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Quantitative Approaches to Upland Game Bird Conservation in Kansas

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QUANTITATIVE APPROACHES TO UPLAND GAME BIRD CONSERVATION IN KANSAS

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the Graduate School of
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In Partial Fulfillment
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Master of Science
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by
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Accepted by:
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ABSTRACT

The Great Plains region of North America is an important ecosystem supporting many plant and animal species, but one of the most endangered ecosystems in the world. Loss and degradation of grasslands in the Great Plains region has resulted in major declines in abundance of grassland bird species. To ensure future viability of grassland bird populations, it is crucial to evaluate specific effects of environmental factors across species to determine drivers of population decline and develop effective conservation strategies. In this study, I took a multi-species approach to understanding effects of environmental factors on four species of upland game birds in Kansas. I quantified effects of land cover and weather changes, as well as conservation practice implementation on lesser and greater prairie-chickens (*Tympanuchus pallidicinctus* and *T. cupido*, respectively), northern bobwhites (*Colinus virginianus*), and ring-necked pheasants (*Phasianus colchicus*). I found evidence of nonlinear, threshold effects of land cover change on abundance of all four focal species, though specific effects differed by species and spatial scale. I then focused on lesser prairie-chicken conservation and used a decision support tool to select candidate areas to prioritize conservation efforts and evaluate tradeoffs between conservation and economic objectives in different conservation scenarios. I found difference in conservation approach led to different optimal solutions, though some areas were important regardless of conservation approach. It is important to study drivers of species population decline, and apply ecological findings of such studies to improve conservation management. In this study, I both examined effects of land cover and weather on abundance of multiple species of upland game birds, and applied ecological knowledge to systematic conservation planning for an upland game bird species of conservation concern. My results provide valuable context to managers for optimizing conservation management for grassland birds in Kansas.

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CHAPTER ONE

A MULTI-SPECIES APPROACH TO MANAGING EFFECTS OF LAND COVER AND WEATHER ON UPLAND GAME BIRDS

INTRODUCTION

The Great Plains region of the United States is an important ecosystem supporting many plant and animal species. However, this region is one of the most endangered grassland systems in the world, and among the most endangered ecosystems in North America (Samson and Knopf 1994, Samson et al. 2004). Grasslands in this region, consisting of short, mixed, and tallgrass species along a west to east precipitation gradient, have been steadily declining in quantity and quality since large-scale European settlement of the area beginning after enactment of the first of the Homestead Acts in 1862. By the early 2000s, estimates of Great Plains grassland loss totaled around 70% of the bioregion (Samson et al. 2004). Such losses in habitat have caused dramatic declines in grassland bird populations, including losses of many endemic species (Knopf 1994, Coppedge et al. 2001). Grassland birds in the United States are declining faster than any other avian guild (Brennan and Kuvlesky 2005, BirdLife International 2018). Grassland bird habitat losses are in large part caused by anthropogenic effects. As agriculture became more prevalent and intensified in the Great Plains, vast areas of grasslands were converted to croplands. This conversion of land cover paired with practices such as pesticide use, intensive unmanaged grazing, invasive plants, declining nutritional quality, and improper burning tactics lead to changes in the landscape that negatively affect both habitat quantity and quality (Samson et al. 2004).

In addition to landscape characteristics, climatic drivers also affect avian populations in the Great Plains region. Climate change can directly affect avian populations through physiological limitations leading to changing survival and recruitment rates (Root 1988, Grisham et al. 2016), and indirectly affect avian populations through modifying potential habitat, leading to shifts in species' ranges, reductions in population abundance, and, eventually, local extinctions (Root et al. 2003, Thomas et al. 2006, Virkkala et al. 2008). However, conservation priorities often do not take projected climate change into consideration (Langham 2015). More research is needed to understand specific species' responses to differing weather conditions in order project species' responses to potential changes in climate. Climate change further affects species demographics through interactive effects with the landscape, yet these two effects are often studied independently (Selwood et al. 2015). Quantifying the interactive effects of landscape and climate change are important in the Great Plains region where there is evidence for both factors influencing wildlife populations (Samson et al. 2004). For example, there is evidence of near-surface temperature change in the region due to land cover change (Mahmood et al. 2006). These interactive effects can also alter avian population dynamics. For example, decreases in grassland cover caused significant decreases in resilience of lesser prairie-chickens (*Tympanuchus pallidicinctus*) to extreme drought events (Ross et al. 2016a). Incorporating both the effects of landscape and climate change into conservation and management is therefore important for ensuring the viability of future avian populations (Nichols et al. 2011).

Conservation management often must attempt to develop management strategies that benefit many species simultaneously, especially in areas such as the Great Plains region, which has numerous species of conservation concern. Therefore, to develop conservation plans, the concept often labeled as umbrella species, focal species, surrogate species, or indicator species is often utilized. These terms refer to species that have habitat requirements similar to those of many other species, but have more extensive spatial needs (Suter et al. 2002). In theory, developing management strategies to conserve habitat of one of these species would therefore indirectly benefit many other species as well. While this tool is useful in some regions under some specific conservation goals, habitat and resource needs of most species never perfectly overlap and this conservation strategy will therefore never provide an ideal solution for all species of interest (Crosby et al. 2015, Carlisle et al. 2018). Instead, an alternative solution may be to manage for an optimal landscape that may not be the optimal scenario for any one species but beneficial to most species. This approach may prove useful in the Great Plains region, where many species of conservation concern have different, and oftentimes conflicting, resource and habitat needs.

The lesser and greater prairie chicken (*Tympanuchus pallidicinctus* and *T. cupido*, respectively), northern bobwhite (*Colinus virginianus*), and ring-necked pheasant (*Phasianus colchicus*) are all important economically as upland (non-waterfowl) game birds in the state of Kansas. Kansas is generally in the top 3-4 pheasant hunting states, top 3 quail hunting states, and one of the top states for greater prairie-chicken hunting, as it has the greatest density of greater prairie-chickens in the United States (KDWPT 2017a,b,c). A hunting season was previously held for lesser prairie-chickens in Kansas, but harvest was discontinued when the species was federally listed as threatened in 2014 due to population declines. This listing rule was vacated by judicial decision in 2015, but lesser prairie-chicken harvest is still not allowed in the species range in the state (Haukos et al. 2016). Populations of all four of these species are currently in decline in the state (Hernández et al. 2013; Hefley et al. 2013; Hiller et al. 2015; Pabian et al. 2015; Ross et al. 2016a,b). There are a variety of factors of both anthropogenic and natural origin causing these population declines, but specific causes may not be similar among species. However, these individual mechanisms are all related to a combination of changes in habitat quantity, habitat quality, and weather variables (Brennan 1991, Lusk et al. 2001, Sauer et al. 2013, Grisham et al. 2016), the effects of which likely vary by spatial scale. For example, difference in spatial scale resulted in differing responses of lesser prairie-chicken populations to habitat loss and fragmentation (Fuhlendorf et al. 2002). However, no study has taken a multi-species approach to examining effects of both land cover and weather for upland game birds in this region across multiple spatial scales. In attempts to conserve habitat for these and other wildlife species, several Farm Bill practices, including the Environmental Quality Incentives Program (EQIP), have been put into practice in Kansas to incentivize farmers to conserve grasslands on their private property. While these conservation practices are expected to benefit wildlife species, more research is needed to assess the success of private-lands conservation on improving targeted populations.

I examined effects of land cover and weather on populations of upland game birds in Kansas. I quantified effects of grassland:cropland ratio, edge density of grassland patches, summer temperature and drought, and winter temperature and precipitation on abundance of lesser and greater prairie-chicken, northern bobwhite, and ring-necked pheasant populations using hierarchical models in a Bayesian framework across a gradient of fine to broad spatial scales. I hypothesized that abundances of all species have a threshold response to the landscape variables, though prairie-chickens would be more sensitive to increasing cropland and edge density than northern bobwhites and ring-necked pheasants. Northern bobwhites and ring-necked pheasants often select multiple types of adjoining habitat with distinct "edge" components and habitat in or adjacent to cropland, whereas prairie-chickens select mid to tall grasses at larger spatial scales, using shorter grasses only for lekking (NRCS 1999a,b,c; NRCS 2005). I hypothesized that abundances of all four species would be negatively affected by increasing summer temperature, drought, and winter precipitation, and decreasing winter temperatures, though the degree of response would vary by species. Prior studies have demonstrated prairie-chickens to be particularly susceptible to drought (Svedarsky et al. 2000; Pitman 2014; Hovick et al. 2015; Grisham et al. 2016; Ross et al. 2016a,b), northern bobwhites to drought, winter precipitation, and temperature in the summer (Guthery et al. 2000a,b; Lusk et al. 2001; Janke et al. 2015), and ring-necked pheasants to winter temperature and winter precipitation (Perkins et al. 1997, Homan et al. 2000, Prendergast 2018b).

I also examined effects of several EQIP practices on these four species of upland game birds. I quantified effects of brush management, prescribed burning, cover crop, prescribed grazing, and upland wildlife habitat management on abundances of lesser and greater prairie-chickens, northern bobwhites, and ring-necked pheasants. I hypothesized that all four species would be positively affected by increased area of each implemented practice, as all five of these practices involve restoring natural plant communities to the ecosystem, thus likely improving habitat (NRCS 2011, 2013, 2014, 2015, 2017). In combination, these analyses will provide valuable context to managers and aid in optimizing conservation and management efforts for multiple species.

METHODS

Study Area

Study sites were Kansas Department of Wildlife, Parks, and Tourism survey routes for lesser prairie-chickens, greater prairie-chickens, northern bobwhites, and ring-necked pheasants. These survey routes occurred across Kansas, representing the majority of land use and habitat types found in the state (Figure 1.1). Survey routes for ring-necked pheasants, greater prairie-chickens, and lesser prairie-chickens did not extend into the southeast portion of the state, which did not include ranges of these species. Habitat in Kansas primarily consisted of grassland and cropland land cover types. Grasslands included both native grasslands and croplands removed from production and converted back to grasslands under the U.S. Department of Agriculture Conservation Reserve Program (CRP).

The lesser prairie-chicken range in Kansas was comprised of three distinct habitat ecoregions (McDonald et al. 2014). The Short-Grass Prairie/CRP Mosaic Ecoregion was comprised of short-grasses including buffalograss (*Bouteloua dactyloides*) and blue grama (*B. gracilis*), and mid-grasses including sideoats grama (*B. curtipendula*), little bluestem (*Schizachyrium scoparium*), sand dropseed (*Sporobolus cryptandrus*), and western wheatgrass (*Pascopyrum smithii*). The Mixed-Grass Prairie Ecoregion consisted of mixed-grasses and agricultural lands, including grazing and cropland. The Sand Sagebrush Prairie Ecoregion primarily contained mid and tall-grasses, including sand bluestem (*Andropogon hallii*), little bluestem, switchgrass (*Panicum virgatum*), prairie sandreed (*Calamovilfa longifolia*), and sand dropseed (McDonald et al. 2014, Spencer et al. 2017). Much of the greater prairie-chicken range in Kansas occurred in the Flint Hills and Smoky Hills ecoregions (Pitman 2014). The dominant grass species in the Flint Hills Ecoregion included big bluestem (*Andropogon gerardii*), switchgrass, and Indian grass (*Sorghastrum nutans*; Küchler 1975). The Smoky Hills Ecoregion was composed of mixed-grass prairies, as short-grass species of the west transition to tall-grass prairies in the east, including a mix of buffalograss, blue grama, big bluestem, Indian grass, switchgrass, little bluestem, tall dropseed, side-oats grama, as well as woody species including hackberry (*Celtis occidentalis*), sand plum (*Prunus angustifolia*), and smooth sumac (*Rhus glabra*; Kansas Geological Survey 2018).

The ring-necked pheasant and northern bobwhite ranges in Kansas occurred in the High Plains, Smoky Hills, South Central Prairies, Flint Hills, Glaciated Plains, and Osage Cuestas ecoregions, although the ring-necked pheasant range did not extend far into the

Osage Cuestas Ecoregion in southeastern Kansas. The High Plains Ecoregion contained short-grass species such as buffalograss and blue grama, as well as some small cacti species and yucca (*Asparagaceae* spp.; Kansas Geological Survey 2018). The South Central Prairies also contained mixed-grass species similar to the Smoky Hills Ecoregion. The Glaciated Plains and Osage Cuestas ecoregions included dense stands of tall-grass prairie, including big bluestem, switchgrass, little bluestem, and Indian grass in the western portions, and forests with oak (*Quercus* spp.) and hickory (*Carya* spp.) species (Kansas Geological Survey 2018).

Count Surveys

The Kansas Department of Wildlife, Parks, and Tourism conducted count surveys for all four of the focal species each year during spring and summer using roadside surveys (Table 1.1). Biologists conducted surveys for lesser and greater prairie-chickens twice each year and surveys for northern bobwhites and ring-necked pheasants once a year. To conduct a roadside survey, observers drove a transect, stopping at 11 locations along the transect to conduct auditory surveys (Pitman 2014; Prendergast 2018a,b). In prairie-chicken surveys, observers conducted additional flush counts for identified leks. If a lek (defined as 3 or more chickens on a display site) was identified by calls of booming males at a stop, and determined to be within 1 mile of the route, the observer went back after the auditory counts, flushed the lek, and counted all individual prairie-chickens in that lek (Pitman 2014). Time of day, survey period, listening duration, and transect length varied between species (Table 1.1). Prairie-chicken surveys began in

1964, but I only used data beginning in 1978 due to limited number of routes in prior years. Northern bobwhite surveys began in 1996, but I used data beginning in 1998 due to limited number of routes in prior years. Ring-necked pheasant surveys began in 1997. To better assess effects of land cover and weather on avian populations on the appropriate scales, I summed count data of all 11 stops on each route for each visit, in the case of prairie-chicken surveys, and of all 11 stops on each route for each year, in the case of northern bobwhite and ring-necked pheasant surveys.

Environmental variables

To assess effects of land cover change on these four focal species, I acquired land cover data from several sources. For land cover in the lesser prairie-chicken range, I used LandSat imagery for 1978, 1985, 1988, 1994, 2003, and 2013. Land cover for these data was classified as grassland, cropland, urban, or water at a 30-m resolution (Spencer et al. 2017). For land cover in the ranges of the other three focal species, I obtained data for the entire state of Kansas from the National Land Cover Database for 2001, 2006, and 2011. I similarly classified land cover in this dataset as grassland, cropland, urban, or water at a 30-m resolution. Grassland classification in all data sets consisted of both native grassland and land enrolled in the CRP. Enrollment in CRP occurred in 1986 and 1987 with re-enrollment in 1996 and 1997 and again in 2006 or 2011, so these data provide information on land cover in years prior to and following these contracts.

I estimated effects of land cover by calculating the proportion of land covered by grassland to land covered by cropland as well as the edge density of grassland patches in varying buffer sizes around each survey route for each year of land cover data. I used buffer sizes of 3, 5, and 10 km around the survey routes to assess the effects of land cover on populations at a variety of spatial scales, ranging from fine to broad scale with respect to the home ranges of the focal species. I used Fragstats version 4 (McGarigal et al. 2012) to calculate the ratio of grassland:cropland by comparing the amounts of each land cover type in the buffered area around each route in each year. I also used Fragstats to calculate edge density (ED) of grassland patches by summing the lengths of all edge segments of grassland and dividing by the total grassland area in each buffered landscape for each year. Changes in land enrolled in CRP are the major drivers of land cover change in upland game bird habitat in Kansas (Spencer et al. 2017). I therefore assumed land cover (i.e., grassland:cropland ratio and edge density) in the buffered areas remained constant between CRP contract years to fill in gaps in land cover data.

To assess effects of weather on the four focal species, I obtained historical weather data from the National Climatic Data Center for each of the 9 climate regions in the state of Kansas. Weather data contained information on drought, severity of summer temperatures, and severity of winter temperatures and precipitation. I used the Palmer Drought Severity Index (PDSI) to quantify drought during the breeding seasons of each focal species. I created a drought covariate by averaging the PDSI values for June, July, and August each year, for each climate region, and implemented a 1-year lag effect (e.g., PDSI values in 2016 would influence population numbers in 2017). I created a covariate for summer temperature severity by selecting the highest monthly maximum temperature (TMAX) from values in June, July, and August each year, for each climate region, and applied a similar 1-year lag effect. I created a covariate for winter temperature severity by selecting the lowest monthly minimum temperature (TMIN) from values during December, January, and February preceding a breeding season (e.g., values in December of 2016 and January and February of 2017 would influence population numbers in 2017). I also used the Precipitation Index (PCP) to quantify severity of winter precipitation. I created a covariate for winter precipitation by averaging the PCP values for December, January, and February preceding each breeding season (e.g., values in December of 2016 and January and February of 2017 would influence population numbers in 2017) for each climate region.

