

An assessment of grassland loss, woody encroachment, and pesticide use on North American
grassland bird populations

by

Rachel Helen Rusten

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Approved by:

Co-Major Professor
Andrew Ricketts, PhD

Approved by:

Co-Major Professor
Daniel Sullins, PhD

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Abstract

The North American Breeding Bird Survey monitoring effort has consistently documented grassland bird declines since 1966. Within Kansas, greater prairie-chicken (*Tympanuchus cupido*) surveys suggest declines over the past 30 years. Grassland loss has largely been cited as the main driver of declines; however, I predicted other factors, such as pesticide application and woody encroachment, may have induced additive and interactive effects. I evaluated grassland loss, woody encroachment, and pesticide use as potential threats to meadowlarks (*Sturnella magna* and *S. neglecta*) and greater prairie-chickens (*Tympanuchus cupido*) within the Great Plains and Kansas, respectively.

I examined meadowlark abundance throughout the Great Plains of the United States to assess the relative influence of each potential threat between 1996 and 2019. The linear least square regressions showed strong relationships between observed meadowlarks and the three factors. With consideration of different starting populations between Bird Conservation Regions, the grassland model performed best based on the slope of the beta coefficient ($\hat{\beta}_1 = 0.513$, 95% CI [0.463, 0.562]; $\Delta\text{RMSE} = 3.46$), while the woody encroachment model reported the highest predictive accuracy ($\hat{\beta}_1 = -0.453$, [-0.487, -0.419]; $\Delta\text{RMSE} = 6.52$). The pesticide model had a negative relationship with observed meadowlarks ($\hat{\beta}_1 = -0.292$, [-0.328, -0.256]; $\Delta\text{RMSE} = 0.09$), and indicated negative effects beyond grassland loss alone, the only correlated factor ($r = -0.70$).

To evaluate these three threats on greater prairie-chickens in Kansas, I used a random forest modeling approach to compare grassland, woody cover, and pesticide use between occupied ($n = 5,552$) and “pseudoabsent” ($n = 11,223$) greater prairie-chicken lek site locations from 1993 to 2020. My best-supported variables were the amount of grassland at 5 km, woody

cover percentage within 800 m, and pesticide application rates within 5 km of lek locations. I validated the model fit using independent holdout data (~25%) and ROC curve (AUC = 0.956); the model correctly predicted the use or absence of leks with 89.95% accuracy. My analysis suggests that though the amount of grassland has slightly declined, the amount of available habitat could have increased on average in Kansas. However, I estimated a 15% loss of greater prairie-chicken habitat in the Flint Hills coinciding with woody encroachment and increased pesticide use. My greater prairie-chicken species distribution model likely overestimates expected habitat without considering finer-scale factors (i.e. anthropogenic features) and vegetation structure but provides a foundation for local assessments. These results supported my hypotheses that grassland loss, woody encroachment, and pesticide use all present negative demographic consequences for meadowlarks, greater prairie-chickens, and other grassland birds within Kansas, and across the Great Plains. Current trends may be altered with grassland conservation action. The findings emphasize the importance of reducing the consequences of agricultural intensification and promoting bountiful grasslands for wildlife and producers alike.

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Dedication

I dedicate this thesis to the women in my family. You laid a foundation for me that you did not have, and I am forever grateful. This education and my individuality would not have been possible without the sacrifices and generosity of my grandma Karen and my great aunt Ruth, among many others.

Preface

Throughout this thesis, the use of “I” does not indicate that I am the sole contributor for data collection, effort, and results presented. My use of the first person throughout is for style purposes and to relinquish anyone from commitment to my writing and ideas expressed herein.

Chapter 1 - Avian and Landscape Data Appraisal

Remotely sensed products and large monitoring datasets hold ample information that can be used with correct statistical approaches to glean knowledge of trends and examine specific relationships. Understanding the intended purpose of each product is essential to selecting the best products for each research question. The objectives of my study are to highlight the large regional trends of grassland birds and three potential threats: grassland loss, woody encroachment, and pesticide use. Here, I examine multiple freely available datasets. The qualifiers I investigated for these data sources were temporal availability, spatial scales, and associated published accuracy (Table 1.1). Many fine-scale data are provided for large areas, such as entire states, but may only be available for select years (Paull et al. 2015, Holifield Collins et al. 2023). I was interested in products spanning the Great Plains region of the United States over a ~30-year period.

Examined Data Sources

Bird Monitoring Datasets

Many field techniques have been developed and successfully conducted to examine bird communities and trends. In the interest of efficiency, roadside point counts are the keystone survey method selected by many state breeding bird atlases and monitoring efforts. The North American Breeding Bird Survey (BBS) uses the standardized roadside approach and maintains over 50 years of bird survey data in the United States (Sauer et al. 2017). The BBS is commonly used to index bird abundance trends within multiple strata: country, state, bird conservation region, or hybrid (<https://eesc.usgs.gov/MBR/>). While the BBS is a valuable resource, there are limitations in this long-term dataset. Biases are immediately induced by road-based surveys and

fluctuating observers influence detection probabilities. In addition, routes can change over time or may be surveyed temporarily. Users of BBS data need to be diligent about route selection. Studies have introduced complex statistical methods to account for biases (Link and Sauer 2015).

Audubon Christmas Bird Count (CBC) is an honored tradition in many communities across North America and globally. The CBC is unique as it offers an incredibly long-term dataset (starting in 1908) to document non-breeding habitat use and occupancy by birds (<https://audubon.maps.arcgis.com/apps/webappviewer/index.html?id=90a19cde26c44ad8808190219e79fd4e>). Unfortunately, survey efforts center around urban areas, and some core Plains states have few surveys. Additional bias is also introduced as the number and experience of observers vary greatly, resulting in fluctuations in sampling effort (Meehan et al. 2019), which is common in community-collected data.

While citizen-science data are important, I did not consider any broad, user-based datasets such as CBC, eBird, or iNaturalist. Unlike community-collected data, BBS volunteers undergo training and surveys must be completed based on environmental conditions specified in the collection protocol (Sauer et al. 2017, Ziolkowski et al. 2023). It is important to recognize the uses of community-collected data. However, due to the extreme volatility of the sampling effort, high false negative rates, and resulting high-powered statistical inference necessary (Altwegg and Nichols 2019), community-collected data were not suitable for this project.

Land Cover

Broad Classification Products

Land Change Monitoring, Assessment, and Projection (LCMAP) is a large assemblage of products defining land cover change provided by the U.S. Geological Survey's Earth Resources

Observation and Science (EROS) Center. I am interested specifically in the primary land cover product within LCMAP Collection 1.3 (USGS 2022). The land cover classification relies on the National Land Cover Database (NLCD; Yang et al. 2018) land cover layers. Using the machine learning algorithm, Continuous Change Detection and Classification (CCDC; Zhu and Woodcock 2014), LCMAP advances the accuracy of NLCD data and predicts land cover change between reported years (Brown et al. 2020). In addition, LCMAP used Analysis Ready Data, which is cloud-omitted, high-quality spatial data in addition to a digital elevation model (DEM, 10 m) and wetland potential index sites (Brown et al. 2020). An independent reference dataset containing 27,000 sampled points was used to evaluate the model and assess accuracy (Pengra et al. 2020). In addition, each year's layer reports pixel-level predictive confidence (Brown et al. 2020). A quick assessment of the trends within LCMAP's vegetation cover product shows that grassland and cropland directly reflect each other's pattern (Figure 1.1).

The United States Forest Service (USFS) Land Change Monitoring System (LCMS) provides an annual land cover product between 1985 and 2020 (USFS 2022). Version 2022.8 is trained with Landsat Collection II and Sentinel-2 with 70,000 aerial photos and 1,100 ground samples and matches NLCD resolution. This product offers 11 separate land cover classes, differentiating grass/forb from shrubland and forest. Notably, LCMS does not define the distinction between trees and shrubs. An accuracy assessment was not conducted due to a low number of samples. However, it uses innovative classification algorithms, Multi-Index Integrated Change Analysis (MIICA) and LandTrendr to predict land cover (USFS 2022). The LCMS land cover reports a high correlation in land cover prediction with the LANDFIRE product.

The LANDFIRE product also utilizes MIICA, which may explain results similar to LCMS predictions. While LCMS focuses on change between years, LANDFIRE targets fire-

related disturbance. It was created in partnership among the U.S. Forest Service, Department of Interior, and The Nature Conservancy. This product provides layers starting in 2001, updated every two years. There is no reported accuracy, only "agreement" due to the small reserved sample size of ground plots, shifting field sampling methods, and the overall emphasis on timely production (USDA et al. 2019). Reported agreement values from the earliest product, 2001, are as follows: Northern Plains report a 68% agreement on existing vegetation type, lowering to 63% average agreement across all zones east of Rocky Mountains in the contiguous United States (USDA et al. 2006). Currently, the LANDFIRE products emphasize identifying annual disturbances rather than projecting further into the past, as with LCMS and LCMAPS.

Cropland Data Layer (CDL) - also known as Cropscape through its portal name- is a popular tool that provides finer classification of crop field to crop type (Lark et al. 2021), and spatially estimates >100 crop types and several land classes (USDA 2022). However, the provided annual data varies between states; Kansas's earliest layer reflects conditions in 2006. After the initial start, CDL reports annual layers every year up to 2022, which is appropriate for detecting changes in producer crop choice over time but may underestimate planted acreage in earlier years (Lark et al. 2021). The CDL has many applications and can be derived into a pesticide use footprint (Budreski et al. 2016).

Continuous Cover Products

Multi-Resolution Land Characteristics Consortium comprises multiple federal agencies and provides many products spanning different regions across North America. Most products are only available for the western United States (e.g., Rangeland, Condition, Monitoring, Assessment, and Projection; Shi et al. 2022), but it also hosts products across the contiguous

United States, like the immensely popular NLCD. Another continuous United States product includes a special NLCD Tree Canopy Cover annual product for 2011-2021 presented in 30 m resolution, with each pixel as a continuous value representing the percent cover (USFS 2023). This tree cover product predicts land cover values from multiple remote sensed products and two algorithms: 1) LandTrendr with Landsat and Sentinel imagery sources and 2) random forest with CDL and LiDAR 3DEP (USGS 2019, Lark et al. 2021). Additional data cleaning was conducted by masking areas defined as urban, water, or crop (Houseman et al. 2023). Accuracy was calculated with 30% holdout data (RMSE 13%). This tree canopy product intends to provide a wide temporal scale with annual time steps. Tree canopy is defined by species and the extent of foliage (Houseman et al. 2023).

Another tool largely used for documenting percent woody cover is maintained in the Rangeland Analysis Platform (RAP). The RAP provides two annual products, biomass and land cover, between 1986 and 2022 (Allred et al. 2021). The land cover products are partially derived from Landsat and a collection of 75,000 field samples from the Natural Resource Conservation Service (NRCS) and Bureau of Land Management 2004-2019. In the development of RAP spatial layers, 10% of field samples were retained for model validation and calculating Root Mean Square Error (Allred et al. 2021). Land cover within RAP comprises five data products representing different plant functional types: annual forbs and grasses, perennial forbs and grasses, shrubs, trees, and bare ground. Like USFS's tree product, it is a fractional product and provides a percentage (0-100%) of vegetation cover rather than categorical classification (e.g., grassland, cropland, woodland) at 30 m resolution. The most recent product, version 3, uses Landsat Collection II imagery, additional ground samples and an artificial neural network to predict each plant functional type (Allred et al. 2021). The tree layer reports the lowest

inaccuracies of all plant functional types (MAE 2.6%, RMSE 6.7%). With that, there are accuracy errors of stark contrast objects on the landscape (i.e., center pivot crop fields predicted as dense woody vegetation; Figure 1.2). Fortunately, there are solutions to minimize error at the consumer level by connecting with other landcover products. To address this, I masked pixels of each year's RAP tree prediction within the United States Great Plains labeled as cropland for each respective year's LCMAP landcover product. Plotted together, the extracted tree layer estimated more total woody cover than the corrected "cropless" layer yet follows the same positive trend over time (Figure 1.2).

Pesticide Use

National Pesticide Synthesis Project provides annual estimates of each compound applied within the contiguous United States at the county scale (Thelin and Stone 2013, Baker and Stone 2015). Included compounds can be categorized as insecticides, herbicides, fungicides, or rodenticides. Application rates are surveyed at the farm level by a third-party consultant and stratified by pesticide compound, crop type, and year (Baker and Stone 2015). In addition, it provides two estimates that are calculated separately in response to missing survey data from crop districts. Crop districts are counties grouped within each state by similar geography and agriculture practices (USDA 2018), and all are not always surveyed. For non-reported crop districts, EPest Low reports zero application, while EPest High extrapolates data from neighboring districts to estimate application for each year. Some of the published literature using these data take the average of the two values (Toda et al. 2021), while others use the high estimate (Ertl et al. 2018). The high estimate may reflect trends more accurately and depicts an overall increase despite not including seed coating in the total estimated application post-2014 (Figure 1.3).

Resolution

After compiling multiple options, I selected the following spatial products: LCMAP (grassland), RAP (woody vegetation), and National Pesticide Synthesis (Table 1.1). The utilization of these land cover and use data are demonstrated in the following two chapters. Chapter 2 applies BBS to examine trends in meadowlark (*Sturnella neglecta* and *S. magna*) breeding season abundance throughout the United States Great Plains. Chapter 3 explores the role of each factor within one Great Plains state and species. Kansas is one of two states with a robust population of greater prairie-chickens (*Tympanuchus cupido*). Coupled with an extensive database maintained by the Kansas Department of Wildlife and Parks, Chapter 3 links lek site selection to possible threats 1993-2020. Both chapters explore the relationship and strength of three threats to grassland bird populations: grassland loss, woody encroachment, and pesticide use. Concurrently, my research emphasizes the significance of monitoring efforts and publicly available data in developing management strategies and conservation planning.

Figures

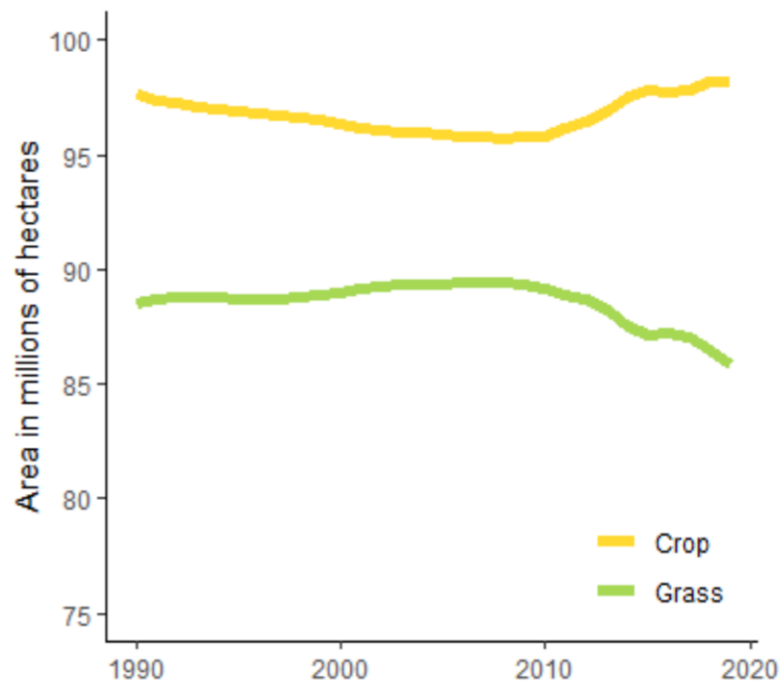


Figure 1.1. Comparison of changes in the two largest land cover types, cropland and grassland, in the Great Plains, USA 1992-2019. Cover types were classified and estimated from data provided by Land Change Monitoring, Assessment, and Projection (LCMAP) annual land cover product. Notably, trends mirror each other after 2010.

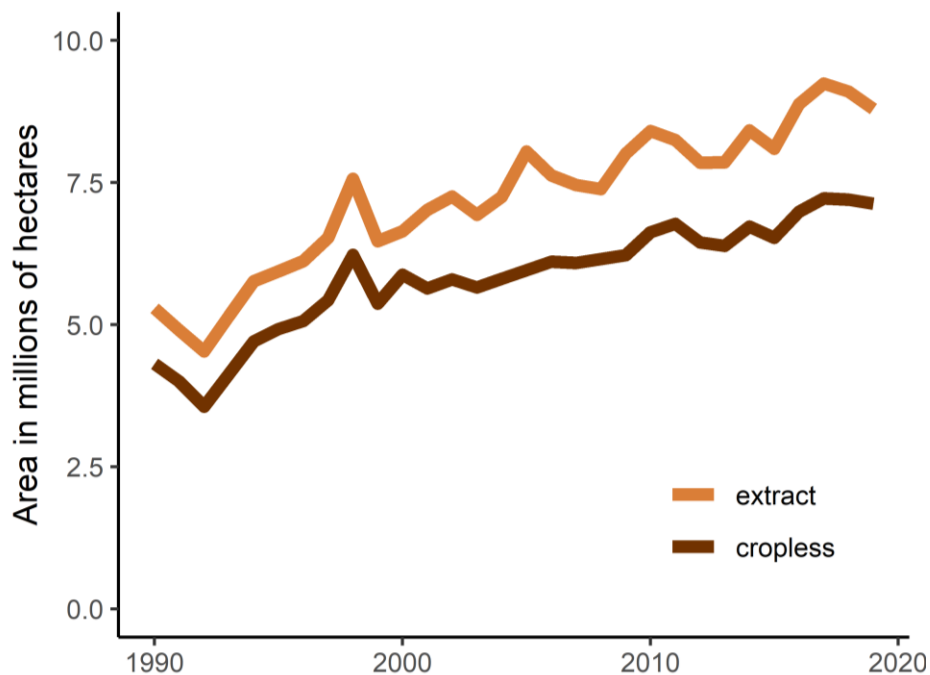


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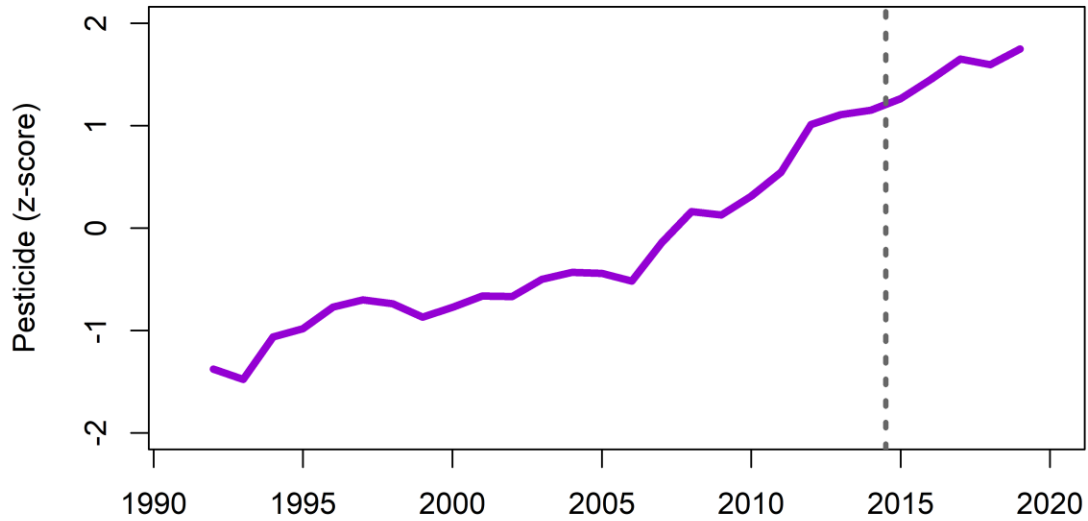


Figure 1.3. Estimated applied pesticide values (standardized by z-score) in the United States Great Plains between 1992-2019 (Thelin and Stone 2013, Baker and Stone 2015). Values are derived from the ‘high estimate’ provided by the National Pesticide Synthesis Project. Plotted are the sum of 61 selected compounds and families including glyphosate; 2, 4-D; organophosphates, and neonicotinoids. The vertical dotted line depicts when data collection efforts change and no longer include seed coatings after the line.

