properties in the surrounding landscape.

Data Layer Conversions

After downloading all GIS data, I used ArcGIS Pro (v. 2.6.0, Esri, Redlands, CA) to clip the railroad and power transmission line layers to include data only from the states of interest. Then I combined TIGER Primary and Secondary Roads and TIGER Streets into one Feature layer called "Roads." Then I changed all GIS layer projection coordinate systems to match, except for Roads, Railroads, and power transmission lines. I combined all state "Cropland Data Layer by State" (hereon "Landcover") into 1 raster. I also combined all NRCS Conservation Easement Areas by State (hereon "Conservation Easements") and National Elevation Dataset 30 Meter (hereon "Elevation"; NRCS 2020) into respective merged layers. I also converted Conservation Easements into a raster format to ensure compatibility with the modeling software.

In order to promote accurate density measurements, I changed the projection coordinate system for Roads, Railroads, and power transmission lines to Albers equal area (Hovick et al. 2015). Next, I used the Near tool in ArcGIS Pro to calculate the distance to the nearest power transmission line and road (respectively) relative to capture locations. I then used the Line Density tool to convert the linear polyline layers into density rasters. Power transmission lines and roads are potential sources of mortality and may affect FGS habitat selection.

I used the Slope tool in ArcGIS Pro to construct a raster slope layer with degree values from 0–90 to measure maximum change between a certain cell and all 8 neighbor cells. I only modified the projection coordinate system of MODIS to match the rest of the layers similar to the methods used in Hovick et al. (2015).

Instead of using the default 10,000 random points from MaxEnt, I created my own bias file for the generation of 10,000 custom pseudo-absence points. I created custom points instead of using the points generated by default because custom background points from only sampled habitat types helps MaxEnt focus on differentiating between presence and background distributions instead of sampling bias (Mutascio et al. 2018). I generated these points in the command prompt using this bias file (Philips 2017). In order to incorporate areas of habitat only, I used the USDA CropScape tool (v. 2020, National Agricultural Statistical Service, Washington, D.C.) to extract only the Grass/Pasture landcover type. MaxEnt assumes the biologists who collected the data did not sample every possible habitat patch in the five-state area (Fourcade et al. 2014). My bias file assessed the sampling bias from MaxEnt assuming the observers surveyed only areas of known presence (Stolar and Nielsen 2015, Young et al. 2011).

To finish the use of ArcGIS Pro, I used the Raster to ASCII tool to ensure all data layers matched in extent (combined state boundaries) and resolution (same as MODIS, the variable with the coarsest resolution), a requirement for MaxEnt (Young et al. 2011). I removed all data columns from the attributes table of the observation data point before creating two new fields in the attribute table for longitude and latitude. Then I calculated the longitude and latitude geometry, in that order, for every animal observation point. I proceeded by converting the attribute table of the points layers to a .csv input file (Excel v. 2016, Microsoft, Redmond, WA).

MaxEnt for Data Analysis

To analyze the data, I used maximum entropy modeling (MaxEnt; v. 3.4.1, Philips et al. 2020) to create SDMs of suitable FGS habitat across the sampled states. MaxEnt does not require known (and expensive-to-collect), absence data because MaxEnt uses 10,000 pseudo-absence points from background pixels (Philips and Dudík 2008, Hovick et al. 2015).

To determine model covariates and assess the importance of different covariates based on parsimony, I used four models based on results in Hovick et al. (2015) to determine which combination of variables best predicts suitable FGS habitat. The first model (the "Universal model”) included all data layers (Landcover, MODIS, Conservation Easements, Elevation, Slope, Roads, power transmission lines, and wind turbines). The second model (the “Environmental model”) included only environmental factors (Table 3.1). The third model (“Anthropogenic-Landcover”), included anthropogenic layers (Table 3.1) and the Landcover layer. The final model (“Anthropogenic-MODIS”) included the anthropogenic layers and the MODIS layer. I created the additional models that did not include all variables based on FGS behavior to develop more simplistic models, easily-interpreted by a wide variety of stakeholders (i.e. politicians, farmers, land managers, DNRs, etc.; Hovick et al. 2015).

Table 3.1. Data from IN, IL, IA, WI, and MO, USA involved in species distribution modeling to predict likelihood of presence (suitability) for Franklin's ground squirrel (*Poliocitellus franklinii*, FGS) at Kankakee Sands and the surrounding landscape in northwest IN and northeast IL.

Data	Explanation	Source(s)
FGS capture/observation location	Animal data	IN, IL, IA, WI, and MO Natural History Inventories/Databases
FGS observation locations	Additional FGS observations	Global Biodiversity Information Facility
NRCS State by State	State boundaries	Natural Resources Conservation Service (NRCS) Geospatial Data
Cropland Data Layer by State	Detailed landcover	NRCS GDS
NRCS Conservation Easement Areas by State	Additional habitat	NRCS GDS
National Elevation Dataset 30 m	Influences burrow site selection	NRCS GDS
Slope	Influences burrow site selection	Created in ArcGIS Pro (Slope tool)
Gridded Soil Survey (gSSURGO)	Influences burrow site selection	NRCS GDS
TIGER Primary and Secondary Roads	Fragments habitat; Inhibits metapopulation	NRCS GDS
TIGER Streets	Fragments habitat; Inhibits metapopulation	NRCS GDS
MOD13Q1 v006	Additional general landcover	Application for Extracting and Exploring Analysis Ready Samples
Power transmission lines	Provide raptor perch sites	Homeland Infrastructure Foundation-Level Data (HIFLD)
Railroads	Positively-associated with FGS presence and	HIFLD

The modelling format I used is known as samples with data (SWD). SWD format creates models based on species sample locations and background points that contain environmental data values for the respective sets of data files. Users can then project these models with GIS environmental data to obtain pictures of model results across the geographic area of interest. This method is useful for reducing time running models of large environmental datasets (Philips 2017) and perform better for bias correction than the regular MaxEnt method of using a bias file (Mutascio et al. 2018).

The settings I used were: random seed, 5,000 iterations, a convergence threshold of 10^-5, cross-validation replicated run type, and a regularization multiplier of 3 (Mutascio et al. 2018). Using a regularization multiplier of three, instead of one, lowers the risk of model overfitting and also smooths model output across the area of interest (Elith et al. 2011, Merow et al. 2013). Subsample replicated run type samples the data without replacement based on the random test percentage, while cross-validation uses all the data to develop and test the models (Elith et al. 2011, Merow et al. 2013). Cross-validation is better for small datasets because it requires using all data for evaluation (Philips 2017). I created 10 replicates for each model to estimate errors of fitted functions and predictive performance on held-out data (Elith et al. 2011, Mutascio et al. 2018).

Model Evaluation

To evaluate these models, I used the area-under-the-curve (AUC) test, omission error, and one-tailed binomial tests. AUC measures the probability of a presence data point being ranked higher than a random absence point (Hovick et al. 2015), which I generated using my bias file. An AUC score of > 0.75 is considered useful for modeling purposes (Hovick et al. 2015).

MaxEnt generates omission error rates using suitability values ranging from 0 to 1 and then converts these into binary predictions of present or absent (Hovick et al. 2015). I obtained omission error rates (the proportion of presences in testing data associated with areas of predicted absences) based on the 10th percentile training presence threshold. This threshold provides omission of 10% of training presence data points to account for possible errors (Hovick et al. 2015). The one-tailed binomial test evaluated whether the models predicted present FGS locations better than random chance with the same area proportion predicted as “present” by the models (Hovick et al. 2015).

I chose to perform this project at the multi-state scale because of the decline and resultant scarcity of this species necessitate the inclusion of many states to obtain an adequate amount of data. I concluded the MaxEnt analysis by creating maps of the ″useful″ model outputs in ArcGIS Pro, as well as a map showing overlapping model outputs, that show binary, present/absent (present = can occupy, values ≥ 0.5; absence = cannot occupy, values < 0.5) predictions for the raster cells in this area of interest. I also created a model overlay map. I converted the ASCII files for model replicate averages created by MaxEnt to a "fuzzy membership" raster type using the Fuzzy Membership tool. In the Fuzzy Membership tool, I used the Linear membership type. I finished this portion by combining the fuzzy membership rasters for each model into an overlaid layer using the Fuzzy Overlay tool. In the fuzzy overlay tool I used the "Or" overlay type, which returned the maximum values out of the input raster sets.

Logistical and Operation Information Gathering

To obtain logistical and operational information for TNC to the potential translocation of FGS to Kankakee Sands, I used published literature, web resources, email, and telecommunications. This information was requested by John Shuey, the Director of Conservation Science for TNC. He requested information on necessary federal and state-level permits, locations of source populations, facilities (i.e. zoological parks) that could and/or are willing to house a captive source, release strategies, and an overview of potential genetic concerns. This information is located in Appendix B.2.

3.3 Results

After applying the spatial filter, I used 44 FGS locations from 2010–2021 in modeling attempts. Only the Universal and Anthropogenic-MODIS models were useful, with AUC > 0.75 (Table 3.2). The test omission error rates for both models with passing AUC scores were identical and low (0.0775; Table 3.2). However, the p-values for MaxEnt's binomial tests were not significant ($p > 0.05$), meaning the model predictions were not better than random.

Table 3.2. Statistics for Franklin's ground squirrel MaxEnt models of relative likelihood of occurrence at Kankakee Sands in northwest Indiana, USA. Omission error rates and p-values based on 10th percentile training presence training omissions as per Hovick et al. (2015).

	Mean Training AUC	Standard Deviation	Omission Error Rate	One-tailed Binomial Test P-value
Universal	0.753	0.127	0.0775	0.2778
Environmental	0.733	0.114	0.0775	0.2918
Anthropogenic-landcover	0.699	0.131	0.0723	0.2897
Anthropogenic-MODIS	0.757	0.107	0.0775	0.2421

The best predictor in the Universal model was Soils (28.2%; Table 3.3). The second best predictor for this model was Landcover (23.4%; Table 3.3). The third most influential variable was Slope (17.0%; Table 3.3). The least informative variable for the Universal model was Transmission Line Density (0.1%; Table 3.3).

Table 3.3. Percent contributions of environmental variables for MaxEnt models predicting Franklin's ground squirrel relative likelihood of occurrence at Kankakee Sands in northwest Indiana, USA based on state heritage program occurrence data.

	Elevation	**Slope**	**Soils**	**Landcover**	**Cons. Ease.**	**MODIS**	**Transmission Line Density**	**Road Density**	**Railroad Density**
Universal	1.7	17	28.2	23.4	1.1	9.0	0.1	9.9	9.6
Environmental	1.9	16.6	36.4	32.2	1.7	11.2	-	-	-
Anthropogenic-landcover	-	-	-	59.3	-	-	0.1	20.7	19.9
Anthropogenic-MODIS	-	-	-	-	-	30.0	0.1	44.4	25.5

The best predictor variable for the Anthropogenic-MODIS model was Road Density (44.4%; Table 3.3). The second best predictor was MODIS (30.0%), followed by Railroad Density (25.5%; Table 3.3). The least informative predictor was again Transmission Line Density (0.1%; Table 3.3).

Using ArcGIS Pro, I created a potential distribution map for each model (Fig. 3.3, Fig. 3.4). The maps are based on relative likelihood of occurrence on a zero to one scale, with zero least likely to support FGS. The overlay map displays areas of highest predictions for all models (Fig. 3.5).

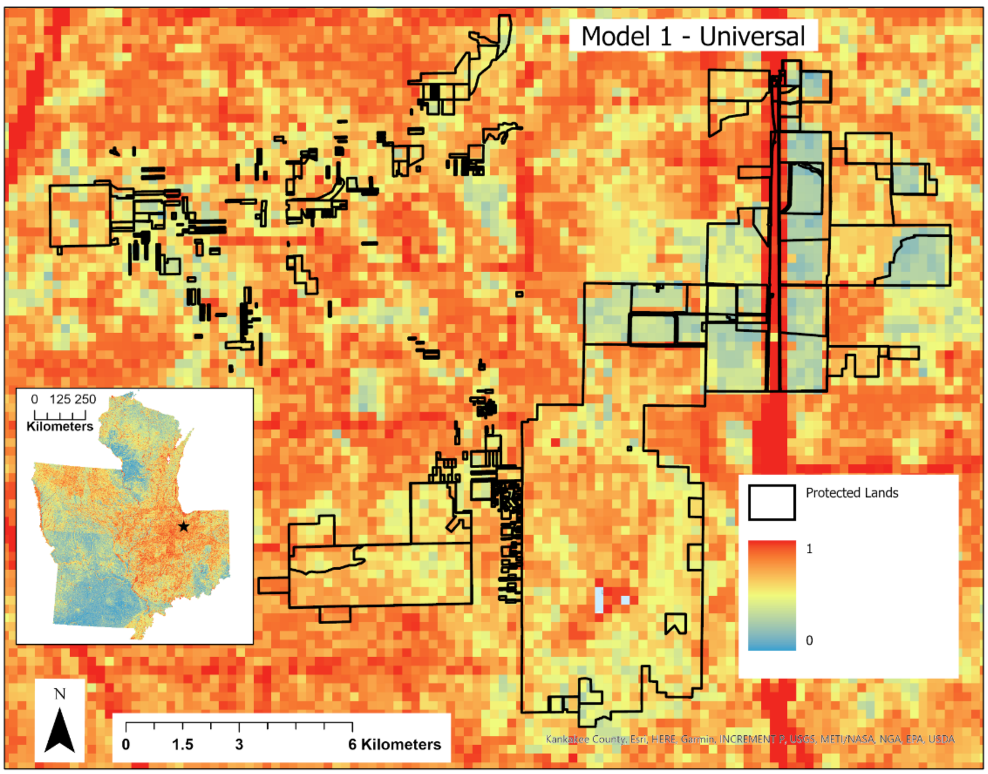

Figure 3.3. Map of the MaxEnt Universal model results for relative likelihood of Franklin's ground squirrel occurrence at Kankakee Sands and the surrounding protected lands in northwest Indiana and northwest Illinois, USA. The Universal model incorporated all environmental variables in its predictions. Warmer colors (shades associated with the number 1) indicate a higher relative likelihood of occurrence. Use of roadside surveys resulted in roads being predicted as "high relative likelihood of occurrence". Linear red predictions should be interpreted as high relative likelihoods along roadsides because Franklin's ground squirrels inhabit unmowed roadsides (Duggan et al. 2011). All boundaries are from Becker (2018).

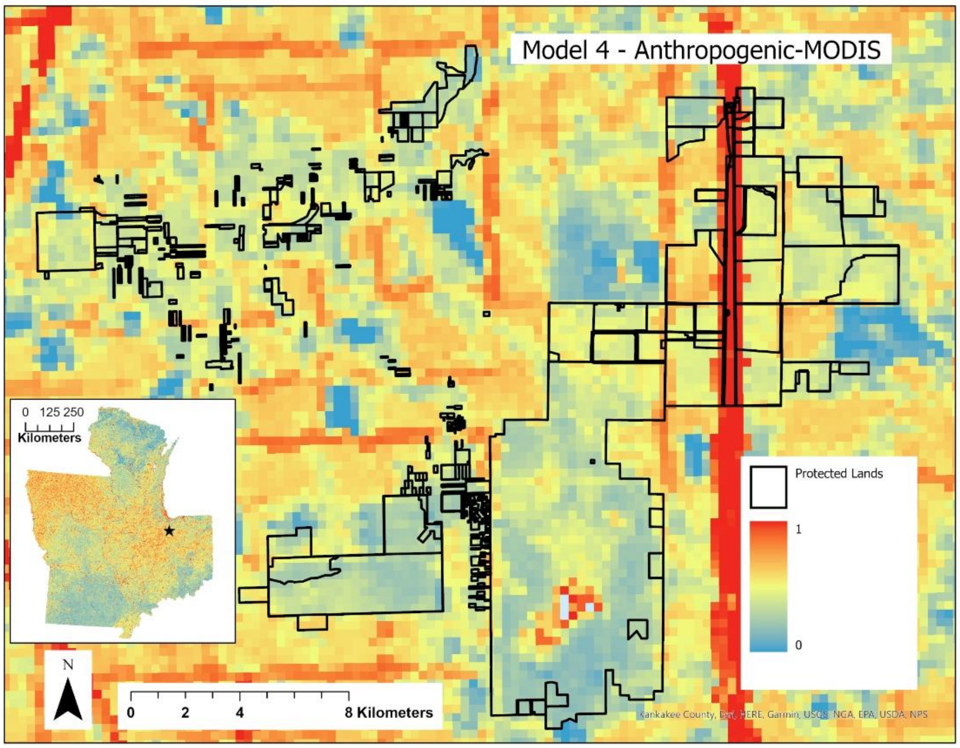

Figure 3.4. Map of the MaxEnt model results for relative likelihood of Franklin's ground squirrel occurrence at Kankakee Sands and the surrounding protected lands in northwest Indiana and northwest Illinois, USA. This model, Anthropogenic-MODIS, incorporated all anthropogenic environmental variables along with the MODIS land cover variable in its predictions. Warmer colors (shades associated with the number 1) indicate a higher relative likelihood of occurrence. Roadside surveys resulted in roads being predicted as "high relative likelihood of occurrence". Linear red predictions should be interpreted as high relative likelihoods along roadsides because Franklin's ground squirrels inhabit unmowed roadsides (Duggan et al. 2011). All boundaries are from Becker (2018).