Environmental Quality Incentives Program Practices

To assess effects of conservation practices on the four focal species, I obtained data on the implementation of five conservation practices in the EQIP. These conservation practices included brush management (code 314), prescribed burning (code 338), cover crop (code 340), prescribed grazing (code 528), and upland wildlife habitat management (code 645). Data for these practices included the estimated number of acres of each practice implemented within a 3, 5, and 10-km buffered area around surveyed leks, for each year between 2004 and 2016. To assess long-term effects of these five conservation practices on populations of the four focal species, I created a covariate for each practice for each buffer size by taking the cumulative sum of the number of hectares of that practice within that buffer size around each survey route implemented up until and including that year (e.g., the value for brush management at the 3-km spatial scale for a particular route in 2006 would be the sum of acres of the brush management practice implemented within a 3-km buffer around that route in 2004-2006).

Statistical Modeling

I implemented hierarchical models in a Bayesian framework (Royle 2004) to estimate and quantify effects of land cover, weather, and EQIP practice on lesser and greater prairie-chicken lek abundance across the respective ranges of these species in Kansas. Data in these models were specified as coming from a binomial distribution

$$y_{i,j,t} \sim \mathrm{Bin}(N_{i,t}, p_{i,j,t}) \tag{1}$$

where the $y_{i,j,t}$, consisting of count data from lek surveys at route i, visit j, and year t, are distributed binomially with parameters $N_{i,t}$, the estimated abundance on leks, and $p_{i,j,t}$, the probability of detection. Due to count survey methodology, I was not able to separate detection probability associated with identifying leks from detection probability associated with detecting individuals on a lek. The probability of detection in all prairie-chicken models therefore referred to this combined detection.

I constructed process models to describe change in male prairie-chicken abundance on leks, which I assumed to be representative of changes in prairie-chicken population as a whole. I modeled the abundance of leks as coming from a negative binomial distribution to account for overdispersion in the count data

$$N_{i,t} \sim \mathrm{NegBin}(p.nb_{i,t}, r) \tag{2}$$

where r is the overdispersion parameter, $\mu_{i,t}$ is the mean parameter, and $p.nb = r/(r + \mu_{i,t})$. I implemented four models: two piecewise linear models (Qian and Cuffney 2012, Qian

2014) that allowed for thresholds or change points along the gradient of 1) grassland:cropland and 2) edge density, respectively, with linear effects for all weather covariates, 3) a linear model with interactions between land cover and weather covariates, and 4) a model with linear effects for all EQIP practice covariates. The piecewise linear models, or “threshold models,” were defined as

$$z_{i,t} = \log(\mu_{i,t}) = \beta_0 + \beta_1 x_{\mathrm{PDSI}} + \beta_2 x_{\mathrm{PCP}} + \beta_3 x_{\mathrm{TMAX}} + \beta_4 x_{\mathrm{TMIN}} + (\beta_5 + \delta I(x_{\mathrm{LAND}} - \phi))(x_{\mathrm{LAND}} - \phi) + \varepsilon_{i,t} \tag{3}$$

and

$$z_{i,t} = \log(\mu_{i,t}) = \beta_0 + \beta_1 x_{\mathrm{PDSI}} + \beta_2 x_{\mathrm{PCP}} + \beta_3 x_{\mathrm{TMAX}} + \beta_4 x_{\mathrm{TMIN}} + (\beta_5 + \delta I(x_{\mathrm{ED}} - \phi))(x_{\mathrm{ED}} - \phi) + \varepsilon_{i,t} \tag{4}$$

the linear model with interaction terms, or “interaction model,” was defined as

$$z_{i,t} = \log(\mu_{i,t}) = \beta_0 + \beta_1 x_{\mathrm{PDSI}} + \beta_2 x_{\mathrm{PCP}} + \beta_3 x_{\mathrm{TMAX}} + \beta_4 x_{\mathrm{TMIN}} + \beta_5 x_{\mathrm{LAND}} + \beta_6 x_{\mathrm{ED}} + \beta_7 x_{\mathrm{PDSI*LAND}} + \beta_8 x_{\mathrm{LAND*PCP}} + \varepsilon_{i,t} \tag{5}$$

and the linear model for EQIP practice covariates, or “EQIP model,” was defined as

$$z_{i,t} = \log(\mu_{i,t}) = \beta_0 + \beta_1 x_{314} + \beta_2 x_{338} + \beta_3 x_{340} + \beta_4 x_{528} + \beta_5 x_{645} + \varepsilon_{i,t} \tag{6}$$

where in the piecewise linear models β_0 was an intercept, β_1 the coefficient for PDSI, β_2 the coefficient for PCP, β_3 the coefficient for TMAX, β_4 the coefficient for TMIN, and β_5 the coefficient for grassland:cropland (designated as LAND) or the coefficient for ED. In the linear model with interaction terms, β_0, β_1, β_2, β_3, and β_4 were again coefficients for the intercept, PDSI, PCP, TMAX, and TMIN, respectively, β_5 was the coefficient for grassland:cropland, β_6 the coefficient for ED, β_7 the coefficient for the interaction between PDSI and grassland:cropland, and β_8 the coefficient for the interaction between

PCP and grassland:cropland. In the EQIP model, β_0 was an intercept, β_1 the coefficient for brush management, β_2 the coefficient for prescribed burning, β_3 the coefficient for cover crop, β_4 the coefficient for prescribed grazing, and β_5 the coefficient for upland wildlife habitat management (each practice designated by practice codes). I then modified these coefficients for grassland:cropland and ED, β_5, in the threshold models by adding or subtracting (depending on a positive or negative estimated effect) the intensification coefficients, δ, after the change points or thresholds, ϕ. The intensification coefficient describes the change in β_5 after the threshold value is reached. The indicator function, $I(a) = 0$ when $a < 0$ ($x - \phi < 0$, i.e., before reaching the threshold value) and $I(a) = 1$ when $a \geq 0$ ($x - \phi \geq 0$, i.e., after reaching the threshold value). Random effects, $\varepsilon_{i,t}$, were specified as coming from a normal distribution

$$\varepsilon_{i,t} \sim \mathrm{N}(0, \sigma) \tag{7}$$

and were random effects for site and time. I assessed the fit of each model by comparing the residuals and predicted values on a 1-to-1 line. For the linear models with interaction terms and linear EQIP models, I used stochastic search variable selection to evaluate importance of each environmental variable within the model. I based inference on variables with a probability of inclusion of 0.5 or greater (Walli 2010, Malsiner-Walli and Wagner 2011).

The northern bobwhite and ring-necked pheasant surveys did not have repeated counts, so I could not estimate detection probability in a hierarchical model. Instead, in both these models, data were defined as

$$y_{i,t} \sim \mathrm{NegBin}(p.nb_{i,t}, r) \tag{8}$$

where the $y_{i,t}$, consisting of count data from auditory surveys at route i and year t, are distributed with a negative binomial distribution with r as the overdispersion parameter, $\mu_{i,t}$ as the mean estimated abundance, and $p.nb = r/(r + \mu_{i,t})$. I similarly implemented piecewise linear models as in equations 3 and 4, a linear model with an interaction between land cover and weather as in equation 5, and a linear EQIP model as in equation 6.

I used Markov Chain Monte Carlo and a Gibbs sampler in JAGS 4.3.0 (Plummer 2017) with the package runjags (Denwood 2016) in program R version 3.4.3 (R Core Development Team 2017) to obtain posterior distributions for all model parameters. I discarded at minimum the first 200,000 samples as burn-in, used a thinning rate of 5, and saved 10,000 samples from 3 chains for all models. I evaluated convergence of chains with a Gelman-Rubin statistic (R <1.05). I specified prior distributions as $\beta_0, \beta_1, \beta_2, \beta_3, \beta_4, \beta_5, \beta_6, \beta_7, \beta_8 \sim N(0,10)$, $\delta \sim N(0,10)$, $\varepsilon_{i,t} \sim N(0, 15)$, $r \sim \text{Gamma}(1, 1)$, and $\phi \sim U(l, u)$ where l and u are the lower and upper values of the standardized grassland:cropland ratio or edge density. I repeated all modeling for all four species using land cover data from the 3, 5, and 10 km buffer sizes.

RESULTS

Population Trends

From 1978 to 2014, 25,877 lesser prairie-chickens were observed on 17 routes. Estimated abundance indicated lesser prairie-chickens on observed leks decreased from 1,665 males in 1978 (95% CI = 946; 2,536) to 845 males in 2014 (95% CI = 403; 1,364), a decline of 49%. Estimated lesser prairie-chicken abundance on observed leks was greatest in 1979 with 2,040 males (95% CI = 1,164; 3,167), and lowest in 2013 with 334 males (95% CI = 248; 483; Figure 1.2).

From 1996 to 2014, 32,507 greater prairie-chickens were observed on 33 routes. Estimated abundance indicated greater prairie-chickens on observed leks decreased from 2,045 males in 1996 (95% CI = 1,440; 2,800) to 1,425 males in 2014 (95% CI = 1,003; 1,897), a decline of 30%. Estimated greater prairie-chicken abundance on observed leks was greatest in 1998 with 2,685 males (95% CI = 2,207; 3,263), and lowest in 2013 with 1,198 males (95% CI = 793; 1,682; Figure 1.2).

From 1998 to 2015, 24,069 northern bobwhites were observed on 74 routes. Estimated abundance from my models indicated abundance of calling northern bobwhite males on surveyed routes decreased from 2,367 males in 1998 (95% CI = 2,101; 2,650) to 1,648 males in 2015 (95% CI = 1,500; 1,807), a decline of 30%, although there was considerable inter-annual variation in this trend. Abundance of calling northern bobwhite males on surveyed routes was greatest in 1998 with 2,367 males (95% CI = 2,101; 2,650), and lowest in 2001, with 895 males (95% CI = 771; 1,016; Figure 1.2).

From 1997 to 2015, 144,507 ring-necked pheasant crowing calls were recorded on 66 routes. Estimated abundance from my models indicated the number of ring-necked crowing calls on surveyed routes decreased from 11,319 in 1997 (95% CI = 9,950; 12,712) to 8,736 in 2015 (7,954; 9,599), a decline of 23%, although there was considerable inter-annual variation in this trend. The number of ring-necked pheasant crowing calls on surveyed routes was greatest in 2011 with 13,611 calls (95% CI =

11,937; 15,373), and lowest in 2012 with 6,072 calls (95% CI = 5,377; 6,780; Figure 1.2).

Interaction Models

All interaction models for each species fit these data with Bayesian p-values between 0.13 and 0.61. Inclusion probabilities in the lesser prairie-chicken interaction models indicated grassland:cropland ratio as the only important variable for inference; only at the 5 and 10 km scales (Figure 1.3). Grassland:cropland ratio had a significant, negative effect on abundance of lesser prairie-chickens at observed leks, meaning as amount of grassland increased with respect to cropland, lesser prairie-chicken abundance at observed leks decreased at both spatial scales (Table 1.2). All weather variables, edge density, and interactions between landscape and weather variables had low probabilities of inclusion.

Inclusion probabilities in the greater prairie-chicken interaction models indicated grassland:cropland ratio as an important variable for inference at the 3 and 10 km spatial scales and edge density as important for inference at the 5 and 10 km spatial scales (Figure 1.3). Grassland:cropland ratio had a significant, negative effect on abundance of greater prairie-chickens at observed leks at both spatial scales (Table 1.2). Edge density also had a significant, negative effect on abundance of greater prairie-chickens at observed leks at both spatial scales, meaning as amount of edge per unit of area of grassland patches increased, greater prairie-chicken abundance at observed leks decreased (Table 1.2). All weather variables and interactions between landscape and weather variables had low probabilities of inclusion.

Inclusion probabilities in the ring-necked pheasant interaction models indicated both grassland:cropland ratio and edge density as important variables for inference at all three spatial scales (Figure 1.3). Both grassland:cropland ratio and edge density had significant, negative effects on ring-necked pheasant abundance on surveyed routes (Table 1.2). All weather variables and interactions between landscape and weather variables had low probabilities of inclusion.

Inclusion probabilities in the northern bobwhite interaction models indicated none of the environmental variables were important for inference. All landscape, weather, and interactions between landscape and weather variables at all spatial scales had low probabilities of inclusion (Figure 1.3).

Threshold Models

All interaction models for each species fit these data with Bayesian p-values between 0.07 and 0.61. No piecewise-linear model indicated any significant effects of any weather variable for any species-buffer combination. All four species exhibited a significant threshold response to both grassland:cropland ratio and edge density, though the specific effect varied by species and buffer size.

Lesser prairie-chickens had a significant threshold response to grassland:cropland ratio at the 3 and 5 km buffer sizes, greater prairie-chickens at the 5 and 10 km buffer sizes, northern bobwhites at the 3 km buffer size, and ring-necked pheasants at the 3, 5, and 10 km buffer sizes. At the 3 and 5 km spatial scales for lesser prairie-chickens, the 10 km scale for greater prairie-chickens, and 3 km scale for northern bobwhites, grassland:cropland ratio initially had a significant, positive effect on abundance until a threshold point, and then a significant, negative effect after the threshold point (Table 1.3, Figure 1.4). Thus, in these cases, abundance initially increased with increasing grassland with respect to cropland until the threshold points, after which abundance decreased with increasing grassland.

At the 5 km spatial scale, grassland:cropland ratio initially had a significant, negative effect on greater prairie-chicken abundance until a threshold point, and then a significant, positive effect after the threshold point (Table 1.3, Figure 1.4). At this scale, greater prairie-chicken abundance at observed leks therefore initially decreased with increasing grassland with respect to cropland until a threshold point, after which abundance at observed leks increased with increasing grassland.

Ring-necked pheasants initially had a significant, negative response to grassland:cropland ratio until a threshold point, and then had a less severe, but still negative response to grassland:cropland ratio after the threshold point at all three spatial scales (Table 1.3, Figure 1.4). Thus, ring-necked pheasant abundance initially decreased with increasing grassland with respect to cropland until a threshold point, after which ring-necked pheasant abundance more slowly declined with increasing grassland.

Lesser prairie-chickens had a significant threshold response to edge density of grassland patches at the 10 km buffer size, greater prairie-chickens at the 10 km buffer size, northern bobwhites at the 3, 5, and 10 km buffer sizes, and ring-necked pheasants at the 3, 5, and 10 km buffer sizes. Abundances of all species at all significant buffer sizes initially had significant positive responses to increasing edge density of grassland patches until specific threshold points, and then had significant negative responses after the threshold points, though these threshold points varied by species and buffer size (Table 1.4, Figure 1.5).

Environmental Quality Incentives Program Models

All EQIP models for each species fit these data with Bayesian p-values between 0.09 and 0.65. Inclusion probabilities in the lesser prairie-chicken EQIP models indicated cover crop as the only important variable for inference, and only at the 3 and 5 km spatial scales (Figure 1.6). Cover crop had a significant, negative effect on abundance of lesser prairie-chickens at observed leks at both scales, meaning as the cumulative amount of cover crop implemented over time increased, lesser prairie-chicken abundance at observed leks decreased (Table 1.5). The 95% credible intervals of cover crop at the 5 km scale did overlap 0, but there was a 94% probability of a negative effect ($P(\beta_3 < 0) = 0.94$).

Inclusion probabilities in the greater prairie-chicken EQIP models indicated brush management as an important variable for inference at the 10 km spatial scale, prescribed burning as important at the 3 and 10 km scales, prescribed grazing as important at the 3, 5, and 10 km scales, and upland habitat management as important at the 3 and 10 km scales (Figure 1.6). Brush management and upland wildlife habitat management had a significant, positive effect on greater prairie-chicken abundance at observed leks, meaning as the cumulative amount of these two practices implemented over time increased, greater prairie-chicken abundance at observed leks increased at all relevant spatial scales (Table 1.6). Prescribed burning and prescribed grazing had a significant, negative effect on greater prairie-chicken abundance at observed leks, meaning as the cumulative amount of these two practices implemented over time increased, greater prairie-chicken abundance at observed leks decreased at all relevant spatial scales (Table 1.6). The 95% credible intervals of prescribed burning at the 3 km spatial scale and prescribed grazing at the 5 km spatial scale did overlap zero, but there was a 95% and 97% probability of a negative effect, respectively ($P(\beta_2 < 0) = 0.95$, $P(\beta_4 < 0) = 0.97$).