Tables

Table 1.1. Spatial layers considered for the study. All datasets originated from sources that are available and free to the public.

Highlighted products were selected as data sources for this thesis.

	Dataset Name	Authoring Org.	Product	Spatial Scale	Temporal Scale	Data Source	Methods/Algorithm	Accuracy
Bird Pop. Trend	North American Breeding Bird Survey (BBS)	USGS (Robbins et al. 1986)	Species count by route	25mi long routes	1966-present	Trained volunteers conducting 3-min point counts following protocol	-	-
	Christmas Bird Count (CBC)	Audubon	Species count by location	15mi diameter circles	1900-present	Number of observers(volunteers) vary - tally birds along established routes	-	-
Land Cover	Rangeland Analysis Platform (RAP)	NRCS	Version 3.0	30m resolution	1986-2022	Landsat Collection II Field Samples: NRCS + BLM + NCPN (n=75k)	Artificial neural network with corrected remotely sensed imagery and 90% ground sites	Withheld 10% (n=7,500) RMSE: Perennial (14%) Shrub (8.8%) Tree (6.7%) Annual (11%)
	Landscape Change Monitoring System (LCMS)	USDA: U. S. Forest Service	Version 2020.5: Land cover annual layers	30m resolution	1985-2020	Landsat Collection II + Sentinel-2 70k aerial photos + 1,100 ground sites	LandTrendr and CCDC	NA
	Land Change Monitoring, Assessment, and Projection (LCMAP)	USGS: EROS	CONUS Collection 1.2	30m resolution	1985-2020	Landsat Collection 1 + DEM + WPI + NLCD + 25k ground sites	CCDC	grass/shrub v. cropland 90%; grass/shrub v. tree 87%

Land Cover	NLCD Tree Canopy Cover (TCC)	USDA: U. S. Forest Service	Version 2021.4	30m resolution	2011-2021	Landsat Collection II + Sentinel-2 + 3DEP + CDL	LandTrendr and random forest Masked crop, water, and urban areas	Withheld 30% of response RMSE: 13% MAE: 8%
	LandFIRE	USGS: EROS (USDA et al. 2006)	LF 2.0.0 Existing Veg Cover (EVC)	30m resolution	2001-2019	Landsat + NLCD + FIA + PRISM	MIICA	Agreement N. Plains: 68% (n=621) National: 63% (n=4,210)
	Cropland Data Layer (CDL)	USDA: NASS	CropScape	30m resolution	2006-2022	Landsat + Sentinel-2	NLCD training data	Varies by state avg ~85%
Pesticide	National Pesticide Synthesis Project	USGS: NAWQA	CONUS	County	1992-2019	20,000 surveyed farm operations + NASS harvested crop acreage	EPest Low – Extrapolates within CRD EPest High – Extrapolates within CRD and to neighboring non-reported CRD	-

Chapter 2 - A Broad Evaluation of Landscape Threats to North American Great Plains Meadowlark Populations

Human activity and modification have triggered different extents of degradation, impacting nearly half of global grasslands (Gang et al. 2014, Bardgett et al. 2021). In North America, the Great Plains lost 70% of its grasslands by the beginning of the 21st century (Samson et al. 2010). Since then, economic factors have induced greater grassland conversion to cropland (Rashford et al. 2011, Lark et al. 2019), with a conversion rate rivaling South American deforestation rates by 2010 (Wright and Wimberly 2013).

Accelerated rates of land conversion, shifts in producer crop choice, and heavy use of chemical fertilizers and pesticides are all components of agricultural intensification (Matson et al. 1997). This process is amplified to support higher global food production, yet has driven much of the biodiversity loss in the North American Great Plains (Green et al. 2005, Sánchez-Bayo and Wyckhuys 2019). Sources have highlighted the unintended consequences of human activity on ecosystems and their wildlife since the 1960s (Carson 1962). Long-term monitoring efforts are required to understand the lasting impacts on wildlife populations. One ongoing example is the establishment of the North American Breeding Bird Survey (BBS) in 1964 (Robbins et al. 1986, Sauer et al. 2017). Since the program's inception, trends have indicated an alarming decline in native bird abundance (Sauer et al. 2017). Current projections show North America has lost >2.5 million native birds, and the grassland bird guild has suffered the most loss (Correll et al. 2019, Rosenberg et al. 2019). Numerous anthropogenically caused threats, such as the agricultural conversion of temperate grassland, the invasion of woody vegetation, and

potentially the increased application of pesticides across the Great Plains, can result in declines in the occupancy and density of grassland birds (Bernath-Plaisted et al. 2023).

Grassland conversion and development for agricultural purposes contribute to a direct loss of breeding habitat and are the most attributed causes of grassland bird population declines (Bernath-Plaisted et al. 2023, Douglas et al. 2023). Over the past decade, an additional 50 million hectares of grassland were converted in the Great Plains to cropland (Augustine et al. 2021). Additional loss of grassland results from energy development and infrastructure construction, which continues to expand (Allred et al. 2015, Thompson et al. 2015). Multiple factors have likely led to a loss of continuous breeding habitat for grassland bird species across North America and continue to threaten their persistence (Knopf 1994, Londe et al. 2019a).

The remaining parcels of grassland in the North American Great Plains are at greater risk of diminishing habitat quality for grassland birds. Degradation is inevitable without proactive land management practices involving disturbance (Van Auken 2009). Historically, the Great Plains had disturbances caused by large free roaming native herbivores and fire that interacted with periodic drought (Vickery 2000, Archer et al. 2017). Without natural disturbance, woody-stemmed plants invade grassland-dominated systems in a process termed woody encroachment. The woody cover potential is greatest in high annual precipitation areas but subject to change with shifts in precipitation patterns and global rise in atmospheric CO₂ caused by climate change (Sankaran et al. 2005, Archer et al. 2017). Across North America, annual expansion of woody cover is greater than five times larger in the Great Plains than surrounding areas (Barger et al. 2011). Occupation of woody vegetation leads to the loss of ground cover for many grassland birds and can cause their population distributions to shift (Graves et al. 2010, Lautenbach et al. 2017, Roberts et al. 2022).

Fragmented grasslands can be further weakened by agricultural practices in surrounding landscapes. Even when not directly treated, grasslands are susceptible to foliar chemical deterrent application (hereby, pesticide use) from nearby agricultural fields or pastures due to runoff (Struger et al. 2008) and drift (Egan et al. 2014). Agricultural intensification is commonly correlated with pesticide use; the amount of applied pesticides continues to rise across the Great Plains (Thelin and Stone 2013, Baker and Stone 2015). Chemical pesticide applications may directly affect survival of birds by ingestion (Prosser and Hart 2005, Roy et al. 2019) and reproductive behaviors (Fry 1995); or indirectly, by reduction of resources. Food resources such as arthropods and non-target plants may be depleted below levels that sustain broods (Boatman et al. 2004, Gibbons et al. 2015). Drastic increases in pesticide practices are especially concerning as pesticides' acute or long-term effects on grassland bird vital rates remain unclear.

The effects of grassland conversion and woody encroachment on grassland bird populations are well documented. However, how these threats rank compared to pesticide use has yet to be determined. Here, I asked if these three factors affect grassland birds at large, regional scales. I hypothesized that grassland conversion, woody encroachment, and pesticide use have negatively affected grassland bird abundance in the North American Great Plains. To examine these associations, I evaluated each purported threat's relationship and relative contribution to the abundance of two grassland obligate passerine species across the Great Plains, USA, during a 23-year period (1996-2019).

Methods

Study Area

This study focused on the United States extent of the North American Great Plains. Across the Great Plains, a precipitation gradient (30 – 80 cm annual rainfall) combined with topographic variation creates diverse vegetation structure and composition (Adler and Levine 2007, Augustine et al. 2021). Under the vegetation lies nutrient-rich soil, with areas close to irrigable water sources experiencing greatest tillage risk (Samson et al. 2010, Sullins et al. 2021). This has led to much of the Great Plains becoming an irregular landscape reflected by the differing land use operations. As a result, abundance and occupancy of grassland bird species vary across the range. To account for this, I stratified the Great Plains into five existing regions delineated across North America by similar landscape patterns, bird communities, and management goals called Bird Conservation Regions (BCR; Figure 2.1; Bird Studies Canada and NABCI 2014). Stratification by BCR of selected bird abundance data is appropriate for estimating population change at a landscape scale (Sauer et al. 2003).

Data Preparation

Bird Abundance

I selected the North American Breeding Bird Survey (BBS) dataset as it has the broadest temporal and spatial scale available for estimating trends in bird abundance in the United States. The BBS was designed for long-term monitoring collected by trained volunteers and organized by the US Fish and Wildlife Service and later the US Geological Survey (Robbins et al. 1986, Sauer et al. 2003). Annual surveys were conducted along 25-mile-long transects consisting of 50 three-minute roadside point counts documenting number of individuals audibly and visually

detected during the breeding season (late May – Early July; Robbins et al. 1986, Sauer et al. 2017).

I designated eastern and western meadowlark species (*Sturnella magna* and *S. neglecta*) as indicators of grassland bird population change. Both species are grassland obligates (Vickery et al. 1999) that exhibit a high tolerance for a variety of plant associations (Lanyon 1956).

Meadowlarks can be linked to pesticide use through their direct use of row-crop fields (Best et al. 1997) and have a diet that predominately relies on insects in the breeding season (Skutch 1996). These birds are characterized by bold natures (Davis and Lanyon 2020, Jaster et al. 2022) and contribute consistently high detection rates during roadside surveys (Rotenberry and Knick 1995, Wellicome et al. 2014). Physically, both species are similar in stature and behavior (Davis and Lanyon 2020, Jaster et al. 2022). To avoid potential observer misidentification, I totaled eastern, western, and ‘unidentified meadowlark’ counts to estimate meadowlark abundance.

Surveys that did not meet quality criteria (i.e., weather, route completion, date, or time; Ziolkowski et al. 2023), and routes that fell < 50% within the delineated study area were removed (Figure 2.2).

Data from the BBS are used with the assumption that changing land cover did not change meadowlark detection across Great Plains BBS routes. I summarized count data by route to examine the strength of the relationship among land cover variables and meadowlark abundance. These values were log-transformed (with the addition of a small constant, 1) to combat the right skew of count data, retain a simplistic statistical approach, and demonstrate low false positive rates in linear models (Ives 2015). While excellent for examining beta coefficients (Ives 2015), log-transformed data is difficult to interpret predictions compared to untransformed data (O’Hara

and Kotze 2010). Observations from BBS data were bulk downloaded using `bbsAssistant` package in R (Burnett et al. 2019, R Core Team 2023).

Vegetation Data

To relate threats to route-level trends in BBS abundance data, it is essential to use local land cover data (Weber and Theberge 1977, Bart et al. 1995). I used a remote-sensed product from USGS's Landscape Change Monitoring Assessment and Project (LCMAP) to estimate grassland loss (Brown et al. 2020). The land cover product has a predicted classification at 900 m² (30 x 30 m pixels) for eight broad categories (e.g., grassland/shrub, cropland, forest, etc.) on an annual basis (Brown et al. 2020). I estimated the amount of grassland (ha) within an 800 m buffer along routes that had completed surveys between 1996 and 2019. An 800 m radius was used in previous studies linking land cover variables to meadowlark occurrence on BBS routes (Niemuth et al. 2017).

It is difficult to remotely sense the gradient of woody encroachment on grassland-dominated systems with broad classifications. A solution to define change on very small scales is to use fractional cover products that provide a percentage cover of a vegetation type within each pixel (Jones et al. 2020). I used the Rangeland Analysis Platform's (RAP) tree vegetation cover product to estimate woody encroachment (Allred et al. 2021). Like LCMAP, this remotely sensed product is annually available at 30 x 30 m but additionally provides percent cover instead of a categorical classification. Because of the error associated with certain land cover types, I corrected by masking pixels of RAP's layer that LCMAP classified as developed, cropland, or water (Allred et al. 2021). Using the same approach as grassland, I estimated the amount of woody cover (ha) within 800 m.

Pesticide Use

I used the National Pesticide Synthesis Project's dataset to estimate trends in herbicide, insecticide, fungicide, and overall pesticide use (Thelin and Stone 2013, Baker and Stone 2015). The finest spatial resolution that is publicly available for this product is provided at the county scale. County totals are the product of crop production by county (provided by USDA and NASS 1998) and local compound application rate by crop collected by a third-party contractor (Thelin and Stone 2013). This product provides two estimates; the high estimate approximates values for unreported counties by extrapolating rates to neighboring, non-reporting crop districts (Baker and Stone 2015) and has been used in previous studies relating wildlife populations to land use (Ertl et al. 2018). I used a subset of the pesticide compounds due to a change in the data collection protocol starting in 2019. To resolve this, I narrowed total pesticides across all years included in this analysis to the compounds reported in 2019. The resulting sum of pesticide application (kg) includes compounds of interest such as glyphosate; 2, 4-D; neonicotinoids; and organophosphates. Application values were associated with routes using an area-weighted approach, effectively providing an index of local pesticide use change across time and space. In practice, I inputted 800 m buffered routes in the 'summarize within' tool in ArcGIS Pro 2.9.0.

All purported threats were standardized into z-scores to allow for direct comparison between each model's derived regression parameters. Data were prepared using ArcGIS Pro 2.9.0 (ESRI 2023) and R (R Core Team 2023).

Data Analysis

Model Coefficient Evaluation

This chapter's objective was to evaluate the relationship among grassland composition, woody cover, and pesticide use to meadowlark counts using large monitoring datasets. I sought to

simplify the modeling approach, thereby reducing assumptions. One large assumption with modeling is normally distributed error terms, a common condition with many modeling approaches. Many surveys are zero-inflated, meaning most will result in heavy-tailed residuals that violate this assumption. To account for this, I log transformed the meadowlark count data and estimated the relationship to standardized predictor variables using least squares regression. Nearly all routes included in this analysis were sampled more than once (Figure 2.2), and each survey is considered unique, regardless of observer or preceding year's count. With the large sample size ($n = 9,323$), I could stratify surveys into respective BCRs without losing statistical power (Murtaugh 2007, Ives 2015).

Before fitting the model, I assessed correlations among predictor variables using Pearson's correlation coefficient (Benesty et al. 2009). Due to the high correlation between grassland and pesticide use, I fit only single-variable models to avoid multicollinearity issues. Single-variable models are directly comparable because all independent variables are z-scores.

For model fitting, I retained a random selection of samples representing 75% of each region's surveys. With this subset, I estimated β_1 by minimizing three loss functions to rank each purported threat's relationship with grassland bird populations. The models did not include a year covariate, including a time covariate can obscure the landscape information I was attempting to extract from time series datasets (Hefley et al. 2016). The number of surveys that met conditions across regions remained stable after 1996 (Figure 2.3). I used the linear model

$$\mathbf{y}_i = \beta_0 + \beta_1 \mathbf{x}_{1,i} + \beta_2 z_2 + \beta_3 z_3 + \beta_4 z_4 + \beta_5 z_5 + \varepsilon_i,$$

where $\mathbf{y}_i$ is the observed number of total meadowlarks counted on each survey, i ($1, 2 \dots 6,993$). The threat, $\mathbf{x}_{1,i}$, represents the predictor variable of each model (i.e. grassland, woody vegetation, and applied pesticide) summarized for each route surveyed. The intercept term (β_0)

represents surveys conducted in the lowest performing region, BCR#22 (with the lowest average meadowlark counts). The remaining terms $\beta_2 z_2, \beta_3 z_3, \dots \beta_5 z_5$ represent the additional difference for each region (BCR#11, BCR#17, BCR#18, and BCR#19, respectively) in comparison to the intercept, accounting for the variance in starting populations. The error term, ε_i , represents the unexplained variability from the constrained part of the model and is assumed to be independent (Faraway 2014). I included a linear model without the independent term ($\beta_1 x_{1,i}$) as a null model containing only regional variation for comparison.

Predictive Accuracy

I ranked the four models based on two metrics. First, the strength of regression coefficient $\hat{\beta}_1$ or slope of each fitted line. I utilized the bootstrap method with 1,000 replicates to estimate 95% confidence intervals surrounding $\hat{\beta}_1$ using the R package mosaic (Efron and Tibshirani 1986, Pruim et al. 2017). Second, I ranked the predictive accuracy of each model estimated with the holdout data left out from model fitting (25%). I selected root mean squared error (RMSE) as the model evaluation metric (Chai and Draxler 2014). The metric is calculated

$$RMSE = \sqrt{\frac{1}{n} \sum_{i=1}^n \epsilon_i^2},$$

where n is the total number of routes used to test the model ($n = 2,330$), and ε_i represents the model's residuals. In simpler terms, RMSE is calculated by taking the square root of the average resulting from the 2,330 differences between the model's predicted and observed meadowlarks. To compare models, I used the change between the null and each predictive model to calculate $\Delta RMSE$. Data analysis was conducted using R (R Core Team 2023).