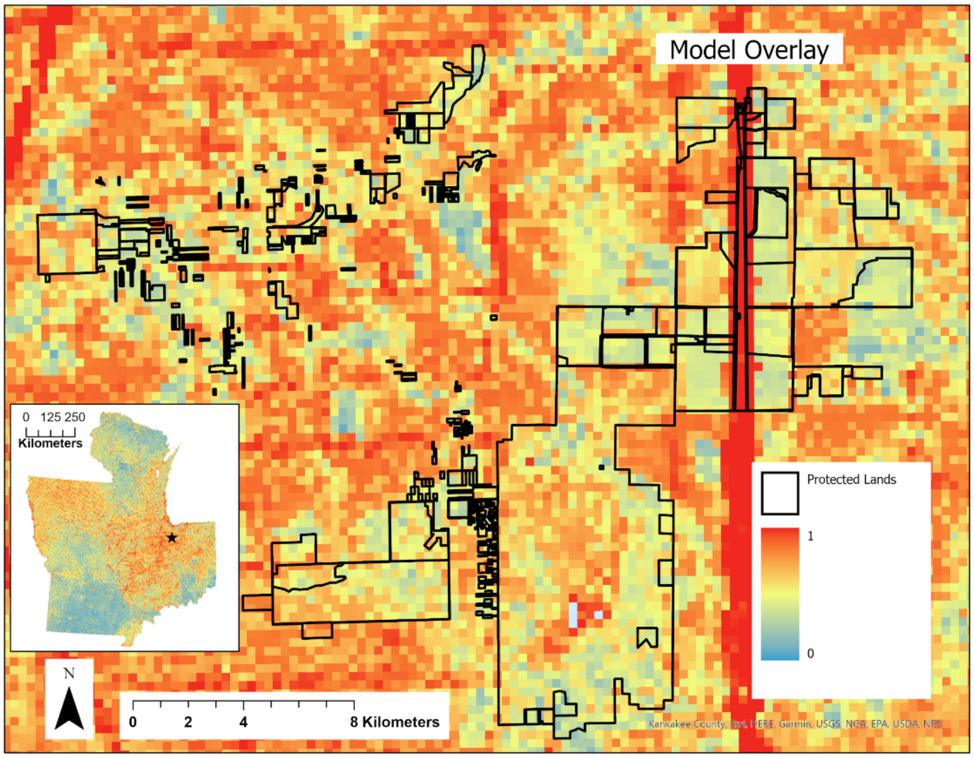

Figure 3.5. Map of the overlay of two MaxEnt models: Universal and Anthropogenic-MODIS, that predict relative likelihood of occurrence of Franklin's ground squirrel. Warmer colors (shades associated with the number 1) indicate a higher relative likelihood of occurrence. Roadside surveys resulted in roads being predicted as "high relative likelihood of occurrence". Linear red predictions should be interpreted as high relative likelihoods along roadsides because Franklin's ground squirrels inhabit unmowed roadsides (Duggan et al. 2011). All boundaries are from Becker (2018).

3.4 Discussion

Results for my prediction that the majority of the Kankakee Sands property and surrounding landscape would be suitable (suitability values < 0.5; arbitrary cutoff) for a FGS translocation attempt were inconclusive because my models performed poorly. Only two of four models were barely above the AUC cutoff threshold of ≥ 0.75. No model performed better than the null model according to the binomial tests (Table 3.2). Given that I only had 44 data points, more data would likely improve model outputs.

Soils and slope were the highly informative predictor for both of the passing models because soil type and slope influence burrow site selection in that FGS dig easier in sandy soils, while slopes promote soil drainage (Duggan et al. 2011; Table 3.3). The second best predictor was Landcover, the categorical habitat variable of the model. Landcover was likely important because

FGS are associated with grasslands and marshes (Martin and Heske 2005, Ostroff and Finck 2003). However, these most influential environmental variables were excluded from the Anthropogenic-MODIS model, Road Density became the best predictor variable. The second best predictor was the MODIS land cover variable, followed by Railroad density. Most data points were located along roads and railroad rights-of-way as a result of survey styles, but still Road and Railroad Densities were not very influential in model results when I included Soils, Slope, and Landcover in the Universal model (Table 3.3). FGS use railroad rights-of-way because they are often sloped and unmowed (Martin et al. 2003, Huebschman 2007, Young 2012). Lack of railroad rights-of-way prevalence throughout the five-state study area is likely a reason for low influence in the Universal model.

MODIS likely influenced Universal model results less than the Landcover variable because the Landcover variable consisted of clearly-defined categories while MODIS consisted of satellite imaging of NDVI spatial and temporal comparisons of vegetative land cover in blue, red, and near-infrared reflectance (Didan 2015, Hovick et al. 2015). Therefore, the Landcover layer had a clearer separation of habitat types versus the MODIS layer. Conservation Easements and Transmission Line Density in both models, where applicable, contributed minimally to model results. This is also likely due to low prevalence of conservation easement areas and transmission lines throughout the study area. Elevation in the Universal model was low because elevated areas present greater predation risk via greater visibility of FGS.

A majority of FGS occurrences were in areas with predicted relative likelihood of occurrence scores ≥ 0.5, MaxEnt′s baseline prevalence modeling input. I considered 0.5 minimum score of potential translocation success (Mutascio et al. 2018, Fitzpatrick et al. 2013). Given this, units B, E, G, M, N, and O of Kankakee Sands (Fig. 3.1, Fig. 3.5) would be the best sites to translocate FGS. Outside of Kankakee Sands, areas with predicted relative likelihood of occurrence ≥ 0.5 were Beaver Lake Nature Preserve, Conrad Savanna Nature Preserve, the north and central portions surrounding the wetland in Willow Slough Fish and Wildlife Area (WSFWA), the southern portion of the Iroquois County State Wildlife Area to the west of WSFWA, and scattered, smaller parcels of restored prairie northwest of Kankakee Sands (Fig. 3.1, Fig. 3.5). However, it is important to note that in both model outputs (Fig. 3.3, Fig. 3.4), the linear red predictions (indicating high relative likelihood of occurrence), are roads or roadsides. Biologists should interpret these areas of the map as roadsides with higher predicted relative likelihoods of occurrence because FGS inhabit unmowed roadsides (Duggan et al. 2011). Moreover, high relative likelihood predictions for the agricultural matrix are interesting because they usually prefer to cross through agricultural areas via unmowed roadside and railroad rights-of-way instead of open fields (Duggan et al. 2011). However, they will cross fields of corn, soybeans, wheat, and sunflowers (Choromanski-Norris et al. 1989, Niva 2010, Duggan et al. 2011; Fig. 3.3, Fig. 3.5).

It is worth mentioning that the environmental data did not reflect microhabitat assessments. Rather, they represent broad, macrohabitat classifications and characteristics. Therefore, I did not incorporate fine-scale habitat characteristics into my modeling and biologists should interpret results on a macrohabitat scale. Future modeling efforts that incorporate fine-scale habitat would likely have more informative results for a similar sample size.

The reintroduction of animals onto isolated habitat patches could pose challenges in terms of establishing a metapopulation between nearby habitat patches. The FGS supposed maximum dispersal limit of 5 km is a boon for this species in terms of promoting population connectivity between habitat islands in the region of proposed translocation (Duggan et al. 2011). I presume there is ongoing effort by TNC in both Indiana and Illinois, along with the Departments of Natural Resources of both states, to connect existing habitat patches to create larger, contiguous, or otherwise connected tallgrass prairie habitat patches in the area of proposed reintroduction. There is definitely potential for increased connectivity in the area via land acquisition by conservation organizations and landowner participation in Farm Bill programs such as the Conservation Reserve Program. Additionally, establishing captive populations in nearby zoological parks could help assuage the need for frequent population input by minimizing distance traveled between the translocation site and stable, wild source populations.

In general, adult FGS (males: 0.1–3.6 km; females: 65–450 m) typically disperse less than juveniles, who are more motivated to locate burrow sites and forage (3.6 km; Martin and Heske 2005). It is important to recognize the differences in dispersal distances for different age classes (juvenile vs. adult) to promote accurate assessment of landscape connectivity and promote population connectivity in a fragmented landscape.

Another concern specific to Kankakee Sands is the presence of a major highway bisecting the property. This busy road would present a higher mortality risk for translocated FGS trying to travel between halves of the property. Other, smaller, gravel roads fragment individual sections

(Fig. 3.1) of Kankakee Sands, and may present mortality risk and discourage dispersal as well. Kankakee Sands and the surrounding grassland properties are islands in a largely corn and soybean agricultural matrix. Although FGS mostly disperse via linear habitat patches along roadsides instead of open fields (Choromanski-Norris et al. 1989), my models predicted high relative likelihoods of occurrence in the non-grassland areas (Fig. 3.3). Duggan et al. (2011) observed that FGSs perceived higher predation risk in agricultural fields than in prairie patches based on giving-up densities (GUDs). They also observed that while FGSs mostly used unmowed roadsides (maximum observed distance for travelling along an unmowed roadside was 130 m), this species will cross open fields (maximum observed distance across an open field 240 m) when energetic constraints are greater than the risk of predation. Moreover, FGS will cross fields of corn, soybeans, wheat, and sunflowers (Choromanski-Norris et al. 1989, Niva 2010, Duggan et al. 2011). These facts may explain model predictions of high relative occurrence likelihoods in agricultural fields.

There are several potential solutions to improve predictive power of future modeling attempts. The first one relates to the species' cryptic nature (i.e., their occurrence at low densities in the eastern portion of their range, secretive behavior, occupation of burrows located in dense vegetation, and tendency to hibernate 7 – 8 months out of the year; Ostroff and Finck 2003, Duggan et al. 2011). Cryptic species are, by nature, more difficult to detect. Although absence data are generally more difficult and expensive to collect on a usable, state-wide scale (Philips and Dudík 2008, Hovick et al. 2015), biologists should use a SDM modeling approach that incorporates both presences and absences, such as Generalized Linear Models, to better predict occurrence probabilities for cryptic species (Brotons et al. 2004, Carlos-Júnior et al. 2020). Second, state natural history inventory biologists might expand their FGS sampling efforts to new areas and beyond roadside surveys, as possible, in order to generate more FGS data points and consequently increase future sample sizes. Second, biologists could expand modeling efforts to range-wide for the species. Finally, biologists could use FGS data points from before my sample period of 2010 – 2020 while checking to ensure all points are within suitable habitat classes. Or, if a given point is within an agricultural field, verifying that the landcover class did not change over time from the time of data point entry based on earlier versions of the landcover layer.

I acknowledge limitations behind SDMs. The first is that undetected individuals are not accounted for. The second is that mortality sources such as predation and vehicle collisions are not accounted for. The third is the assumption of a detection probability that is constant across the landscape (Merow et al. 2013). These factors have to be excluded due to the simplicity of the modeling design, which only accounts for land use to predict habitat suitability. However, I was satisfied with primarily evaluating existing habitat and land use as the predictor for species occupancy because habitat destruction due to increased resource depletion and land use conversion from a growing human population is the main factor in global biodiversity loss (Singh 2002). Additionally, the lack of data of undetected individuals and the inability to incorporate mortality sources into evaluations should not result in inaction in regard to modeling for conservation purposes. With respect to these concerns, other researchers have successfully used MaxEnt for purposes similar to mine (Cilliers et al. 2013, D'Elia et al. 2015).

Use of elevation and slope as predictor variables was a concern because the topography of northwest Indiana and northeast Illinois is relatively flat, therefore I assumed elevation and slope would not have much local predictive effect. However, I ultimately included these predictors given that the study area includes all of Indiana, Illinois, Iowa, Missouri, and Wisconsin. Incorporating the five-state area meant that the elevation and slope varied more over such a large distance and had a greater predictive input toward model results than only examining local conditions.

Ideally, I would reduce spatial autocorrelation among environmental variables when performing SDM. Although I was unable to successfully account for spatial autocorrelation using the diagnostic measure, I was confident with continuing the project given my incorporation of the biologically-relevant spatial filtering distance used to promote independence of FGS observation points. I believe my results are still useful given that other studies used MaxEnt analyses to assess species distributions for potential future reintroductions and did not account for potential spatial autocorrelation whatsoever (Adhikari et al. 2012, Hendricks et al. 2016), whereas I performed 2 extensive attempts to detect and minimize this issue. Moreover, regularization in MaxEnt increases its stability regarding correlated environmental variables, which is related to spatial autocorrelation (Elith et al. 2011).

An a-posteriori observation is that over 220 m of raster resolution was lost changing the soils layer resolution from 10 m to 231.66 m to match the MODIS layer resolution. This may explain why the correlograms' and semivariograms' spatial filtering distances were as large as they were. In future modeling efforts for FGS, I recommend excluding MODIS as a predictor in order to preserve as much data and resolution as possible in all environmental data layers.

I did not correct for spatial autocorrelation because these potential spatial filtering distances were large enough to leave only three data points. However, regularization in MaxEnt increases its stability regarding correlated environmental variables, which is related to spatial autocorrelation (Elith et al. 2011). Instead, I used the maximum suspected dispersal distance as a spatial filter to ensure independence of data points between the FGS location data used in MaxEnt.

MaxEnt estimates absolute occupancy only if the user provides presence-absence data (Guillera-Arroita et al. 2014). When working with presence-only and background data, like in this project, MaxEnt estimates relative likelihood of occurrence (Guillera-Arroita et al. 2014, Merow and Silander Jr. 2014, Perkins-Taylor and Fray 2020). Relative likelihoods of occurrence are proportional to actual probabilities of occurrence, so therefore are still useful (Guillera-Arroita et al. 2015, Merow and Silander Jr. 2014). Moreover, biologists can interpret my map results as an index of habitat suitability on a zero to one scale (Perkins-Taylor and Fray 2020; Fig. 3.3, Fig. 3.4, Fig. 3.5).

Modeling with a finer-scale soils layer from not including MODIS as well as more data in future modeling attempts would be beneficial to obtaining more conclusive results. No models performed better than the null, despite having passing AUC scores. From the results of this study, the feasibility of FGS translocation to Kankakee Sands and the surrounding landscape in terms of my macro-habitat assessment is inconclusive. Despite the model results, I believe reintroduction the area would be worthwhile in terms of establishing a new population of a declining Indiana state-endangered mammal, which would bolster nongame conservation efforts for this species, along with continuing to complete the restoration of the tallgrass prairie ecosystem already begun by TNC.