Inclusion probabilities in the ring-necked pheasant EQIP models indicated brush management as the only important variable for inference, at the 3, 5, and 10 km spatial scales (Figure 1.6). Brush management had a significant, negative effect on ring-necked pheasant abundance at all three spatial scales (Table 1.7). All EQIP practices for northern bobwhite EQIP models at all spatial scales had low probabilities of inclusion (Figure 1.6).

DISCUSSION

I quantified effects of land cover, weather, and EQIP practice implementation on abundances of four species of upland game bids in Kansas. Specifically, I found 1) abundance declined for lesser and greater prairie-chickens but remained stable for ring-necked pheasants and northern bobwhites across the survey periods, 2) more support for grassland:cropland ratio and edge density affecting abundance of the four focal species than weather covariates, 3) both grassland:cropland ratio and edge density had threshold effects on abundance that varied across species and spatial scale, and 4) EQIP practices had effects on abundance of lesser and greater prairie-chickens and ring-necked pheasants that varied across species and spatial scale.

Abundance Estimates

I found evidence of an overall decline in abundance of lesser and greater prairie-chickens between the 1978-2014 and 1996-2014 survey periods, respectively, although there was inter-annual variation in population trends over this period. This result supports previous assessments of prairie-chicken declines in the state (Jensen et al. 2000, Pitman 2014, Nasman 2018). I did not find evidence of an overall decline in abundance of northern bobwhite or ring-necked pheasant between the 1997-2015 and 1998-2015 survey periods, respectively, though there was large interannual variability. While both northern bobwhites and ring-necked pheasants have experienced declines historically in Kansas and throughout much of the species' respective ranges (Hernández et al. 2013, Sauer et al. 2013,), my results support the characterization of these species' populations in Kansas as being relatively stable during my survey years (1997-2017 for ring-necked pheasants and 1998-2017 for northern bobwhites), and not experiencing similar declines as other regions during this time period (Hernández et al. 2013; Sauer et al. 2013; Prendergast 2018a,b).

Land Cover and Weather Effects on Abundance

In general, land cover affected abundance of the focal species more than weather. Prior studies have demonstrated effects of extreme summer and winter weather on abundance of these four species (Perkins et al. 1997; Guthery et al. 2000a,b; Homan et al. 2000; Svedarsky et al. 2000; Lusk et al. 2001; Hovick et al. 2015; Grisham et al. 2016; Ross et al. 2016a,b). However, few studies have simultaneously compared relative effects of weather and land cover on abundance. In my models, I measured the linear effects of weather covariates on these populations, which may not be linear. For example, PDSI may only affect abundance in years of extreme drought (i.e., PDSI greater than a certain threshold; Guthery et al. 2000a,b; Lusk et al. 2001; Ross et al. 2016a,b) and have no effect on populations in years with average PDSI values. It is also possible that these extreme weather events occurred over shorter time periods than I measured, causing these effects to be missed in my models.

Threshold Effects of Land Cover on Abundance

I found evidence of threshold effects of grassland:cropland ratio and edge density of grassland patches in all four species, although specific effects varied by species and spatial scale. Abundance of lesser prairie-chickens initially increased until a threshold of increasing grassland with respect to cropland before decreasing in both fine and intermediate spatial scales. Abundance of greater prairie-chickens and northern bobwhites had similar threshold responses at the broad and fine spatial scales, respectively. Ring-necked pheasant abundance decreased with increasing grassland with respect to cropland until a threshold, at which point increasing grassland had a weaker, but still negative effect on abundance.

Lesser prairie-chickens are generally characterized as selecting medium grasses (NRCS 1999a, Haukos and Zavaleta 2016) and conversion of grassland to cropland is often attributed to the decline in abundance of the species (NRCS 1999a, Hagen et al. 2004). However, results of this and other studies suggest that lesser prairie-chickens may utilize croplands to some limited extent, and the presence of some cropland at a finer spatial scale can benefit abundance of this species (Ross et al. 2016b). Greater prairie-chickens are characterized as selecting medium to tall grasses (Jones 1963, NRCS 2005), but similarities in life history and habitat selection to lesser prairie-chickens may result in greater prairie-chickens receiving similar benefits by access to limited amounts of cropland. Northern bobwhites and ring-necked pheasants do often select habitat in or near agricultural land (NRCS 1999b,c). The relationships I found between abundance and grassland:cropland ratio further highlight the importance of cropland as a habitat source to these species.

I expected the threshold point to be at a higher proportion of grassland in the prairie-chicken models than the northern bobwhite model, as northern bobwhites often select habitat in close proximity to cropland (NRCS 1999b, Janke et al. 2015), whereas prairie-chickens are characterized as selecting habitat primarily containing mid to tall grasses (NRCS 1999a, Haukos and Zavaleta 2016). Differences in response to grassland:cropland ratio may be due to differences in spatial scale. Both home ranges and dispersal capabilities of lesser and greater prairie-chickens are much higher than that of northern bobwhites (Robel et al. 1970, Terhue et al. 2006, Janke and Gates 2013, Patten et al. 2016). As these species select habitat at different spatial scales, they likely also respond to changes in grassland:cropland ratio at different scales. Differences in response to land cover changes between species make it challenging to manage the landscape for multiple species. In this case, managing for an optimal landscape for multiple species may be an effective tool. Achieving such an optimal landscape may involve maintaining different amounts of grassland and cropland at different spatial scales. For example, effective management of a landscape for both northern bobwhites and prairie-chickens might involve considering grassland:cropland ratio at the 3-km spatial scale for northern bobwhites and the 5-km or 10-km spatial scale for prairie-chickens. As the habitat needs of these species do not perfectly overlap, this may result in a landscape with a proportion of grassland:cropland that does not necessarily create optimal conditions for any one species (i.e., the threshold point of response to grassland:cropland ratio for a particular species), but optimizes conservation potential for all species (i.e., achieves the maximum combined abundance of all managed species).

Abundance of all four species initially increased with increasing edge density of grassland patches, and then decreased with increasing edge past a threshold point. Northern bobwhite and ring-necked pheasant abundance demonstrated this effect at all three spatial scales, while lesser and greater prairie-chickens only demonstrated this effect at broad spatial scales. The threshold effects of grassland:cropland ratio showed the potential benefit of some amount of cropland with grassland in the landscape for these four species. It is therefore not surprising that these species also select some amount of grassland edge, as this allows for interactions of the species with cropland. Grassland edge is also related to patch size. For example, the presence of many small patches will increase the amount of edge in a landscape compared to the presence of fewer, larger patches of the same total area. Northern bobwhites and ring-necked pheasants are generally categorized as more edge-dwelling than prairie-chickens, often selecting habitat in close association with cropland (NRCS 1999a,b,c; Smith et al. 1999; NRCS 2005; Janke et al. 2015). This characterization is supported in my models, where northern bobwhite and ring-necked pheasant had positive increases with increasing edge density of grassland patches to a larger threshold point than both prairie-chicken species. While all four species did respond similarly to increasing edge density of grassland patches, differing optimal amounts of grassland edge among species makes it challenging to manage a landscape for all four species. Managing for an optimal landscape may again be an effective tool. Achieving an optimal landscape may involve a configuration of grassland patches with both edge and patch interior components that do not necessarily create optimal conditions for any one species (i.e., the threshold point of response to increasing edge for a particular species), but optimizes conservation potential for all species (i.e., achieves the maximum combined abundance of all managed species).

Effects of Environmental Quality Incentives Program Practices on Abundance

Cover crop had a negative effect on lesser prairie-chicken abundance at multiple spatial scales. The cover crop practice involves planting crops including grasses, legumes, and forbs for seasonal cover in an agricultural system (NRCS 2017). This negative relationship between lesser prairie-chicken abundance and cover crop implementation suggests that the planting of cover crops may not restore the vegetative structure to agricultural systems at a large enough scale to be relevant for lesser prairie-chicken populations. Brush management had a positive effect on greater prairie-chicken abundance at the broad spatial scale, but a negative effect on ring-necked pheasant abundance at all scales. Woody plant encroachment on grasslands decreases habitat quality for prairie-chickens (Svendarsky et al. 2000, Hagen et al. 2004, Lautenbach et al. 2017), and brush management likely helps decrease this encroachment, leading to higher quality grassland habitat and increasing abundance. However, brush removal may also result in the removal of important vegetative cover for ring-necked pheasants. Prescribed burning and prescribed grazing had negative effects on greater prairie-chicken abundance at several spatial scales. The intent of these two practices is to maintain desired plant communities, with any potential benefits to wildlife occurring indirectly as a consequence of restoring the natural plant community balance (NRCS 2011, 2014). However, both prescribed grazing and prescribed burning implemented to improve grazing forage, and may therefore result in increased grazing intensity (NRCS 2011, 2014). The negative effects of these practices on greater prairie-chicken abundance may be a result of increased grazing intensity. Upland wildlife habitat management had a positive effect on greater prairie-chicken abundance at the fine and broad spatial scales. This practice involves directly managing land to improve habitat for wildlife, with focus on enabling movement or providing shelter and food (NRCS 2015).

Differences in response to these conservation practices among species pose a challenge to conservation managers. Optimizing a landscape simultaneously for multiple species may be an effective strategy for managers to consider. For example, if lesser prairie-chickens were used as an umbrella species for upland game bird conservation management, brush management may be used across the landscape to increase abundance, but may result in declines in ring-necked pheasants. Instead, there may be a level of brush removal that maximizes abundance for all upland game bird species in the landscape, while not resulting in optimal conditions for any one species. More research is needed to better understand specific effects of these EQIP practices on individual species at different spatial scales to quantify such optimal landscapes.

CONCLUSION

No study has previously taken a multi-species approach to examining the effects of land cover and weather variables on abundance of upland game birds across multiple spatial scales. I took such an approach to quantify effects of land cover and weather on lesser and greater prairie-chickens, northern bobwhites, and ring-necked pheasants across these species' respective regions in Kansas. I found that land cover variables were the most important drivers in abundance changes in all four species. I found threshold responses in abundance of all four species to grassland:cropland ratio and edge density of grassland patches, suggesting that presence of both grassland and cropland are important in the landscape to simultaneously maintaining these populations. I also quantified effects of five different EQIP practices. At least one EQIP practice had a significant effect on lesser and greater prairie-chicken and ring-necked pheasant abundance. However, only upland wildlife habitat management and brush management had a positive affect on abundance, and only on greater prairie-chickens. This highlights the need for further research to consider the specific effects of each practice on each species to determine the viability of the practice as a conservation measure. Spatial scale is also important to consider in conservation management of these species, as responses to change in both land cover and EQIP practice implementation differed across spatial scales.

Conservation managers are often faced with the difficult task of managing multiple species simultaneously. In such circumstances, the best strategy may involve managing for an optimal landscape that may not be the optimal scenario for any one species, but beneficial to the most species. My results are an important first step in understanding the effects of land cover, weather, and conservation practice implementation on abundances of four different species of upland game birds in Kansas. Understanding how these effects compare across species and spatial scale will help develop conservation strategies to optimize conservation simultaneously for multiple species, thus improving conservation potential across a landscape.

TABLES

Table 1.1 Descriptions of annual upland game bird count surveys conducted by the Kansas Department of Wildlife, Parks, and Tourism for lesser and greater prairie-chickens (LEPC and GRPC), northern bobwhites (NOBO), and ring-necked pheasants (RNEP). Observers conducted surveys along transects consisting of 11 stops at 1-2 mile intervals. Transects were located across each of the species' respective ranges in Kansas. Shown are the species counted in each set of surveys, years the surveys were conducted, range of dates over which surveys were conducted each year, range of times over which surveys were conducted each day over the annual survey period, duration of each count at each stop along a transect, and each transect length.

Species	Survey Years	Dates of Survey	Time of Surveys	Listening Duration (min)	Transect Length (km)
LEPC & GRPC	1978 – 2017	20 Mar – 20 Apr	30 min before sunset – 90 min after sunset	3	16
NOBO	1998 – 2017	1 June – 16 June	Sunrise – completion of transect	5	16
RNEP	1997 – 2017	25 Apr – 15 May	45 min before sunset– completion of transect	2	32

Table 1.2 Results of the interaction models for each species (LEPC for lesser prairie-chicken, GRPC for greater prairie-chicken, RNEP for ring-necked pheasant, and NOBO for northern bobwhite) and buffer size combination. Shown are mean effects of significant (effect sizes not overlapping 0) variables, and 95% credible intervals of effects of significant variables. Significant variables included grassland:cropland ratio (LAND) and edge density (ED).

Model Combination		LAND			ED		
species	buffer (km)	mean	lower 95% CI	upper 95% CI	mean	lower 95% CI	upper 95% CI
LEPC	3	-	-	-	-	-	-
LEPC	5	-0.339	-0.655	-0.030	-	-	-
LEPC	10	-0.271	-0.572	-0.019	-	-	-
GRPC	3	-0.288	-0.455	-0.111	-	-	-
GRPC	5	-	-	-	-0.398	-0.525	-0.256
GRPC	10	-0.464	-0.645	-0.275	-0.588	-0.718	-0.454
RNEP	3	-0.302	-0.404	-0.194	-0.280	-0.358	-0.201
RNEP	5	-0.345	-0.440	-0.238	-0.296	-0.376	-0.221
RNEP	10	-0.349	-0.446	-0.243	-0.318	-0.393	-0.243
NOBO	3	-	-	-	-	-	-
NOBO	5	-	-	-	-	-	-
NOBO	10	-	-	-	-	-	-

Table 1.3 Results of the threshold models for grassland:cropland ratio (LAND) for each species (LEPC for lesser prairie-chicken, GRPC for greater prairie-chicken, RNEP for ring-necked pheasant, and NOBO for northern bobwhite) and buffer size combination. Shown are mean effects of significant (effect sizes not overlapping 0) variables and 95% credible intervals of effects of significant variables. The intensification coefficient is designated δ, and the threshold point is ϕ.

Model Combination		LAND			δ			ϕ		
species	buffer (km)	mean	lower 95% CI	upper 95% CI	mean	lower 95% CI	upper 95% CI	mean	lower 95% CI	upper 95% CI
LEPC	3	4.069	2.414	5.752	-4.522	-6.255	-2.913	-0.278	-0.398	-0.172
LEPC	5	3.811	2.038	5.642	-4.380	-6.215	-2.537	-0.289	-0.370	-0.192
GRPC	5	-0.440	-0.692	-0.204	1.923	0.409	4.301	2.883	1.413	4.094
GRPC	10	0.236	-0.095	0.595	-1.272	-1.783	-0.696	1.263	0.443	2.114
NOBO	3	0.134	0.065	0.202	-1.748	-4.733	-0.018	3.879	3.037	4.488
RNEP	3	-0.860	-1.006	-0.710	0.705	0.488	0.927	1.229	0.886	1.556
RNEP	5	-0.957	-1.108	-0.801	0.788	0.579	0.996	1.198	0.877	1.496
RNEP	10	-0.811	-0.933	-0.694	1.630	0.484	3.747	4.008	2.122	5.673

Table 1.4 Results of the threshold models for edge density (ED) for each species (LEPC for lesser prairie-chicken, GRPC for greater prairie-chicken, RNEP for ring-necked pheasant, and NOBO for northern bobwhite) and buffer size combination. Shown are mean effects of significant (effect sizes not overlapping 0) variables and 95% credible intervals of effects of significant variables. The intensification coefficient is designated δ, and the threshold point is ϕ.