Isolating the Influence of Pesticide Use

I followed two approaches to isolate any correlated variables between the single fixed effect models. First, I examined relationships between a model's residuals and a correlated variable in question to determine if unexplained variation of one predictor may be related to a different factor. Then, I compared adjusted *R*-squared values between singular fixed effect model and an additive model of the two correlated variables.

Results

I fit four linear models with 6,993 completed routes, with an average route recording 83.3 individual meadowlarks (range = 0-871) totaled from 50 count surveys along each route. Routes were surveyed on average 12.1 times (range = 1-24). There was variation in the number of surveys among regions (BCR #11 = 1,204 routes, BCR #17 = 947 routes, BCR #18 = 1,346 routes, BCR #19 = 1,052 routes, and BCR#22 = 2,444 routes; Figure 2.3).

Model Coefficient Evaluation

Each regression coefficient direction supported my hypotheses; the amount of grassland was positively associated with more meadowlarks recorded, while greater surrounding woody vegetation and pesticide use was related to lower observed birds. Comparing derived, scaled parameters, both grassland ($\hat{\beta}_1 = 0.513$, 95% CI [0.463, 0.562]; Figure 2.5a) and woody vegetation ($\hat{\beta}_1 = -0.453$, [-0.487, -0.419]; Figure 2.5b) models produced strong relationships with meadowlark counts. The pesticide model produced a negative relationship with bird abundance ($\hat{\beta}_1 = -0.292$, [-0.328, -0.256]; Figure 5c). All three derived parameters have

confidence intervals that do not overlap zero, indicating each influenced meadowlark abundance (Figure 2.6). These relationships were exhibited across BCRs (Figure 2.5a-c). With the implementation of the regional additive effect, each BCR effectively worked as a replicate, further supporting the relationship (Figure 2.5d).

Predictive Accuracy

I estimated predictive accuracy to further compare meadowlarks relationships between the factors. The resulting ranking differed from those based on beta coefficient slope. The region-only or null model showed a predictive accuracy value of 81.07 RMSE (Figure 2.7). All three models of interest produced increased predictive accuracy (Figure 2.7). Compared to the null, the woody vegetation model had the least predictive error (74.54 RMSE, Δ RMSE = 6.52). The next ranked model by accuracy was the grassland model (77.60 RMSE, Δ RMSE = 3.46), followed by the pesticide use model (80.97, Δ RMSE = 0.09).

Isolating the Influence of Pesticide Use

Across the three covariates, only one pair recorded a significant correlation coefficient ($r > 0.4$). Woody cover was not correlated with other covariates (Figure 2.4, $r < 0.3$). However, grassland cover and pesticide use were negatively correlated (Figure 2.4, $r = -0.698$). Due to this, I sought a method to discern if the two covariates explained the same variance in meadowlark abundance. To isolate the influence of applied pesticides irrespective of grassland composition, I fit a least squares linear regression between residuals (unexplained variation) of a grassland model and standardized pesticide values. The resulting slope parameter indicated a negative trend and was considered informative as it did not overlap zero at the 99% confidence interval ($\hat{\beta} = -0.103$,

99% CI [-0.132, -0.074]; Figure 2.8). This relationship was confirmed with bootstrap samples of different sizes ($n = 10,000, 1,000, 100, \text{ and } 10$) to confirm 99% confidence intervals. Then, I compared adjusted R -squared values between the grassland model ($R^2 \text{ Adj} = 0.298$) and a model containing grassland and pesticide as an additive effect ($R^2 \text{ Adj} = 0.309$). The informative relationship with residuals and positive change of adjusted R -squared indicated that pesticides have explanatory power outside the variation explained by the grassland model.

Discussion

My study examined three potential factors influencing meadowlark populations at large, regional scales in the North American Great Plains. Among routes distributed throughout the central United States, models indicated declines in recorded meadowlarks are related to grassland loss, woody encroachment, and pesticide use. Results verify these three factors as threats to grassland birds while accounting for differences in distribution within regions. I used two metrics to confirm a relationship between grassland loss, woody encroachment, pesticide use and meadowlark abundance. Overall, grassland composition surrounding routes had the greatest influence on meadowlark abundance based on regression coefficients, while woody encroachment was the strongest predictor ranked by accuracy estimates (RMSE). The significance of these factors and applied pesticide to recorded abundances of grassland obligate species was emphasized through the bootstrap method.

With increased certainty, land cover and use relationships with grassland bird abundance exist at large, regional scales. In current trends, continental eastern meadowlark populations are expected to decline by 50% within two decades (Panjabi et al. 2017). Meadowlarks respond to land cover and land use factors at multiple scales (Bakker et al. 2002, Niemuth et al. 2017,

Schindler et al. 2023), resulting in lower local occupancy and abundance. With threats contributing to large-scale landscape alteration, grassland bird populations across the study area are declining (Correll et al. 2019, Rosenberg et al. 2019). Many grassland bird species once common are now a conservation concern throughout their range, including bobolink (*Dolichonyx oryzivorus*), chestnut-collared longspur (*Calcarius ornatus*), grasshopper sparrow (*Ammodramus savannarum*), and greater prairie-chicken (*Tympanuchus cupido*; USFWS 2021). These species are declining despite adapting and using agricultural fields as a proxy for grassland (Elliott and Johnson 2018). It is likely that other grassland wildlife species trend and respond similarly to regional changes in land cover and use.

When examining trends over a large, diverse region, I found that the amount of grassland was consistently the most important predictor of grassland bird abundance. My results follow other studies conducted at the route level correlating less grassland with lower grassland bird abundance and species richness (Hamer et al. 2006, Haroldson et al. 2006, Mahony et al. 2022, Schindler et al. 2023). As grass-dominated areas are a requirement for grassland birds during the breeding season, land conversion is direct habitat loss and will negatively affect grassland bird populations (Vickery et al. 1999, Askins et al. 2007). Continued grassland conversion is evident across the Great Plains (Waisanen and Bliss 2002, Samson et al. 2010, Rashford et al. 2011, Augustine et al. 2021) and the major association with the continental collapse of grassland bird populations (Rosenberg et al. 2019).

My results indicate that declines in grassland obligate populations have been tied to increasing woody encroachment at landscape scales, as other studies have reported (Keyser et al. 2020). At smaller scales, these findings are well established. Lower grassland bird occupancy has been documented relative to woody patch growth (Bakker et al. 2002, Grant et al. 2004). In

addition, pastures with greater woody vegetation cover report lower passerine nest survival (Monroe et al. 2019). This may be tied to the expansion of mesopredator distributions, which can also be related to woody encroachment (Barding and Nelson 2008). Woody encroachment can alter the surrounding ecosystem across the Great Plains, impacting producers (Barger et al. 2011). In rangelands, woody vegetation affects the producer's bottom line, totaling >\$4 billion (USD) in rangeland production loss over the past 30 years (Morford et al. 2022).

In efforts to maximize production and remain competitive as producers, row-crop agricultural pesticide application is rising (+150% in the Great Plains since 1992), trending negatively with grassland bird abundance. Other long-term monitoring datasets have resulted in similar relationships with pesticide use (Hallmann et al. 2014, Ertl et al. 2018). Although my approach does not account for pesticide application in pastures or hayfields, other studies have shown that chemically treating fields can decrease grassland passerine use (Buckingham et al. 2006), disregarding the lack of producer benefit (O'Meilia 2003, Fuhlendorf et al. 2009). Pesticide use is a symptom of ongoing agricultural intensification, and related activities have been considered the main threat to grassland birds (Stanton et al. 2018, Correll et al. 2019, Mahony et al. 2022, Douglas et al. 2023). Historically, mortalities have been reported from insecticide ingestion by feeding on grain from recently planted row crop fields (Griffin 1959), and direct mortality rates may still be underestimated (Linz et al. 1997). More recently, there are growing concerns surrounding non-lethal pesticide ingestion that may affect songbird demography, including lowered fertility rates (Fry 1995, Humann-Guillemot et al. 2019), and gut microbiota (Drovetski et al. 2022, Roy and Chen 2023). In addition, grassland insectivores and granivores may be experiencing indirect effects of pesticide use on vegetation and invertebrates up to 2 km from application (Krupke et al. 2012).

While examining land use trends, it is difficult to parse effects of fine-scale changes in vegetation cover from broad land cover variables. First, users must be aware of the error in data derived from remote-sensed products. All data have inherent error, and therefore, the resulting models are a simplification of reality (Grimm et al. 2005). However, examining trends over a large space and time allows insight into patterns that are otherwise hard to describe with smaller sample sizes. Secondly, land cover is all related. Specifically, the Great Plains grassland composition is not entirely independent of pesticide use sourced from cropland. In turn, I attempted to isolate the effect of pesticide use from grassland loss. Although slight, the results indicated that pesticide use provides additional consequences for meadowlarks independent of grassland loss.

Large monitoring datasets are very useful when considering inherent limitations; biases of BBS data are well documented (Droege and Sauer 1990). Many publications support complicated statistical models to overcome biases related to detection probabilities (Smith and Edwards 2021, Mimmagh et al. 2022) or provide evidence for data removal (Kendall et al. 1996). Most reiterate the importance of using local land cover concerning route level counts (Weber and Theberge 1977, Bart et al. 1995). Detection radii can be significantly reduced from open terrain to forested areas (Weber and Theberge 1977, Veech 2006a), so the detected number of birds may have been lower. Yet, the scale of the study and nature of indicator species minimize bias going into data analysis. Densely forested routes almost only occurred around one area: the Black Hills, South Dakota (<3% grassland within 800 m of route compared to 33% average across study area), and were anticipated to have low meadowlark recordings, respectively. Additionally, each route surveyed was assumed to be independent from preceding years and not impacted by changes in the observer. Fortunately, survey-wide detection rates of meadowlark species are

unaffected by the observer, weather, or time of year surveyed (CV = 0.12; Weber and Theberge 1977, Sauer et al. 1994).

Despite the observational nature and simple regression approach of this study, results are significant and provide multiple lines of evidence for a causal link. In addition, results follow five of seven considerations in support of causal evidence coined by Hill (1965), including strength, consistency, temporality, gradient, and plausibility (Faraway 2014). These traits exemplify the high performance of simplistic statistical approaches and further emphasize effects of grassland loss, pesticide use, and woody encroachment on grassland birds.

A growing volume of evidence linking declines in biodiversity due to modern agricultural practices highlights a needed shift to sustainable practices, including investment into ‘biopesticides’ and mindful application of current synthetic pesticides (Al-Ahmadi 2019, Lykogianni et al. 2021). Biopesticides are derived from natural materials and offer many benefits, including low residual outputs and host-specific applications (Gupta and Dikshit 2010). Still, producers are wary due to their low efficacy in changing weather conditions and high start-up costs compared to synthetic pesticides (Al-Ahmadi 2019). Synthetic pesticides are popular as producers reported larger return margins with low cost and ease of use (Osteen and Fernandez-Cornejo 2013). Appropriate synthetic pesticide application may be necessary; without pesticides, global food production yield may experience up to a 40% net loss (Lykogianni et al. 2021). However, overapplication leads to many risks, including reduced future efficacy through pesticide resistance and negatively affecting human, terrestrial, and aquatic health (Dodds et al. 2004, Sexton et al. 2007, Bøhn et al. 2014, Cuhra et al. 2016).

To protect all biodiversity, wildlife-conscious farming is vital, such as limiting pesticide application near protected areas and retaining patches of minimally altered grasslands (Green et

al. 2005, Sánchez-Bayo and Wyckhuys 2019). Global land protection is the most popular action called on to reverse negative impacts on the environment (Douglas et al. 2023). Land retirement programs, such as the Conservation Reserve Program (CRP), have positively affected meadowlarks and other grassland obligate species populations across the Great Plains (Veech 2006b, Riffell et al. 2008, Sullins et al. 2018). Avian species reported using CRP grasslands to nest responded positively, reporting increased nest success rates (>89%) across the Midwest (Best et al. 1997). Conservation Reserve Program grasslands can provide habitat for species of concern and range of grassland species may have expanded into landscapes following maturation of CRP grasslands (Sullins et al. 2018, Harryman et al. 2019). Land sparing through enrollment of CRP is beneficial but requires mindful land management to become habitat for grassland bird species (Reiley et al. 2019, Shew et al. 2019).

Management within remaining grassland is important, and effectiveness can be multiplied by cooperative action of neighbors across the Great Plains. Agricultural intensification has large effects outside the target application by altering the availability of surrounding resources and removing ecological drivers that sustain grasslands (Matson et al. 1997). To support high-quality grasslands, good management practices and diverse disturbances, including mechanical woody removal and fire, are necessary (Bielski et al. 2021). Haying or moderate-heavy grazing should be postponed to promote grassland birds until nests fledge, typically in mid-July (Campomizzi et al. 2019, Gruntorad et al. 2021). Fire treatment intensity can be lessened by high intensity grazing and promote woody vegetation in grassland systems (Wilcox et al. 2018), and frequent fire intervals can prevent woody transitions (Ratajczak et al. 2014). Prescribed burns on grazed pastures with rest periods to build fine fuels simultaneously reduce the spread of woody species and increase forage in patch-burn grazing systems (Wilcox et al.

2022). Deliberate and active disturbances through patch-burn grazing and rotational grazing systems promote vegetation structure and diversity, benefiting all grassland fauna without loss in agricultural productivity (Campomizzi et al. 2019, Lituma et al. 2022).

Conclusion

Grassland birds are in decline across North America. Associated factors related to diminishing grassland birds include habitat loss and degradation. Meadowlarks, grassland indicator species, are negatively trending with grassland loss, woody encroachment, and pesticide application across the US Great Plains. My results demonstrate the importance and utility of publicly available field monitoring and remotely sensed datasets. With simple statistical approaches, patterns between grassland bird abundance and broadscale threats are strong enough to meet five of seven criteria to be considered causal links. To alleviate the trends, cooperative and fast action is necessary to prevent further habitat loss. A collective effort of diversifying rangeland practices, restoring grasslands, and reducing reliance on synthetic pesticides will enable the Great Plains to benefit human and wildlife populations.

Figures

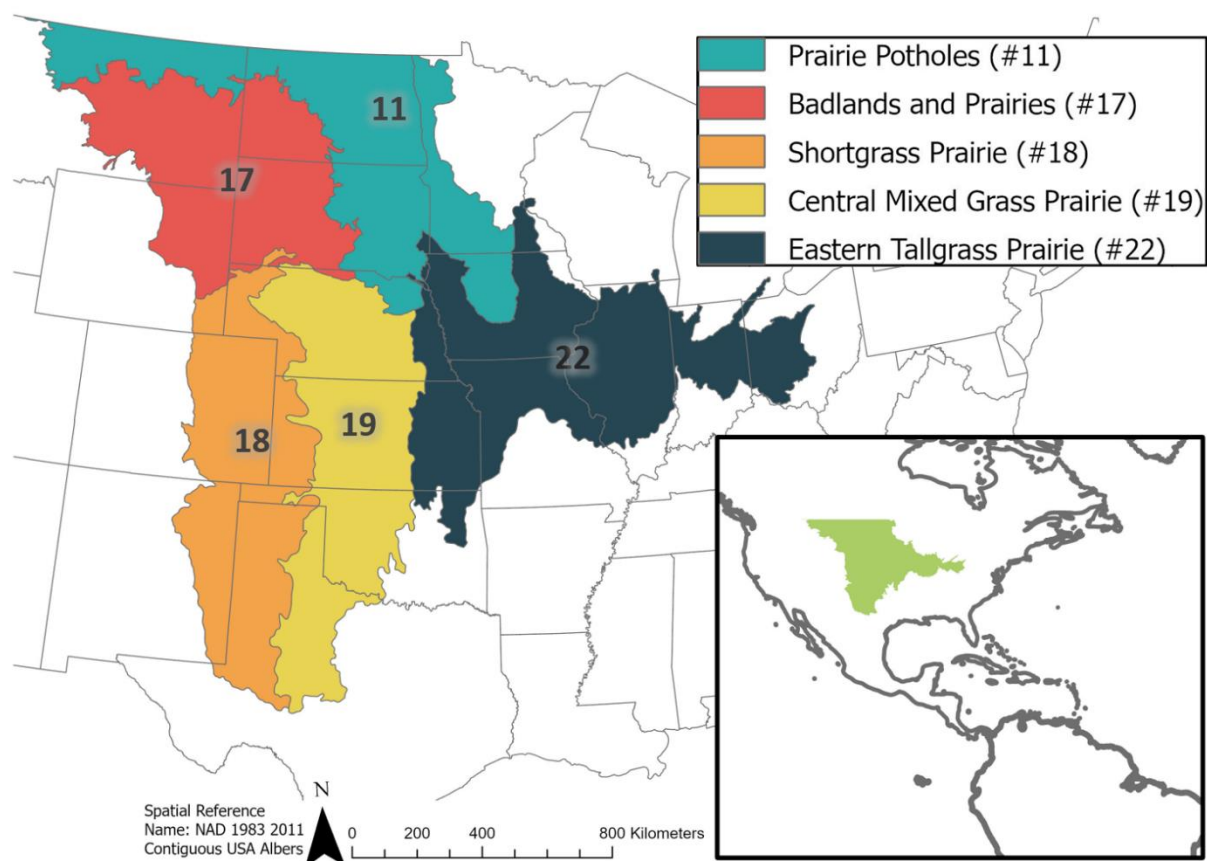


Figure 2.1. Map of United States extent of the North American Great Plains. The study area consists of five Bird Conservation Regions: Prairie Potholes (BCR#11), Badlands and Prairies (BCR#17), Shortgrass Prairie (BCR#18), Central Mixed Prairie (BCR#19), and Eastern Tallgrass Prairie (BCR#22). The color scheme is retained for all figures.