3.5 Management Implications

I had five site-specific management practices of Kankakee Sands and one region-specific suggestion regarding a potential translocated population of FGS. The first suggestion to consider is removing all fencerows and unused power lines as possible in order to minimize available perch sites for predatory birds. Second, it may be beneficial to remove or close (permanently or temporarily) any county roads in order to increase habitat connectivity, but the political feasibility of this is low. Furthermore, given Kankakee Sands is divided into multiple fragments by numerous county roads, these roads would present an additional mortality hazard for any translocated FGS.

Third, managers might want to avoid spring season prescribed burning or limit the application area because FGS awake from hibernation from April to early June and will need food upon emerging from their winter dens.

Fourth, related to available food after hibernation emergence, I suggest planting native, cool-season grasses to provide additional spring food sources. Cool-season grass selection is important because potential FGS habitat is positively influenced by smooth brome (*Bromus inermis*) cover, a non-native, invasive cool-season grass (Duggan et al. 2011). Some alternative, native cool season grasses include Canada wildrye (*Elymus canadensis*) in uplands, Virginia wildrye (*Elymus virginicus*) in bottomlands, and bottlebrush grass (*Elymus hystrix*, INDNR 2004, Southwestern Indiana Master Gardener Association 2021). An additional suggestion is to continue acquiring land and connect habitat patches as possible to further facilitate population/habitat connectivity in Kankakee Sands and the surrounding area. My final suggestion was the implementation of a wildlife underpass under US-41, if possible, to reduce FGS mortality risk, as ground squirrels use these artificial crossings (Murphy 2011).

Our regional management suggestion is to work with county-level road maintenance to leave roadsides unmowed to promote landscape connectivity. I suggest this because FGS use linear habitat patches (unmowed roadsides) to disperse more than crossing open fields (Choromanski-Norris et al. 1989) and are critical for dispersal (Martin et al. 2003). It is worth reiterating that FGS will cross fields of corn, soybeans, wheat, and sunflowers (Choromanski-Norris et al. 1989, Niva 2010, Duggan et al. 2011). Given this, encouraging local farmers to plant wheat as an off-season cover crop would boost landscape connectivity for this species where grassland restoration is not yet feasible. Land managers could work with the Illinois and Indiana Departments of Natural Resources to implement other, similar management practices as deemed possible and appropriate. Finally, in the event that FGS are translocated to the Kankakee Sands area, Greenwood et al. (1998) suggests FGS might benefit from food provisioning in late summer when they accumulate most of their fat reserves that helps increase overwinter survival.

CHAPTER 4. EXTENSION DELIVERABLE

4.1 Restore Prairie on Your Property to Protect History, Wildlife, and Humanity

Landowners in northwest and west-central Indiana have the opportunity to create significant real-world change on their properties. This can be done through the restoration of tallgrass prairie habitat in its historical range. But what exactly is special about tallgrass prairie?

Tallgrass prairie is defined as “a fire-dependent ecosystem distinguished by tall grasses (up to 10 feet tall), and deep, rich soils” (USFWS 2020). “Fire-dependent” means that fire prevents conversion of prairie to forest or other habitat dominated by woody species. This habitat provides numerous benefits for humanity and the environment. These processes largely stem from the deep root systems of prairie plants (University of Northern Iowa 2020). Said processes include: erosion control, rainfall and runoff filtration, invasive weed control, and carbon storage (University of Northern Iowa 2020).

Tallgrass prairie once covered 170 million acres of North America (National Park Service 2018). It ranged north to south Manitoba, Canada, south to north Texas, west to central Oklahoma, and east to northwest and west-central Indiana (Fig. 4.1). Because of conversion to agricultural fields beginning around 150 years ago, less than 6.8 million acres remain intact (National Park Service 2018, National Park Service 2020). That is an equivalent decrease in land cover of over 7 times the size of Indiana to only 3/10 of Indiana.

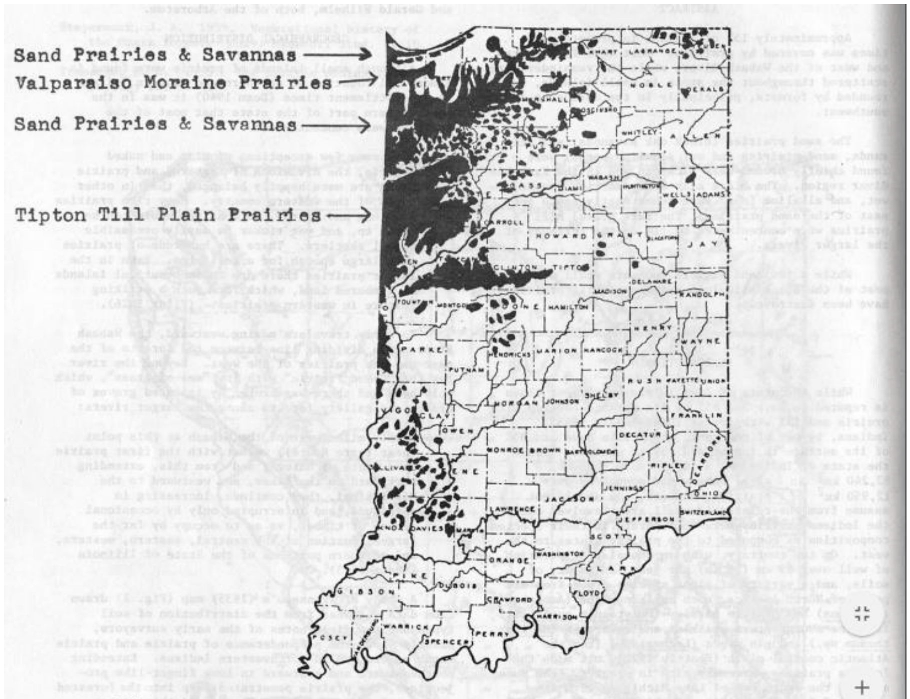

Fig. 1. Map of Indiana showing the distribution of major types of prairies and savannas in presettlement times. Modi´_ l from E.N. Transeau (1935), Ecology 16: 423-457.

Figure 4.1. Distribution of major types of prairies and savannas in presettlement times. Source: Indiana Soils: Evaluation and Conservation Manual. Used with permission (Betz 1976).

Most of what remains of this habitat is in the Kansas Flint Hills (National Park Service 2020). In Indiana, our tallgrass prairies once covered 15% of the state, or about 3.5 million acres (Fig. 4.2; Betz 1976).

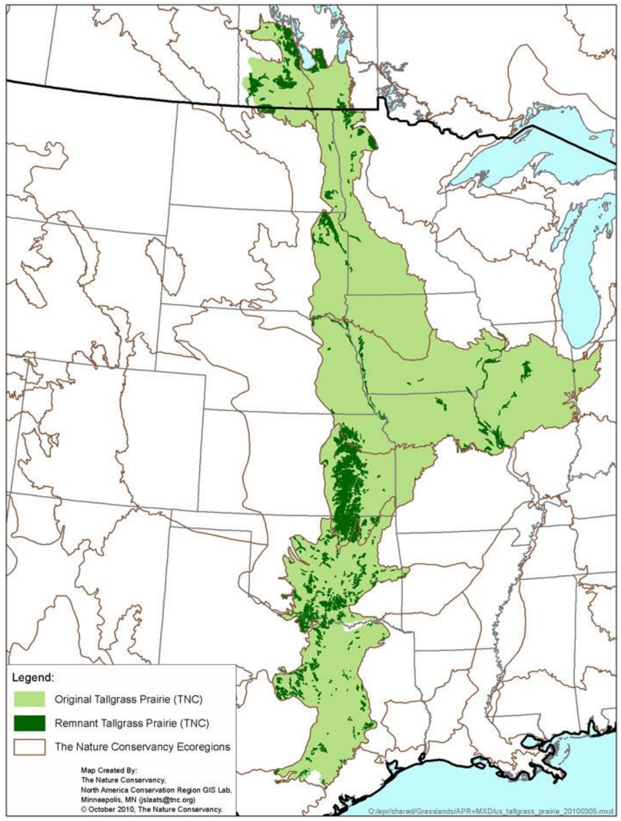

Figure 4.2. Current and historical distribution of the Tallgrass Prairie. Map courtesy of The Nature Conservancy, Minneapolis, MN (Jacques 2019).

In 2009, there was only a few hundred acres of tallgrass prairie in Indiana (Runkel and Roosa 2009). Today, there are over 22,500 acres in Indiana due to restoration efforts at Oak Ridge Prairie, Kankakee Sands, Willow Slough Fish & Wildlife Area, Beaver Lake Nature Preserve, Conrad Savanna Nature Preserve, Hoosier Prairie, and several smaller private land restorations (Lake County Park and Recreation 2020, TNC 2020, TNC 2020). These restorations are fragmented and separated by many miles of agricultural, urban, and suburban land.

The tallgrass prairie was also an ecosystem inhabited by many wildlife species. Larger animals including bison, wolves, elk, and deer once roamed these prairies. Moreover, so did smaller animals such as greater prairie-chicken and pocket gophers, to name a few (Jacques 2019, National Park Service 2018). These animals have the potential to be reintroduced into our state.

As was alluded to earlier in this article, reintroduction is possible only by first restoring the tallgrass prairie habitat within its historical range. Hoosier landowners have the ability to facilitate the return of these animals and processes to their land. This would thereby help to restore and protect Indiana's natural heritage. Moreover, the restoration of tallgrass prairie could directly benefit humanity by increasing food security. This would be accomplished by providing habitat for pollinators (declining insect species that pollinate crops such as cranberries, apples, plums, and more), game species (such as the greater prairie-chicken and elk), and "livestock" (i.e. American bison; Tangley and NWF 2020).

One method for landowners to restore tallgrass prairie is through a federal program that assists landowners in restoring prairie (and other) habitat on their own land. Said program is the Conservation Reserve Program (or CRP for short). According to the USDA Farm Service Agency's website, "CRP is a land conservation program" where "in exchange for a yearly rental payment, farmers [and landowners] enrolled in the program agree to remove environmentally sensitive land from agricultural production and plant species that will improve environmental health and quality" (USDA Farm Service Agency 2020).

Alan Mathew, who is a professor in and head of the Department of Animal Sciences at Purdue University in West Lafayette, IN, landowner in northwest Indiana, and former farmer of about 20 years, is also an active participant in the CRP.

Mathew owns 4 properties totaling 320 acres in the highly-agricultural White County Indiana. "Most of the land is under crop production under cash rent leases … with the remainder in prairie wildlife habitat, ponds, or woodlands." He enrolled 16 acres in CRP and CREP (Conservation Reserve Enhancement Program; another program that is part of the CRP). He has decided to keep 31 acres as wildlife habitat without participating in CRP. Twenty-one of these acres he keeps as tallgrass prairie and savanna – the transition from woodland to prairie – habitat.

As an avid hunter, he decided to enhance and restore wildlife habitat on his own properties. In 2011, he reached out to a local wildlife biologist for advice. The wildlife biologist told him about the CRP, the CREP, respective program benefits, as well as proper plant species and management tips. Since enhancing and restoring wildlife habitat in a land mostly devoted to crop and wind energy production, he noticed an increase in wildlife.

He has observed an increase in the number and species of songbirds and mammals. Moreover, he noticed a more consistent presence of ring-necked pheasant, northern bobwhite, and red fox. He also noticed more white-tailed deer in his plots that are amidst row crops and far from any other cover. Additionally, plants such as milkweed have established themselves in areas that he does not burn, mow, or spray. As a result, he has noted a greater variety of insects, including:

monarch butterflies, praying mantis, and fireflies in those areas. Lastly, the presence of a vernal pond (present only during periods of heavy rainfall) and a permanent pond on two of his properties provided habitat for new populations of amphibians and aquatic turtles.

Mathew also explained some perceived pros and cons of participating in the CRP. His main pro was the return of wildlife. Other pros included consistent annual payments throughout the contract compared to uncertainties in crop production conditions and crop sale prices. He also said CRP payments are comparable to cash rent values in his area. Another pro was the optimization of income potential from what would have been marginal farmland.

Cons included potential lost profit during optimal farming years and/or on high quality farmland (compared to marginal farmland). Another con is the required maintenance. Such maintenance includes invasive plant species control by annual selective spraying, prescribed burning every few years, and occasional tilling and replanting as needed to maintain forb – non-woody, flowering plants that are not grass- or reed-like – populations. Regardless of these pros and cons, Mathew stated that the economic situation is unique to each landowner.

Mathew shared some advice for landowners interested in participating in the CRP. For maintenance, he recommended that landowners consider whether they will perform the maintenance themselves or hire others instead. Other advice included to consider one's own goals, type(s) of land owned. This would be in regard to which conservation program(s) they would qualify for and which land would have the greatest impact for wildlife, soil, and water conservation/enhancement. Finally, he recommended that aspiring CRP participants contact their local Soil and Water Conservation Service, county Extension office, wildlife biologist, and local active CRP participants for an abundance of free information and personal insights.

"Once you've made the decision, enrolled in and implemented the program: Enjoy the benefits, including a greater variety and number of wild animal and plant species, and the satisfaction of contributing to the conservation and biodiversity on your own land, as well as that of the surrounding area." – Alan Mathew

Individuals who are interested in learning more about CRP and other conservation programs, view this webpage:

https://www.fsa.usda.gov/programs-and-services/conservation-programs/index

Purdue University posted a webpage with numerous sources of prairie seed vendors:

https://www.purdue.edu/hla/sites/yardandgarden/prairie-wildflowers/sources-of-seed-and-plants-of-prairie-wildflowers/

Information on prescribed burning is here: https://www.tallgrassrestoration.com/ourservices/prescribedburns

CHAPTER 5. CONCLUSION

Part of the solution to the global biodiversity crisis is performing reintroductions of declining species to their historical ranges where these species have been extirpated. Continuing to assess and model habitat suitability for target species is one of the first steps toward getting animals on the ground. I attempted such an assessment for my two species of interest, GPC and FGS. More data likely would have resulted in statistically-stronger models all around. On this note, I am disappointed in the lack of conclusive results for the FGS. However, I am satisfied that at least the Universal model for the GPC is considered useful and statistically significant. I am hoping that TNC will take these promising results, my management suggestions, and the species life history information as well as logistical and operational constraint information regarding reintroductions that I have compiled for them in order to implement a reintroduction program for the GPC.

Related to a lack of species location data, I am also hoping that my extension deliverable article will inspire landowners in northwest Indiana to restore all or a portion of their lands to tallgrass prairie. Restoration of habitat is another crucial step toward enabling successful reintroductions. All-in-all, I am proud to have had the opportunity to potentially advance conservation efforts through my work here at Purdue University.

APPENDIX A. LITERATURE REVIEWS

A.1 Greater Prairie-Chicken

GPC Decline

The GPC is a North American grassland bird that is experiencing population declines (Johnson et al. 2011). Most subspecies of GPC are classified as "vulnerable" to extinction on IUCN's Red List (BirdLife International 2016). However, one subspecies, Attwater's prairie-chicken (*Tympanuchus cupido attwateri*) is listed as "endangered" under the U.S. Endangered Species Act (U.S. Fish and Wildlife Service 2020). Prior to European settlement, the GPC's range centered in the tallgrass prairie habitat throughout Minnesota, south to northern Texas, east to northwestern and west-central Indiana, and west to Nebraska and Kansas, USA (Johnson et al. 2011). Post-European settlement, the GPC expanded its range to Alberta, Canada, south to northern Mexico, west to Colorado, and east to Ohio, due to land clearances for agricultural purposes (Johnson et al. 2011). This species is currently distributed in Colorado, Kansas, Nebraska, South Dakota, and North Dakota, as well as smaller ranges in Minnesota, Oklahoma, Missouri, and Illinois, USA (Johnson et al. 2011).