Model Combination		ED			δ			ϕ		
Species	buffer (km)	mean	lower 95% CI	upper 95% CI	mean	lower 95% CI	upper 95% CI	mean	lower 95% CI	upper 95% CI
LEPC	10	4.455	0.380	7.723	-4.696	-7.855	-0.696	-1.218	-1.500	-0.898
GRPC	10	2.730	1.456	4.075	-3.194	-4.602	-1.933	-1.204	-1.319	-1.083
NOBO	3	0.140	0.088	0.196	-7.457	-9.929	-4.825	1.758	1.707	1.811
NOBO	5	0.142	0.083	0.200	-2.954	-5.484	-0.456	1.681	1.476	1.932
NOBO	10	0.125	0.066	0.183	-2.404	-5.073	-0.406	1.709	1.357	1.964
RNEP	3	-0.028	-0.129	0.074	-1.084	-1.293	-0.877	0.359	0.230	0.562
RNEP	5	-0.033	-0.137	0.070	-1.015	-1.216	-0.816	0.280	0.163	0.462
RNEP	10	0.138	-0.050	0.328	-0.921	-1.147	-0.710	-0.222	-0.456	0.040

Table 1.5 Results of the EQIP models for lesser prairie-chickens (LEPC) for each buffer size. Shown are the mean effects and 95% credible intervals of variables with high probabilities of inclusion (>0.5) from the stochastic search variable selection. Cover crop (code 340) was the only variable with high probability of inclusion.

Model Combination		340		
species	buffer (km)	mean	lower 95% CI	upper 95% CI
LEPC	3	-0.815	-1.296	-0.099
LEPC	5	-0.464	-1.053	0.047
LEPC	10	-	-	-

Table 1.6 Results of the EQIP models for greater prairie-chickens (GRPC) for each buffer size. Shown are the mean effects and 95% credible intervals of variables with high probabilities of inclusion (>0.5) from the stochastic search variable selection. Brush management (code 314), prescribed burning (code 338), prescribed grazing (code 528), and upland wildlife habitat management (code 645) were the variables with high probabilities of inclusion.

Model Combination		314			338			528			645		
species	buffer (km)	mean	lower 95% CI	upper 95% CI	mean	lower 95% CI	upper 95% CI	mean	lower 95% CI	upper 95% CI	mean	lower 95% CI	upper 95% CI
GRPC	3	-	-	-	-0.334	-0.793	0.035	-0.634	-0.926	-0.173	0.661	0.122	1.101
GRPC	5	-	-	-	-	-	-	-0.361	-0.862	0.005	-	-	-
GRPC	10	0.738	0.459	1.023	-0.785	-1.127	-0.414	-0.852	-1.146	-0.536	0.630	0.295	0.921

Table 1.7 Results of the EQIP models for ring-necked pheasant (RNEP) for each buffer size. Shown are the mean effects and 95% credible intervals of variables with high probabilities of inclusion (>0.5) from the stochastic search variable selection. Brush management (code 314) was the only variable with high probability of inclusion.

Model Combination		314		
species	buffer (km)	mean	lower 95% CI	upper 95% CI
RNEP	3	-0.358	-0.472	-0.227
RNEP	5	-0.287	-0.401	-0.174
RNEP	10	-0.267	-0.374	-0.163

FIGURES

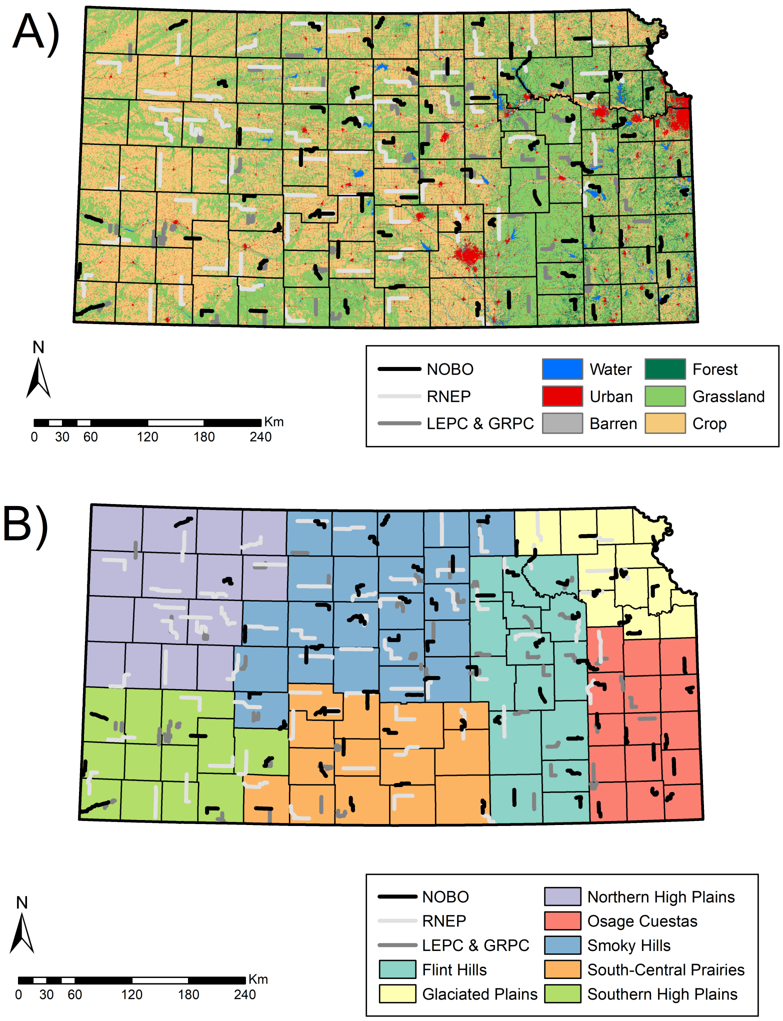

Figure 1.1 Distribution of transect locations from annual count surveys conducted by the Kansas Department of Wildlife, Parks, and Tourism in Kansas for lesser and greater prairie-chickens (LEPC & GRPC), northern bobwhites (NOBO), and ring-necked pheasants (RNEP) conducted by the Kansas Department of Wildlife, Parks, and Tourism across A) land cover (U.S. Geological Survey 2014) and B) small game management regions (KDWPT 2018) in the state.

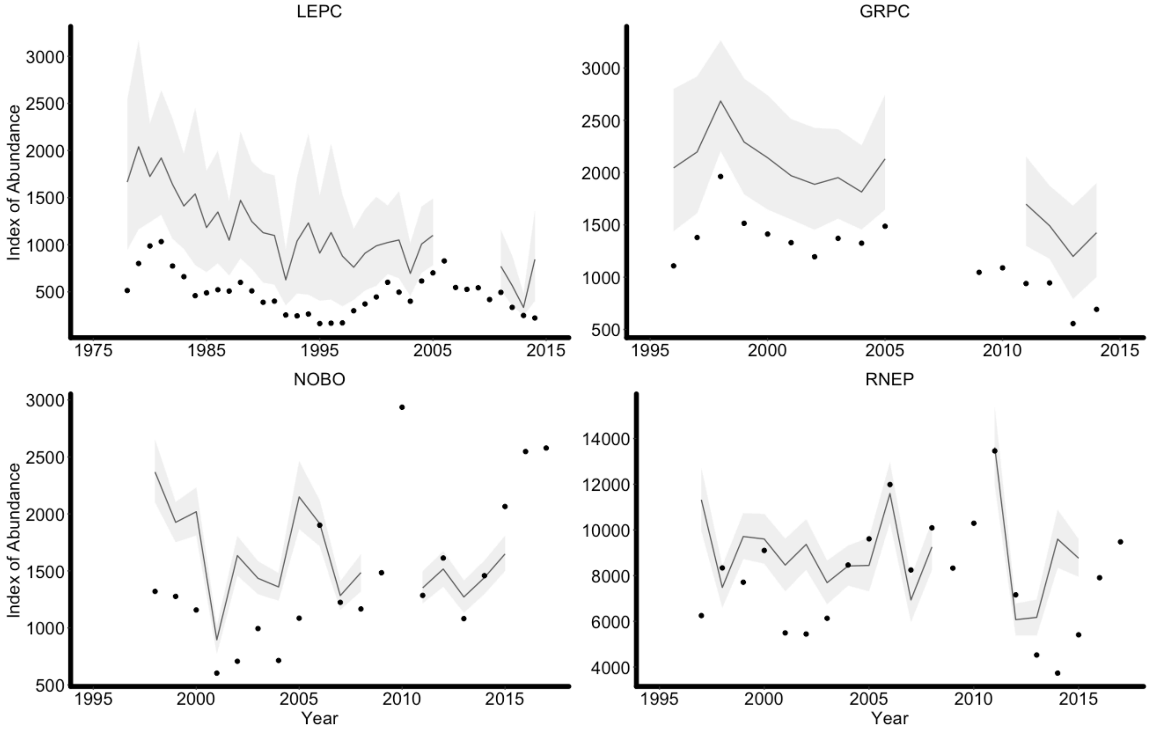

Figure 1.2 Estimates of total abundance from the threshold models for lesser prairie-chickens (LEPC) and northern bobwhites (NOBO), and from the interaction models for greater prairie-chickens (GRPC) and ring-necked pheasants (RNEP) on surveyed transects in Kansas. Counts from the surveys are shown as points and the 95% credible intervals of population estimates are shown in gray. Population estimates were not calculated in years where a lack of data prevented covariate estimation.

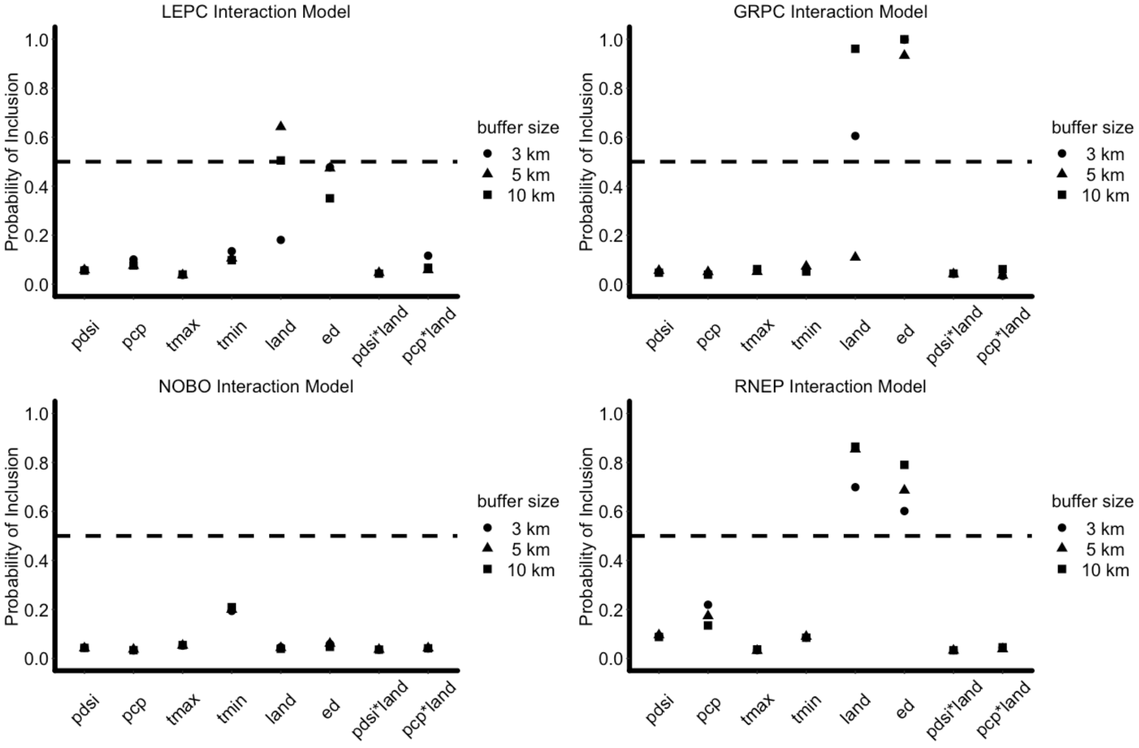

Figure 1.3 Inclusion probabilities from stochastic search variable selection used in the models with linear effects of landscape and weather variables and interaction terms for the interaction between landscape and weather variables. Variables likely affected abundance of lesser prairie-chicken (LEPC), greater prairie-chicken (GRPC), northern bobwhite (NOBO), or ring-necked pheasant (RNEP) populations if the inclusion probabilities were above 0.5 (dashed line). Variables considered were the Palmer Drought Severity Index (pdsi) of summer months with a 1-year lag effect, precipitation index (pcp) of winter months in the winter prior to survey season, maximum temperature (tmax) of summer months with a 1-year lag effect, minimum temperature (tmin) of winter months in the winter prior to survey season, grassland:cropland ratio (land), edge density of grassland patches (edge density), the interaction between the Palmer Drought Severity

Index variable and the grassland:cropland ratio variable (pdsi*land), and the interaction between the precipitation index variable and the grassland:cropland ratio variable (pcp*land). Shape of the point represents the spatial scale of landscape variables used in the model.

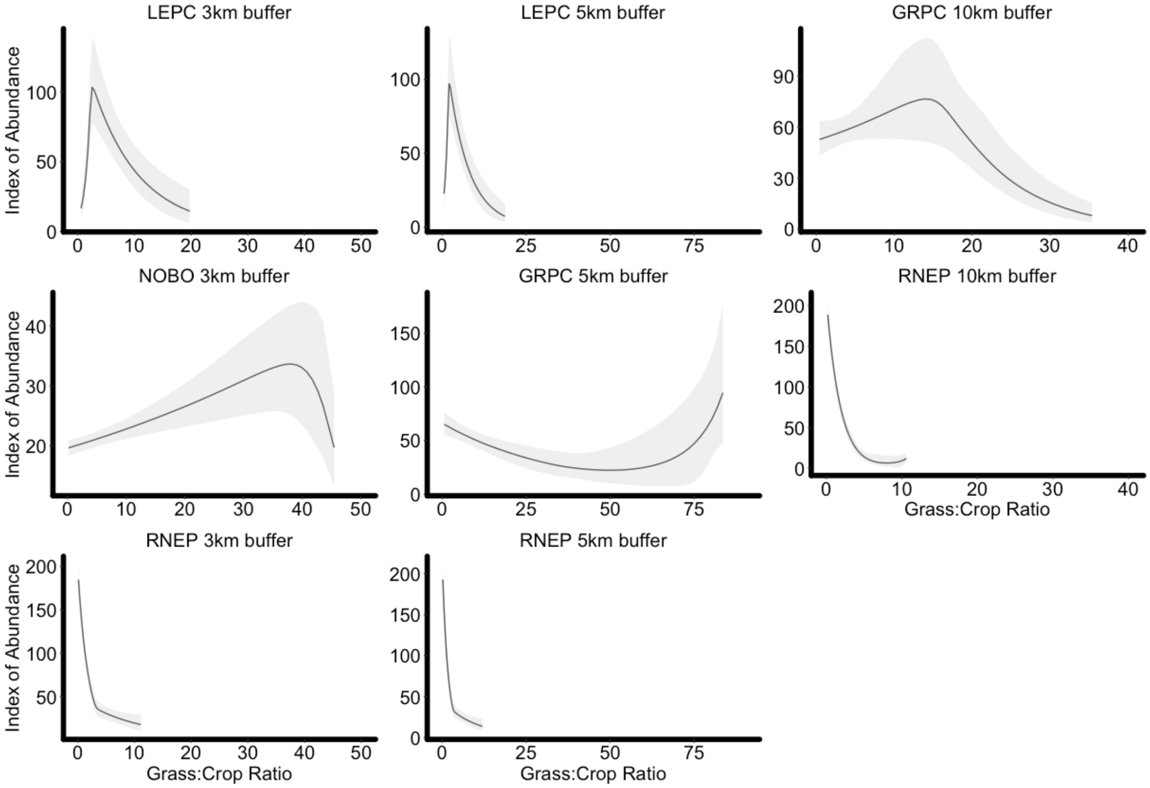

Figure 1.4 Changes in relative abundance of lesser prairie-chicken (LEPC) males, greater prairie-chicken (GRPC) males, calling northern bobwhite (NOBO) males, and number of ring-necked pheasant (RNEP) crowing calls in Kanas in response to an estimated threshold of grassland:cropland ratio, with 95% credible intervals shown in gray.