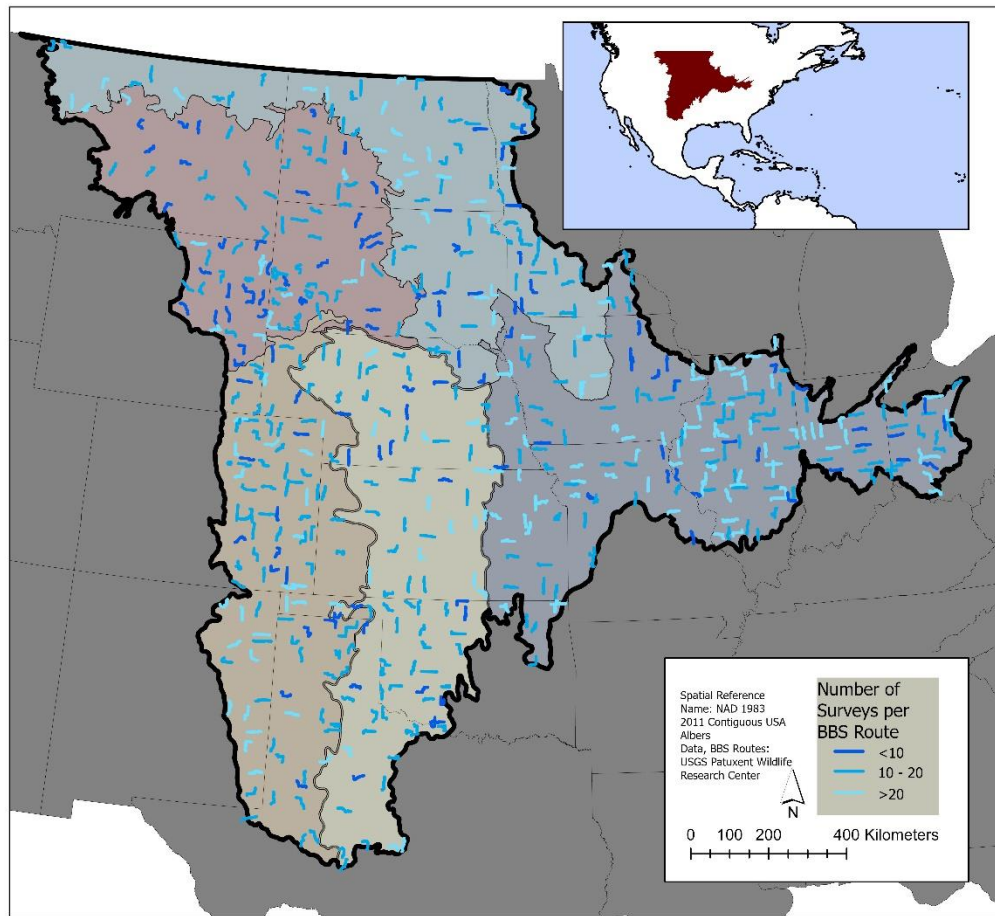


Figure 2.2. The study area describing the distribution and coverage of Breeding Bird Survey routes that were >50% within the delineated Great Plains ($n = 614$). Routes were grouped based on the number of surveys. Surveys were conducted once yearly (1996-2019). Most routes had >10 surveys conducted, with a large percentage of >20 surveys. The study area is outlined and subdivided by Bird Conservation Region.

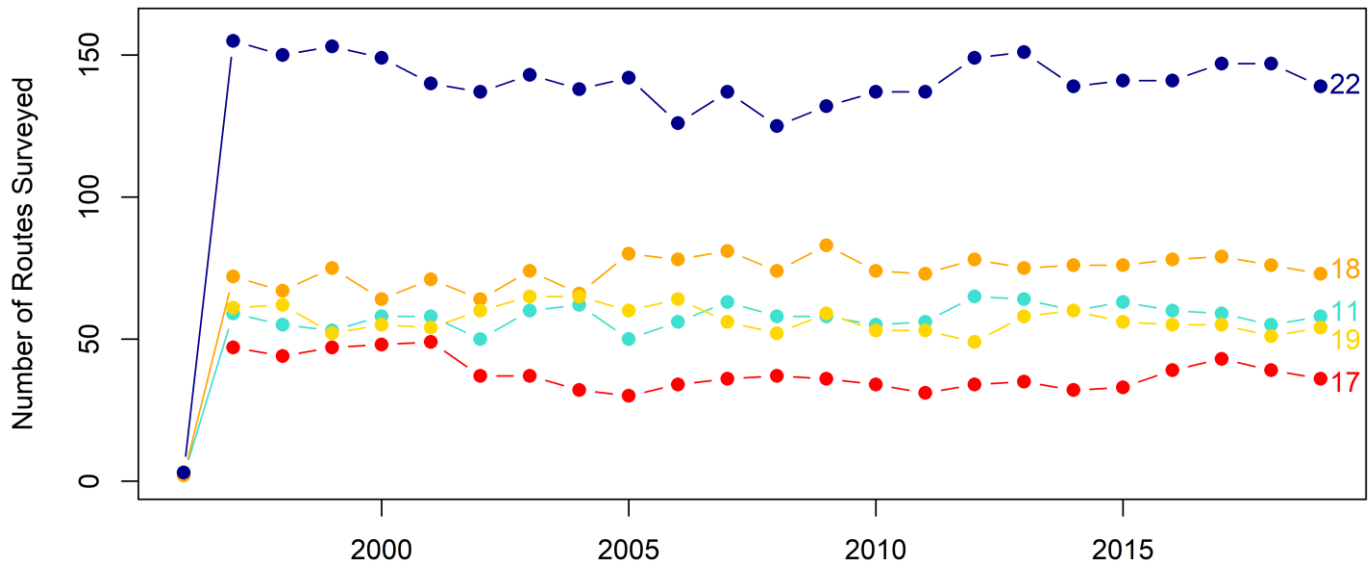


Figure 2.3. Survey effort by number of Breeding Bird Survey routes completed annually per region between 1996 and 2019. Few surveys met the quality condition in 1996. These surveys were retained as the data analysis approach did not include a year covariate. Trends remained stable between 1997 and 2019. The study area consists of the following regions: Prairie Potholes (BCR#11), Badlands and Prairies (BCR#17), Shortgrass Prairie (BCR#18), Central Mixed Prairie (BCR#19), and Eastern Tallgrass Prairie (BCR#22).

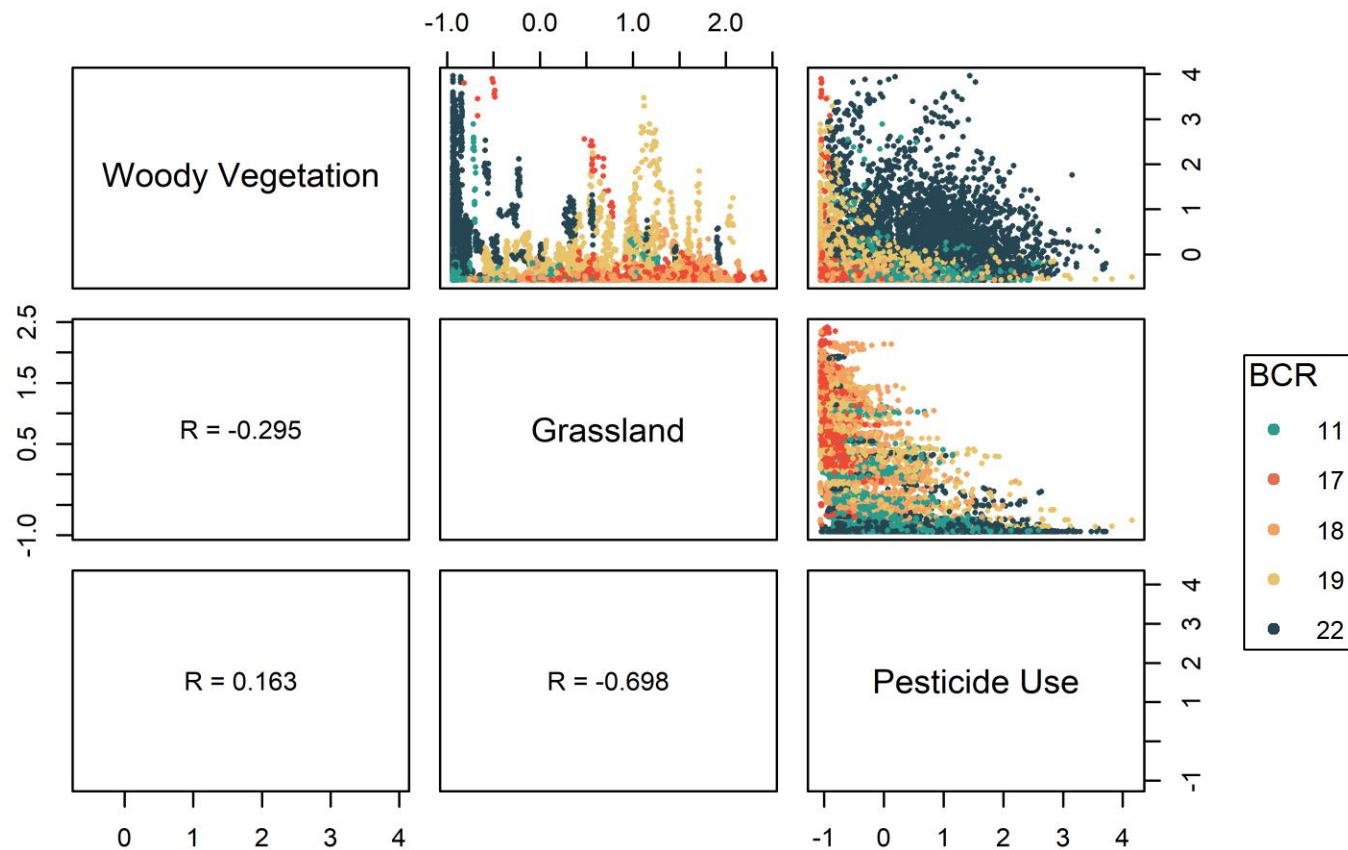


Figure 2.4. Correlation matrix of three standardized (z-scores) independent variables, woody vegetation, grassland, and pesticide use surrounding Breeding Bird Survey routes in 1996-2019 in the United States Great Plains (n=9,211). To better examine covariate trends with woody vegetation, z-scores greater than 4 were removed only for correlation assessment. A slight relationship existed between woody vegetation and grassland ($r = -0.295$). Grassland and pesticide use were negatively correlated ($r = -0.698$). There was no correlation between the remaining pair. Each survey was colored based on the following regions: Prairie Potholes (BCR#11), Badlands and Prairies (BCR#17), Shortgrass Prairie (BCR#18), Central Mixed Prairie (BCR#19), and Eastern Tallgrass Prairie (BCR#22).

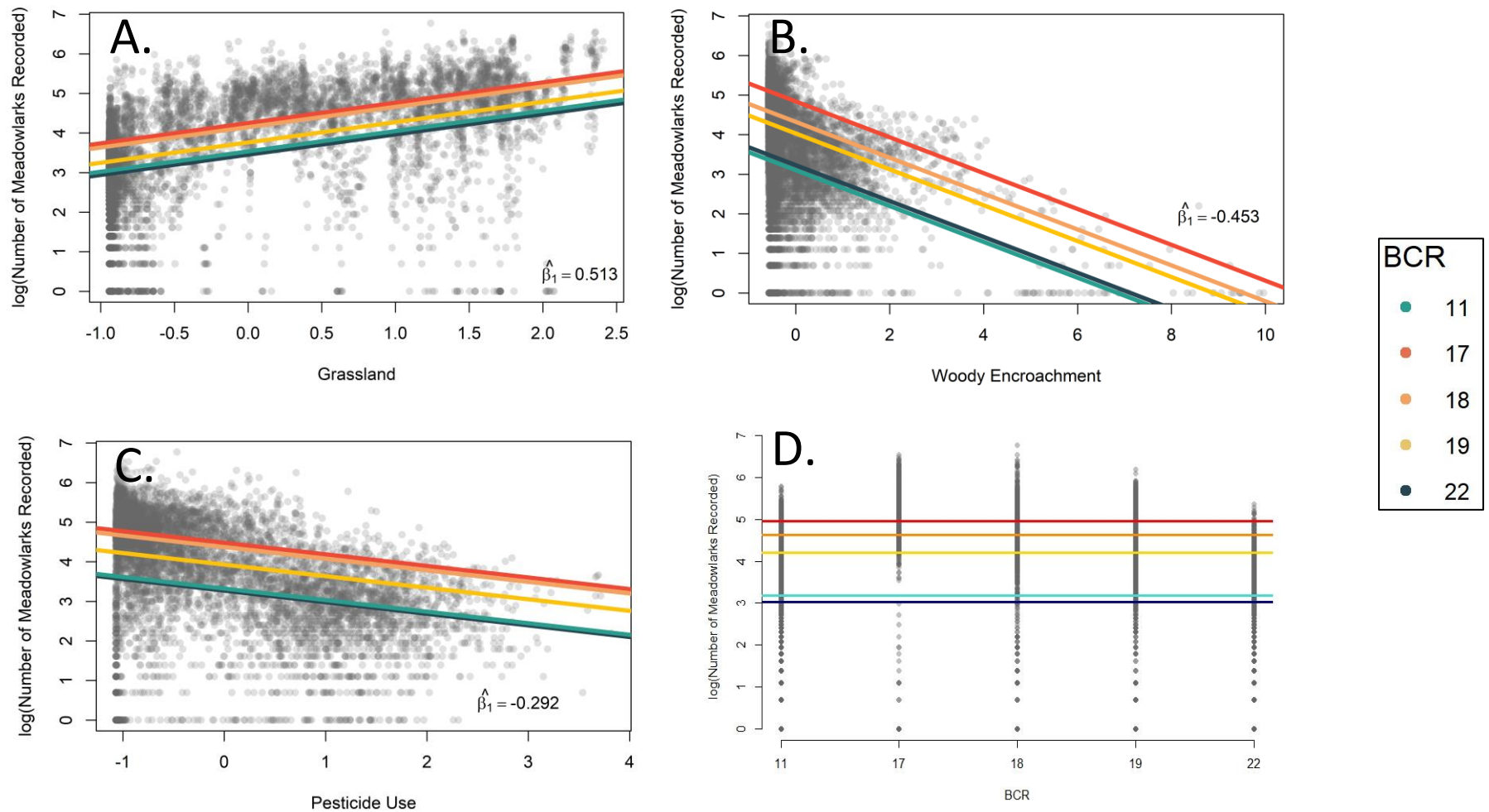


Figure 2.5 (a-d). Relationships of purported threat (z-score; a: grassland, b: woody vegetation, c: pesticide use) in the 800m surrounding United States Breeding Bird Survey routes to meadowlark (*Sturnella* spp.) abundance in the Great Plains (n = 6,993). The null, or region-only model, documents differences in average reported meadowlarks between regions (d). Meadowlark count data were log transformed and analyzed by fitting least squares regression functions. Each color represents a different Bird Conservation Region (see Figure 2.1).

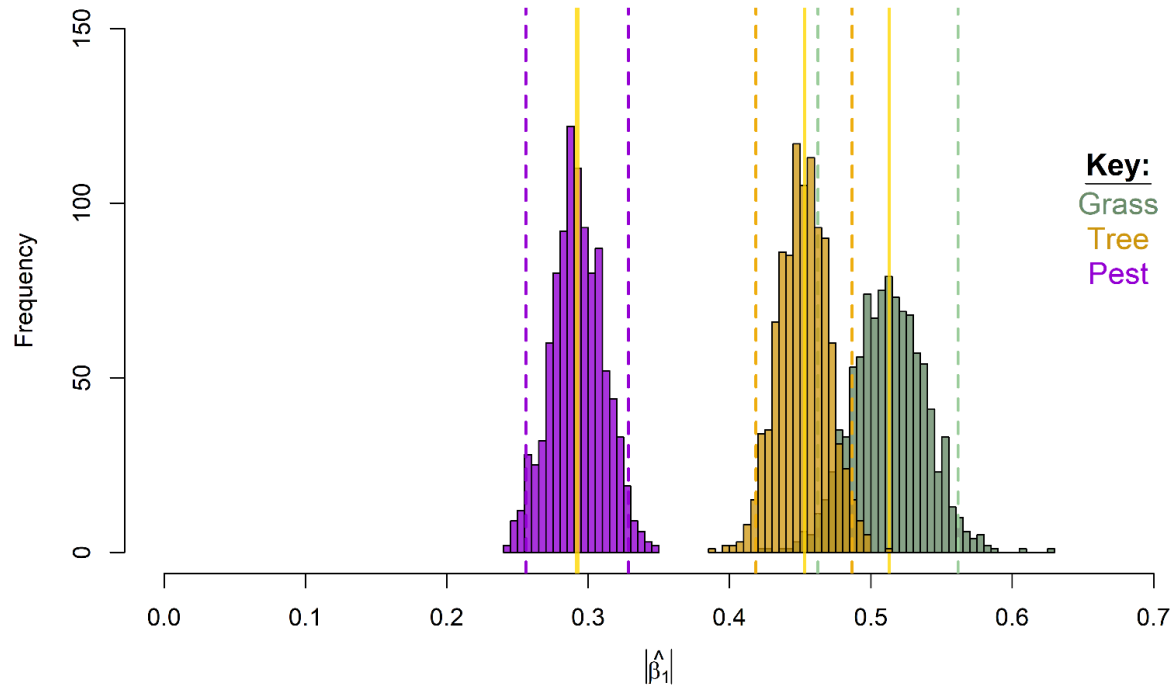


Figure 2.6. The empirical distribution of bootstrapped results ($n = 1,000$) for $|\hat{\beta}_1|$ for each least squares regression represented by color. Each distribution has the related 95% confidence intervals (dashed lines) and the estimated regression coefficient $\hat{\beta}_1$ (gold line). Distribution widths vary by model but estimated 95% confidence intervals did not overlap with zero.