Although historical abundances are mostly unknown, GPC numbers declined from 500,000 in the early 1970s to 200,000 birds in the late 1990s (Johnsgard 2002). Johnson et al. (2011) stated that habitat loss via fragmentation, fire suppression, agricultural practices, and urban/suburban sprawl. However, the GPC persisted in marginal farmland and prairie remnants in and around Kankakee Sands in Newton County, Indiana, until around the 1970s (Indiana Department of Natural Resources Division of Fish and Wildlife [INDNRDFW] 1999, Mumford et al. 2019). Today, the estimated density is 0 – >10 birds/100 ha for males and 2.5 birds/ha for females in prairies where the species remains (Johnson et al. 2011).

General Habitat

The GPC primarily inhabited tallgrass prairie, oak-savanna, and aspen parkland habitats (Aldrich 1963). Of these three habitat types, tallgrass prairies and oak-savannas were historically habitat for GPC in large areas of northwestern Indiana (Homoya et al. 1985). Today this area of

Indiana is mostly agricultural fields and wind farms, aside from a few tallgrass prairie and oak-savanna remnants and restorations. However, undisturbed (tall and dense) tallgrass prairies were likely not ideal habitat for GPCs (Svedarsky et al. 2003). This suggests that these birds relied on disturbances, such as fire and/or large ungulate grazing to maintain optimal habitat.

Within tallgrass prairie and oak-savanna habitats, GPCs require specific resources and features of these habitats in order to fulfill their life histories. Individuals and populations in different parts of their range need different total areas of habitat (Table A.1). It is unclear if Walk and Warner (1999) are referring to the required area for one bird or more than one. Because the area is small, I assumed it was a recommendation of minimum area per bird. It is also important to state that GPCs in the eastern portion of their range can survive on smaller habitat patches than GPCs in the west (Svedarsky et al. 2003). Currently in Illinois, there are ~200 GPCs of both sexes persisting, with human assistance, on 2 intensively managed reserves of 165 ha and 65 ha, respectively (Illinois Department of Natural Resources [ILDNR] 2018, ILDNR 2020, ILDNR 2020, Svedarsky et al. 2003). Moreover, each total reserve area is divided among 3 habitat fragments. GPCs in these reserves are threatened with low genetic diversity, lek disruption and nest parasitism by ring-necked pheasants (*Phasianus colchicus*), and stochastic events such as hail storms and droughts (ILDNR 2020). These threats have warranted numerous supplemental introduction efforts in Illinois (ILDNR 2020).

Table A.1. Various suggested minimum habitat areas based on different studies. Recommended minimum areas for individuals and populations of Greater-Prairie Chickens are based on observations of the birds in Wisconsin and Illinois, USA.

Study	Location	Individual/Population?	Area (ha)
Walk and Warner (1999)	Illinois, USA	Individual	65
Svedarsky et al. (2003)	Wisconsin, USA	Population	1020-6410
Svedarsky et al. (2003)	Illinois, USA	Population	610
Ryan et al. (1998)	Missouri, USA	Population	65

In Illinois, Sanderson et al. (1973) recommended a ‘‘scatter-pattern’’ approach to managing the GPC, where they targeted establishment of 16- to 64-ha grasslands. They speculated that a single grassland tract would support fewer prairie-chickens than scattered grasslands. Both prairie-chickens (*Tympanuchus* spp.) make extensive use of intervening agricultural lands for foraging and brood-rearing, and therefore scattered grassland tracts likely support more individuals of these species than a single tract might.

Lekking and Lek Sites

There are several subclasses of habitat with varying vegetation heights that must be present in order for the GPC to complete its life history (Table A.2). To preface the first subclass, GPC reproduce through a polygynous system known as lekking (Svedarsky et al. 2003), which is a reproductive strategy that involves multiple males occupying a "booming ground" where they use a variety of visual and auditory displays to attract females (Johnson et al. 2011). Lek sites are defended by dominant males and females (Johnson et al. 2011). This behavior typically occurs March-May (Robb and Schroeder 2005).

Table A.2. Vegetation structure summary for different habitat subclasses. I summarized the following information from Svedarsky et al. (2003). They obtained this information by summarizing multiple studies that performed vegetation surveys across the range of the Greater Prairie-Chicken in North America. I listed measurements from tallgrass prairies in an effort to retain relevance to my project.

Habitat Subclass	Vegetation Height (cm)
Lek	0-15
Nesting	36-74
Brood-rearing	Unspecified
Roosting (spring and fall)	3.9-5.3
Roosting (summer)	31.8 (within vegetation 37-52)
Roosting (snow present)	13.2 (within vegetation 31.9-35.4)
Foraging	Any (provided food is present)

Lek sites ("booming grounds") are selected based on maximizing visibility to females, with areas of bare to low ground cover (< 15 cm) being best while proximity to nest habitat is also important (Svedarsky et al. 2003). According to Winder et al. (2015), lek longevity and population viability depend upon the availability and proximity (< 5 km) of nesting habitat to lek sites. The most suitable lek sites are on higher elevation relative to the rest of the area. Moreover, since grouse species tend to avoid roads, power lines, and wind turbines, proximity to these may negatively influence lek site selection (Hovick et al. 2015).

Nesting and Nesting Habitat

In Illinois, nesting season typically occurs May-July (Svedarsky et al. 2003). GPC select nesting sites, based on cover characteristics: 36-74 cm average, vertically dense, and surrounded by open area for optimal vigilance (Svedarsky et al. 2003). Typical clutch size is ~12 eggs for first nest attempts and ~10 eggs for renests (Robb and Schroeder 2005). Incubation is 23–25 days and performed by females only (Johnson et al. 2011). Relative to environmental hazards to nesting, proximity to wind turbines does not affect nest site selection or nesting success (Harrison et al. 2017).

Brood-rearing and Brood-rearing Habitat

After the eggs hatch, GPC females and chicks will move anywhere from 0.4 – 10.3 km from the nesting site to brood-rearing habitat, based upon several criteria (Svedarsky et al. 2003). Such criteria for this subclass include lowland topography and recent (< 1 year ago) disturbance through burning, grazing, or haying (Svedarsky et al. 2003). Given that chicks primarily eat insects and are more susceptible to various mortality sources than adults, brood-rearing habitat must also provide: access to the nesting area, areas for ground-level movement, areas for sunning and dusting, adequate insect abundance, and protection from weather and predators. Moreover, lowlands are associated with lower predation risk compared to uplands (Svedarsky et al. 2003). The most important factors in GPC population longevity are nest establishment, hatching success, and chick survival (Robel, R. J., Zilkha Renewable Energy, unpublished report), so these habitat subclasses should receive the greatest management attention.

Roosting and Roosting Habitat

GPC also exhibits day-resting and night-roosting behavior (Jones 1963). Day-resting areas preferred by GPC are areas of continuous cover (Jones 1963). This gregarious bird forms mixedsex flocks, with higher aggregations in colder seasons (Johnson et al. 2011). However, night roosting consists of individual birds (Lehmann 1941). These birds may be spaced at least 30 cm apart (Lehmann 1941), but little information is known otherwise about spacing of these birds (Johnson et al. 2011). Night-roosting habitat varies by season (Jones 1963). This habitat is almost exclusively on the ground (Svedarsky et al. 2003). Significantly, GPCs spend more than ½ of their lives in their night roost areas (Svedarsky et al. 2003). They also shift night roost sites successively, averaging almost 1 km apart (Svedarsky et al. 2003).

In the spring and fall, the roost vegetation is short (a range of 3.9-5.3 cm), as is the surrounding vegetation (mean ~50 cm; Jones 1963). When snow is on the ground (winter and early spring), GPCs night roost in areas of vegetation averaging 13.2 cm tall surrounded by higher vegetation (a range of 31.9-35.4 cm; Jones 1963). Summer night roosts averaged 31.8 cm surrounded by vegetation ranging 37-52 cm high (Jones 1963). GPCs will also fly up to 1.61 km or more to arrive at wetland areas to use as night roosts, though lowlands with dry areas are adequate (Svedarsky et al. 2003). GPCs seek denser vegetation to avoid temperature and precipitation extremes during summer and winter. They also seek denser vegetation to reduce predation risk (Svedarsky et al. 2003). GPCs are also known to roost in snow drifts, often by burrowing (Jones 1963, Svedarsky et al. 2003). The recommended arrangement of these habitat subclasses is that they should be located within 8 km of each other.

Diet, Feeding, Flocking, and Associated Habitat

GPCs are most likely to feed in flocks during dawn and dusk (Robb and Schroeder 2005). In seasons where snow is present on the ground, foraging habitat must possess adequate shrub and tree buds as well as various seeds (Svedarsky et al. 2003). It is worth mentioning that GPCs will often visit adjacent agricultural fields during these seasons to glean leftover crops (Svedarsky et al. 2003). GPCs will also travel multiple kilometers to reach supplemental feeding areas. This increases mortality risk (Svedarsky et al. 2003). It would be worthwhile to supplement their diet if snow-season mortality is high until adequate food sources become available in reintroduction areas. It is also important to note that GPCs can perform flights >11 km between habitat patches (Leopold 1931). During non-snow seasons, foraging habitat must provide adequate greens, forbs, arthropods, and supplemental agricultural crops, depending on the season (Svedarsky et al. 2003).

According to Hamerstrom et al. (1941), Korschgen (1962), and Svedarsky et al. (2003), natural winter diet items include browse such as catkins of American hazel (*Corylus americana*) and birch (*Betula* spp.), buds of maple (*Acer* spp.) and blueberry (*Vaccinium pennsylvanicum*). Fruits and mast include rose hips (*Rosa* spp.), berries of smooth sumac (*Rhus glabra*), and acorns (*Quercus* spp.). Greens include leaves from *Rubus* spp., clovers (*Trifolium* spp.), and grasses. GPCs also consume various seeds of species considered agricultural weeds including ragweed (*Ambrosia* spp.) and false buckwheat (*Polygonum* spp).

Rose hips and the buds and catkins of woody species compose the majority of GPC natural winter food items (Svedarsky et al. 2003). Agricultural foods include corn and soybean gleanings (Svedarsky et al. 2003). These food items can make up the majority of GPC winter diet when present, which makes them limiting food items for the population during this season (Svedarsky et al. 2003). However, another natural alternative does exist. Svedarsky et al. (2003) explained that native prairie forbs are major winter food sources for GPCs in tallgrass prairies. However, forbs have been increasingly targeted by herbicide applications and frequent prescribed burns that shifted prairie composition to mostly grasses (Svedarsky et al. 2003).

Regarding diet during the remaining 3 seasons, corn is a leading food item. In terms of natural foods, GPCs will consume rose hips and buds and catkins and buds of woody species (Svedarsky et al. 2003). In spring, GPCs will primarily consume sedges (*Carex* spp.), green grasses, soybeans, and broom-corn (*Sorghum bicolor*; Korschgen 1962). In the summer, in order of prevalence, GPCs will eat wild seeds, fruits, browse, insects, grain, vegetable debris, and mast (Korschgen 1962). However, arthropods composed > 80% of GPC diet from June-August in the tallgrass Sheyenne National Grassland of North Dakota (Rumble et al. 1988). This suggests GPCs take advantage of the most abundant food sources in an area. So, the relative importance of food appears to change depending on population location.

In fall, from most consumed to least, GPCs will consume: grasshoppers (Acrididae), ragweed, oats (*Avena sativa*), clover, greenbrier (*Smilax* spp.), dogwood (*Cornus* spp.), crickets (Grylidae), buckwheat (*Fagopyrum sagitattum*), *Rubus* spp., blueberries, rose, hawkweed (*Hieracum canadense*), and chokeberry (*Aronia* spp.; Korschgen 1962, Svedarsky et al. 2003). Year-round, GPCs obtain most of their needed water through their diet and dew on vegetation, except in droughts when surface water is used (Svedarsky et al. 2003).

Exotic Species Impacts on GPC

Short (2009) claimed that it is important to consider the impact of non-native species in order to improve the success of a reintroduction effort. In response to this, I outlined competition of GPCs with a non-native game bird: the ring-necked pheasant (RNP). RNP hens are nest parasites and cocks harass displaying GPC males at booming grounds (Vance and Westemeier 1979). Svedarsky et al. (2003) stated that RNP hens parasitize GPC nests, decreasing the productivity of nests due to embryo mortality, nest abandonment, or shortening incubation period. They stated that shortening of incubation occurred due to RNP eggs hatching sooner than GPC eggs. Harassment of displaying GPC males by RNP males has significant effects regarding disruption of dominance hierarchies and copulation success at smaller leks (Vance and Westemeier 1979). It is recommended that RNPs not occur within 16.1 km of areas occupied by GPCs (Vance and Westemeier 1979). If elimination of RNP is not an option, reduction in RNP density is also beneficial (Robb and Schroeder 2005).

Predation and Competition

GPCs have various predators. According to Svedarsky et al. (2003) and Johnson et al. (2011), nest predators include American crow (*Corvus brachyrhynchos*), great horned owl (*Bubo virginianus*), striped skunk (*Mephitis mephitis*), Virginia opossum (*Didelphis virginiana*), coyote (*Canis latrans*), northern raccoon (*Procyon lotor*), and red fox (*Vulpes vulpes*). They also stated that predators of adult GPCs include various hawks (*Accipiter* spp. and *Buteo* spp.), and other raptors such as great horned owl, peregrine falcon (*Falco peregrinus*), and bald eagle (*Haliaeetus leucocephalus*). Furthermore, land-based predators include red fox and coyote, in addition to feral cats *(Felis catus*) and dogs (*Canis lupus familiaris*). GPC populations in fragmented grasslands with more woody and non-woody edges experience higher predation rates, meaning that predator control may be of importance in establishing new populations in these areas (Svedarsky et al. 2003).

Disturbances

Other threats to GPCs include natural and prescribed fires, grazing (Svedarsky et al. 2003), and wind energy development (Robel, R. J., Zilkha Renewable Energy, unpublished report). Spring burns facilitate growth of forbs and nesting habitat while fall burns in willow (*Salix* spp.)

lowlands improve brood-rearing habitat (Svedarsky et al. 2003). However, fire suppression leads to the encroachment of woody species and cool-season grasses while overburning removes all nesting and brood-rearing habitat until the plants regrow to a height suitable for these GPC behaviors (Svedarsky et al. 2003).

Grazing affects the vegetation community and vegetation vertical structure (Svedarsky et al. 2003). Svedarsky et al. (2003) recommends leaving as much as ⅔ of the prairie unburned during nesting season. American bison (*Bison bison*) graze in a way that affects both vegetative community composition and vertical structures (Vinton et al. 1993). Bison grazing facilitates growth of nesting and brood-rearing habitat for GPCs (Svedarsky et al. 2003). Vinton et al. (1993) explained that bison graze non-randomly based on plant palatability, and prevent dominant species from forming matrices. This leads to more diverse and spatially/temporally heterogeneous community and vertical structure, which enables more species to occupy grasslands. Larger patches of grazing occur in previously-burned areas, but bison graze on both burned and unburned grasslands (Vinton et al. 1993). Mowing may also be used as a substitute for grazing (Svedarsky et al. 2003). Cattle grazing may serve as a suitable alternative to bison grazing because both grazing styles increased spatial heterogeneity while increasing biodiversity compared to ungrazed prairies (Towne et al. 2005).

However, mowing without haying can leave a thick thatch layer that restricts the movement and consumption of seeds and invertebrates for GPC chicks (Harper 2007). Haying also results in greater GPC exposure to predators (Svedarsky et al. 2003). Also, intense burning and grazing/mowing combined results in little to no nesting habitat and reduced grasshopper densities for GPCs (Svedarsky et al. 2003).

Wind Farms

Wind energy is expected to provide 20% of U.S. energy by 2030 (McNew 2014). Regarding wind energy development, McNew (2014), in north-central Kansas, concluded that nest site selection and nest survival were strongly related to vegetative cover, with no correlation to wind energy development. However, in Oklahoma, Pruett et al. (2009) observed that GPCs crossed powerlines less often than would be expected if their movements were random. This likely has to do with avoidance of perceived potential perch sites for raptors and other avian predators of GPCs.