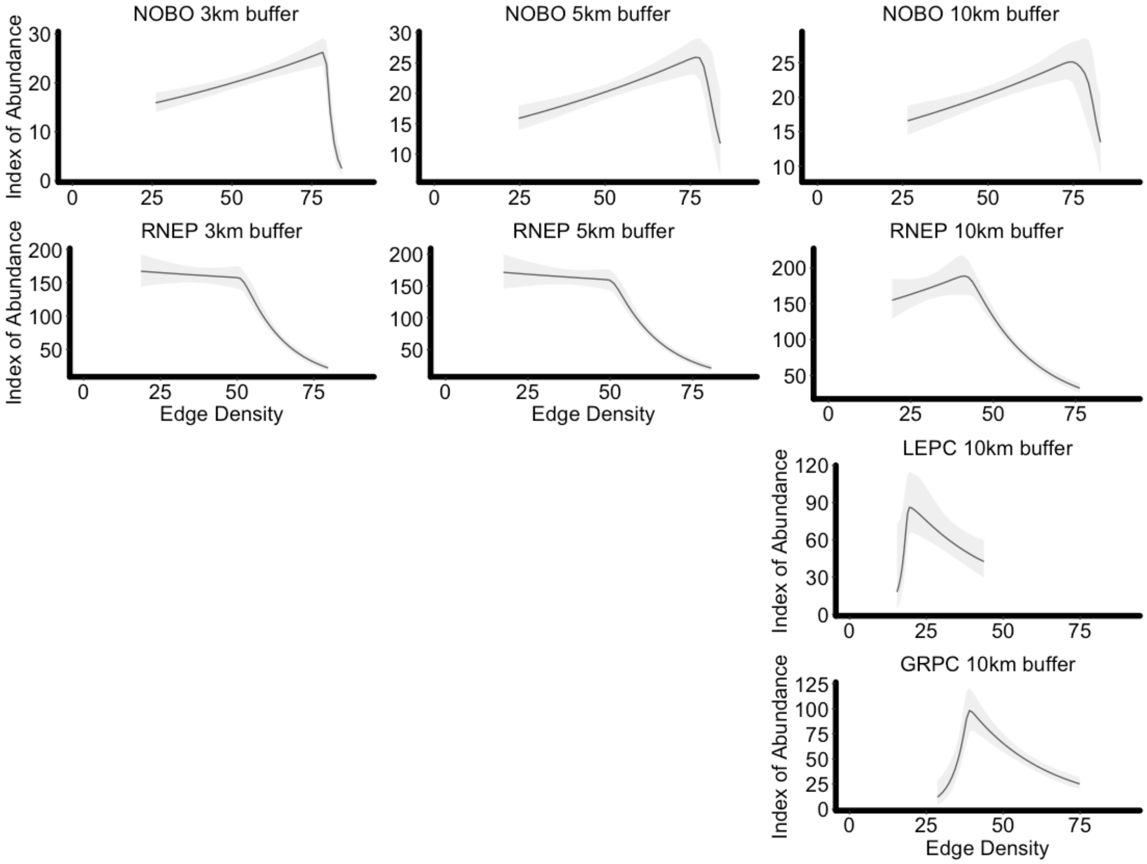

Figure 1.5 Changes in relative abundance of lesser prairie-chicken (LEPC) males, greater prairie-chicken (GRPC) males, calling northern bobwhite (NOBO) males, and number of ring-necked pheasant (RNEP) crowing calls in Kanas in response to an estimated threshold of edge density of grassland patches, with 95% credible intervals shown in gray.

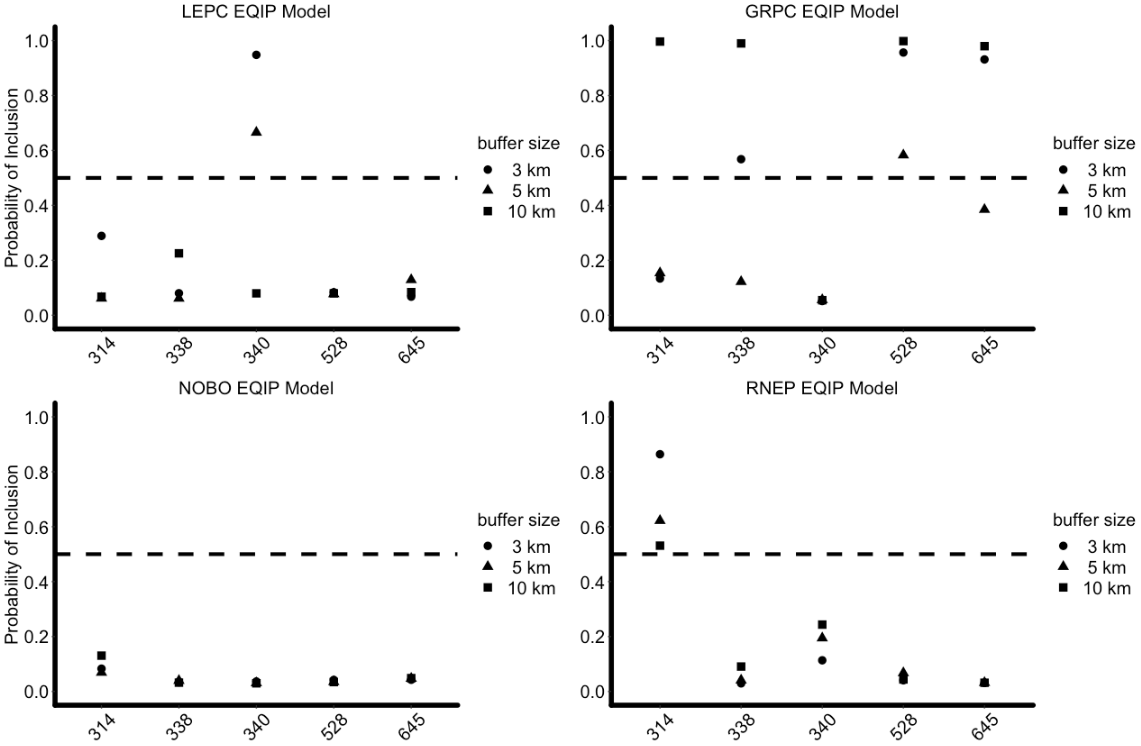

Figure 1.6 Inclusion probabilities from stochastic search variable selection used in the models with linear effects of Environmental Quality Incentives Program (EQIP) practices. Variables likely affected abundance of lesser prairie-chicken (LEPC), greater prairie-chicken (GRPC), northern bobwhite (NOBO), or ring-necked pheasant (RNEP) populations if the inclusion probabilities were above 0.5 (dashed line). EQIP practices considered were brush management (code 314), prescribed burning (code 338), cover crop (code 340), prescribed grazing (code 528), and upland wildlife habitat management (code 645). Shape of the point represents the spatial scale of landscape variables used in the model.

CHAPTER TWO

THE USE OF A DECISION SUPPORT TOOL TO PRIORITIZE CANDIDATE AREAS FOR LESSER PRAIRIE-CHICKEN CONSERVATION

INTRODUCTION

Conservation planning is the process of developing strategies to effectively minimize loss of biodiversity or some other valued aspect of the natural world (Pressey and Bottrill 2009). Traditional *ad-hoc* methods of conservation planning often led to inefficient allocation of resources and prioritizing areas based on criteria not related to conservation value, such as location or aesthetic qualities (Pressey et al. 1993). Recent development of many systematic methods for conservation planning, such as spatial conservation prioritization, has improved effectiveness of such planning and led to improved conservation outcomes (Margules and Pressey 2000, Pressey and Bottrill 2009). Spatial conservation prioritization involves using quantitative spatial analysis to identify areas of high conservation value to invest in for future conservation actions (Wilson et al. 2009, Kukkala and Moilanen 2013). This analysis aids managers in identifying important areas for biodiversity, prioritizing these areas so resources can be allocated to efficiently achieve conservation targets, and evaluating tradeoffs between conservation and socio-economic objectives (Pressey and Bottrill 2009, Kukkala and Moilanen 2013).

The lesser prairie-chicken (*Tympanuchus pallidicinctus*) is a prairie grouse species native to the United States found in the High Plains region of the Great Plains in

Kansas, Colorado, Oklahoma, New Mexico, and Texas. Once widely abundant in the region, lesser prairie-chicken population size and range has declined by >90% since the early 1900s (Taylor and Guthery 1980, Hagen et al. 2004). These declines are primarily attributed to decreases in habitat quality caused by agriculture, grazing, and energy development (NRCS 1999a, Hagen et al. 2004, Van Pelt et al. 2013, Spencer et al. 2017). Securing quality habitat is therefore the primary focus of current lesser prairie-chicken conservation efforts (Van Pelt et al 2013). To maximize conservation potential, it is important that quality habitat is secured in ecologically important areas, such as areas near lek locations. Leks are important activity centers for lesser prairie-chickens, and individuals rarely move further than 4.8 km from leks, regardless of time of year (Riley et al. 1994, Woodward et al. 2001, Hagen and Giesen 2005, Kukal 2010, Boal et al. 2014, Grisham et al. 2014, Boal and Haukos 2016). Systematic methods of conservation planning could help managers efficiently select ecologically important areas to prioritize lesser prairie-chicken conservation efforts while minimizing economic affects of conservation plans.

In response to threats to the lesser prairie-chicken population, the Lesser Prairie-Chicken Interstate Working Group developed the Lesser Prairie-Chicken Range-wide Conservation Plan (hereafter range-wide plan) to facilitate effective conservation of the species while minimizing economic affects of conservation. As part of this plan, the Lesser Prairie-Chicken Interstate Working Group identified range-wide and sub-population goals for a ten-year timeframe, desired habitat amounts and conditions to achieve these population goals, and ecologically important areas to focus conservation efforts (Van Pelt et al. 2013). While some of these areas occur on public lands, the majority occur on private lands, highlighting a need for involvement of private landowners in habitat management. The range-wide plan therefore aims to enhance and coordinate implementation of incentive-based landowner programs to increase voluntary participation of management practices that create quality habitat for lesser prairie-chickens (Van Pelt et al. 2013).

The Southern Great Plains Crucial Habitat Assessment Tool (SGP CHAT, hereafter CHAT) is a decision support tool designed in coordination with the Range-wide Plan to designate areas for lesser prairie-chicken conservation and industry development (SGP CHAT 2013). The CHAT is an online tool for managers, industry, and the public that identifies priority areas and connectivity zones for lesser prairie-chicken conservation. The CHAT is used to spatially map ecologically important areas identified in the Range-wide Plan, areas identified in habitat models as currently available as lesser prairie-chicken habitat, and areas identified in models as not currently available for lesser prairie-chicken habitat without further land management. The CHAT is also used to estimate potential affects of development in locations within the lesser prairie-chicken range, thus encouraging avoidance of negative effects on lesser prairie-chicken populations from development or mitigating effects when avoidance is not possible (SGP CHAT 2013, Van Pelt et al. 2013). While the CHAT provides many tools for lesser prairie-chicken conservation management, no decision support tool is currently being used to aid in selection of areas for conservation focus within ecologically important areas identified in the Range-wide Plan, or to facilitate lesser prairie-chicken conservation while minimizing negative economic effects to the agricultural industry.

Marxan is a decision support tool for spatial conservation prioritization and reserve system design (Ball et al. 2009). Marxan is a software program designed to help solve “minimum set problems” where some minimum target of conservation features is to be achieved for the smallest possible cost (McDonnell et al. 2002, Game and Grantham 2008). Marxan is therefore used to help identify a combination of sites that meet user-defined conservation targets for the minimum amount of any user-defined socio-economic cost (Ball and Possingham 2000, Possingham et al. 2000). Marxan could be an effective tool to complement the Range-wide Plan and CHAT for lesser prairie-chicken conservation planning. While the CHAT identifies ecologically important areas for lesser prairie-chicken conservation, it does not aid in the selection of areas to prioritize within these ecologically important areas. The CHAT is used to estimate effects of development within the lesser prairie-chicken range, but it does not estimate effects of lesser prairie-chicken conservation to economic activity in the lesser prairie-chicken range. Using Marxan could help managers accomplish both of these tasks by aiding in the selection of potential conservation areas that minimize economic effects to the region, as well as allowing managers to compare multiple conservation planning scenarios.

While Marxan is an effective tool for spatial conservation prioritization, it does have some limitations. One key limitation is the difficulty in capturing functional connectivity in reserve system designs (Ardron et al. 2010). Functional connectivity is important to persistence of animal populations through the maintaining of ecological processes such as animal movement, genetic diversity, and range shifts (Kareiva and Wennergren 1995, Ricketts 2001, McRae et al. 2008). It is therefore important to pair Marxan results with *post-hoc* connectivity analyses to ensure potential conservation plans are ecologically viable. One way to model connectivity and quantify how landscape features affect connectivity is through use of circuit theory. In circuit theory, animal movement is considered analogous to current in an electrical system, which allows simultaneous evaluation of multiple dispersal pathways (McRae 2006, McRae et al. 2008). Landscapes are represented as conductive surfaces, with landscape features that inhibit animal movement assigned high resistance values, and landscape features more permeable to animal movement assigned low resistance values. Calculated resistances and current flow in the system can then be related to ecological processes of connectivity (McRae 2006, McRae et al. 2008, McRae et al. 2013).

I used Marxan to aid in selecting specific 2.59-km^2 (1-mi^2) sites to prioritize areas for lesser prairie-chicken conservation identified in the Range-wide Plan. I incorporated economic data for both crops and grazing in Kansas to select candidate areas that maximized lesser prairie-chicken conservation potential while minimizing potential economic effects to the agricultural industry in Kansas. I set conservation targets to select enough land in the lesser prairie-chicken range to meet the population goals set in the Range-wide Plan. I adjusted these targets spatially to select sites based on three scenarios: 1) sites primarily from ecologically important areas identified as part of the Range-wide Plan, 2) sites primarily from areas identified from habitat models as current potential habitat for lesser prairie-chickens, and 3) sites primarily from areas within 10 km of current known lesser prairie-chicken lek locations. I hypothesized that scenarios 1 and 3 would differ the most, while scenario 2 would share similarities with scenarios 1 and 3 and selection would be correlated with both scenarios. Scenario 1 was the most restrictive, constraining potential selections to only the most ecologically important areas, while scenario 3 was the least restrictive with no constraints on potential selections in relation to current ecological value within a buffered distance around leks. Restrictions in scenario 2 fell between scenarios 1 and 3, expanding potential selections outside the most ecologically important areas, but constraining selections to areas defined as current potential habitat. I also hypothesized that despite these constraints, all three scenarios would have selected areas in common when the economic value of these areas was low. These Marxan analyses provide important aid to managers in selecting areas to prioritize lesser prairie-chicken conservation and evaluate potential effects of multiple conservation scenarios on the agricultural industry.

METHODS

Study Area

I evaluated different conservation planning scenarios for the lesser prairie-chicken (*Tympanuchus pallidicinctus*) across its range in Kansas. The lesser prairie-chicken range in Kansas was made up of three distinct habitat regions: the Short-Grass Prairie/CRP Mosaic Ecoregion, Mixed-Grass Prairie Ecoregion, and Sand Sagebrush Prairie Ecoregion (Figure 2.1A, McDonald et al. 2014). The Short-Grass Prairie/CRP Mosaic Ecoregion was comprised of short-grasses including buffalograss (*Bouteloua* *dactyloides*) and blue grama (*Bouteloua gracilis*), and mid-grasses including sideoats grama (*Bouteloua curtipendula*), little bluestem (*Schizachyrium scoparium*), sand dropseed (*Sporobolus cryptandrus*), and western wheatgrass (*Pascopyrum smithii*). The Mixed-Grass Prairie Ecoregion consisted of mixed grasses and agricultural lands, including grazing and cropland. The Sand Sagebrush Prairie Ecoregion primarily contained mid and tall-grasses, including sand bluestem (*Andropogon hallii*), little bluestem, switchgrass (*Panicum virgatum*), prairie sandreed (*Calamovilfa longifolia*), and sand dropseed (McDonald et al. 2014, Spencer et al. 2017). Agricultural land was also prevalent across the lesser prairie-chicken range in Kansas. This included cropland to produce crops including corn, cotton, sorghum, soybeans, wheat, and alfalfa as well as pasture for cattle production (Figure 2.1b, USDA et al. 2018). I used 2.59-km^2 (1-mi^2) hexagonal planning units across the lesser prairie-chicken range as the unit of study, as these are the planning units currently used in Southern Great Plains Crucial Habitat Assessment Tool (SGP CHAT 2013).