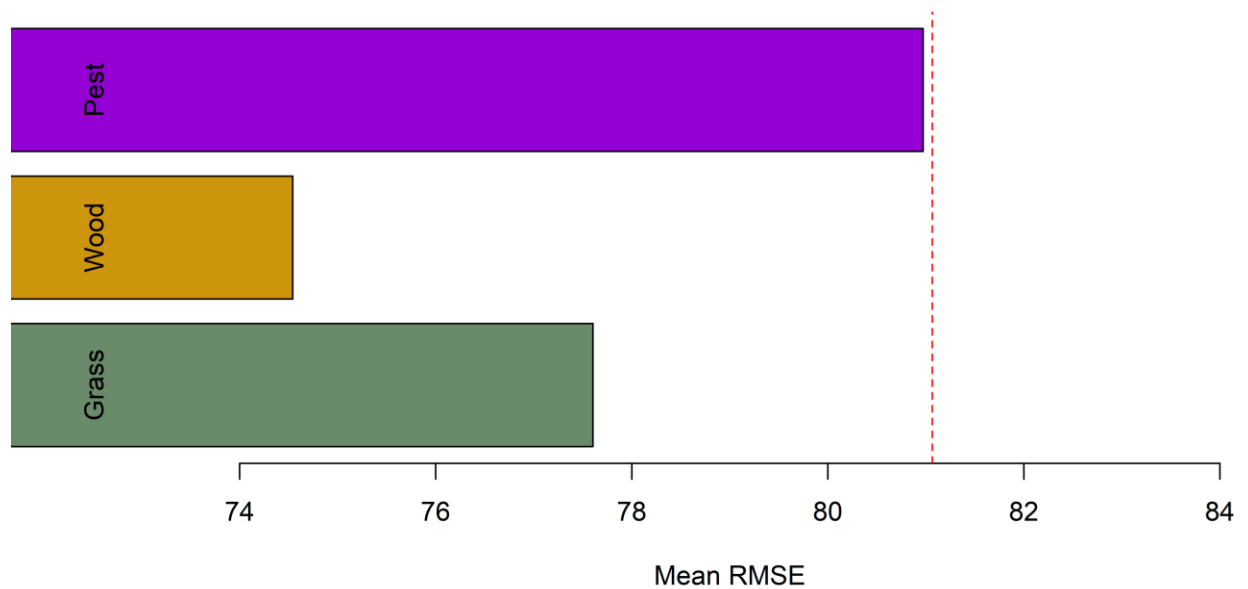


Figure 2.7. The predictive accuracy (RMSE) of fitted regression models using holdout data ($n = 2,330$ routes). Holdout data included 25% of United States Great Plains Breeding Bird Survey routes randomly selected from each of the five Bird Conservation Regions. The red dashed line represents the null or region-only model (RMSE = 81.065 meadowlarks). Each model had additional predictive power than the null model.

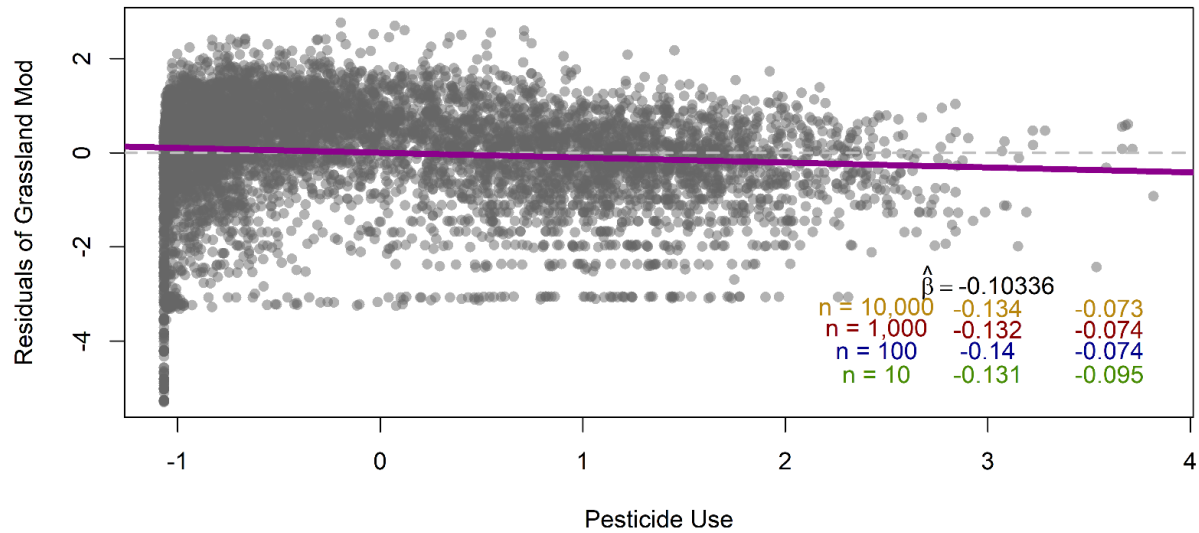


Figure 2.8. Residuals of the grassland model plotted against standardized (z-score) pesticide use values surrounding United States Breeding Bird Survey routes in the Great Plains (n= 6,993). A least square regression line (purple) indicated a relationship of pesticide to the unexplained variance of grassland, with confidence intervals estimated by the bootstrap method ($\hat{\beta}_1 = -0.103$, 99% CI [-0.132, -0.074]). Multiple bootstrapped samples (n = 10,000, 1,000, 100, and 10) were drawn to demonstrate the certainty of 99% CI and the significance of the resulting relationship.

Chapter 3 - Landscape-scale Constraints on Greater Prairie-Chicken Lek-site Selection in Kansas

Introduction

Across North America, bird populations have declined in recent decades, with the grassland bird guild experiencing the sharpest declines (Correll et al. 2019, Rosenberg et al. 2019). The largest contributor to this continent-wide decline is the loss of breeding habitat (Hill et al. 2014).

Temperate grassland within the Great Plains continues to be converted for agricultural purposes, and in some areas, conversion rates have increased (Lark et al. 2019). The quality of remaining grasslands for avian populations has been negatively affected by factors operating at broad landscape scales (Bernath-Plaisted et al. 2023). For example, land use operations including development of anthropogenic structures, land conversion, and pesticide application can influence avian space use (Londe et al. 2019a, Roy et al. 2019). One of the largest threats to grasslands is woody encroachment, a phenomenon described by woody vegetation that invades herbaceous dominated spaces due to a lack of diverse disturbances (Brennan and Kuvlesky 2005).

Many grassland obligate species in the Great Plains have diminished in abundance, including the greater prairie-chicken (*Tympanuchus cupido*; Correll et al. 2019). The greater prairie-chicken is a species of concern due to population declines and the near extinction of a subspecies (*T. cupido attwateri*; Svedarsky et al. 2000, Panjabi et al. 2017, USFWS 2021). In recent decades, Kansas is one of few states to have stable greater prairie-chicken populations (Applegate and Horak 1999). Unfortunately, long-term monitoring efforts indicate steady

declines in portions of Kansas, steep drops in recent estimates (Pitman et al. 2014, Nasman et al. 2021).

Most prairie grouse (*Tympanuchus* spp.) populations are monitored through annual lek surveys (Schroeder and Braun 1992, Haukos and Smith 1999). Lek surveys can be used to evaluate the landscape scale needs of these species (Hovick et al. 2015, Niemuth et al. 2017). Leks are a focal point in breeding season distribution for greater prairie-chickens (Westemeier 1973, Winder et al. 2015a). Males will continue to display throughout the breeding season and remain closer to the lek than females during much of the annual cycle (Robel et al. 1970, Winder et al. 2018). After copulating with a mate, females typically nest within 2 km of a lek (Hovick et al. 2015, Winder et al. 2018). Nesting habitat is reflected by nearby vegetation mainly composed of herbaceous, grass-dominated areas and are attributed to greater prairie-chicken lek occurrence (Niemuth 2005, Runia et al. 2021).

Although grassland-dominated landscapes help define quality greater prairie-chicken breeding habitat, cropland, at low proportions, may be beneficial (Horak and Applegate 1998, Svedarsky et al. 2000). After breeding, female greater prairie-chickens may nest near crop field edges that provide more accessible habitat for small chicks during the brood-rearing period (Sanderson et al. 1973, Matthews et al. 2011). With this proximity to cropland, greater prairie-chickens have an increased pesticide exposure risk and potential sub-lethal effects (Roy and Chen 2023). In addition, pesticides can erode and drift, indirectly influencing forb and invertebrate availability in local grasslands (Krupke et al. 2012, Goulson et al. 2015). Agricultural land use (e.g., row-crop farming and rangelands) directly affects the quality of targeted and neighboring grasslands.

In addition to landscape-scale factors, changes within grassland may also degrade habitat for greater prairie-chickens. For example, changes in vegetation structure and composition from intensive practices of grazing or fire implementations (McNew et al. 2012). Culturally driven management practices conducted simultaneously by many land managers can change the landscape at relevant scales (Robbins et al. 1986, With et al. 2008). Notably, extensive use of fire and grazing limits residual cover and becomes an immediate loss of nesting habitat (Svedarsky et al. 2000). With continued high-intensity grazing, lack of fuel weakens fire effects and promotes woody vegetation (Jessup et al. 2003, Knapp et al. 2008). Woody vegetation is a well-established deterrent of greater prairie-chicken space use (Niemuth 2005, Raynor et al. 2019, Gehrt et al. 2022). But, it is not clear what the woody cover threshold is before a lek will not subsist. While leks represent a focal point in breeding habitat, it is important to understand the landscape criteria of lek locations to better manage populations and target conservation.

To inform more strategic conservation efforts, range-wide assessments of available greater prairie-chicken habitats are necessary (Robb and Schroeder 2005). Research has documented *Tympanuchus* spp. habitat use can be predicted by land cover and land use (Niemuth 2011, Londe et al. 2019a). I focused on landscape features surrounding lek locations to evaluate landscape scale threats affecting greater prairie-chicken populations in Kansas. I aimed to 1) examine the relationship of lek locations with grassland, woody vegetation, and pesticide use at multiple scales, 2) identify potential thresholds of landscape variables related to lek-site selection, and 3) predict the past (1993) and current (2020) distribution of greater prairie-chicken breeding habitat at fine spatial scale (30 x 30 m). These objectives can be obtained using a flexible modeling algorithm, random forest, and long-term lek monitoring dataset provided by the Kansas Department of Wildlife and Parks (KDWP).

Study Area

To determine a study area, I delineated a minimum convex polygon around recorded greater prairie-chicken lek locations (1990 – 2022) in Kansas, USA (Figure 3.1) Lek locations are sourced from monitoring data maintained by Kansas Department of Wildlife and Parks (KDWP) personnel and cover most of Kansas's current occupied greater prairie-chicken range (Svedarsky et al. 1999, Robb and Schroeder 2005). The study area included a majority of an ecoregion of interest, the Flint Hills (US Environmental Protection Agency (EPA) et al. 2012), and represented a large portion (67% or 14,380,784 ha) of the state.

Kansas is an agriculture-focused state. Dominant agricultural crop products for 2006 included winter wheat (*Triticum aestivum*), grain sorghum (*Sorghum bicolor*), corn (*Zea mays*), and soybeans (*Glycine max*; USDA National Agricultural Statistics Service Cropland Data Layer 2006, 2020). The rank order of the top four products changed by 2020 (corn, winter wheat, soybeans, and sorghum, respectively), and combined planted acreage more than tripled (Figure 3.2). Between 1993 and 2020, the study area produced an estimated gain of 1.04% cultivated cropland (75,290 ha) and a loss of 1.9% grassland (119,688 ha; U.S. Geological Survey 2022). Most of the remaining grasslands in Kansas are working landscapes (i.e., rangeland/pasture) for beef cattle production (*Bos taurus*; Middendorf et al. 2009).

Along the west-to-east 50 cm precipitation gradient (Daly et al. 2008), Kansas grasslands host three types of grassland comprised of different dominant herbaceous vegetation: shortgrass prairie (plant height <40 cm; buffalo grass [*Bouteloua dactyloides*], blue grama [*Bouteloua gracilis*]), mixed-grass prairie (20-60 cm; little bluestem [*Schizachyrium scoparium*]), and tallgrass prairie (>50 cm; big bluestem [*Andropogon gerardii*], Indiangrass [*Sorghastrum nutans*], switchgrass [*Panicum virgatum*]; Shelford 1963).

Loss of Kansas grasslands can also be associated with woody encroachment. Native, invigorated species most problematic in the study area include eastern red cedar (*Juniperus virginiana*), honey locust (*Gleditsia triacanthos*), fragrant sumac (*Rhus aromatica*), American elm (*Ulmus americana*), and rough-leaved dogwood (*Cornus drummondii*). Exotic invasive species, including Siberian elm (*Ulmus pumila*), salt cedar (*Tamarix ramosissima*), and Russian olive (*Elaeagnus angustifolia*), have also encroached into Kansas grasslands. Between 1992 and 2020, woody vegetation expanded 78.7% (+285,682 ha) across the study area and 81.5% (+56,876 ha) in Kansas Flint Hills.

Pesticide applications have expanded rapidly over time. In the study area, the estimate of 61 applied pesticide compounds nearly doubled (94.2%), and within Flint Hills, it expanded four times (435%) from 1993 to 2020 (Thelin and Stone 2013, Baker and Stone 2015). Examined pesticide families and compounds include common row crop products that may be applied to rangelands, such as glyphosate (RoundUp®); 2, 4-D; dicamba; and organophosphates (malathion, chlorpyrifos, dicrotophos). Triclopyr (Garlon® 3A) is also considered due to its broad application to roadsides, crop edges, and mitigating woody vegetation (Vander Yacht et al. 2017).

Methods

Datasets

Greater Prairie-Chicken Lek Data

I obtained greater prairie-chicken lek survey data maintained by KDWP (August 2023 iteration). Between 1993 and 2020, lek surveys were conducted by various wildlife biologists, researchers, volunteers, consultants, and land managers between March and early May each year. In spring,

most male greater prairie-chickens engage in daily lekking behavior, and abundance counts are consistent in March-April (Schroeder and Braun 1992). I included leks identified from all efforts, including roadside surveys, aerial transects, scientific permits, and project developer surveys. Among all survey protocols, technicians recorded species, counted individuals, and a GPS location for each lek observation. Most (87%) of lek records were collected via roadside sampling. Road surveys were conducted twice annually for each county on established 16 km routes, with observers conducting ten 3-minute stops at predetermined locations spaced >1.6 km apart.

I analyzed all confirmed greater prairie-chicken surveyed lek entries between 1993 and 2020. I included leks identified as ‘unknown’ species that fell into greater prairie-chicken exclusive range and opportunistic observations that confirmed >2 individuals (Hovick et al. 2015). Available but non-recorded lek locations (pseudoabsences) were randomly generated within the study area as a surrogate for confirmed absence points. The minimum convex polygon was the furthest recorded extent, and the contained area is considered available for use by greater prairie-chickens. I generated 1,200 random points throughout the study area with exemption of areas identified as water, developed, or used (defined as area within 400 m of any observed lek between 1990-2020). Areas classified as water or developed were based on median year (2007).

Landscape Variables

Across Kansas, I examined grassland, woody vegetation, and pesticide use between 1992 and 2020. I compiled and obtained land cover composition data from multiple sources. I obtained percent grassland estimates from the annual land cover product from USGS’s Landscape Change Monitoring Assessment and Project (LCMAP) at a 30 x 30 m resolution (Brown et al. 2020).

This product follows the broad classification scheme of eight classes, for which I used “grass/shrub,” along with the remaining categories for data preprocessing of further datasets. For each lek location and pseudoabsence, I estimated the amount of grassland within a 5-km radius using focal statistics in ArcGIS Pro version 3.1.0. The 5-km radius is a scale of documented importance for *Tympanuchus* species (Winder et al. 2015a, Sullins et al. 2018).

Woody cover required a finer scale to quantify, and therefore I used a fractional cover product—specifically the Rangeland Analysis Platform (RAP) to produce a percent cover of woody vegetation within 30 x 30 m resolution (Allred et al. 2021). Each year, I corrected the tree layer by masking cropland, water, and developed pixels identified by the corresponding year’s LCMAP layer. Furthermore, I explored relationships with woody cover at multiple scales (200 m, 800 m, 1.6 km, and 2.5 km) to identify the most predictive scale. I estimated woody cover at these scales using focal statistics in ArcGIS Pro version 3.1.0.

USGS’s National Pesticide Synthesis Project estimates pesticide application at the county scale of the Great Plains states from 1992 – 2019 (Thelin and Stone 2013, Baker and Stone 2015). To interpret changes in application rates across time and space, I assumed pesticide application rates were consistent within each county. The dataset specifies >500 pesticide compounds, and I retained 61 compounds due to changes in collection procedures following 2018. Because I was most interested in current conditions, I restricted all years to the 61 compounds to get consistent estimates across time. Neonicotinoids are not included due to additional changes in sampling efforts in 2015; seed treatments were no longer inherently added to applied compound values in the dataset and impossible to parse apart at the consumer level. Interestingly, the dataset still exhibits an overwhelming application increase post-2015 regardless of the restriction to foliar applied amounts. I summarized pesticide values into separate

groupings as follows: glyphosate; 2, 4-D; triclopyr, dicamba, organophosphates (malathion, chlorpyrifos, dicrotophos), and all pesticides (sum of 61 compounds collected over time). I conducted focal statistics at a 5-km scale in ArcGIS Pro version 3.1.0 (ESRI 2023). This allowed for lek locations intersecting multiple county boundaries to be spatially weighted in application estimates.

For each recorded lek location, I attached landscape variables for their respective year. Except for pesticide data, due to the pre-growing season annual start of lek displays, I associated each lek location with the previous year's application rate because the bulk of each season's application occurs after April. For each pseudoabsence, landcover and land use variables were attached in the same systematic process as used points for each year, creating a total of 33,600 (1,200 x 28 years) pseudoabsences across the study area and period.

In an examination of lek persistence and predicted habitat, I also considered several anthropogenic features on the landscape. Anthropogenic features are explored for only the year 2020 for this project due to low data temporal accessibility. Structures built before 2018 were assumed to persist on the landscape in 2020. Multiple vertical structures are known to influence greater prairie-chicken space use (Hovick et al. 2014). The following structures were considered: oil and natural gas wells (subsequently labeled oil wells; Londe et al. 2019), wind turbines (Winder et al. 2015b), and transmission lines (Pruett et al. 2009, Londe et al. 2019). Spatial data were sourced from oil wells (HIFLD 2019), wind turbines (Hoen et al. 2018), and transmission lines (HIFLD 2022).

Data Analysis

Relating Landscape Variables to Lek Persistence

Before prediction efforts, I explored 20+ year lek persistence. I focused on leks observed in 1997 that followed the definition described in the previous section. Persistence status was determined if 1997 lek locations overlapped within 1.6 km of surveyed 2020 lek locations. All surveyed counties in 1997 were restricted to roadside sampling, with the same routes surveyed in 2020 (Figure 3.3). An 800-m buffer, or a total diameter distance of 1.6 km, was selected to allow for flexibility in exact lek location yet distinguish distinct leks (>1.6 km; Merrill et al. 1999, Niemuth 2000). Although a lek may be considered locally extinct in this implementation, the status can only convey local extirpation of greater prairie-chickens and probable loss of breeding habitat.