GPCs may perceive wind turbines as potential perches for their predators as well (Pruett et al. 2009). Another study, performed by Winder et al. (2014) in Kansas, observed female GPC avoidance of wind turbines during the breeding season. They observed that the mean home range size of the females in the study doubled and that space use was positively correlated with distance from wind turbines. According to Robel (Robel, R. J., Zilkha Renewable Energy, unpublished report), in their study performed near Rosalia, Kansas, the presence of any wind turbine within 1.61 km of GPC habitat will render that habitat unsuitable for nesting and brood-rearing. It appears that it is best to avoid wind energy development near current and potential GPC habitat if the intent is reintroduction.

A.2 Franklin's ground squirrel

FGS Decline

FGS live in grasslands across the northern and central Great Plains region of North America (Martin and Heske 2005). However, this species has declined by an unknown magnitude in the eastern portion of its range: northern Illinois and northwestern Indiana (Martin et al. 2003, Martin and Heske 2005). A presence/absence survey performed in Indiana from 1984–1990 by Johnson and Choromanksi-Norris (1992) determined that FGSs were present in the following northwestern Indiana counties: Lake, Porter, Laporte, Newton, Starke, Benton, White, Tippecanoe, Warren, and Vermillion. Indiana mammalogists captured most FGS in Benton County. It is unknown whether these populations are still present, but it is likely some or most have died out given the aforementioned decline trend (Martin et al. 2003, Martin and Heske 2005). Furthermore, the Indiana state mammalogist observed continual range retractions of this species in the state through live-capture sampling efforts (B. Westrich, Indiana Department of Natural Resources [INDNR], personal communication).

Habitat loss, fragmentation, the use of the pesticide Dieldrin from the 1950s–70s, and perception as an agricultural pest contributed to FGS decline (Martin et al. 2003; Huebschman 2007). The unknown magnitude of decline is due to the elusiveness and patchy distribution of this animal making accurate population surveys difficult (Martin et al. 2003). The FGS is classified as being of "least concern" on IUCN's Red List (Cassola 2016). However, they are listed as a state-endangered species in Indiana (INDNR 2019). Martin et al. (2003) stated in their study that although this species may be surviving undetected across their “declining” range, it is likely appropriate to enact conservation efforts in Midwestern states, including Indiana. They noted the absence of FGS at various trapping grid sites in Illinois, USA, where the species historically occurred. It is unknown whether FGS experiences cyclical population fluctuations, aside from in the northern portion of their range in Canada (Huebschman 2007), which could explain observations by Martin et al. (2003).

Habitat

Historically, FGS occupied the thick vegetation of tallgrass and mid-grass prairies (Martin and Heske 2005). Today, FGS inhabits tallgrass prairies, savannas, as well as unmowed roadsides and railroad rights-of-way (Martin et al. 2003, Huebschman 2007). FGS typically uses railroad rights-of-way. The slight increase in elevation of these areas promotes drainage to keep their burrows dry (Young 2012). Regardless, FGS also uses brushy areas and savannas; it is not strictly associated with tallgrass prairies (Huebschman 2007). Moreover, nearby wetlands provide additional food opportunities for FGS in terms of ducks, ducklings, and duck eggs (*Anas* spp. and *Aythya* spp.; Ostroff and Finck 2003). In all used habitats, FGS lives in dense areas of high (≥25.4 cm) vegetation (Huebschman 2007).

Additionally, FGSs live in multi-branched, multi-entrance (2–3) burrows, typically on sloping ground, that contain branches for nesting, food storage, and defecation (Jones et al. 1983, Ostroff and Finck 2003). They will either dig their own burrows, reuse old FGS burrows, or use burrows made by plains pocket gopher (*Geomys bursarius*; Choromanski-Norris et al. 1989). FGS may live in these burrows alone or in pairs (Ostroff and Finck 2003).

Home Range Size

According to a few studies, FGS annual home range size is larger for males than females (Table B.1). They also observed that males and females change their home ranges biweekly in search of adequate food (Choromanski-Norris et al. 1989, Ostroff and Finck 2003). Biweekly home range size also varied between sexes (Table B.1). The home ranges within and between both sexes can overlap (Choromanski-Norris et al. 1989). The degree of overlap is limited to 50% of home range core areas. This is because FGSs are largely asocial, live in groups of only

1–2 conspecifics, and tend to avoid unfamiliar individuals (Ostroff and Finck 2003, Zumdahl 2020). The densities of FGS in Alberta, Canada and Manitoba, Canada, where the species is common, is 1.25–2.5 adults/ha and 1.6–2.0 individuals/ha, respectively.

Table B.1. Annual and biweekly home range sizes for Franklin's ground squirrel (*Poliocitellus franklinii*). Both Choromanski-Norris et al. (1989) as well as Ostroff and Finck (2003) studied Franklin's ground squirrels in North Dakota. In Illinois, Breyer (2016) recorded observed home ranges for juvenile males and females.

Study	Males	Females
Annual		
Ostroff and Finck (2003)	24.6 ha	8.7 ha
Choromanski-Norris et al. (1989)	0.24-3.79 ha	0.08-2.36 ha
Breyer (2016)	16.79 ha	1.17 ha
Biweekly		
Ostroff and Finck (2003)	0.5-38.4 ha	0.3-14.1 ha

Hibernation and Post-Hibernation

In addition to nightly use of its burrows, FGS hibernate August/early September until April. This fact, along with inconspicuous retreats through dense vegetation to burrows, contributes to the species' elusiveness in sampling (Jones et al. 1983, Ostroff and Finck 2003). An additional reason for elusiveness is that this species is known to change burrows 3–4 times after emerging from hibernation or within a few weeks to a year after initial residence of a particular burrow (Choromanski-Norris et al. 1989; Haberman and Fleharty 1971). Finally, this species may change burrows in response to the search for adequate food (Choromanski-Norris et al. 1989).

Movements through the Landscape

FGS exhibit limited adult mobility in a fragmented landscape. FGS will travel far (>3.5 km between burrows) provided adequate cover is available (Martin and Heske 2005). However, FGS rarely moves beyond dense cover in tallgrass prairies (Krohnne and Schramm 1994). Moreover, these movements may not begin until May, when adequate density of vegetative cover has accumulated via growing, especially after a prescribed burn. Unmowed roadsides provide corridors to help alleviate this limitation.

Duggan et al. (2012) observed that FGSs perceived higher predation risk in agricultural fields than in prairie patches based on giving-up densities (GUDs). They also observed that while FGSs mostly use unmowed roadsides (maximum observed distance for travelling along an unmowed roadside was 130 m), this species will cross open fields (maximum observed distance across an open field 240 m) when energetic constraints are greater than the risk of predation. However, Choromanski-Norris et al. (1989) observed a FGS that regularly crossed an agricultural matrix of 0.6 km, though this was omitted from their analysis as an outlier. They observed that FGSs mostly limit themselves to field edges where there is adequate cover. Annual mean activity radii were 147.9 ± 69.4 m for males and 87.7 ± 35.4 m for females (Choromanski-Norris et al. 1989).

Martin and Heske (2005) tracked male FGSs that dispersed 0.1–3.6 km as well as female FGSs that dispersed 65-450 m. Some of these FGSs crossed no roads while one crossed up to four. Martin et al. (2003) suggested corridors of linear habitat patches are critical for dispersal. Juvenile dispersal appears to be less limited than adult movements to locate burrows/burrow sites and forage, with individuals observed to disperse 3.6 km (Martin and Heske 2005). Again, FGS rarely leave the heavily-vegetated areas of tallgrass prairies (Krohne and Schramm 1994). So, unmowed roadsides may facilitate movement of both juveniles and adults across a fragmented landscape.

Reproduction

FGSs mate and reproduce from when females emerge from hibernation in April until early June (Ostroff and Finck 2003). Yearling females do not reproduce (Ostroff and Finck 2003). Moreover, the sex ratio of populations is 1:1 (Ostroff and Finck 2003). During mating, males and females may live in burrows together, after which the male departs (Jones et al. 1983). After a 28-day gestation period, FGS will give birth to 2–13 altricial young that remain in the burrow (Jones et al. 1983). Independence occurs 2 weeks later (Jones et al. 1983). Male 9–11 week-old FGS will disperse further than females (Martin and Heske 2005). FGSs also interact less with and avoid littermates more than neighbors and strangers (Hare 2004). This could have implications for the longevity of small populations.

Foraging and Diet

FGS must accumulate fat reserves for hibernation during their few months of activity (April-August/early September; Krohne and Schramm 1994). FGSs will forage in areas of dense grass cover, often along the marsh edges and in forest-prairie transitions (Ostroff and Finck 2003). Peak foraging activity for FGS is typically 4–7 hours after sunrise and 5 hours before sunset (Ostroff and Finck 2003). FGSs in Illinois are inactive on cool, overcast, windy, and rainy days (Ostroff and Finck 2003).

FGS diet consists of 66–75% vegetative matter, while the remainder consists of insects, small mammals, adult ducks, ducklings, and bird eggs as become available (Choromanski-Norris et al. 1989, Ostroff and Finck 2003). Interestingly, FGSs appear to rarely pass up the opportunity to consume duck eggs (Sargeant et al. 1987). In terms of vegetable matter, which is the majority of their spring diet, FGS eat succulent roots, herbaceous shoots, grasses, as well as the leaves, buds, seeds, and blossoms of stinging nettle (*Urtica diolica*), dandelion (*Taraxacum* spp.), clover (*Trifolium* spp.), and sow thistle (*Sonchus* spp.; Ostroff and Finck 2003). In the summer, animal and insect matter is primarily consumed, in addition to vegetable matter, fruits, and seeds (Ostroff and Finck 2003). Animal and insect matter consists of: frogs, toads, fish, bird eggs, young birds, adult ducks, mice, young rabbits, ants, crickets, grasshoppers, and caterpillars (Ostroff and Finck 2003). Additional summer food items include: chokecherry (*Prunus virginiana*), wild pea (*Lathyrus* spp.), and cultivated grains, such as corn (*Zea mays*) and soybeans (*Glycine max*; Ostroff and Finck 2003). FGSs accumulate the majority of the fat reserves for hibernation during the late summer feeding period (Greenwood et al. 1998). FGS will continue to consume available foods until hibernation in late August/early September (Ostroff and Finck 2003). FGS will hydrate themselves mostly through the consumption of succulents (Ostroff and Finck 2003). They will rarely directly drink water (Ostroff and Finck 2003).

Predation and Competition

Numerous predators prey upon FGS. The predator that influences population density the most is the American badger (*Taxidea taxus*; Haberman and Fleharty 1971). Other predators include the American mink (*Neovison vison*), coyote (*Canis latrans*), red fox (*Vulpes vulpes*), and long-tailed weasel (*Mustela frenata*; Haberman and Fleharty 1971, Jones et al. 1983). Predators such as striped skunk (*Mephitis mephitis*), various raptors (*Accipiter* spp. and *Buteo* spp.), and snakes depredate FGS less often (Jones et al. 1983). External parasites include fleas, lice, mites, and ticks while internal parasites include nematodes, cestodes, and protozoans (Jones et al. 1983; Ostroff and Finck 2003).

FGS does not have any major interspecific competitors (Ostroff and Finck 2003). However, intraspecific resource competition and interspecific resource competition with the thirteen-lined ground squirrel (*Ictidomys tridecemlineatus*) commences when population densities of both species are too high on an area of otherwise "high quality" habitat (Martin et al. 2001). Otherwise, these 2 species coexist throughout their overlapping ranges (Martin et al. 2001).

Based on a survey performed at 15 sites in Illinois by Martin et al. (2001), it is unknown whether non-native forbs, woody plants, and grasses negatively impact FGS. However, their only FGS capture was at a portion of a site with little to none of these non-natives. It may be beneficial to eliminate these species from potential reintroduction sites.

Anthropogenic Impacts

Anthropogenic threats to FGS include prescribed fire, grazing, mowing, haying, and pesticides (Huebschman 2007). Again, FGSs prefer areas of dense and high (≥25.4 cm) vegetation (Huebschman 2007). Regarding prescribed fire, FGS will avoid areas affected by this land treatment until the areas are re-vegetated (Huebschman 2007). However, fire maintains prairie and savanna habitats. Avoiding burning all properties simultaneously or all sections of a single property at once would be beneficial. FGS avoids areas of mowing, haying, overgrazing (down to less than 25.4 cm; Huebschman 2007). Historically, pesticide applications, such as Dieldrin, negatively impacted FGS populations by virtually eliminating entire populations (Huebschman 2007). However, pesticides have become less of a threat due to stricter application regulations (Huebschman 2007).

APPENDIX B. OPERATIONAL AND LOGISTICAL INFORMATION

B.1 Greater Prairie-Chicken

A good source of advice on GPC translocation is "A Plan for the Recovery of the Greater Prairie-Chicken in Illinois" (Walk 2004). It goes into more detail on topics I discussed below and provides further information regarding the translocation of this species. This is especially relevant to Kankakee Sands given that the property nearly borders Illinois.

Evaluation of Habitat Types for Translocation

Walk (2004) explained that areas of potential habitat via translocation should be monitored for at least three years prior to translocation to ensure presence of habitats specific to various life history functions. The specific areas of habitat that are critical to the success of a translocation effort are potential nesting, brood-rearing, and roosting habitat. I add to this: ensuring adequate locations for lek site selection.

Source Populations and Genetic Constraints

GPC for translocations should be sourced from states that have adequate populations to withstand a sourcing for another state's translocation effort. Walk (2004) stated that Nebraska and Kansas were sources for Illinois' effort to reintroduce GPC. However, NatureServe Explorer (2021) shows that GPC are only "apparently secure" in South Dakota and Kansas, while vulnerable in Nebraska. It would be best to rely on the more secure source. Biologists must work with each state's Department of Natural Resources to arrange for permits and for the actual translocation of GPC. Walk (2004) explained that birds removed from a population for translocation efforts should total < 5% of the total population.

There are also some genetic constraints and concerns to address when talking about a translocation effort. Outbreeding depression may (but not necessarily) occur when the offspring of (in this case) individuals from geographically- and/or temporally-distant populations experience negative effects on fitness due to different genetic adaptations to their respective environments (Frankham et al. 2017). This fitness reduction is likely to arise only for the first few to dozens of generations. Outbreeding depression is relevant to selecting states to supply source populations as individuals from these places are likely to interbreed in the translocation area. Biologists should select source states with similar environments to the intended translocation site (if animals are already present nearby) in order to help avoid this issue.

A more common and serious concern is inbreeding depression. According to Frankham et al. (2017), inbreeding depression happens when genetic variability in a population is reduced due to inbreeding (including individuals as distantly-related as first cousins). Inbreeding depression occurs when there is an increased prevalence of undesirable reproductive and environmentally-adaptive traits (harmful recessive alleles), which lowers reproductive fitness and resistance to environmental stochasticity, resulting in higher extinction risk. To reduce the risk of inbreeding depression, biologists should facilitate gene flow by increasing habitat connectivity and otherwise augment the genetic variability by supplying new individuals into a metapopulation with low/at-risk genetic diversity (Frankham et al. 2017) until the minimum viable population (MVP) size is reached. MVP sizes for polygynous lekking bird species are based on the effective population size (i.e. the number of breeding males), and are therefore lower than the actual population size (J. Dunning, Purdue University, personal communication). Published MVP size for this species is 200 birds or 100 breeding males (Svedarsky et al. 2000).

Habitat fragmentation and gene flow are relevant to avoiding the two latter genetic concerns. Habitat fragmentation is the separation and parcelization of natural areas/habitat due to human or natural disturbance. To maintain gene flow in an area with a metapopulation, as well as to help avoid inbreeding depression and genetic drift due to population isolation, biologists and land stewards should aim to increase connectivity of the habitat fragments with corridors or habitat restoration if the non-habitat matrix cannot be crossed by the species of interest (Frankham et al. 2017). Connectivity and maintaining at least an MVP, thus avoiding inbreeding depression and genetic drift, will help any translocated population withstand any demographic and environmental stochasticity (Walk 2004).