Identifying Priority Areas

I identified priority areas for lesser prairie-chicken conservation within the lesser prairie-chicken range in Kansas using the decision-support software Marxan (Ball et al. 2009). I used Marxan to identify areas to prioritize for conservation by finding the optimal set of planning units that minimized the following objective function:

$$\text{Objective function} = \mathbf{\Sigma}_{\text{PUs}}\,\text{Cost} + \text{BLM}(\mathbf{\Sigma}_{\text{PU}}\,\text{Boundary}) + \text{SPF}(\mathbf{\Sigma}_{\text{CTs}}\,\text{Penalty}) \qquad (1)$$

where PUs were planning units, cost was the defined cost for selecting a specific planning unit in the conservation reserve design, BLM was the boundary length modifier, boundary was the outer boundary of the selected planning units, SPF was the species penalty factor, CTs were the conservation targets, and penalty was the amount of a conservation target not met in the solution. Marxan thus used this objective function to minimize the cost of all sites included in the conservation reserve design while accounting for penalties for not achieving conservation targets (weighted by the user-defined SPF) and for low connectivity (weighted by the user-defined BLM; Game and Grantham 2008, Hermoso et al. 2012).

Conservation Targets

As part of the Lesser Prairie-Chicken Range-wide Conservation Plan, the Lesser Prairie-Chicken Interstate Working Group (LPC IWG) identified 10-year average population targets for lesser prairie-chickens in each of the ecoregions across the species' overall range (Van Pelt et al. 2013). I adjusted these population targets to calculate population goals for each ecoregion in Kansas by multiplying the total population goal of each ecoregion by the proportion of area of each ecoregion located in Kansas. I then divided the population goals of each ecoregion in Kansas by the estimated lesser prairie-chicken population densities in Kansas from the Range-wide Plan to calculate the area of quality habitat needed to sustain population goals in Kansas (Van Pelt et al. 2013; Table 2.1).

Cost Data

I obtained irrigated crop budgets for corn, cotton, sorghum, soybeans, winter wheat, and alfalfa for western Kansas from AgManager.info (Ibendahl et al. 2018). AgManager.info is a database of information from the Department of Agricultural Economics at Kansas State University, which serves as a source of information, analysis, and decision-making tools for agricultural producers, agribusiness, and others (Kansas State University Department of Agricultural Economics 2019). From these budgets I subtracted the per acre income from total expenses per acre to calculate the crop-specific profit per acre of land. When data were split into northwestern and southwestern regions, I averaged all income and expense data to get values for all of western Kansas.

To calculate per acre profit from cattle production, I also obtained beef farm and pasture rental rate budgets for Kansas from AgManager.info (Reid and Taylor 2016, Reid and Tonsor 2017). From the beef farm budget, I subtracted the per cow income from the total expenses per cow for both cow-calf and stocker operations. I then used the estimated stocking rates for normal years for both types of cattle operations from the pasture rental rate budget to calculate per acre profits of both types of cattle operations. I then averaged these two profit values to get the average profit per acre of pasture for all cattle in Kansas.

To remove negative profit values, I scaled all profit values to between 0 and 1. I then multiplied these values by the areas of the respective cropland or pasture in each planning unit to calculate a “cost” value for each of the planning units. This “cost” value is therefore an index of potential economic productivity in the agricultural industry.

Conservation Scenarios

As part of the Lesser Prairie-Chicken Range-Wide Plan, the LPC IWG proposed a focal area approach to effectively conserve habitat required to meet lesser prairie-chicken population targets. Under this strategy, the LPC IWG identified a set of focal areas consisting of about 36% of the lesser prairie-chicken range, along with connectivity zones to allow linkage among focal areas (Van Pelt et al. 2013). Under the Range-wide Plan, the LPC IWG set a goal of conserving enough habitat (based on estimated population density values in each ecoregion) in the focal areas to sustain 75% of the population targets, with the other 25% of the population targets sustained in the connectivity zones and other areas. To reach these targets, the Range-wide Plan proposed conserving good to high quality habitat in at least 70% of each focal area, as well as in at least 40% of each connectivity zone (Van Pelt et al. 2013). I obtained data on the proposed locations of the focal areas and connectivity zones, as well as locations of modeled habitat and nonhabitat within the lesser prairie-chicken range from the Southern Great Plains Crucial Habitat Assessment Tool (hereafter CHAT), a spatial model designed to help prioritize areas for lesser prairie-chicken conservation and industry development (SGP CHAT 2013). I evaluated three different conservation scenarios, each of which I designed to have conservation targets that involved conserving enough habitat in each ecoregion in the lesser prairie-chicken range in Kansas to sustain population targets set in the range-wide plan. In each scenario, I adjusted the restrictions on where the land conserved was located so that my three scenarios represented a gradient of greater restriction (planning units in the solution primarily selected from the focal areas and connectivity) to lower restriction (planning units did not have to be located in or near focal areas and connectivity zones) to compare efficiency of potential conservation solutions.

In Scenario 1, I set conservation targets of conserving 70% of each focal area (CHAT score of 1, Figure 2.2) and 40% of each connectivity zone (CHAT score of 2, Figure 2.2) located in Kansas. I subtracted these areas in each ecoregion from the total land in each ecoregion needed to sustain lesser prairie-chicken population targets to get the amount of land in modeled habitat outside the focal areas and connectivity zones (CHAT score of 3, Figure 2.2) needed to meet these targets. I compared these areas to the total area of land with a CHAT score of 3 in each ecoregion to convert these areas to proportional targets. This resulted in a target of 2% of the Mixed-Grass Prairie Ecoregion, 18% of the Sand Sagebrush Prairie Ecoregion, and 21% of the Short-Grass Prairie/CRP Mosaic Ecoregion with CHAT score 3.

In Scenario 2, I did not specify targets for focal areas or connectivity zones. Instead I set conservation targets based on land from the combined area of land with CHAT scores 1-3. I compared these areas in each ecoregion needed to sustain the lesser prairie-chicken population targets in Kansas to the total areas of land with CHAT scores 1-3 in each ecoregion to convert these areas to proportional targets. This resulted in a target of 24% of the Mixed-Grass Prairie Ecoregion, 35% of the Sand Sagebrush Prairie Ecoregion, and 47% of the Short-Grass Prairie/CRP Mosaic Ecoregion with CHAT scores 1-3.

In Scenario 3, I instead set conservation targets based on conserving land within 10 km of known lek locations. A 10 km buffer was a large enough distance to account for lesser prairie-chicken home range size and allow the Marxan algorithm enough freedom to efficiently select the amount of land needed to sustain the lesser prairie-chicken population targets (Haukos and Zavaleta 2016). I obtained known lek locations from the CHAT and created a 10 km buffer around them. I then compared the area in each ecoregion needed to sustain the lesser prairie-chicken population targets in Kansas to the total areas of land within 10 km of known lek locations in each ecoregion to convert the areas to proportional targets. This resulted in a target of 21% of land in the Mixed-Grass Prairie Ecoregion, 42% of land in the Sand Sagebrush Prairie Ecoregion, and 31% of land in the Short-Grass Prairie/CRP Mosaic Ecoregion within 10 km of known lek locations.

Calibration and Marxan Analyses

I calibrated the SPF, BLM, number of iterations, and number of runs used for each Marxan scenario, following the guidelines of the Marxan Good Practices handbook (Ardron et al. 2010). Calibration involves conducting sensitivity analyses on these user-defined input parameters to ensure robust results are produced by the Marxan algorithm. I used Zonae Cogito v 1.74 (Segan 2011) to run Marxan v 2.43 (Ball et al. 2009) for all calibration and analyses. To calibrate the SPF, I used the calibration tool in Zonae Cogito to repeatedly run each Marxan scenario with increasing SPF values. I evaluated these results to find the point at which the number of conservation features that do not meet their target approached 0 (i.e., the lowest SPF value that still results in the meeting of all conservation targets).

To calibrate the BLM, I used a weighting method developed to explore multi-objective tradeoffs between optimization in cost and boundary (Fischer and Church 2005). I first ran each Marxan scenario (using the calibrated SPF) with a BLM of 0 and selected the result from the output with the lowest total cost. I calculated the cost and boundary length for that solution and plotted it as point X, which represented the minimum cost solution. I then ran each Marxan scenario with all cost values set to 0 and a BLM of 1 and selected the result from the output with the lowest total boundary. I calculated the cost and boundary length of that solution and plotted it as point Y, which represented the minimum boundary solution. The line connecting these two points was the estimated trade-off curve between the optimal cost and optimal boundary solutions. I used the absolute value of the slope of this line as the BLM in each scenario, as this point represented the optimum on the trade-off curve between minimizing cost and minimizing boundary length (Ardron et al. 2010).

To calibrate the number of iterations used in the Marxan algorithm, I ran each Marxan scenario (using the calibrated SPF and BLM) first with 10^6 iterations, as this is the minimum recommended number of iterations for Marxan problems (Ardron et al. 2010). I then iteratively reran each Marxan scenario while increasing the number of iterations by a factor of 10 (e.g., 10^7, 10^8, etc. iterations). I compared the Marxan "scores" (values calculated by the objective function) of the outputs of each scenario and selected a number of iterations that was large enough where increasing iterations resulted in little improvement to the score, but low enough to still allow a sufficient diversity of solutions.

To calibrate the number of runs (i.e., number of potential solutions calculated) in the Marxan algorithm, I ran each Marxan scenario (using the calibrated SPF, BLM, and number of iterations) first with 100 runs, as this is the minimum recommended number of runs for Marxan problems (Ardron et al. 2010). I then iteratively reran each Marxan scenario while increasing the number of runs by 100 (e.g., 200, 300, etc. runs). I compared the selection frequencies (number of times a particular planning unit was selected out of the total number of runs) and selected a number of runs that was high enough where an increase in the number of runs resulted in little difference in selection frequency of all the planning units in the outputs.

I ran Marxan using the above calibrated parameters for each conservation scenario. I evaluated the outputs to make sure all solutions met conservation targets. I visually examined the "best" solution, or solution with the lowest Marxan score, and the selection frequencies of each scenario.

Scenario Comparisons

To compare scenarios, I calculated the number of planning units selected with each CHAT score in the "best" solution. I also used a Pearson's correlation test to compare the selection frequencies of planning units in each pair of scenarios. To visually compare each pair of scenarios, I created change maps by subtracting the selection frequency of planning units in one scenario from the corresponding planning units of another. Values in the resulting map therefore ranged from -100 (contained in 100% of solutions in one scenario and 0% of solutions in the other) to 100 (reverse of -100 values), with values of 0 representing no difference between the two sets of solutions.

Connectivity Analyses

To test the potential connectivity of habitat patches selected in the "best" solutions of each scenario, I used Circuitscape v 4.0 to run connectivity models (McRae et al. 2013). I used the human footprint index for Kansas in 2009 at a 1-km^2 spatial resolution as a map of resistance to lesser prairie-chicken movement. Indices of human modification of the landscape have been commonly used to represent resistances to animal movement in a landscape in connectivity models (Magle et al. 2009, Alagador et al. 2012, Belote et al. 2016, McClure et al. 2016, Littlefield et al. 2017). The human footprint index is a measure of direct and indirect human pressures on the environment and includes data on the extent of built environments, cropland, pasture land, human population density, night-time lights, railways, roads, and navigable waterways (Venter et al. 2015, 2016). I used habitat patches selected in the "best" solutions of each scenario as focal nodes in the connectivity model. I then ran a pairwise connectivity model and created cumulative current maps to compare connectivity between scenarios.

RESULTS

Calibration and Marxan Analyses

From sensitivity analysis on the input parameters, I determined that a SPF of 2.3, 1.7, and 1.8 was sufficient for scenarios 1, 2, and 3, respectively, and in all cases resulted in solutions where all conservation targets were met. After evaluating the multi-objective tradeoff function between optimizing cost and optimizing boundary length, I determined a BLM of 339.35 in scenario 1, 250.345 in scenario 2, and 161.0795 in scenario 3 best represented the optimum between cost and boundary. In all solutions, the respective BLM values resulted in solutions with planning units selected for conservation adequately clumped together across the landscape. I determined that 10^7 iterations and 100 runs were adequate in all scenarios to achieving near-optimal results while still producing a diversity of solutions.

In Scenario 1, the "best" (i.e., lowest Marxan score calculated by the objective function, Figure 2.3a) solution consisted of 5,345 planning units, the majority of which were located in the focal areas (CHAT score of 1), followed by modeled habitat not located in the focal areas or connectivity zones (CHAT score of 3), connectivity zones (CHAT score of 2), and modeled nonhabitat (CHAT score of 4; Table 2.2). Among the entire set of 100 solutions, planning units selected within the focal areas and connectivity zones had high selection frequencies (i.e., selected in a greater percent of solutions). Areas selected outside the focal areas and connectivity zones generally had low selection frequencies, particularly in the Short-Grass Prairie/CRP Mosaic and Sand Sagebrush Prairie ecoregions. Of the highly selected planning units outside the focal areas and connectivity zones, most were connected to patches of highly selected planning units within the focal areas and connectivity zones, although a patch of highly selected planning units disconnected from those within the focal areas and connectivity zones did occur in the Short-Grass Prairie/CRP Mosaic Ecoregion in Wichita and Pawnee counties (Figure 2.4a).

In Scenario 2, the “best” solution (Figure 2.3b) consisted of 4,633 planning units, the majority of which were located in the focal areas, followed by modeled habitat not in the focal areas or connectivity zones, connectivity zones, and modeled nonhabitat (Table 2.2). Among the entire set of 100 solutions, all three ecoregions contained patches of highly selected planning units both within and outside of the focal areas and connectivity zones, as well as patches of planning units with low selection frequency within and outside of these areas (Figure 2.4b).

In Scenario 3, the “best” solution (Figure 2.3c) consisted of 4,498 planning units, the majority of which were located in modeled nonhabitat, followed by the focal areas, modeled habitat not found in the focal areas or connectivity zones, and connectivity zones (Table 2.2). Among the entire set of 100 solutions, planning areas selected outside the focal areas and connectivity zones had increased selection frequency, particularly in the Short-Grass Prairie/CRP Mosaic and the Mixed-Grass Prairie ecoregions. Overall, most planning units were selected with either high frequency, or not at all. There were some planning units selected with low frequency in the northern Short-Grass Prairie/CRP Mosaic Ecoregion (Figure 2.4c).

Scenario Comparisons

Selected planning units in Scenario 1 were moderately correlated with the selected planning units in Scenario 2 ($r = 0.66$). Selected planning units in Scenario 3 had little correlation to selected planning units in Scenario 1 ($r = 0.03$) or Scenario 2 ($r = 0.16$).

When comparing Scenarios 1 and 2, planning units highly selected in Scenario 1 but not Scenario 2 in all three ecoregions occurred primarily in the focal areas, while planning units highly selected in Scenario 2 but not in Scenario 1 in all three ecoregions occurred in both the connectivity zones and modeled habitat located outside the focal areas and connectivity zones (Figure 2.5a).

When comparing Scenarios 1 and 3, planning units highly selected in Scenario 1 but not Scenario 3 in the Short-Grass Prairie/CRP Mosaic and Mixed-Grass Prairie ecoregions occurred in every focal area and connectivity zone, whereas planning units most selected in Scenario 3 but not Scenario 1 occurred almost exclusively outside of the focal areas and connectivity zones in both modeled habitat and modeled nonhabitat. Differences between Scenarios 1 and 3 in the Sand Sagebrush Prairie Ecoregion primarily followed this same pattern, though differences between the two scenarios were not as pronounced as in the other ecoregions (Figure 2.5b).

When comparing Scenarios 2 and 3, planning units selected in Scenario 2 but not in Scenario 3 in the Short-Grass Prairie/CRP Mosaic Ecoregion primarily occurred in the focal areas and connectivity zones. Planning units selected in Scenario 3 but not in Scenario 2 primarily occurred outside the focal areas and connectivity zones in both modeled habitat and modeled nonhabitat. Differences between Scenarios 2 and 3 in the

Mixed-Grass Prairie Ecoregion primarily followed this same pattern, but planning units selected in Scenario 3 but not in Scenario 2 primarily occurred in the northern extent of the ecoregion. Planning units selected in Scenario 2, but not in Scenario 3 in the Sand Sagebrush Prairie Ecoregion occurred in both inside and outside of the focal areas and connectivity zones, while planning units selected in Scenario 3 but not in Scenario 2 occurred primarily outside of the focal areas and connectivity zones in both modeled habitat and modeled nonhabitat. Differences between Scenarios 2 and 3 in the Sand Sagebrush Prairie Ecoregion were not as pronounced as in other ecoregions (Figure 2.5c).