Land cover and land use variables for 2020 were attached to both ‘locally extinct’ and active lek locations of 1997. Variables included percent land cover at multiple scales and land use across multiple pesticide compounds and anthropogenic vertical structures. Grassland was singularly examined within 5 km, while scales of woody vegetation cover spanned 200 m, 800 m, 1.6 km, and 2.5 km. Pesticide compounds and vertical structures were considered at singular scales (5 km and 2 km, respectively). For structures, I calculated the density and occupancy of each structure: oil wells, wind turbines, and transmission power lines. To determine power pole density, I generated points along transmission lines at 250 m intervals.

Predicting Greater Prairie-Chicken Breeding Habitat

I used a random forest approach to predict current and past greater prairie-chicken habitat across Kansas. Random forest is a flexible predictive tool that explores nonlinear relationships between

the response variable and covariates, as long as use remains within the limits provided in the dataset used to fit the model. To do this, the random forest builds multiple classification trees trained on splits along each covariate's gradient to predict the relative probability of use (Breiman 2001). This approach has been successfully conducted with other grassland grouse species to predict species distribution (Sullins et al. 2019, Picardi et al. 2020).

Machine learning algorithms are especially sensitive to unbalanced datasets, and lek collection efforts vary by year. To balance the dataset and reduce the false positive rate, twice as many pseudoabsence points were randomly selected as used points for each year. Then, I retained 75% of this dataset, selecting proportionally within each year for model fit. The remaining 25% is used for model validation and determining the probability of use thresholds.

I assessed a global model for multicollinearity, removed culpable variables, and identified the most influential covariates using a variable importance measure called model improvement ratio (MIR; Evans and Murphy 2018). Although random forest variable importance measures can be biased towards favoring categorical features, the global model only had continuous predictor variables to rank, avoiding this bias (Strobl et al. 2007). The scale of woody vegetation or compound of pesticide that produced the highest MIR was selected for the pruned model. I executed the final, pruned random forest as a classification (0 – pseudoabsence, 1 – used) by growing 1000 trees.

To validate the model, I reported the out-of-bag (OOB) error rate calculated with the withheld data from the bagging process within the random forest algorithm. Although this algorithm does not inherently require independent validation (Evans and Cushman 2009, Murphy et al. 2010), I conducted external model validation with the reserved holdout data. Using the test dataset (25% reserved), I maximized specificity and sensitivity through the receiver operator

curve (ROC) plot-based approach to identify the best threshold of probability to use in our species distribution prediction (Liu et al. 2005, Robin et al. 2011). Using the ROC threshold reporting the lowest error, I created and compared two binary maps of breeding habitat for greater prairie-chicken for each year (1993 and 2020).

To determine the predictive strength of the final model, I reported the percent correctly classified (model accuracy) of both fitted and reserved data and calculated the area under the curve of the ROC plot. In addition, I used three additional metrics to confirm the occurrence threshold (delta.ss, kappa, and sum.ss).

My model approach does not account for land management practices, fine-scale vegetation structure, or other land use manipulation that influence greater prairie-chicken space use. To understand available habitat in 2020, I created a visualization of anthropogenic features on the landscape at that time (Figure 3.9). Based on previous literature evaluating avoidance effects on greater prairie-chicken space use, I used the following buffers: wind turbines at 1 km (Winder et al. 2015b), transmission lines at 700 m (Pruett et al. 2009, Londe et al. 2019), and oil wells 400 m (Londe et al. 2019).

Results

Lek monitoring in Kansas has been conducted across multiple data sampling efforts between 1993 and 2020. The study area encapsulates all the contemporary (since 1990) greater-prairie chicken range in Kansas (14,380,784 ha). Across the study area, I included 7,456 lek locations (143 – 520 annually) to meet my objectives. Established, active leks that were repeatedly surveyed were counted as individual leks each year. Surveyors recorded lek attendance and locations through different protocols; survey styles varied between established roadside surveys

(n = 5,590, including 898 opportunistic sightings), aerial transects (n = 359), scientific permits (n = 1,502), and project developer surveys (n = 5).

Relating Landscape Variables to Lek Persistence

I determined the status of 195 lek locations (Flint Hills n = 85) from 1997 by comparing recent lek activity in Kansas (Figure 3.3). By 2020, 145 leks were designated inactive (75% decline), or ‘locally extinct’, and 78 remained active. Among lek persistence statuses, mean values and associated standard deviations were summarized, including various landscape scales, pesticide compounds, and anthropogenic structure density and presence. Land cover means differed largely as expected. Active leks were associated with greater percent grassland, while woody cover was lower at persistent leks in all scales besides 200 m (Table 3.1). Pesticide compounds had mixed responses with high variance (Table 3.1). Oil wells and wind turbines were more common for extinct leks, while transmission lines were in greater density near an active lek. Several relationships with anthropogenic features reported standard deviations greater than the affiliated mean values, so I also noted the proportion of leks that had each structure type occupied within 2 km (Table 3.1).

Predicting Greater Prairie-Chicken Breeding Habitat

I randomly selected 75% of used/pseudoabsence entries to fit the species distribution model, totaling 5,552 used entries and 11,223 pseudoabsence locations (Table 3.2). The remaining 25% I retained for testing included 1,904 lek locations alongside 3,689 pseudoabsences.

My final model was well-supported and produced low error rates. The random forest with a fitted dataset returned an OOB rate of 10.63% (Presence had a higher error rate at 15.6%;

Absence had a lower error at 8.2%). The ROC returned the maximum specificity and sensitivity values (0.884, 0.897) and determined a threshold of 0.380 (Figure 3.6). Also using the reserved 25% data, the model reported 89.95% accuracy. My other performance metrics, delta.ss, kappa, and sum.ss, all returned a similar value (0.38 – 0.40). The area under the ROC was 0.956, indicating a reliable model (Figure 3.6; Liu et al. 2005).

Landcover Relationship to Lek Site Selection

All covariates of interest (grassland, woodland, and pesticide use) in the original model had MIR above 0.2, expressing informative power (Figure 3.4). The amount of grassland within 5 km of a lek location was the most important covariate across all available, uncorrelated variables, followed by woodland within 800 m (Figure 3.4). Grassland within 5 km was not correlated with woody vegetation within 800m ($r < 0.1$), but was strongly correlated with cropland (5km; $r = -0.97$). I assessed the correlation between cropland and local pesticides within 5 km and found they were not highly correlated ($r = 0.46$). Pesticide compounds ranked very closely in importance, possibly due to interaction effects. I selected ‘all pesticide’ as it served as the most accurate index of pesticide use on the landscape (Figure 3.4). The original model was pruned to these three variables to form our final model. Partial dependence plots also indicated the importance of predictor variables by examining the strength of the predicted response and visualizing the resulting relationship (Greenwell et al. 2018).

Identify Thresholds of Land Cover Related to Leks

Relationships of landcover variables followed the hypothesized direction to greater prairie-chicken breeding ground use. The relationship of the relative probability of use to grassland was

nonlinear; greater prairie-chicken probability of use remained low in landscapes having less than 60% grassland and peaked near 80% grassland within 5 km of leks (Figure 3.5). The next important variable was woody cover within 800 m of a lek location. The response to woody cover was drastic, showing a steep decline in probability of use from 0-10% woody cover (Figure 3.5). When woody cover exceeded 10%, the model predicted 0% probability of use by greater prairie-chickens, indicating a threshold of woody cover in relation to attended greater prairie-chicken leks (Figure 3.5). The outputted relationship with pesticide application consistently averaged 40% probability of use with any application rate, with use peaking at the lowest application amount (Figure 3.5). The flatness of this partial dependence plot indicates low variable importance (Greenwell et al. 2018).

Predict Current and Past Distribution of Habitat

I used the final model to produce two relative probability of use of greater prairie-chicken breeding habitat across the occupied range of the species for the earliest and most recent years possible with the temporal extent provided by the datasets, 1993 and 2020. Predictions indicated a reduction in probable use within the Flint Hills in 2020, but predicted greater probability of use in west and central Kansas (Figure 3.7)

To ease interpretation, I used the threshold determined by ROC (0.380) to produce binary maps of predicted habitat for each year. I predicted an 8.5% increase in total potential habitat between 1993 and 2020. Within the study area, the model predicted a total of 2,160,000 ha (15%) of potential habitat in 1993 and 2,340,000 ha (16.6%) in 2020 (Figure 3.8). Divided into individual ecoregions, the Kansas Flint Hills experienced the largest loss (-13%) despite having

the most habitat proportionally across all ecoregions in the study area. In 1993, the model predicted 910,000 ha (37.4%), dropping to 792,000 ha (33%) in 2020 (Figure 3.8)

Accounting for Anthropogenic Features when Evaluating Habitat

After accounting for vertical anthropogenic feature avoidance buffers estimated for greater prairie-chickens (and removal of open water and urban areas), estimated potential habitat in 2020 was 6% (-133,222 ha) lower than estimated based on percent grassland, woody cover, and pesticide use alone (Figure 3.9). Total area of breeding habitat was estimated at 2,214,116 ha after accounting for functional habitat loss in areas impacted by anthropogenic features.

Discussion

The random forest algorithm successfully evaluated grassland, woody vegetation, and local pesticide effects on greater prairie-chicken breeding habitat. My model predicted a positive relationship with grassland and a negative relationship with woody vegetation cover. Pesticides provided a slight addition of information, totaling a strong predictive model. Across Kansas at landscape scales, my model predicted an increase in potential habitat over time. In contrast, the Kansas Flint Hills experienced significant predicted habitat declines.

In addition to examining land cover relationships, use of random forest produced thresholds of site use regarding different scales of landscape metrics. Specifically, an 8,100 ha (20,000-acre) area should be at least 60% grassland to be potentially used by greater prairie-chickens. My results are consistent with the findings of other studies. Greater prairie-chickens respond to a majority of grassland within 5 km (Niemuth 2005, Winder et al. 2015a, Runia et al. 2021). The broad definition of grassland does not account for vegetation composition or

structure. As lek sites are a center point in breeding distribution, the surrounding landscape must provide reproductive habitat to support successful nests and broods. A diverse herbaceous structure allows residual cover for nests, provides transversal areas for broods, including bare ground, and protection from irregular precipitation and temperature events (Robel 1967, Buhnerkempe et al. 1984, Matthews et al. 2011, Londe et al. 2021). Reproductive habitat has specific vegetation requirements and therefore the surrounding landscape directly determines lek persistence.

Lek locations are very sensitive to nearby woody vegetation (Roberts et al. 2022). My model identified woody vegetation to be most influential within 800 m of potential lek sites. Other studies have indicated sensitivity to distance from the nearest tree or forest patch (Niemuth 2005, Londe et al. 2019a, Raynor et al. 2019, Gehrt et al. 2022). This follows the innate, avoidant nature of tall structures such as trees. Structures on open grassland provide vantage points for avian predators, and woody cover supports woodland-associated predators (Johnson and Temple 1990, Barding and Nelson 2008). Woody vegetation expansion can be the strongest predictor of space use and nest site selection within greater prairie-chicken home range (Matthews et al. 2013, Raynor et al. 2019, Gehrt et al. 2022).

My study examined the percent cover of woody vegetation at a landscape scale. We did not account for different configurations of woody vegetation. For example, small areas of dense woodland and widely interspersed trees may be categorized similarly within this scale, although ecologically different. Regardless of varying configurations, my results indicated that areas within 800 m that exceed the 10% threshold or contain woody cover totaling >20 hectares (50 acres) will result in local extirpation of greater prairie-chicken leks. This threshold is first to document density of tree threshold to breeding habitat for greater prairie-chickens, and similar

avoidances are reported in other *Tympanuchus* species (1,000m avoidance of trees >1m; Lautenbach et al. 2017).

Although rated as low importance after accounting for vegetation, the pesticide metric provided additional information in the model. Yet, no previous studies have examined the relationship of local pesticide use to greater prairie-chicken lek activity and knowledge of the overall effects of pesticide use on avian populations is limited. Pesticides have been identified in fecal samples of greater prairie-chickens and was not related to the amount of cropland in home range (Roy and Chen 2023). Through ingestion, Galliformes and other grain-eating birds may be subjected to sub-lethal effects of pesticide application (Gibbons et al. 2015, Gaffard et al. 2022, Roy and Chen 2023). Furthermore, effects of pesticides may reach beyond direct ingestion and contamination. Decreases in forb abundance and diversity and declines in associated invertebrates may degrade habitat for *Tympanuchus* spp (Lautenbach 2015). Particularly during the brood-rearing period when habitat includes high proportions of bare ground (Matthews et al. 2011), often provided by crop fields or disturbed pasture. Negative effects could be sourced from two avenues of pesticide use: seed treatments and foliar application. There is a rise in insecticide and fungicide seed treatments on nearly all planted corn in the United States (Roy et al. 2019, Krupke and Tooker 2020, Vojvodić and Bažok 2021). Foliar pesticide application negatively affects most invertebrates (Gunstone et al. 2021) and indirectly affects new plant availability up to 2 km away (Cornish and Burgin 2005, Krupke et al. 2012). Direct and indirect effects on resources cannot be understated, ultimately decreasing the efficacy of reproduction and occupancy by grassland birds (Boatman et al. 2004, Buckingham et al. 2006, Hallmann et al. 2014). My results highlight a growing concern of pesticide use surrounding grassland systems in

Kansas. Pesticide application estimates continue to increase despite projections claiming less use with the availability of genetically modified crops (Coupe and Capel 2016).

Pesticide estimates used in this project were mainly sourced from row-crop agriculture applications but also included some applications to rangeland pastures (Thelin and Stone 2013, Baker and Stone 2015). The lack of accessible and higher-resolution pesticide data is concerning and limits the ability to assess influences on leks. Such limitations to find potential negative consequences with such coarse data highlight the need for further research into the potential effects of pesticides on prairie-chickens and other grassland birds. In this study, pesticides and grassland surrounding leks were not highly correlated ($r = 0.46$). The lack of correlation could stem from different resolutions of each data source (land cover at 30 m, pesticide at county scale). Or, it could reflect the influence of crop choice on application rates. Pesticide application rates vary by crop with corn and soybean plantings having some of the largest rates of application compared to other grain crops such as wheat (Osteen and Fernandez-Cornejo 2013). The increased pesticide application for growing corn and soybeans is relevant as the top crop choice of Kansas producers has changed in the recent decade. Nationally, more corn is being planted with subsidies, and pressure for ethanol biofuel increased (Hill et al. 2006, Wright and Wimberly 2013). Furthermore, emphasis on corn production simplifies crop rotations and reduces soil health and nutrients (Van Eerd et al. 2014, Janovicek et al. 2021). The expansion of genetically modified corn has become available in areas of short growing seasons in the northern Great Plains and has been tied to ungulate mortalities (Butler et al. 2008).

Although my lek-based modeling approach suggested increased prairie-chicken habitat in recent years, these results are based on landscape-scale estimates of percent grassland, woody cover, and pesticide use, so many local features were not considered. Recent disturbances may

inflate lek attendance for a short time and become a misleading indicator of habitat quality (Ross et al. 2019). Though leks move frequently on the landscape between seasons, lek locations are appropriate for habitat selection and population index (Cannon and Knopf 1981) as greater prairie-chicken home ranges are highly associated with lek sites (Robel et al. 1970, Winder et al. 2015a). In addition, I explored lek persistence, as the nature of lek locations is delicate to surrounding land management. Most scales of land cover responded in the hypothesized direction, while pesticide compounds had inconsistent relationships most likely due to course resolution. Vertical structures also reported mixed effects. Transmission lines may have had a response at a smaller scale than evaluated, as suggested by other studies (Pruett et al. 2009). While oil wells and wind turbines reported high variation, averages trended in expected lek statuses. Lek extinction can indicate local population extirpation due to land cover and use.

Greater prairie-chicken leks can be negatively impacted by anthropogenic features including oil wells, transmission lines, cell phone towers, and wind turbines. Structure impacts on greater prairie-chicken space use have been described in multiple studies (Pruett et al. 2009, Winder et al. 2015b, Londe et al. 2019a). With the inclusion of anthropogenic features, low probabilities of habitat were predicted in western Kansas in 2013, despite western lek occurrences used to train the model (Hovick et al. 2015). Unfortunately, these structures could not be brought in at this project's extent due to temporal restrictions in available data. To better understand functionally available habitat within the predicted habitat area, I plotted vertical structures following the impact radii suggested by the sources. I calculated an additional 6% lost habitat for predicted habitat in 2020, including removal of unsuitable land cover. Anthropogenic structures and land use both influence habitat use for greater prairie-chickens at a landscape scale.

The Flint Hills ecoregion retains the largest amount of remaining tallgrass prairie, once considered a stronghold for greater prairie-chickens (Robbins et al. 2002, Robb and Schroeder 2005, With et al. 2008). Despite this, the Flint Hills ecoregion has suffered major losses in population and density of greater prairie-chickens in recent years (Nasman et al. 2021). My results suggest broadscale landscape variables may be at play, considering the predicted surface detailed 13% habitat loss across the ecoregion. Flint Hills pastures can be quickly dominated by eastern red cedar (*Juniperus virginiana*) in areas lacking regular fire, while fire-tolerant woody species may spread in areas of frequent, low intensity fires (Biggs et al. 2015, Li and McKenzie 2020).

Although the amount of grassland remained fairly stable in the Kansas Flint Hills between 1993 and 2020, the quality has been diminished due to severe and frequent disturbances built into the management practices. Historically, the Flint Hills had an average burn interval of ~3 years (Allen and Palmer 2011). Starting in the 1980s, the Flint Hills has been characterized by annual spring burning and extensive grazing, reducing grassland bird populations (including greater prairie-chickens) and overall biodiversity (Svedarsky et al. 2000, Robbins et al. 2002, With et al. 2008). Along with high frequency, the timing of burns centers around the same time of year (April), annually burning ~30% of the total ecoregion (Mohler and Goodin 2012). Although greater prairie-chickens may continue to lek in burned areas, females will not nest in areas following spring burns, and have higher nest success in unburned areas (Sanderson et al. 1973, Westemeier 1973, McNew et al. 2015). Annual burns are generally unsuitable for nesting greater prairie-chickens as resulting residual cover is insufficient (Svedarsky et al. 2000). Instead, breeding female space use peaks in areas spring burned >2 years prior (Gehrt et al. 2022).