Another consideration is the mating system of the species, which I outlined above. GPC are lek-mating birds that require a booming ground that itself needs to meet certain requirements for selection, along with adequate habitat area to meet the rest of its life history needs. In order to maximize mating opportunities for this species, land stewards should avoid altogether or at least minimize and cycle through the area of spring season prescribed burning in order to provide ample vegetative cover for nesting and brood rearing.

Permits

According to the U.S. Fish and Wildlife Service (USFWS; 2021), upland gamebirds such as grouse, which includes GPC, are not protected by the Migratory Bird Treaty Act of 1918. Furthermore, they are not listed as protected under the Act (USFWS 2020). Additionally, GPC is not listed as threatened or endangered under the U.S. Endangered Species Act (Johnson et al. 2011, USFWS 2020), although GPC are listed as a species of conservation concern (USFWS 2021). This means that the permit requirements for moving birds are at the state level. For states with source populations, such as South Dakota, biologists would need to request information on exportation requirements and permits from the South Dakota Animal Industry Board (SDAIB 2021). The South Dakota Department of Game, Fish, and Parks (SDDGFP) also requires a Scientific Collector's Permit that can be applied for with a form available online (SDDGFP 2021). For other source states, such as Kansas, biologists would need to apply for a Collection Permit (Kansas Department of Wildlife, Parks & Tourism 2021).

For states to which the birds are being imported, in this case Indiana, biologists need an importation permit. This permit is available upon request from Linnea Petercheff (Licensing and Permit Supervisor with the Indiana Department of Natural Resources; L. Petercheff, INDNR, personal communication).

Facilities for a Captive Source Population

Two zoological parks within 200 km of Kankakee Sands are willing and able to host a captive population of GPC for supplemental translocations. The first is the Indianapolis Zoo in Indianapolis, Indiana, 197 km from KS. Bill Street (Senior Vice President for Conservation, Education, and Life Sciences with the Indianapolis Zoo) expressed the zoo's willingness and ability to participate (B. Street, Indianapolis Zoo, personal communication). He also explained that there are conservation grants available from the zoo in order to help cover costs. The second zoo is the Brookfield Zoo in Brookfield, Illinois, 114 km from KS. Bill Zeigler (Senior Vice President of Animal Programs with the Chicago Zoological Society [CZS]) expressed similar willingness and ability from the Brookfield Zoo (B. Zeigler, CZS, personal communication). TNC should contact them for next steps.

However, according to Walk (2004), housing captive GPC is not advised. Walk (2004) listed numerous reasons, including: lack of positive association between pen-reared/pen-held GPC and successful releases, including in Illinois, pen-reared/held GPC need their wings clipped to prevent self-infliction of injury and mortality, and loss of muscle mass in pen-held birds from a lack of exercise. It is pertinent for biologists to capture GPC and release them within 72 hours (Walk 2004). If captive-rearing of GPC is a must, given the distance from source population, biologists need to create a protocol for avoiding/minimizing the above-mentioned captivity and genetic issues.

Capture and Release Strategies

Capture methods specific to GPC include drop-nets over or near active feeding areas and walk-in traps near active leks (Schroeder and Braun 1991, Jacobs 2015, Winder et al. 2014). Drop-nets work best when covering the entirety of a small, centralized, and pile-baited (100-200 pounds of grain replenished every few days) previously-identified feeding area (Jacobs 2015). An example of a drop-net system is a 13.4 m diameter net suspended from a 1.4 m steel post and is activated by a pull-pin trigger system (Jacobs 2015). Walk-in traps are created with a cylindrical area of welded wire fence, covered with nylon mesh netting, with a funnel entrance made of chicken wire leading inward, and secured to the ground with metal stakes (Schroeder and Braun 1991, Winder et al. 2014). These traps are best oriented around leks such that GPC can walk into them while going toward the center of the lek (outward-facing) and arbitrarily placed (neutral direction; Schroeder and Braun 1991). Using walk-in traps, Schroeder and Braun (1991) captured more females than males. Moreover, female decoys around these traps are useful for capturing males (Robel et al. 1970). Individual birds can also be captured with mesh dip nets with 10-foot long handles (Robel et al. 1970).

After capture and before release, biologists should develop and implement a quarantine and disease testing protocols. Although Walk (2004) advises release no more than 72 hours after initial capture, Woodford (2000) recommends for birds to quarantine at least 30 days. To compromise between Walk's (2004) and Woodford's (2000) recommendations on housing birds, biologists could perform all disease screening in the 72 hours before release at the translocation site. Screenings include the examination of feathers, feces, and blood for parasites, bacteria, and viruses as applicable to GPC, as well as serological tests for chlamydiosis. If GPC test positive for chlamydiosis, biologists should follow up by culturing cloacal swabs to confirm. If confirmed, biologists must treat said infected bird(s) for around 45 days or discard the bird(s). Moreover, according to Woodford (2000), biologists should perform cloacal swabs for virus identification, serology for avian disease such as avian pox, avian malaria, Newcastle disease, and other diseases as listed and as applicable to GPC.

Additionally, specific to GPC, Walk (2004) recommends discontinuing all put-and-take RNP hunting, along with spreading domestic chicken manure on nearby cropland, within 50 km of GPC translocation efforts in order to minimize the risks of the spread of diseases common among domestic/captive birds.

For release strategies, the initial release should be performed in the summer (Walk 2004). Walk (2004) explained that this season is best given GPC's sedentary tendencies and molting during this season, which limits dispersal abilities. Also, subsequent releases should occur in the fall and late-winter to early spring. These releases should be accompanied by decoy placements and playing recordings of GPC booming at suitable lek sites to promote lekking and discourage dispersal. Further summer release of 20 GPC annually should occur for 3 consecutive years following initial release. Spring releases should include 100 GPC.

Walk (2004) discussed that establishing a metapopulation over a larger area (different habitat patches within < 20 km of each other) should be a goal of biologists as well. He also stated that there is a 60% likelihood of success for booming ground establishment if birds are released within 1.6 km of properly-selected release sites. Translocated populations should be continuously monitored in terms of nesting, brooding, survival and dispersal (Walk 2004).

B.2 Franklin's ground squirrel

Source Populations and Genetic Constraints

FGS for translocations should be sourced from states that have adequate populations to withstand a sourcing for another state's translocation effort. States where this species is stable are Nebraska and South Dakota (NatureServe Explorer 2021). Biologists must work with each state's Department of Natural Resources to arrange for the permits of and for the actual translocation of FGS.

An additional concern when selecting populations to source individuals from is assortative mating. Assortative mating is the phenomenon where individuals prefer to mate with others from their same population (Schindler et al. 2013). This phenomenon occurs in ground squirrel species, such as the Columbian ground squirrel (*Urocitellus columbianus*) in Canada (Schindler et al. 2013). Biologists should select individuals from the same source population to avoid this issue.

There are also some genetic constraints and concerns to address when talking about a translocation effort. Outbreeding depression may (but may not necessarily) occur when the offspring of (in this case) individuals from geographically- and/or temporally-distant populations experience negative effects on fitness due to different genetic adaptations to their respective environments (Frankham et al. 2017). This fitness reduction is likely to arise only for the first few to dozens of generations. Outbreeding depression is relevant to selecting states to supply source populations as individuals from these places are likely to interbreed in the translocation area. To help avoid this concern, biologists should aim to choose source population states with environments similar to each other and to the translocation area if the species of interest is already present nearby.

A more common and serious concern is inbreeding depression. According to Frankham et al. (2017), inbreeding depression occurs when the genetic composition of a population becomes reduced due to inbreeding (out to first cousin breeding). Inbreeding causes inbreeding depression in that: there is an increased prevalence of undesirable reproductive and environmentally-adaptive traits (harmful recessive alleles), which lowers reproductive fitness and resistance to environmental stochasticity, which results in higher extinction risk. To reduce the risk of inbreeding depression, biologists should connect habitat to facilitate gene flow and otherwise supply new individuals to augment genetic diversity of the metapopulation with low/at-risk genetic diversity (Frankham et al. 2017) until the minimum viable population (MVP) size is reached. In the absence of published MVPs for this species, densities for this species range from 3-12 individuals/ha (Olson 2002). These density values could be used as a surrogate for MVP.

Habitat fragmentation and gene flow are relevant to avoiding the two latter genetic concerns. Habitat fragmentation is the separation and parcelization of natural areas/habitat due to human or natural disturbance. To maintain gene flow in an area with a metapopulation, as well as to help avoid inbreeding depression and genetic drift due to population isolation, biologists and land stewards should aim to increase connectivity of the habitat fragments with corridors or habitat restoration if the non-habitat matrix cannot be passed by the species of interest (Frankham et al. 2017). Connectivity and maintaining at least an MVP, thus avoiding inbreeding depression and genetic drift, will help any translocated population withstand any demographic and environmental stochasticity (Walk 2004).

Another consideration is the mating system of the species, which I have outlined above. FGS mate between April and June every year after emerging from hibernation. The males may live in burrows with females for mating and then depart. Females give birth 28 days later, then subsequently care for the offspring. In order to maximize mating opportunities for this species, land stewards should avoid or at least minimize and cycle through areas for spring prescribed burning in order to ensure adequate food and habitat for this species during its mating period.

Permits

Since FGS is not listed as threatened or endangered at the federal level under the U.S. Endangered Species Act, federal permits are not required (U.S. Fish and Wildlife Service 2020). Permits are required at the state-to-state level. Regarding states with potential source populations, such as South Dakota, biologists would need to request information on exportation requirements and permits from the South Dakota Animal Industry Board (SDAIB 2021). The South Dakota Department of Game, Fish, and Parks (SDDGFP) also requires a Scientific Collector's Permit that can be applied for with a PDF available online (SDDGFP 2021). For other source states, such as Nebraska, biologists would need to apply for a Scientific and Educational [Collection] Permit (Nebraska Game and Parks Commission [NEGPC] 2021). Shaun Dunn (Natural Heritage Zoologist for the NEGPC) said a Letter of Authorization may be needed as well (S. Dunn, NEGPC, personal communication). He also said that TNC would need to contact Alicia Hardin (Division Administrator), Will Inselman (Assistant Division Administrator), and himself should TNC decide to continue with this project.

For states where the animals are being imported to, in this case Indiana, biologists would need to request an importation permit. This permit is available upon request from Linnea Petercheff (Licensing and Permit Supervisor with the INDNR; L. Petercheff, INDNR, personal communication). There are no additional requirements for translocations even though FGS is a state-endangered mammal in Indiana.

Facilities for a Captive Source Population

There are two facilities within 200 km of Kankakee Sands that could house a captive population of FGS for supplemental translocations. These are the Brookfield Zoo in Brookfield, Illinois, at 114 km away and the Indianapolis Zoo in Indianapolis, Indiana, at 197 km away from KS. Bill Zeigler (Senior Vice President of Animal Programs with the Chicago Zoological Society [CZS]) expressed the Brookfield Zoo's willingness and ability to participate in a project like this (B. Zeigler, CZS, personal communication). Likewise, Bill Street (Senior Vice President for Conservation, Education, and Life Sciences with the Indianapolis Zoo) expressed similar willingness and ability from that zoo (B. Street, Indianapolis Zoo, personal communication). They both request that TNC contact them. Bill Street further requests that TNC apply for one of the Indianapolis Zoo's conservation grants to help cover costs.

For the captive population itself, Turner et al. (1976) examined postnatal growth and development of 87 first-generation captive FGS from 9 litters (~ 9 pregnant females; see Appendix A for reproduction information). The facility holding the population housed the female squirrels in roughly 53 by 43 by 22 cm cages with sawdust bedding. They fed the animals with Purina lab chow, oats, supplemental hay, and water. Juvenile FGS reach adult weight by day 65 and are ready for release by day 47-67 after birth (Turner et al. 1979). The captive population size depends on the ability of the facility to house a certain amount of FGS and maintain adequate genetic diversity. Males and females should likely be kept separate and alone except for breeding purposes as per their solitary nature.

Capture and Release Strategies

To capture FGS, a useful method is live-trapping (Duggan et al. 2011). This method involves setting transect lines of baited, non-lethal box or cage traps in an effort to capture the animal of interest. As per Duggan et al. (2011), good places to attempt to capture FGS for translocation would be known-presence and historical-presence areas, with discretion toward the latter in terms of time-since-last-observed. Duggan et al. (2011) also explain the need to cover each trap with local vegetation for shade and concealment. Bait used was peanut butter and sunflower seeds, added to the traps in the hours of 0700 hr – 0900 hr. They checked traps every three hours until 1500 hr –700 hr before closing them for the night.

After capture and before release, biologists should develop and enact quarantine and health screening protocols. According to Woodford (2000), rodents should be quarantined 35 days at minimum. During this quarantine, biologists should examine fur, feces, respiratory tracts, and blood for parasitic and bacterial infections. Specifically for members of Sciuridae, biologists should perform a serology for parapoxvirus antibodies, if present in the source location. Only individuals that test positive for antibodies should be released. Final tests include abdominal palpations for impaction and lesions. They also necessitate pre-release immunizations for rabies as well as ear tagging and/or microchipping.

In the absence of published information regarding release strategies for FGS, Matějů et al. (2012) and Gedeon et al. (2012) published articles regarding release strategies for the European ground squirrel (*Spermophilus citellus*). This species, like FGS, is diurnal, creates complex burrows, hibernates a large portion of the year, August until March or April, and begins reproductive behaviors upon ending hibernation (Ramos-Lara et al. 2014). However, it is different from FGS in that the European ground squirrel is colonial while FGS is solitary (Ostroff and Finck 2003, Ramos-Lara et al. 2014). A redeeming fact is that home ranges for conspecifics of both species overlap (Ostroff and Finck 2003, Ramos-Lara et al. 2014).

Specific release strategy suggestions to promote translocation success from Matějů et al. (2012) are to use soft release strategies (i.e. enclosure and/or artificial burrows) to ensure that the squirrel stays in the intended release area. They also explained that translocation projects of the European ground squirrel were most successful when at least 23 squirrels were released per season, with a total released population remaining at or above 60 individuals. Guidelines for effective artificial burrows from Gedeon et al. (2012) were to angle burrow entrances at 30° to facilitate digging while surrounding the entrance with grass around 18 cm ± 1.5).

APPENDIX C. DOCUMENT ACCESSIBILITY

According to Purdue University Policy, Electronic Information, Communication and Technology Accessibility (S-5): As a public university and federal contractor, Purdue University is required to adhere to Sections 504 and 508 of the Rehabilitation Act of 1973 and Title II of the Americans with Disabilities Act. This standard specifies the means by which the University ensures compliance with these laws. To the best of my knowledge, I created this Master's thesis as an accessible document that is in compliance with sections 504 and 508 of the Rehabilitation Act of 1973 and Title II of the Americans with Disabilities Act.

REFERENCES

Adhikari, D., S. K. Barik, and K. Upadhaya. 2012. Habitat distribution modelling for reintroduction of *Ilex khasiana* Purk., a critically endangered tree species of northeastern India. Ecological Engineering 40:37–43.

Aldrich, J. W. 1963. Geographic orientation of American tetraonidae. Journal of Wildlife Management 529–545.

Barnes, W. B. 1952. Zoogeographic regions of Indiana. American Midland Naturalist 48:694–699.

Becker, F. 2016. Kankakee Sands Newton County, Indiana. The Nature Conservancy. <https://www.nature.org/content/dam/tnc/nature/en/documents/Kankakee-Sands-Driving-Tour.pdf>. Accessed 27 May 2021.

Becker, F. 2018. Kankakee Sands basic story map. The Nature Conservancy. <https://www.arcgis.com/apps/mapviewer/index.html?webmap=96b8c2f9074440d48106 7f807fc6a925>. Accessed 26 May 2021.

BirdLife International. 2016. IUCN red list of threatened species 2016. *Tympanuchus cupido*. <https://dx.doi.org/10.2305/IUCN.UK.2016-3.RLTS.T22679514A92817099.en>. Accessed 16 March 2021.