Connectivity Analyses

Overall, there was high connectivity between habitat patches in each ecoregion in all three scenarios (Figure 2.6). In the Short-Grass Prairie/CRP Mosaic Ecoregion, connectivity was greatest in the interior of the region, with areas of lower connectivity existing in the outer portions of the region. In the Sand Sagebrush Prairie Ecoregion, connectivity exhibited a north/south gradient, with connectivity greatest in the northern extent of the region and decreasing southward. In the Mixed-Grass Prairie Ecoregion, connectivity exhibited an east/west gradient, with connectivity highest in the western extent of the region and decreasing eastward.

DISCUSSION

I used the decision support tool Marxan to develop a model to select potential areas to prioritize for conservation of the lesser prairie-chicken while also minimizing negative economic effects on the agricultural industry. I adjusted the spatial restrictions of potentially selected areas in my model to evaluate three alternative conservation strategies for meeting population goals set by the Lesser Prairie-Chicken Interstate Working Group in the Lesser Prairie-Chicken Range-wide Conservation Plan. I also used Circuitscape to build connectivity models to test potential connectivity of habitat patches selected in my Marxan models. I found that 1) differences in conservation approach in each Marxan scenario led to different conservation solutions, though these differences varied by ecoregion, 2) despite these differences, some patches of planning units were highly selected in all three scenarios, and 3) the landscape in the lesser prairie-chicken range had high permeability in all three scenarios, thus allowing for adequate connectivity between habitat patches selected by Marxan.

Scenario Differences

Selection frequency is a valuable indicator of irreplaceability in a conservation design (Trombulak et al. 2008). The more a planning unit is selected, the more irreplaceable it is, and the more important it is to creating an efficient conservation reserve design. When a planning unit has high irreplaceability, the unselected planning units around it likely have much greater cost values. Selecting one of these suboptimal planning units instead of the irreplaceable one would therefore add large costs to the conservation design. When many planning units are selected less frequently in place of fewer planning units selected more frequently, there is more flexibility in which planning units can be selected from the area to meet conservation goals and create an efficient solution. These replaceable planning units all likely have similar cost values, so selecting one over another would have little influence on the cost of the conservation design.

Under the focal area and connectivity zone scenario (Scenario 1), selected planning units in the focal areas and connectivity zones had high selection frequency, and therefore, were more irreplaceable in potential conservation solutions. This means that it is important to carefully consider which planning units within the focal areas and connectivity zones are included in potential conservation plans, as selecting suboptimal planning units in these areas would likely have large effects on the cost of the solution to the agricultural industry. Areas outside the focal areas and connectivity zones in this scenario were more replaceable. High replaceability indicates that while conserving land outside of the focal areas and connectivity zones is still important to achieving lesser prairie-chicken population goals set under the Range-wide Plan, there is more flexibility in which specific planning units can be selected in regards to potential effects on the agricultural industry. Solutions in this scenario also included more planning units than in the other two scenarios, meaning it took more land to efficiently meet the conservation targets.

Similarly to the focal area and connectivity zone strategy, selected planning units in the most efficient solutions with the modeled habitat conservation scenario (Scenario 2) included a large number of planning units in the focal areas. Selected planning units in the focal areas in this scenario were also more irreplaceable, while planning units outside of the focal areas were more replaceable. High irreplaceability indicates that if lesser prairie-chicken conservation efforts focused on prioritizing areas of current potential habitat instead of taking the focal areas and connectivity zones approach, many of the same focal areas would need to be prioritized to ensure an efficient solution with low cost to the agricultural industry. Irreplaceability was lower outside of the focal areas, meaning that while these planning units are still important to meeting conservation goals, there is more flexibility in which of these planning units can be selected while maintaining the lowest effects on the agricultural industry.

The lek-centric conservation scenario (Scenario 3) resulted in very different solutions from the other two scenarios. Selected planning units in the most efficient solutions contained few planning units in the focal areas and connectivity zones. The majority of planning units were located in areas identified by prior models as currently not lesser prairie-chicken habitat. Through conservation practices, these areas could be converted into lesser prairie-chicken habitat, though changing the landscape could involve implementation of strict conservation practices. Differences between the lek-centric conservation scenario (Scenario 3) and the other scenarios are a result of differences in conservation strategy. While some consideration of existing populations of lesser prairie-chickens is taken into consideration in the delineation of focal areas, other criteria including existing habitat conditions, amount of existing fragmentation, presence of selected ecological sites, location of public and conservation lands, extent of conflicting demands for alternative land uses, and other local biologist knowledge are also considered (Van Pelt et al. 2013). As a result, Scenarios 1 and 2 are based on the strategy of focusing on areas with existing habitat for lesser prairie-chickens, while

Scenario 3 is based on the strategy of focusing on known lek locations and adding habitat accordingly.

Differences between scenarios were most pronounced in the Short-Grass Prairie/CRP Mosaic and Mixed-Grass Prairie ecoregions. In these ecoregions, selection of focal areas and connectivity zones had greater effects on the agricultural industry than in other areas, leading to different planning units selected in the lek-centric scenario, where the model was not constraining selections to these priority areas to meet conservation targets.

Scenario Similarities

While there were differences between the scenario solutions, there were some patches of planning units highly selected in all three scenarios. Particularly interesting are the planning units highly selected in all three scenarios not occurring in the focal areas and connectivity zones. Such patches of planning units occurred in Wichita, Pawnee, Rush, and Ellis counties, for example. These patches may be particularly valuable areas for lesser prairie-chickens, as conservation in these areas would have relatively low economic effect on the agricultural industry. Incorporating spatial heterogeneity of costs associated with conservation planning improves efficiency and effectiveness of conservation plans, and oftentimes is as important as incorporating spatial heterogeneity of environmental benefits (Polasky et al. 2001, Ferraro 2003, Naidoo et al. 2006). In my Marxan models, incorporating costs to the agricultural industry not only improved the efficiency of conservation plans, but also helped identify areas of conservation importance that were not identified when ecological factors were alone considered. Without the use of decision support tools to evaluate economic effects, these areas may have been missed in conservation plans.

Connectivity

I found high levels of connectivity between habitat patches selected in the most efficient solutions in each of the three scenarios. The overall high connectivity in all scenarios indicates that the landscape in the lesser prairie-chicken habitat is relatively permeable, even through areas in the system that are dominated by agriculture. I used the Human Footprint Index as an index of resistance to lesser prairie-chicken movement (Venter et al. 2015, 2016). The Human Footprint Index quantifies the human effect on the environment, and considers human population density, railroads, major roads, rivers, urban areas, and agriculture (Venter et al. 2015, 2016). While lesser prairie-chicken movement is likely influenced by some factors not included in this index, negative effects of roads, anthropogenic structures, and large amounts of croplands to lesser prairie-chickens and the species' movement have been documented (Hagan et al. 2004, Pruett et al. 2009, Hagan et al. 2016, Spencer et al. 2017). Marxan models are generally sufficient in clumping selected planning units together by penalizing solutions with high boundary lengths. However, one major criticism of Marxan is that it often fails to adequately provide functional connectivity between clumps of selected planning units (Ardron et al. 2010). By incorporating a *post-hoc* connectivity model, I was able to test potential connectivity between selected habitat patches. Demonstrating high connectivity potential in the solutions for all three scenarios is an important first step to showing that these solutions are viable ecologically. Maintaining structural connectivity of habitat through the shape, size, and location of habitat patches is important to maintaining the functional connectivity of a population, consisting of biological factors like behavior and patterns of gene flow (Brooks 2003). Despite shifts in range size and declines in abundance, lesser prairie-chickens have retained relatively high levels of neutral genetic diversity (Van den Bussche et al. 2003, Bouzat and Johnson 2004, Hagen et al. 2010, Corman 2011, Pruett et al. 2011). However, if current trends continue, fragmentation and isolation of populations will become a threat to lesser prairie-chicken genetic diversity (DeYoung and Williford 2016). It is therefore important to consider connectivity among habitat patches across the three ecoregions when managing for the lesser prairie-chicken in Kansas.

CONCLUSION

The Lesser Prairie-Chicken Interstate Working Group identified ecologically important areas to prioritize conservation focus. However, no decision support tool has been used to aid in selection of areas for conservation focus within these ecologically important areas, or attempt to minimize economic effect of lesser prairie-chicken conservation on the agricultural industry. I developed such a tool using the decision support software Marxan, and used this tool to evaluate tradeoffs between scenarios involving different conservation strategies. I found that these different conservation strategies did result in different optimal solutions, though some areas were highly selected in all scenarios. I also found the landscape in the lesser prairie-chicken range to be relatively permeable among habitat patches selected for conservation focus in my models, which is important ecologically for these areas of potential conservation focus.

An important aspect of conservation planning is evaluating tradeoffs between alternative conservation strategies. In the case of the lesser prairie-chicken, it may be important for managers to evaluate tradeoffs between prioritizing previously identified ecologically important areas and potential economic effects of conservation to the agricultural industry. In some cases, establishing quality habitat in known areas of ecological importance may be the most important objective, and mitigating the economic effects of conservation may play a secondary role. In other cases, quality habitat could be established in the areas of lowest economic importance, thus limiting effects on agricultural producers. My results help provide a tool for managers to evaluate tradeoffs between ecological and economic objectives in lesser prairie-chicken conservation. The use of such decision support tools are important for managers to create optimal conservation plans that effectively maximize conservation potential while minimizing economic effects to the region.

TABLES

Table 2.1 Descriptions of each lesser prairie-chicken habitat ecoregion found in Kansas (KS), including the total area of each ecoregion across the entire lesser prairie-chicken range, the area in Kansas of each ecoregion, and the percent of each ecoregion's total area across the lesser prairie-chicken range found in Kansas. Also shown is the total lesser prairie-chicken population goal for each ecoregion from the Lesser Prairie-Chicken Range-wide Plan (Van Pelt et al. 2013), as well as the Kansas population goal for each ecoregion used to create conservation targets for Marxan analysis.

Ecoregion	Total area (km^2)	Area in KS (km^2)	% in KS	Total Pop. Goal	KS Pop. Goal
Mixed-Grass Prairie	51225	16577	32%	24000	7767
Sand Sagebrush Prairie	32516	15916	49%	10000	4895
Short-Grass Prairie/CRP Mosaic	34978	34978	100%	25000	25000

Table 2.2 Total number of planning units selected in each Marxan scenario, as well as the number and percent of planning units selected in areas with each Crucial Habitat Assessment Tool (CHAT) score (1-4) (SGP CHAT 2013). Scenario 1 is the focal area and connectivity zone approach, Scenario 2 the modeled habitat approach, and Scenario 3 the lek-centric approach. Areas with a CHAT score of 1 are the focal areas, CHAT score of 2 are the connectivity zones, CHAT score of 3 are modeled habitat not found in the focal areas and connectivity zones, and CHAT score of 4 are areas modeled as current non-habitat.

Scenario	Selected Planning Units	CHAT 1	CHAT 2	CHAT 3	CHAT 4
1	5345	4135 (77%)	309 (6%)	798 (15%)	103 (2%)
2	4633	2527 (55%)	389 (8%)	1614 (35%)	101 (2%)
3	4498	736 (16%)	125 (3%)	593 (13%)	3044 (68%)

FIGURES

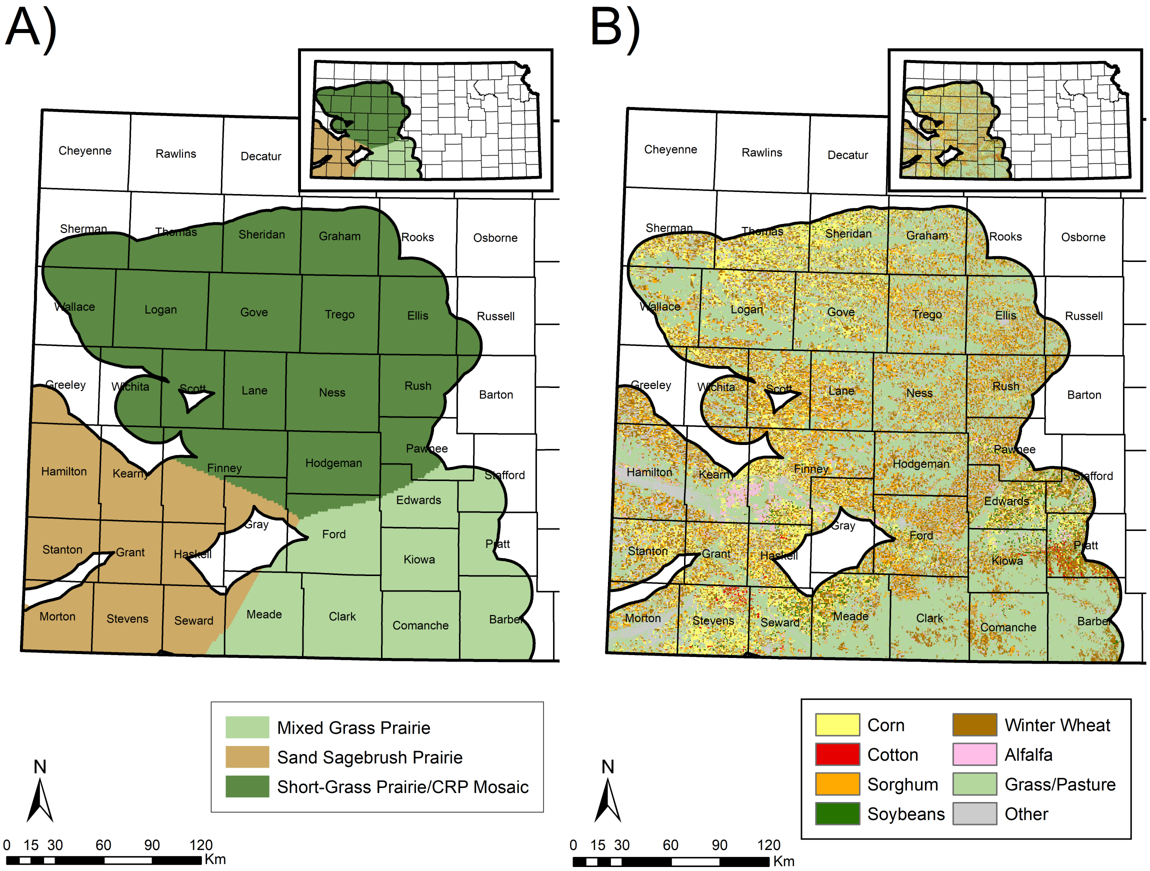

Figure 2.1 Map of the lesser prairie-chicken range in Kansas showing the distribution of A) the three habitat ecoregion types and B) different agricultural land uses (USDA et al. 2018).

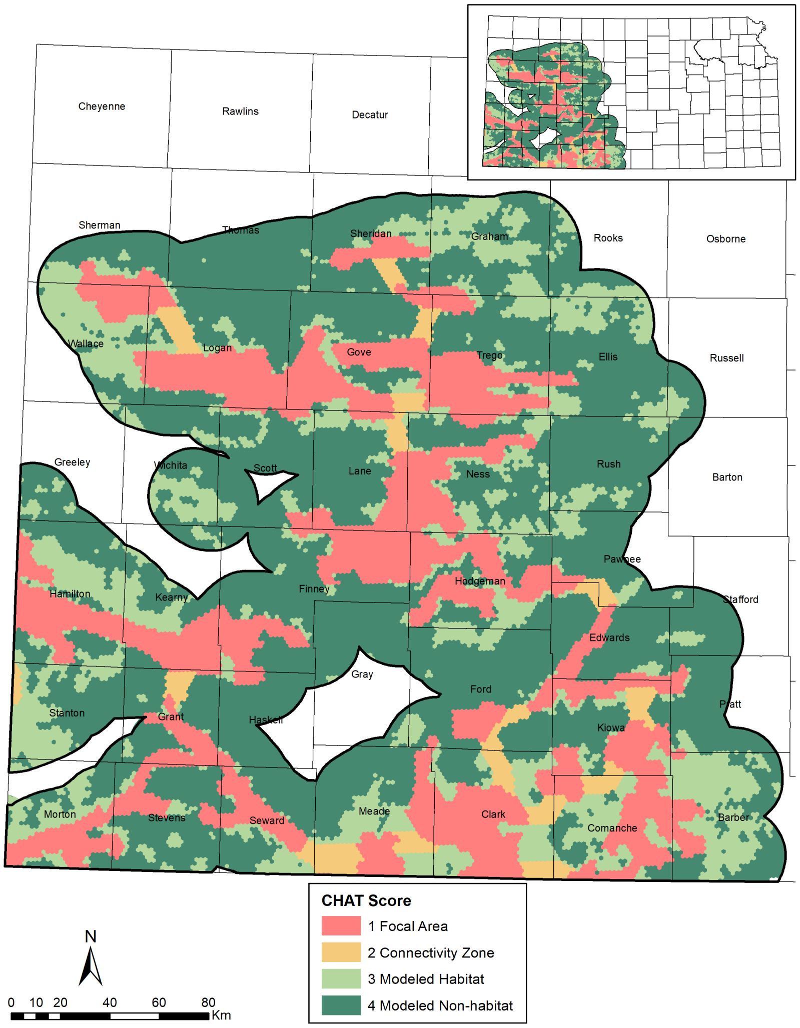

Figure 2.2 Map of the lesser prairie-chicken range in Kansas showing the distribution of land with each Crucial Habitat Assessment Tool score (SGP CHAT 2013).