North American grassland bird populations continue to decline despite literature calling for action and conservation (Samson and Knopf 1994, Vickery et al. 1999, Brennan and Kuvlesky 2005, Askins et al. 2007, Rosenberg et al. 2019). Grasslands require disturbance to remain grasslands. However, high fire frequency and overgrazing may promote woody vegetation in grassland systems by reducing fire fuel load or litter (Briggs et al. 2002, Jessup et al. 2003, Fuhlendorf et al. 2008, Knapp et al. 2008, Wilcox et al. 2018). Patch-burn grazing options with a three-year fire return interval may provide the optimal level of disturbance to promote nesting habitat for greater prairie-chickens (McNew et al. 2015, Winder et al. 2018).

Management Implications

Greater prairie-chickens conduct hierarchical decision-making for breeding habitat. Habitat selection is influenced by factors at multiple scales (Mayor et al. 2009). When establishing leks and home ranges, greater prairie-chickens require at least 60% grassland at an 8,100 ha (20,000-acre) scale. Various grassland disturbances, including high-intensity prescribed burns on three-year fire return intervals and mechanical removal, are required to support herbaceous-dominated ecosystems (Knopf 1994, Bielski et al. 2021). Fall burns provide attractive nesting cover for the following year's breeding season, although areas rested from burns should be accessible across the landscape (Sanderson et al. 1973, Westemeier 1973). Within grasslands, I encourage management choices for diverse herbaceous vegetation structure rather than species composition (Buhnerkempe et al. 1984, Anderson et al. 2015). Greater prairie-chickens are sensitive to woody vegetation. Active management should focus on keeping woody cover below 10% within 200 ha (500 acres). Pesticide use negatively affected the predicted use of greater prairie-chickens in Kansas, and should be further considered, especially for crop fields that border grassland.

Figures

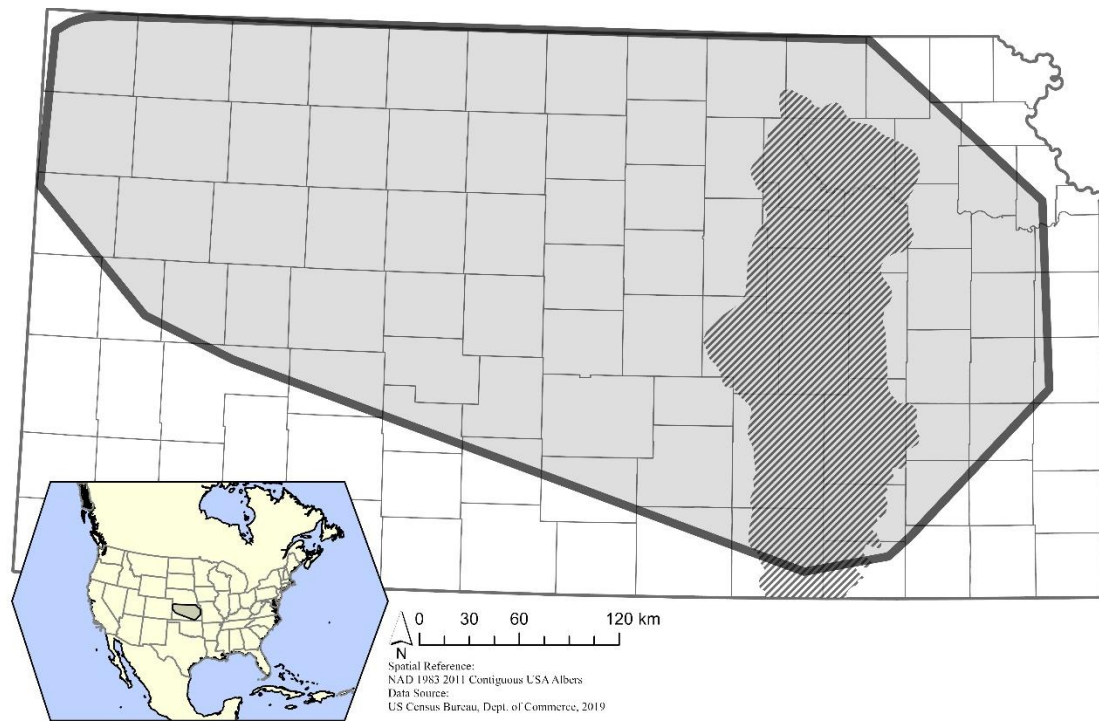


Figure 3.1. The study area within Kansas, USA, 1993-2020. The striped area delineates the Flint Hills ecoregion that extends further south of the area of interest. The study area boundary was defined by a minimum convex polygon of surveyed greater prairie-chicken leks between 1990-2020.

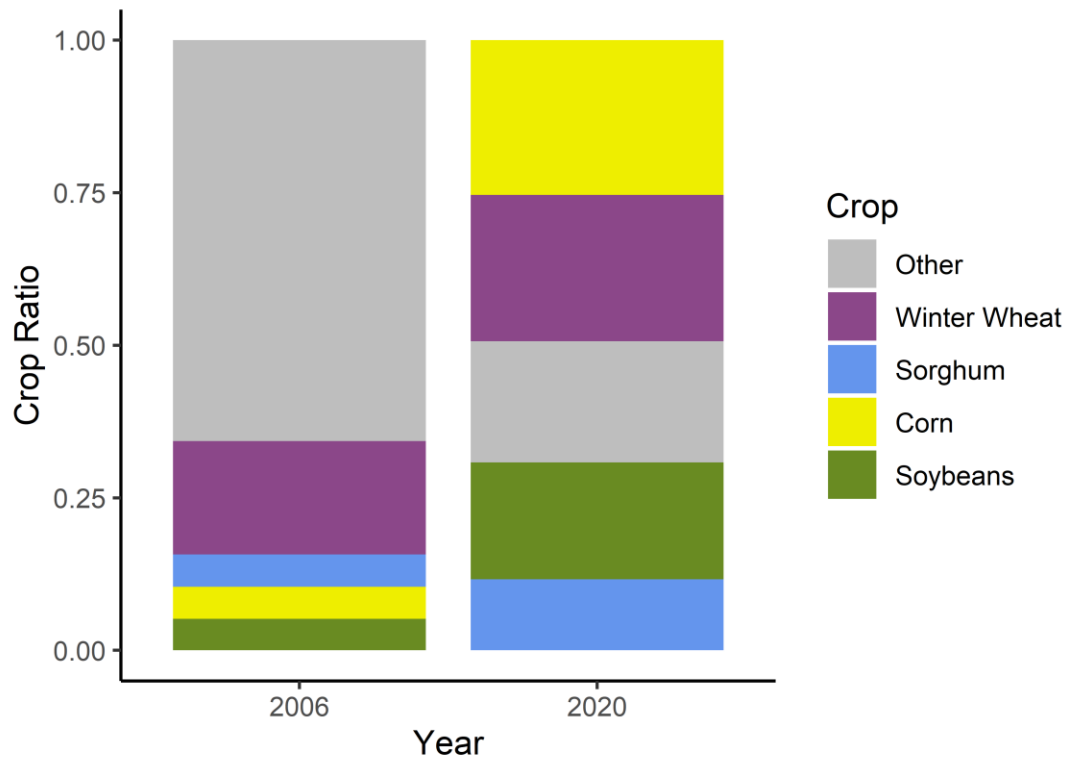


Figure 3.2. Stacked bar chart describing the change of crop selection and proportion of planted acreage of cropland within the study area of Kansas’s top four agricultural products. The order of stacked colors reflects the planted frequency of each crop for each year. In 2006, top products were finer grain crops: winter wheat (purple) and sorghum (blue). The ranking changes in 2020 with corn (yellow) becoming the top product in Kansas, and soybeans (green) outranking sorghum. Crop category ‘other’ includes any other distinguished cultivated crop, including unidentified or fallow fields.

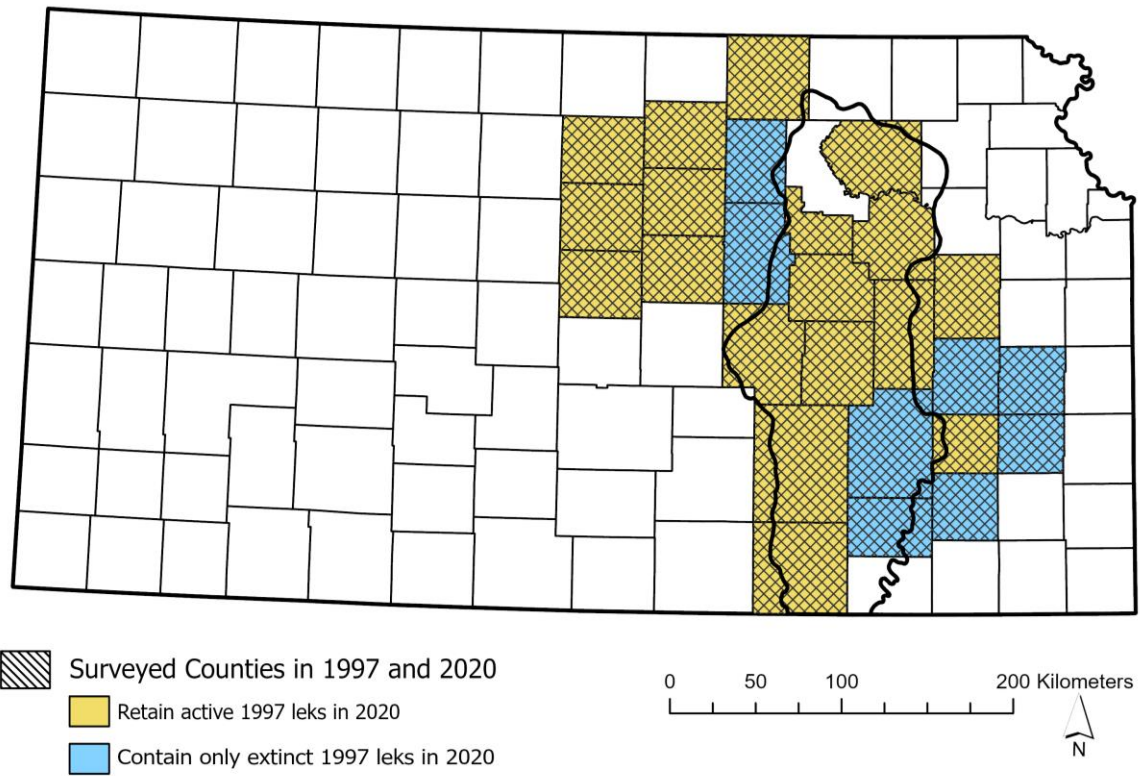


Figure 3.3. Locations of survey efforts to describe lek persistence. Crosshatched counties were surveyed in 1997 and 2020 and reported leks for both years. Counties are colored by retention of active 1997 leks. Lek status was determined active if within 1.6 km of 2020 lek locations. Orange counties retained active 1997 leks, while blue counties had no 1997 lek persistence in 2020.

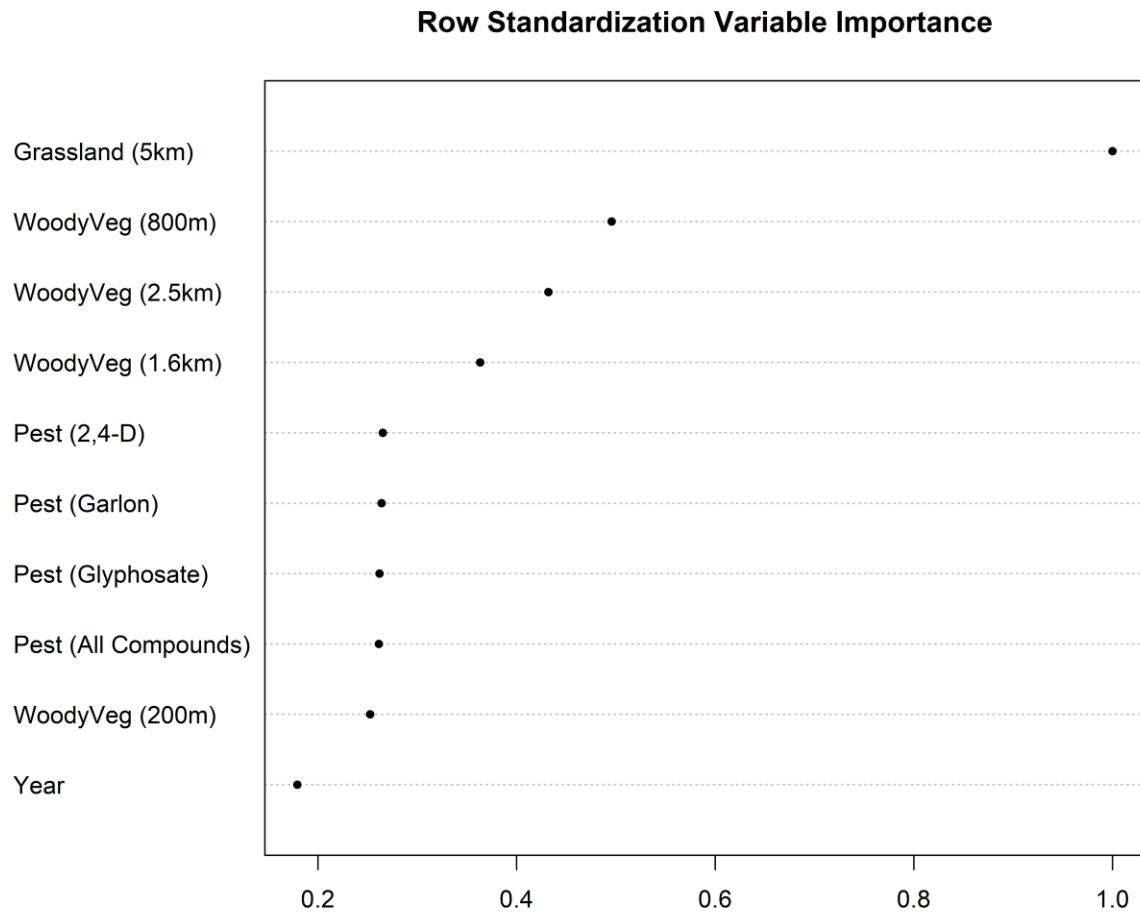


Figure 3.4. Random forest scaled variable importance plot using MIR as a feature selection approach to consider varying landscape scales and pesticide compounds. All included covariates were determined to be uncorrelated ($r < 0.4$).

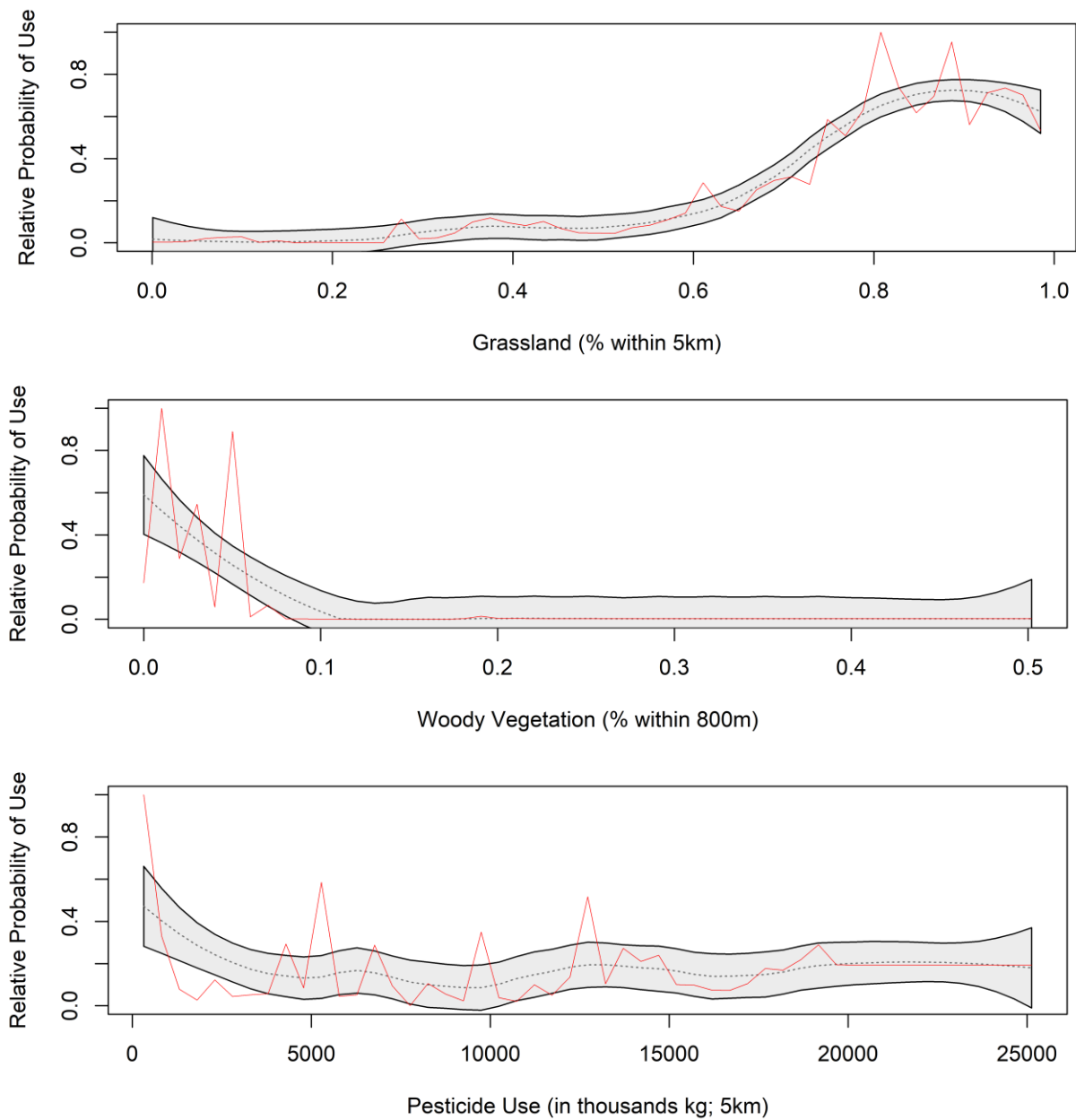


Figure 3.5. Partial dependent plots of the three factors present in the pruned random forest model. Each plot represents a relationship of relative probability of greater prairie-chicken lek use to a landscape variable: the amount of grassland within 5 km (top), percentage of woody vegetation within 800 m (middle), and estimated local pesticide use surrounding 5 km (bottom). Order of plots conveys variable importance in the model.