Boryan, C., Z. Yang, R. Mueller, and M. Craig. 2011. Monitoring US agriculture: the US Department of Agriculture, National Agricultural Statistics Service, Cropland Data Layer Program. Geocarto International 26:341–358.

Breyer, M. 2016. Conservation plan for Franklin’s ground squirrel (FGS) at Wabash Commercial Park, Sangamon County, Illinois Incidental Take Authorization Submittal. Marine Bank.

Brotons, L., W. Thuiller, M. B. Araújo, and A. H. Hirzel. 2004. Presence-absence versus presence-only modelling methods for predicting bird habitat suitability. Ecography 27:437–448.

Carlos-Júnior, L. A., J. C. Creed, R. Marrs, R. J. Lewis, T. P. Moulton, R. Feijó-Lima, and M. Spencer. 2020. Generalized Linear Models outperform commonly used canonical analysis in estimating spatial structure of presence/absence data. PeerJ 8:9777.

Cassola, F. 2016. The IUCN Red List of Threatened Species 2016. *Poliocitellus franklinii*: <https://www.iucnredlist.org/species/41787/22265037>. Accessed 16 March 2021.

Choromanski-Norris, J., E. K. Fritzell, and A. B. Sargeant. 1989. Movements and habitat use of Franklin's ground squirrels in duck-nesting habitat. Journal of Wildlife Management 324–331.

Cilliers, D., S. Evans, H. Coetzee, and L. Van Rensburg. 2013. Developing a site selection tool to assist reintroduction efforts for the southern ground-hornbill *Bucorvus leadbeateri*. Ostrich 84:101–111.

D'Elia, J., S. M. Haig, M. Johnson, B. G. Marcot, and R. Young. 2015. Activity-specific ecological niche models for planning reintroductions of California condors (*Gymnogyps californianus*). Biological Conservation 184:90–99.

Didan, K. 2015. MOD13Q1 MODIS/Terra Vegetation Indices 16-Day L3 Global 250m SIN Grid V006 [Data set]. NASA EOSDIS Land Processes DAAC. <https://doi.org/10.5067/MODIS/MOD13Q1.006>. Accessed 14 December 2020.

Diniz-Filho, J. A. F., and L. M. Bini. 2005. Modelling geographical patterns in species richness using eigenvector-based spatial filters. Global Ecology and Biogeography 14:177–185.

Diniz-Filho, J. A. F., L. M. Bini, and B. A. Hawkins. 2003. Spatial autocorrelation and red herrings in geographical ecology. Global Ecology and Biogeography 12:53–64.

Duggan, J. M., E. J. Heske, R. L. Schooley, A. Hurt, and A. Whitelaw. 2011. Comparing detection dog and livetrapping surveys for a cryptic rodent. Journal of Wildlife Management 75:1209–1217.

Duggan, J. M., R. L. Schooley, and E. J. Heske. 2011. Modeling occupancy dynamics of a rare species, Franklin's ground squirrel, with limited data: are simple connectivity metrics adequate?. Landscape Ecology 26:1477–1490.

Dunn, S. 23 February 2021. Personal communication.

Dunning, J. 24 March 2021. Personal communication.

eBird. 2020. Kankakee Sands Nature Preserve. <https://ebird.org/hotspot/L724814>. Accessed 13 July 2020.

Elith, J., C. H. Graham, R. P. Anderson, M. Dudík, S. Ferrier, A. Guisan, R. J. Hijmans, F. Huettmann, J. R. Leathwick, A. Lehmann, and J. Li. 2006. Novel methods improve prediction of species' distributions from occurrence data. Ecography 29:129–151.

Elith, J., S. J. Phillips, T. Hastie, M. Dudík, Y. E. Chee, and C. J. Yates. 2011. A statistical explanation of MaxEnt for ecologists. Diversity and Distributions 17:43–57.

Frankham, R., J. D. Ballou, K. Ralls, M. D. B. Eldredge, M. R. Dudash, C. B. Fenster, R. C. Lacy, and P. Sunnucks. 2017. Genetic management of fragmented animal and plant populations. First edition. Oxford University Press, Oxford, NY.

Finn, Z. 2020. Restore prairie on your property to protect history, wildlife, and humanity. Indiana Woodland Steward. <http://static1.1.sqspcdn.com/static/f/776357/28389134/1609694127610/308036_WSI_VOL29_NO2_WEB_LO-RES.pdf?token=fmztiSRP%2BtcMJv04CyQDBCzrecE%3D>. Accessed 26 May 2021.

Fitzpatrick, M. C., N. J. Gotelli, and A. M. Ellison. 2013. MaxEnt versus MaxLike: empirical comparisons with ant species distributions. Ecosphere 4:1–15.

Fourcade, Y., J. O. Engler, D. Rödder, and J. Secondi. 2014. Mapping species distributions with MAXENT using a geographically biased sample of presence data: a performance assessment of methods for correcting sampling bias. PloS One 9:97122.

Gedeon, C. I., G. Boross, A. Németh, and V. Altbäcker. 2012. Release site manipulation to favour European ground squirrel *Spermophilus citellus* translocations: translocation and habitat manipulation. Wildlife Biology 18:97–104.

Gedir, J. V., T. Everest, and A. Moehrenschlager. 2004. Evaluating the potential for species reintroductions in Canada. Pages 2–6 in proceedings of the Species at Risk 2004 Pathways to Recovery Conference.

Getis, A. 2010. Spatial Autocorrelation. Handbook of applied spatial analysis. Springer, Berlin, Heidelberg, Germany.

Gillespie, R. 2016. *Tympanuchus cupido* Greater Prairie-Chicken. <https://explorer.natureserve.org/Taxon/ELEMENT_GLOBAL.2.102531/Tympanuchus_cupido>. Accessed 27 May 2021.

Global Biodiversity Information Facility [GBIF]. 2020. Global Biodiversity Information Facility homepage. <https://www.gbif.org/en/>. Accessed 16 December 2020.

Greenwood, R. J., D. G. Pietruszewski, and R. D. Crawford. 1998. Effects of food supplementation on depredation of duck nests in upland habitat. Wildlife Society Bulletin 219–226.

Griffith, D. A. 2005. Effective geographic sample size in the presence of spatial autocorrelation. Annals of the Association of American Geographers 95:740–760.

Guillera-Arroita, G., J. J. Lahoz-Monfort, and J. Elith. 2014. Maxent is not a presence–absence method: a comment on Thibaud et al. Methods in Ecology and Evolution 5:1192–1197.

Guillera-Arroita, G., J. J. Lahoz-Monfort, J. Elith, A. Gordon, H. Kujala, P. E. Lentini, M. A. McCarthy, R. Tingley, and B. A. Wintle. 2015. Is my species distribution model fit for purpose? Matching data and models to applications. Global Ecology and Biogeography 24:276–292.

Guisan, A., and W. Thuiller. 2005. Predicting species distribution: offering more than simple habitat models. Ecology Letters 8:993–1009.

Haberman, C. G., and E. D. Fleharty. 1971. Natural history notes on Franklin's ground squirrel in Boone County, Nebraska. Transactions of the Kansas Academy of Science 76–80.

Hamerstrom, Jr., F. N., F. Hopkins and A. J. Rinzel. 1941. An experimental study of

browse as a winter diet for prairie chickens. Wilson Bulletin 53:185–195.

Hamerstrom, Jr., F. N., and F. Hamerstrom. 1973. The prairie chicken in Wisconsin. Wisconsin Department of Natural Resources Technical Bulletin 64.

Hare, J. F. 2004. Kin discrimination by asocial Franklin's ground squirrels (*Spermophilus franklinii*): is there a relationship between kin discrimination and ground squirrel sociality? Ethology, Ecology & Evolution 16:157–169.

Harper, C. A. 2007. Strategies for managing early succession habitat for wildlife. Weed Technology 21:932–937.

Harrison, J. O., M. B. Brown, L. A. Powell, W. H. Schacht, and J. A. Smith. 2017. Nest site selection and nest survival of greater prairie-chickens near a wind energy facility. The Condor: Ornithological Applications 119:659–672.

Hendricks, S. A., P. R. S. Clee, R. J. Harrigan, J. P. Pollinger, A. H. Freedman, R. Callas, P. J. Figura, and R. K. Wayne. 2016. Re-defining historical geographic range in species with sparse records: implications for the Mexican wolf reintroduction program. Biological Conservation 194:48–57.

Hernandez, J. M. 2015. Ecological niche modeling of *Pteronotropis hubbsi*, the bluehead shiner: Evaluating the effects of spatial filtering and maxent features across various spacial extents. Thesis, University of Texas at Tyler, USA.

Hijmans, R. J. 2021. Raster package in R. <https://rspatial.org/raster/pkg/RasterPackage.pdf>. Accessed 27 May 2021.

Homeland Infrastructure Foundation-Level Data [HIFLD]. 2020. Electric power transmission lines. <https://hifld-geoplatform.opendata.arcgis.com/datasets/electric-power-transmission-lines?geometry=-89.308%2C39.909%2C-85.100%2C41.368>. Accessed 23 November 2020.

Homeland Infrastructure Foundation-Level Data [HIFLD]. 2020. Railroads. <https://hifld-geoplatform.opendata.arcgis.com/datasets/2a9677db741d4a78bd221586fe9a61f5_0>. Accessed 23 November 2020.

Homoya, M. A., D. B. Abrell, J. R. Aldrich, and T. W. Post. 1985. The natural regions of Indiana. Indiana Academy of Science 94:245–268.

Hovick, T. J., D. K. Dahlgren, M. Papeş, R. D. Elmore, and J. C. Pitman. 2015. Predicting greater prairie-chicken lek site suitability to inform conservation actions. PloS one 10.

Huebschman, J. J. 2007. Distribution, abundance, and habitat associations of Franklin's ground squirrel (*Spermophilus franklinii* Sabine 1822). Illinois Natural History Survey Bulletin 38:1.

Illinois Department of Natural Resources [ILDNR]. 2018. Final report for T-85-D-1 recovery of greater prairie chickens in Illinois. Illinois Department of Natural Resources, Springfield, Illinois. <https://www2.illinois.gov/dnr/conservation/IWAP/Documents/SWGReports/T-85-D-1%20Final%20Report.pdf>. Accessed 16 March 2021.

Illinois Department of Natural Resources [ILDNR]. 2020. Illinois Nature Preserves Commission, Illinois Endangered Species Protection Board, and the Natural Heritage Database. Data released 12 November 2020.

Illinois Department of Natural Resources [ILDNR]. 2020. Jasper County Prairie Chicken Sanctuary. <https://www2.illinois.gov/dnr/INPC/Pages/Area8JasperJasperCountyPrairieChickenSanctuary.aspx>. Accessed 16 March 2021.

Illinois Department of Natural Resources [ILDNR]. 2020. Marion County Prairie Chicken Sanctuary. <https://www2.illinois.gov/dnr/INPC/Pages/Area8MarionMarionCountyPrairieChickenSanctuary.aspx>. Accessed 16 March 2021.

Illinois Department of Natural Resources [ILDNR]. 2020. The case of the greater-prairie chicken. <https://www2.illinois.gov/dnr/education/Documents/BioBasics_Activity_3-2.pdf>. Accessed 16 March 2021.

Indiana Department of Natural Resources [INDNR], Division of Fish and Wildlife [DFW]. 1999. Nongame and Endangered Wildlife Program Division of Fish and Wildlife Indiana Department of Natural Resources September 1998 - August 1999. <https://www.in.gov/dnr/fishwild/files/annual.pdf>. Accessed 16 March 2021.

Indiana Department of Natural Resources [INDNR]. 2004. Cool season grass establishment. Division of Fish and Wildlife. <https://www.in.gov/dnr/fishwild/files/HMFSCoolSeason.pdf>. Accessed 16 February 2021.

Indiana Department of Natural Resources [INDNR]. 2019. Endangered and special concern species list. Division of Fish and Wildlife. <https://www.in.gov/dnr/naturepreserve/files/fw-Endangered_Species_List.pdf>. Accessed 16 March 2021.

Indiana Natural Heritage Data Center [INNHDC]. 2020. State of Indiana, Department of Natural Resources, Division of Nature Preserves. Data released 20 November 2020.

Iowa Department of Natural Resources [IADNR]. 2021. State of Iowa, Department of Natural Resources, Natural Areas Inventory. Data released 3 February 2021.

International Union for Conservation of Nature [IUCN]. 2016. Guidelines for reintroductions and other conservation translocations. <https://portals.iucn.org/library/sites/library/files/documents/2013-009.pdf>. Accessed 16 March 2021.

Jacobs, K. F. 2015. A drop-net trapping technique for greater prairie chickens. In Proceedings of the Oklahoma Academy of Science 38:154–157.

Johnsgard, P. A. 2002. Grassland grouse and their conservation. Smithsonian Institution. Washington, D.C., USA.

Johnson, J. A., M. A. Schroeder, and L. A. Robb. 2011. Greater prairie-chicken (*Tympanuchus cupido*). The Birds of North America. Cornell Lab of Ornithology, Ithaca, NY, USA. <https://doi.org/10.2173/bna.36>. Accessed 9 September 2020.

Johnson, S. A., and J. Choromanski-Norris. 1992. Reduction in the eastern limit of the range of the Franklin's ground squirrel (*Spermophilus franklinii*). American Midland Naturalist 325–331.

Jones, R. E. 1963. Identification and analysis of lesser and greater prairie chicken habitat. Journal of Wildlife Management 757–778.

Jones, J. K., Jr., D. M. Armstrong, R. S. Hoffman, and C. Jones. 1983. Mammals of the northern Great Plains. University of Nebraska Press, Lincoln.

Kansas Department of Wildlife, Parks, and Tourism [KSDWPT]. 2021. Education, exhibition, collection, and salvage permits. <https://ksoutdoors.com/Services/Education-Exhibition-Collection-and-Salvage-Permits>. Accessed 16 February 2021.

Korschgen, L. J. 1962. Food habits of greater prairie-chickens in Missouri. American Midland Naturalist 68:3.

Krohne, D. T., and P. Schramm. 1994. Hibernation and life history of the Franklin's ground squirrel (*Spermophilus franklinii*) in a restored prairie. In Proceedings of the Thirteenth North American Prairie Conference. RG Wickett, et al. eds. Windsor, Ontario, Canada.

Lehmann, V. W. 1941. Attwater's prairie-chicken - Its life history and management. U.S. Department of Interior, Fish and Wildlife Service. North American Fauna 57.

Leopold, A. 1931. Report on a game survey of the North Central States. Madison, WI: American Game Association 7–318.

Maes, D., S. Ellis, P. Goffart, K. L. Cruickshanks, C. A. van Swaay, R. Cors, M. Herremans, K. R. R. Swinnen, C. Wils, S. Verhulst, L. De Bruyn, E. Matthysen, S. O'Riordan, D. J. Hoare, and N. A. Bourn. 2019. The potential of species distribution modelling for reintroduction projects: the case study of the Chequered Skipper in England. Journal of Insect Conservation 23:419–431.

Martin, J. M., E. J. Heske, and J. E. Hofmann. 2001. A status survey of the Franklin's ground squirrel (*Spermophilus franklinii*) in Illinois. Illinois Natural History Survey.

Martin, J. M., E. J. Heske, and J. E. Hofmann. 2003. Franklin's ground squirrel (*Spermophilus franklinii)* in Illinois: a declining prairie mammal? American Midland Naturalist 150:130–138.

Martin, J. M., and E. J. Heske. 2005. Juvenile dispersal of Franklin's ground squirrel (*Spermophilus franklinii*) from a prairie "island". American Midland Naturalist 153:444–449.

Matějů, J., Š. Říčanová, S. Poláková, M. Ambros, B. Kala, K. Matějů, and L. Kratochvíl. 2012. Method of releasing and number of animals are determinants for the success of European ground squirrel (*Spermophilus citellus*) reintroductions. European Journal of Wildlife Research 58:473–482.