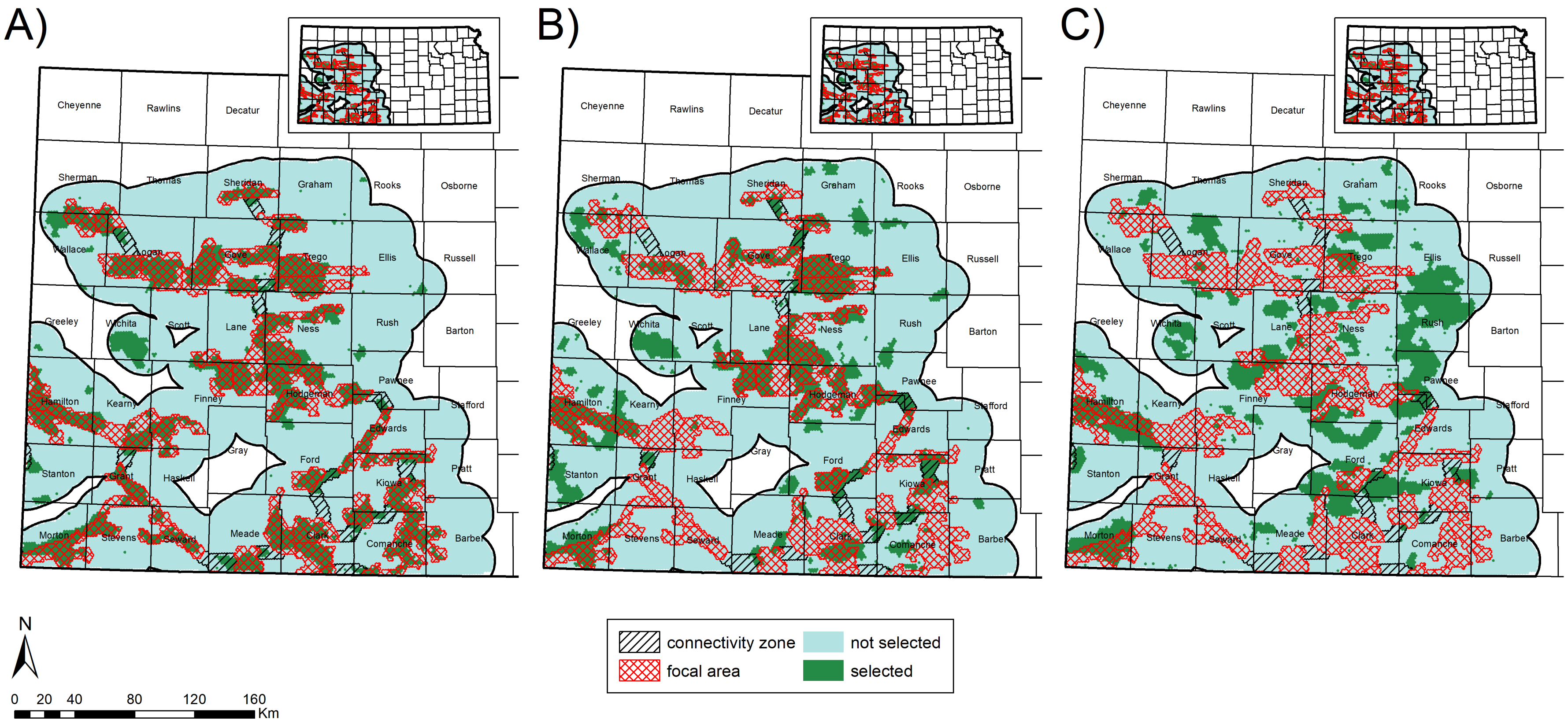

Figure 2.3 Maps of the selected planning units in the "best" (i.e., lowest score calculated by the Marxan objective function) solution from Marxan analyses for A) Scenario 1 (focal area and connectivity zone approach) B) Scenario 2 (modeled habitat approach), and C) Scenario 3 (lek-centric approach). Also shown are the focal areas and connectivity zones identified as part of the Lesser Prairie-Chicken Range-wide Plan (Van Pelt et al. 2013).

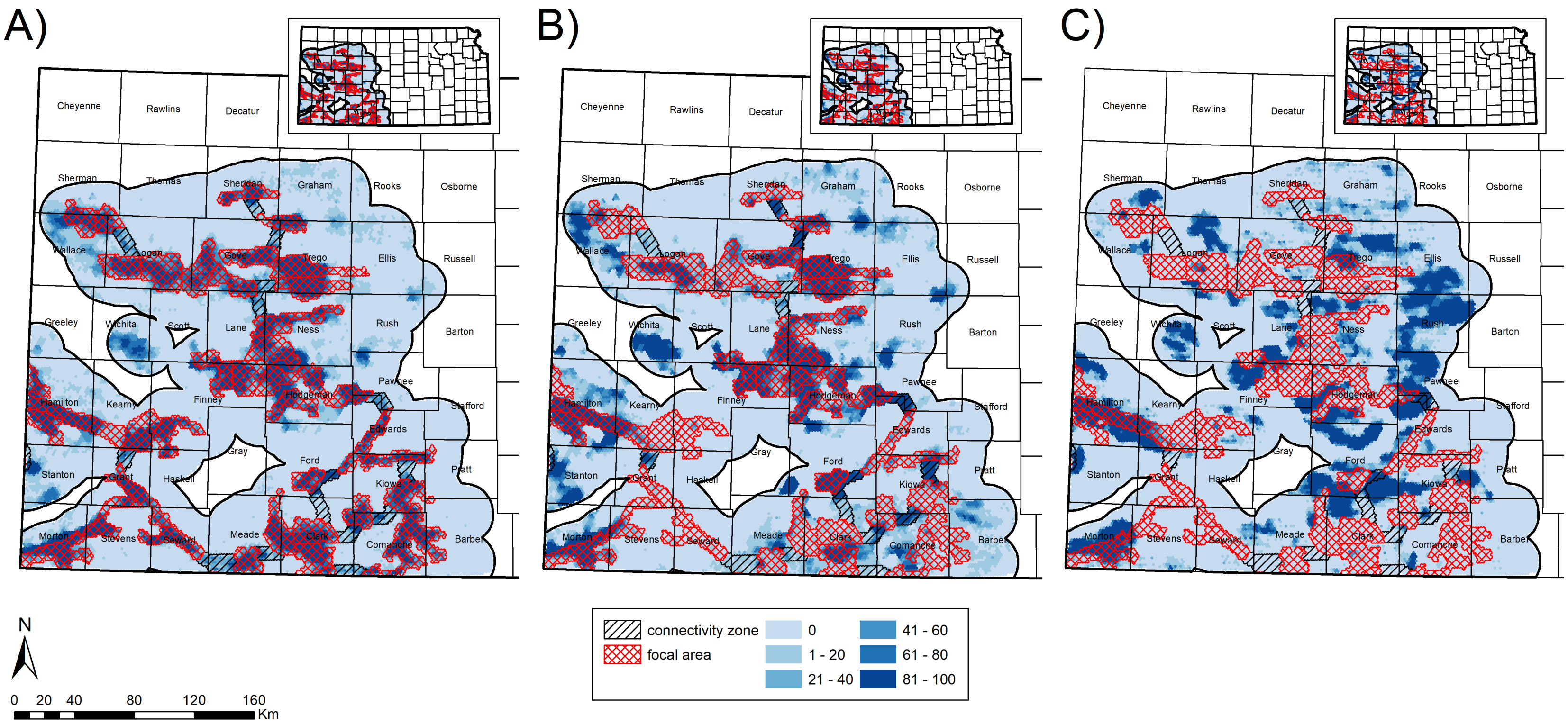

Figure 2.4 Maps of the selection frequency of planning units across all solutions from Marxan analyses for A) Scenario 1 (focal area and connectivity zone approach), B) Scenario 2 (modeled habitat approach), and C) Scenario 3 (lek-centric approach). Also shown are the focal areas and connectivity zones identified as part of the Lesser Prairie-Chicken Range-wide Plan (Van Pelt et al. 2013).

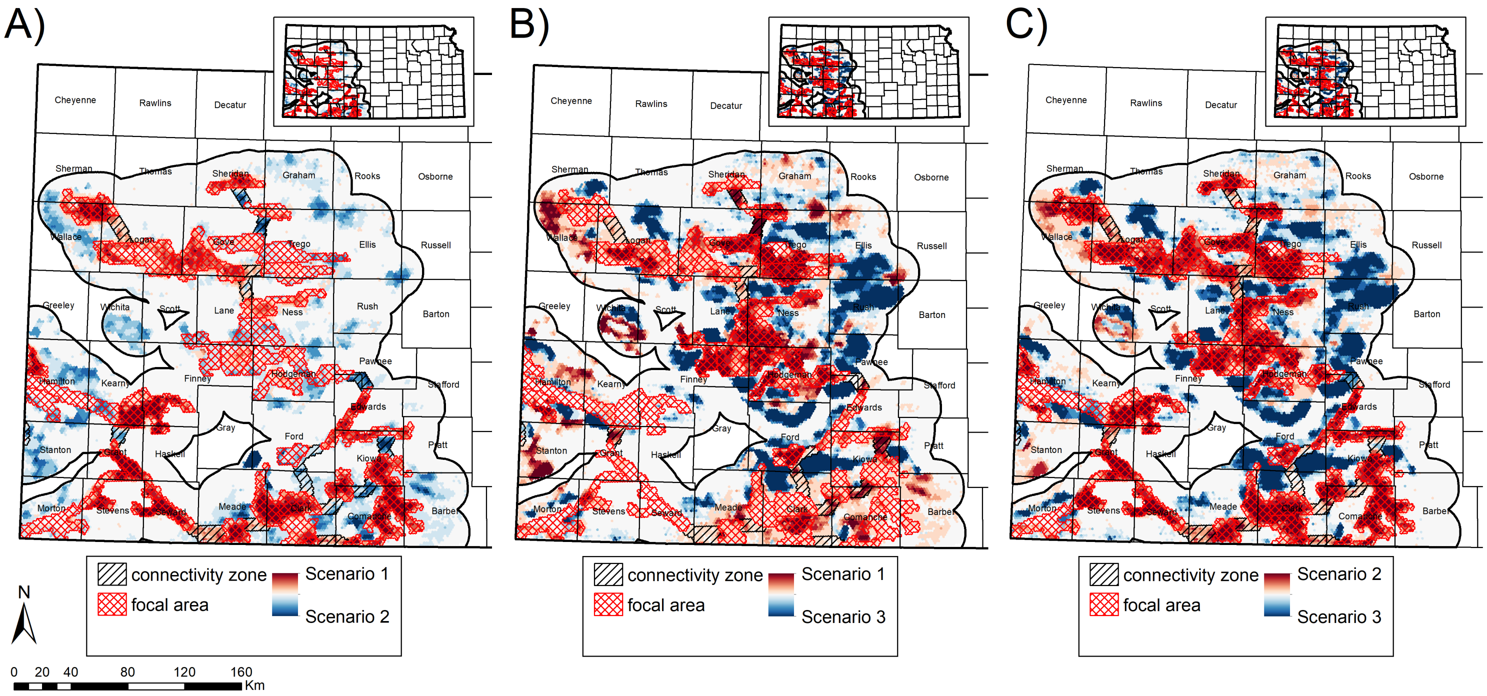

Figure 2.5 Maps of the difference in selection frequency of planning units across all solutions from Marxan analyses between A) Scenarios 1 (focal area and connectivity zone approach) and 2 (modeled habitat approach), B) Scenarios 1 (focal area and connectivity zone approach) and 3 (lek-centric approach), and C) Scenarios 2 (modeled habitat approach) and 3 (lek-centric approach). Also shown are the focal areas and connectivity zones identified as part of the Lesser Prairie-Chicken Range-wide Plan (Van Pelt et al. 2013).

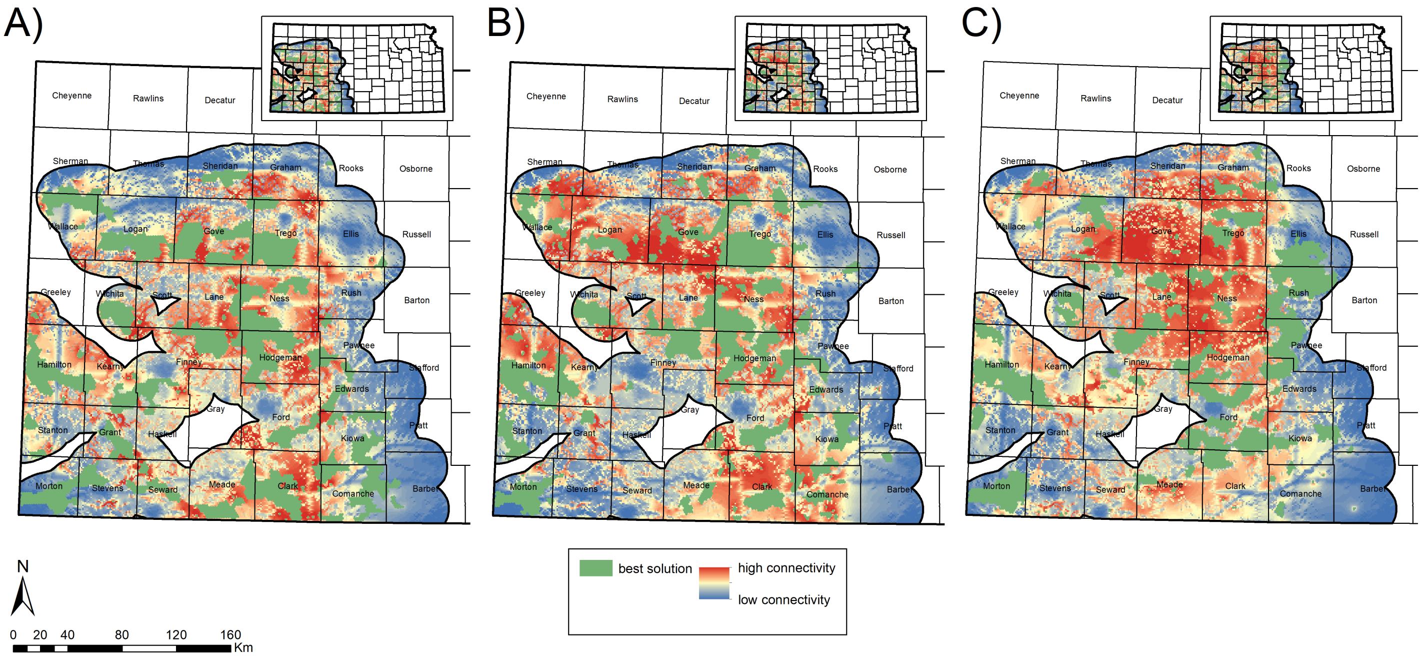

Figure 2.6 Results of the connectivity analyses from Circuitscape (McRae et al. 2013) models for A) Scenario 1 (focal area and connectivity zone approach), B) Scenario 2 (modeled habitat approach), and C) Scenario 3 (lek-centric approach). Also shown are the selected planning units in the "best" (i.e., lowest score calculated by the Marxan objective function) solution from Marxan analyses for each scenario. Red areas indicate high levels of current, and therefore high connectivity in the models, while blue areas indicate low current, and therefore low connectivity.

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