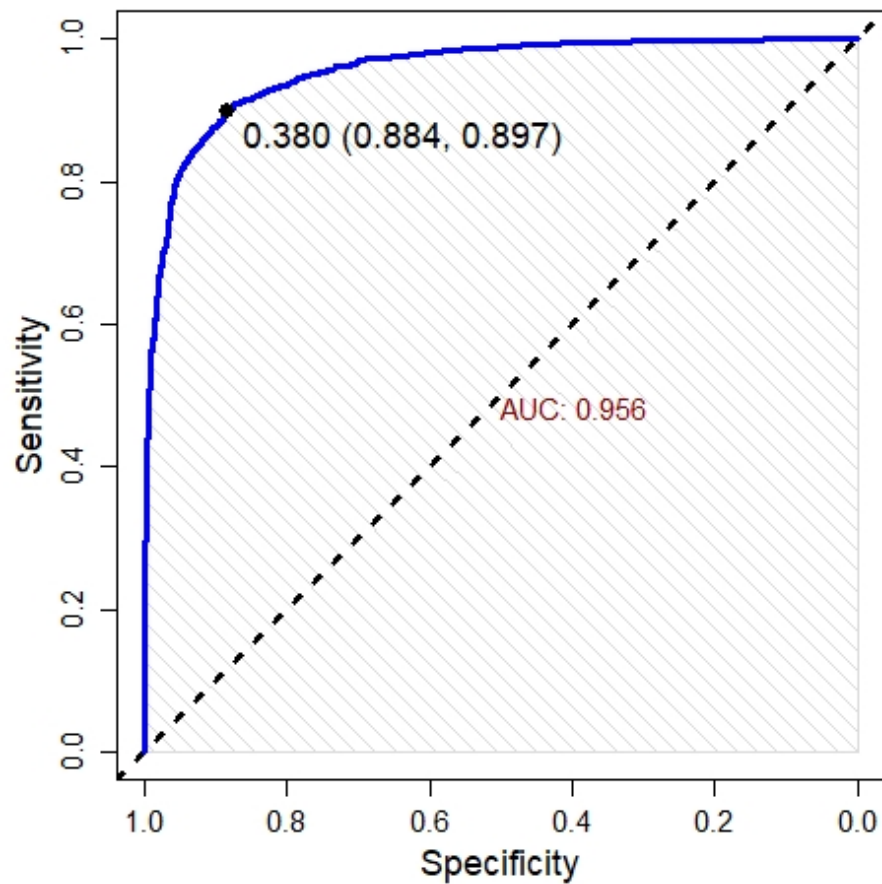


Figure 3.6. Receiver operator curve (ROC) predicted using holdout data (25% of each year) and the final, pruned model. The point plotted in the closest value to the top left corner was the determined threshold (0.380), followed by values for specificity and sensitivity in parentheses. The shaded zone is the area under the curve (AUC = 0.956).

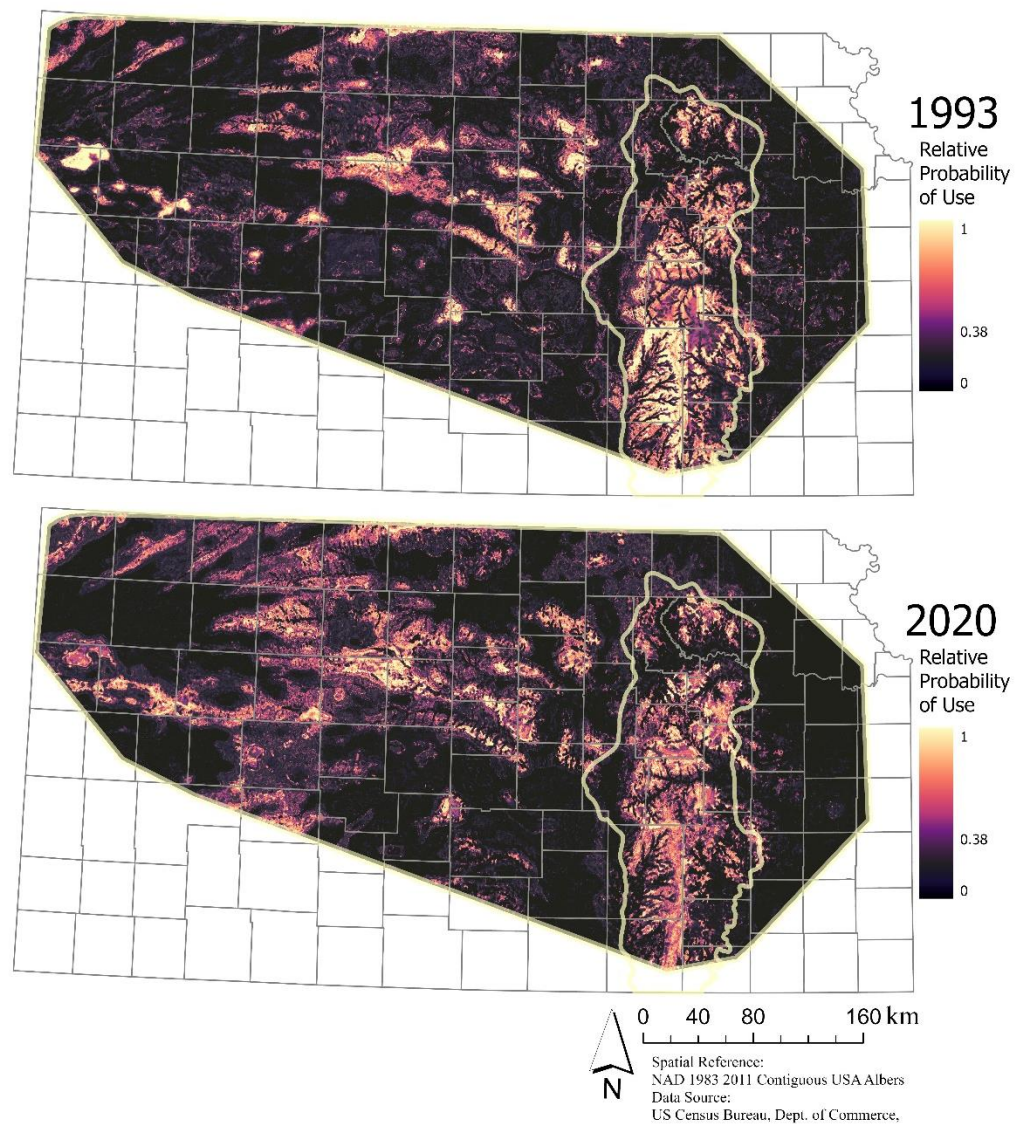


Figure 3.7. Maps describing the spectrum of relative probability of use by greater prairie-chickens in Kansas between 1993 and 2020 as predicted using lek location data and a random forest algorithm. Color ramps are the same between years, and highlight the threshold described by the ROC (0.380).

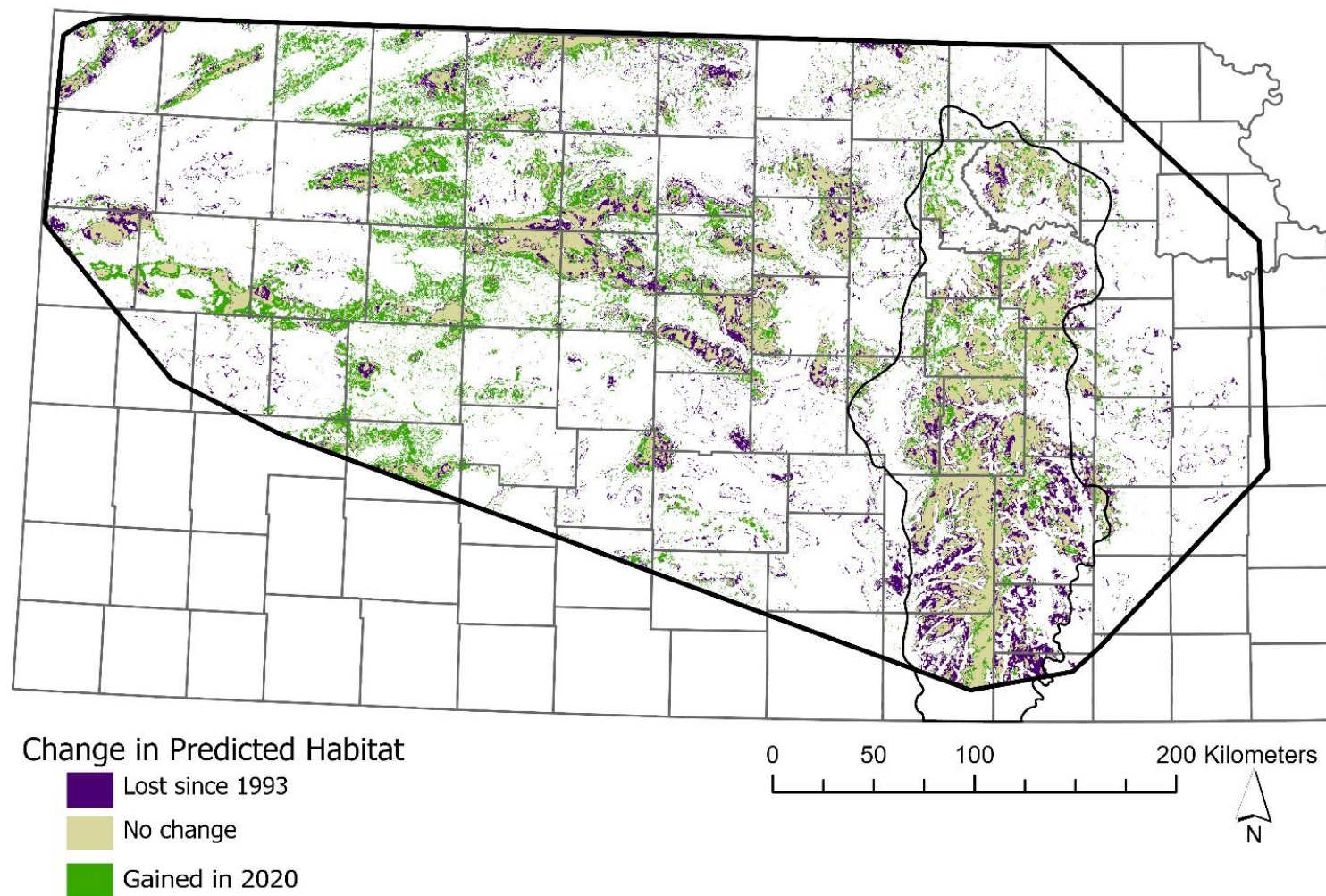


Figure 3.8. Differences in predicted greater prairie-chicken habitat between 1993 and 2020 based on lek use data. The threshold determined from ROC was used to decide along the probability spectrum a binary habitat layer for each year. Areas in green are new areas of potential habitat in 2020 while areas in purple were lost since 1993. The Flint Hills ecoregion is also highlighted.

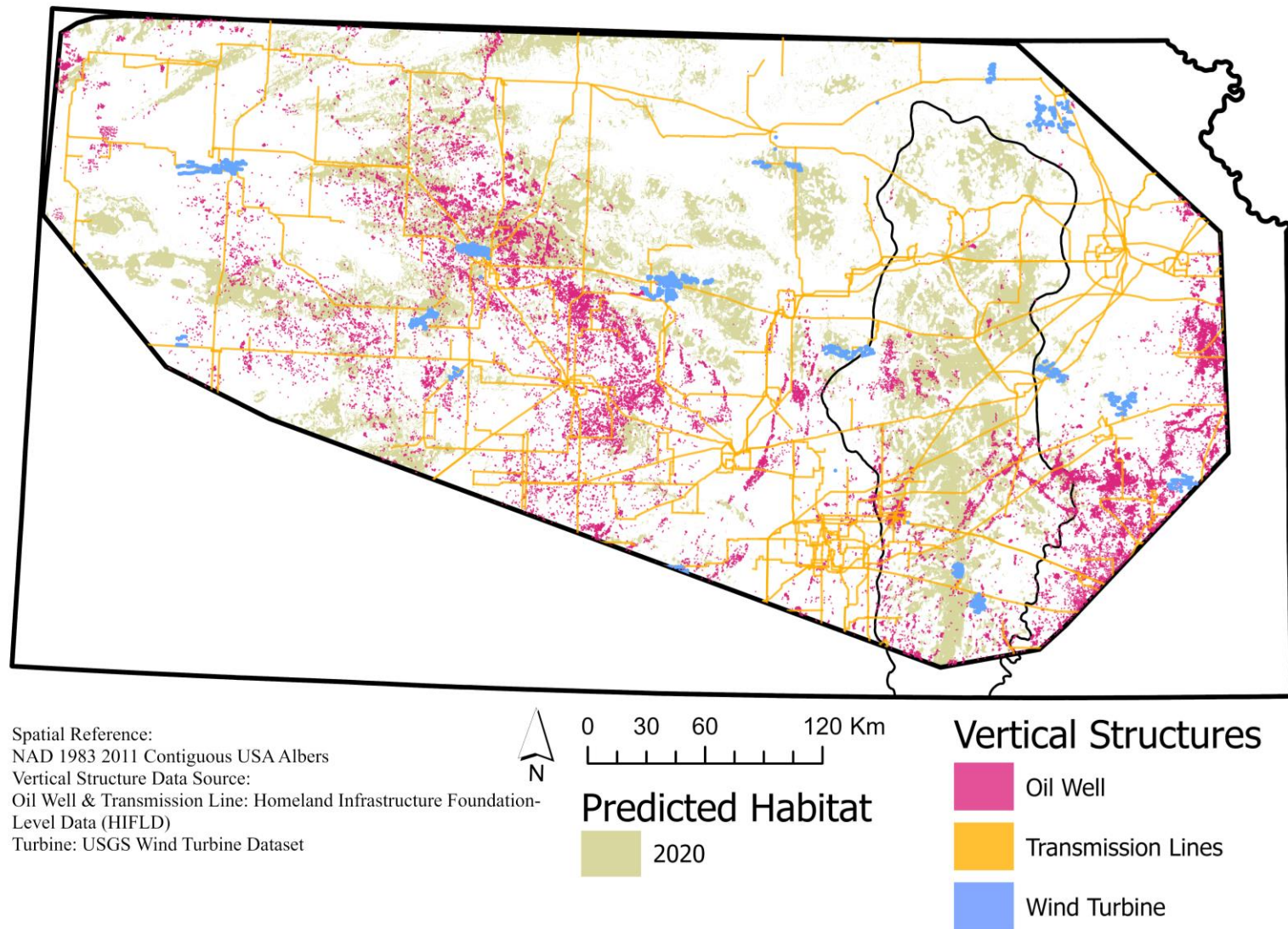


Figure 3.9. An evaluation of impacted predicted available habitat in 2020 based on lek use data considered with three types of vertical anthropogenic structures: oil wells, transmission lines, and wind turbines.

Tables

Table 3.1. Mean (SD) land cover and use variables between two statuses of 1997 greater prairie-chicken leks. Active status was determined if 1997 lek location was within 1.6 km of a 2020 recorded lek. Otherwise, the lek is determined to be ‘locally extinct.’ Values are estimated as average area (ha), application (kg), vegetation cover (%), or density across various scales. The bottom three rows (italicized) are the proportion of lek locations with structure occurrence.

	Active (n = 78)	Locally Extinct (n = 145)
Grassland (ha scale)		
Within 5 km	5,917.9 (1,340)	5,324.1 (1,907)
Within 5 km (%)	75.4 (17)	67.8 (24)
Woody Vegetation (ha scales)		
Within 200 m	0.261 (0.4)	0.18 (0.1)
Within 200 m (%)	1.89 (2.6)	1.29 (1.1)
Within 800 m	3.64 (2.4)	3.87 (3.0)
Within 800 m (%)	1.77 (1.2)	1.88 (1.5)
Within 1.6 km	19.78 (9.33)	22.64 (13.7)
Within 1.6 km (%)	2.40 (1.1)	2.75 (1.7)
Within 2.5 km	60.50 (27.9)	66.91 (33.6)
Within 2.5 km (%)	3.03 (1.4)	3.35 (1.7)
Applied Pesticide (kg within 5 km)		
Glyphosate	2,390.8 (1,153)	2,221.4 (1,450)
Dicamba	561.3 (379)	538.5 (372)
2, 4-D	428.2 (380)	374.9 (306)
Organophosphates	39.8 (33)	41.0 (29)
Garlon	3.1 (1)	4.8 (3)
All Pesticides	6,778.3 (2,770)	6,090.9 (3,790)
Vertical Structures		
Oil Well	0.9 (3.3)	4.1 (13.6)
Wind Turbine	0.1 (0.8)	1.1 (3.0)
Transmission Line Poles	5.1 (9.8)	3.6 (9.5)
<i>Oil Well</i>	<i>14%</i>	<i>28%</i>
<i>Wind Turbine</i>	<i>3%</i>	<i>12%</i>
<i>Transmission Line Poles</i>	<i>26%</i>	<i>17%</i>

Table 3.2. Greater prairie-chicken lek locations and “pseudoabsence” points were included in fitting the species distribution model with landcover and use values. Land cover was reported at multiple scales and reflective of the year each lek location was observed. Pesticide values are associated with the total application of the previous year.

<u>Variable</u>	Use (n = 5,570)		Pseudoabsence (n = 11,223)	
	<u>Mean (SD)</u>	<u>SE</u>	<u>Mean (SD)</u>	<u>SE</u>
Grassland (ha within 5 km)	5,671.7 (1,555)	20.86	3,300.4 (1,906)	17.99
Grassland (% within ~8,000 ha)	72.2 (20)	<0.01	42.0 (24)	<0.01
Woody Vegetation (ha scaled)				
Within 200 m	0.09 (0.2)	<0.01	0.37 (1.0)	0.01
Within 800 m	2.00 (1.9)	0.03	5.18 (9.3)	0.09
Within 1.6 km	13.35 (10.1)	0.14	21.05 (31.9)	0.30
Within 2.5 km	42.78 (29.8)	0.40	51.94 (67.9)	0.64
Applied Pesticide (kg within 5 km)				
Glyphosate	1,487.0 (1,256)	16.85	1,925.3 (1,515)	14.30
2, 4-D	453.5 (358)	4.80	576.3 (409)	3.86
Organophosphates	62.2 (62)	0.83	78.7 (72)	0.68
Garlon	35.2 (116)	1.55	18.7 (72)	0.68
All Compounds	4,153.2 (2,555)	34.28	5,289.8 (3,161)	29.84

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