Matthews, T. W., A. J. Tyre, J. S. Taylor, J. J. Lusk, and L. A. Powell. 2013. Greater prairie-chicken nest success and habitat selection in southeastern Nebraska. The Journal of Wildlife Management 77:1202–1212.

McNew, L. B., L. M. Hunt, A. J. Gregory, S. M. Wisely, and B. K. Sandercock. 2014. Effects of wind energy development on nesting ecology of greater prairie-chickens in fragmented grasslands. Conservation Biology 28:1089–1099.

Merow, C., and J. A. Silander Jr. 2014. A comparison of Maxlike and Maxent for modelling species distributions. Methods in Ecology and Evolution 5:215–225.

Merow, C., M. J. Smith, and J. A. Silander Jr. 2013. A practical guide to MaxEnt for modeling species' distributions: what it does, and why inputs and settings matter. Ecography 36:1058–1069.

Missouri Department of Conservation [MODC]. 2021. State of Missouri, Department of Conservation, Natural Heritage Program. Data received 13 February 2021.

Morris, K. M., E. C. Soehren, M. S. Woodrey, and S. A. Rush. 2020. Habitat-suitability model for the yellow rail (*Coturnicops noveboracensis*) in the northern gulf coast of Alabama and Mississippi, USA. Remote Sensing 12:848.

Mumford, R. E., C. E. Keller, and W. Zimmerman. 2019. The birds of Indiana. Greater prairie-chicken. Indiana University Press, Bloomington, IN.

Murphy, M. L. 2011. Evaluating wildlife corridor linkages: do freeway underpasses connect the peninsular and transverse mountain ranges?. Thesis, University of California, Riverside, CA, USA.

Mutascio, H. E., S. E. Pittman, P. A. Zollner, and L. E. D'Acunto. 2018. Modeling relative habitat suitability of southern Florida for invasive Burmese pythons (*Python molurus bivittatus*). Landscape Ecology 33:257–274.

Naimi, B. 2017. Uncertainty analysis for species distribution model. CRAN. <https://cran.r-project.org/web/packages/usdm/usdm.pdf>. Accessed 5 August 2021.

Naimi, B., N. A. Hamm, T. A. Groen, A. K. Skidmore, A. G. Toxopeus, and S. Alibakhshi. 2019. ELSA: Entropy-based local indicator of spatial association. Spatial Statistics 29:66–88.

Natural Resources Conservation Service [NRCS]. 2020. Geospatial data gateway. <https://datagateway.nrcs.usda.gov/>. Accessed 23 November 2020.

Natural Resources Conservation Service [NRCS Soils]. 2021. County level soil surveys in digital format — SSURGO. <https://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/office/ssr12/tr/?cid=nrcs142p2_010596>. Accessed 27 May 2021.

National Resources Conservation Service Soils [NRCS Soils]. 2021. Description of gridded soil survey geographic (gSSURGO) database. <https://www.nrcs.usda.gov/wps/portal/nrcs/detail/soils/survey/geo/?cid=nrcs142p2_053628#:%7E:text=The%20gridded%20SSURGO%20(gSSURGO)%20dataset,and%20analysis%20of%20soils%20data.&text=The%20tabular%20data%20represent%20the,Soil%20Information%20System%20(NASIS)>. Accessed 27 May 2021.

NatureServe. 2014. The NatureServe network. <https://www.natureserve.org/natureserve-network>. Accessed 23 November 2020.

NatureServe Explorer. 2021. *Poliocitellus franklinii*. <https://explorer.natureserve.org/Taxon/ELEMENT_GLOBAL.2.105759/Poliocitellus_franklinii>. Accessed 10 February 2021.

Nebraska Games and Parks Commission [NEGPC]. 2021. Scientific and education permit. <https://apps.outdoornebraska.gov/sep>. Accessed 16 February 2021.

Niva, L. M. 2010. Habitat use and coexistence of ground squirrels in northern Missouri. University of Central Missouri, Warrensburg, USA.

Nyberg, A. 2019. A bouncing baby badger? Nature notes from Kankakee Sands June 2019.

Nyberg, A. 11 September 2020. Personal communication.

Olson, E. 2002. *Spermophilus franklinii*. Animal Diversity Web. <https://animaldiversity.org/accounts/Spermophilus_franklinii/>. Accessed 16 February 16, 2021.

Ostroff, A. C., and E. J. Finck. 2003. *Spermophilus franklinii*. Mammalian Species 724:1–5.

Perkins-Taylor, I. E., and J. K. Frey. 2020. Predicting the distribution of a rare chipmunk (*Neotamias quadrivittatus oscuraensis*): comparing MaxEnt and occupancy models. Journal of Mammalogy 101:1035–1048.

Petercheff, L. 21 January 2021. Personal communication.

Phillips, S. J. 2017. A Brief Tutorial on Maxent. <http://biodiversityinformatics.amnh.org/open_source/maxent/>. Accessed on 21 April 2021.

Phillips, S. J., and M. Dudík. 2008. Modeling of species distributions with Maxent: new extensions and a comprehensive evaluation. Ecography 31:161–175.

Phillips, S. J., M. Dudík, and R. E. Schapire. 2004. A maximum entropy approach to species distribution modeling. Page 83 in Proceedings of the twenty-first international conference on machine learning.

Philips, S. J., M. Dudík, R. E. Schapire. 2020. Maxent software for modeling species niches and distributions (Version 3.4.1). <http://biodiversityinformatics.amnh.org/open_source/maxent/>. Accessed 8 December 2020.

Pruett, C. L., M. A. Patten, and D. H. Wolfe. 2009. Avoidance behavior by prairie grouse: implications for development of wind energy. Conservation Biology 23:1253–1259.

Ramos-Lara, N., J. L. Koprowski, B. Kryštufek, and I. E. Hoffmann. 2014. *Spermophilus citellus* (Rodentia: Sciuridae). Mammalian Species 46:71–87.

Robb, L. A., and M. A. Schroeder. 2005. Greater prairie-chicken (*Tympanuchus cupido*): A technical conservation assessment. Prepared for the USDA Forest Service, Rocky Mountain Region, Species Conservation Project.

Robel, R. J., J. N. Briggs, J. J. Cebula, N. J. Silvy, C. E. Viers, and P. G. Watt. 1970. Greater prairie chicken ranges, movements, and habitat usage in Kansas. Journal of Wildlife Management 286–306.

Rodríguez, J. P., L. Brotons, J. Bustamante, and J. Seoane. 2007. The application of predictive modelling of species distribution to biodiversity conservation. Diversity and Distributions 13:243–251.

Rumble, M. A., J. A. Newell, and J. E. Toepfer. 1988. Diets of greater prairie chickens on the Sheyenne National Grasslands. In: Bjugstad, Ardell J., tech. coord. Prairie chickens on the Sheyenne National Grasslands: September 18, 1987; Crookston, Minnesota. Gen. Tech. Rep. RM-159. Fort Collins, CO: US Dept. of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station 159:49–54.

Ryan, M. R., L. W. Burger, D. P. Jones, and A. P. Wywialowski. 1998. Breeding ecology of greater prairie-chickens (*Tympanuchus cupido*) in relation to prairie landscape configuration. American Midland Naturalist 140:111–121.

Sargeant, A. B., M. A. Sovada, and R. J. Greenwood. 1987. Responses of three prairie ground squirrel species, *Spermophilus franklinii*, *S. richardsonii*, and *S. tridecemlineatus*, to duck eggs. Canadian Field-Naturalist 101:95–97.

Schindler, S., P. Neuhaus, J. M. Gaillard, and T. Coulson. 2013. The influence of nonrandom mating on population growth. The American Naturalist 182:28–41.

Schmal, R. C. 2005. Mysterious Beaver Lake. Lowell Tribune. Lowell Public Library. <http://www.lowellpl.lib.in.us/s2005may.htm>. Accessed 25 October 2021.

Schoof, M. 14 September 2020. Personal communication.

Schroeder, M. A., and C. E. Braun. 1991. Walk-in traps for capturing greater prairie-chickens on leks. Journal of Field Ornithology 378–385.

Schroeder, M. A., and G. C. White. 1993. Dispersion of greater prairie-chicken nests in relation to lek location: evaluation of the hot-spot hypothesis of lek evolution. Behavioral Ecology 4:266–270.

Soil Survey Staff. 2017. Gridded soil survey geographic (gSSURGO) database for the conterminous United States. United States Department of Agriculture, Natural Resources Conservation Service. <https://gdg.sc.egov.usda.gov/>. Retrieved 4 February 2020.

Southwestern Indiana Master Gardener Association [SWINMGA]. 2021. Native grass garden. <https://www.swimga.org/swimga-gardens/native-grass-garden/>. Accessed 16 February 2021.

Short, J. 2009. The characteristics and success of vertebrate translocations within Australia. Wildlife Research and Management Pty Ltd, Perth, and Australian Government Department of Agriculture, Fisheries and Forestry, Canberra.

Shuey, J. 17 November 2020. Personal communication.

Smith, T. Thompson, D. H. Wolfe, and B. K. Sandercock. 2015. Factors affecting female space use in ten populations of prairie chickens. Ecosphere 6:166.

South Dakota Animal Industry Board [SDAIB]. 2021. Export. <https://aib.sd.gov/export.html>. Retrieved 16 February 2021.

South Dakota Department of Game, Fish, and Parks [SDDGFP]. 2021. Scientific collector's permit. <https://gfp.sd.gov/scientific-collector/>. Retrieved 16 February 2021.

Stohlgren, K. M., S. F. Spear, and D. J. Stevenson. 2015. A status assessment and distribution model for the eastern diamondback rattlesnake (*Crotalus adamanteus*) in Georgia. The Orianne Society.

Stolar, J., and S. E. Nielsen. 2015. Accounting for spatially biased sampling effort in presence-only species distribution modelling. Diversity and Distributions 21:595–608.

Street, B. 18 March 2021. Personal communication.

Svedarsky, W. D., R. L. Westemeier, R. J. Robel, S. Gough, and J. E. Toepher. 2000. Status and management of the greater prairie-chicken *Tympanuchus cupido pinnatus* in North America. Wildlife Biology 6:277–284.

Svedarsky, W. D., J. E. Toepfer, R. L. Westemeier, and R. J. Robel. 2003. Effects of management practices on grassland birds: Greater Prairie-Chicken. Northern Prairie Wildlife Research Center, Jamestown, ND.

The Nature Conservancy [TNC]. 2020. Kankakee Sands. <https://www.nature.org/en-us/get-involved/how-to-help/places-we-protect/kankakee-sands/>. Accessed 4 February 2020.

Towne, E. G., D. C. Hartnett, and R. C. Cochran. 2005. Vegetation trends in tallgrass prairie from bison and cattle grazing. Ecological Applications 15:1550–1559.

Turner, B. N., S. L. Iverson, and K. L. Severson. 1976. Postnatal growth and development of captive Franklin's ground squirrels (*Spermophilus franklinii*). American Midland Naturalist 93–102.

U.S. Climate Data. 2007-2019. Climate Rensselaer - Indiana. <https://www.usclimatedata.com/climate/rensselaer/indiana/united-states/usin0557>. Accessed 4 February 2020.

U.S. Fish and Wildlife Service [USFWS]. 2020. Migratory bird treaty act protected species (10.13 List). U.S. Fish & Wildlife Service - Department of the Interior. <https://www.fws.gov/birds/management/managed-species/migratory-bird-treaty-act-protected-species.php>. Accessed 16 September 2021.

U.S. Fish and Wildlife Service [USFWS]. 2020. U.S. species. <https://www.fws.gov/endangered/species/us-species.html>. Accessed 10 February 2021.

U.S. Fish and Wildlife Service [USFWS]. 2021. Birds of conservation concern. U.S. Fish & Wildlife Service - Department of the Interior. <https://www.fws.gov/birds/management/managed-species/birds-of-conservation-concern.php>. Accessed 16 September 2021.

U.S. Fish and Wildlife Service [USFWS]. 2021. Migratory bird permits. U.S. Fish and Wildlife Service Migratory Birds & Habitat Program. <https://www.fws.gov/pacific/migratorybirds/permits/>. Accessed 17 September 2021.

U.S. Wind Turbine Database [USWTD]. 2020. Data and Web Services. <https://eerscmap.usgs.gov/uswtdb/data/>. Accessed 23 November 2020.

USDA National Agricultural Statistics Service cropland data layer. 2020. Published crop-specific data layer. USDA-NASS, Washington, DC. <http://nassgeodata.gmu.edu/CropScape/>. Accessed 15 December 2020.

Vance, D. R., and R. L. Westemeier. 1979. Interactions of pheasants and prairie chickens in Illinois. Wildlife Society Bulletin 221–225.

Vinton, M., D. Hartnett, E. Finck, and J. Briggs. 1993. Interactive effects of fire, bison (*Bison bison*) grazing and plant community composition in tallgrass prairie. The American Midland Naturalist 129:10–18.

Walk, J. W. 2004. A plan for the recovery of the greater prairie-chicken in Illinois. University of Illinois.

Walk, J. W., and Warner, R. E. 1999. Effects of habitat area on the occurrence of grassland birds in Illinois. The American Midland Naturalist 141:339–344.

Western Regional Climate Center. 2020. Rensselaer, Indiana. <https://wrcc.dri.edu/cgi-bin/cliMAIN.pl?in7298>. Accessed 13 July 2020.

Westrich, B. J. 10 April 2020. Personal communication.

Westwood, R., A. R. Westwood, M. Hooshmandi, K. Pearson, K. LaFrance, and C. Murray. 2020. A field-validated species distribution model to support management of the critically endangered Poweshiek skipperling (*Oarisma poweshiek*) butterfly in Canada. Conservation Science and Practice 2:163.

Winder, V. L., K. M. Carrlson, A. J. Gregory, C. A. Hagen, D. A. Haukos, D. C. Kesler, L C. Larsson, T. W. Matthews, L. B. McNew, M. A. Patten, J. C. Pitman, L. A. Powell, J. A. Smith, T. Thompson, D. H. Wolfe, and B. K. Sandercock. 2015. Factors affecting female space use in ten populations of prairie chickens. Ecosphere 6:1–17.

Winder, V. L., L. B. McNew, A. J. Gregory, L. M. Hunt, S. M. Wisely, and B. K. Sandercock. 2014. Space use by female greater prairie-chickens in response to wind energy development. Ecosphere 5:1–17.

Wisconsin Department of Natural Resources [WIDNR], Wisconsin Natural Heritage Program [WINHP]. 2020. State of Wisconsin, Department of Natural Resources, Natural Heritage Program database. Data released December 7, 2020.

Woodford, M. H. 2000. Quarantine and health screening protocols for wildlife prior to translocation and release into the wild. Wildlife Disease and Zoonotics, University of Nebraska-Lincoln. <https://digitalcommons.unl.edu/zoonoticspub/32/>. Accessed 7 April 2021.

World Climate. 2020. Average weather data for Rensselaer, Indiana. <http://www.worldclimate.com/climate/us/indiana/rensselaer>. Accessed 13 July 2020.

Young, C. 2012. Response of the Franklin's Ground Squirrel to Recreational Trail Development in Springfield, Illinois. Thesis, University of Illinois Springfield, USA.

Young, N., L. Carter, and P. Evangelista. 2011. A MaxEnt model v3. 3.3 e tutorial (ArcGIS v10). Natural Resource Ecology Laboratory, Colorado State University and the National Institute of Invasive Species Science. <http://ibis.colostate.edu/webcontent/ws/coloradoview/tutorialsdownloads/a_maxent_model_v7.pdf>. Accessed 16 March 2021.

Zeigler, B. 18 March 2021. Personal communication.

Zumdahl, K. 2020. Franklin's ground squirrel (*Poliocitellus franklinii*) social distancing: home range size and overlap of a relatively asocial ground squirrel. MSpace. University of Manitoba. <https://mspace.lib.umanitoba.ca/handle/1993/35037>. Accessed 16 March 2021.