EVALUATING HABITAT SUITABILITY FOR LESSER PRAIRIE-CHICKEN CONSERVATION IN THE MIXED-GRASS PRAIRIE ECOREGION

by

Morgan Jean Solomon

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ABSTRACT

Populations of lesser prairie-chickens (*Tympanuchus pallidicinctus*; hereafter "prairie-chicken") in the southern Great Plains have declined by an estimated 85% and the species is currently being reconsidered for protections under the federal Endangered Species Act. Despite efforts to increase the quantity, quality, and connectivity of available habitat, prairie-chicken populations in the mixed-grass prairie ecoregion have remained relatively stable-to-declining. To provide information that will assist in providing more appropriate qualifications of available prairie-chicken habitat, I used ensemble modeling approaches and a least-cost path analysis to develop spatially-explicit predictions of prairie-chicken habitat and assess connectivity of identified habitat within the mixed-grass prairie ecoregion. In addition, I provided a critical comparison of the Western Association of Fish & Wildlife Agencies (WAFWA) Habitat Evaluation Guide and research-based field indices used to quantify the amount and quality of habitat for prairie-chicken conservation on a proprety participating in an incentive-based conservation program. I also explored the potential for using ecological site descriptions and relative condition (similarity index) to monitor reproductive habitat for prairie-chickens. Predictions from our ensembled model identified ~4,576 km^2 of potentially suitable prairie-chicken habitat both occupied and unoccupied. Least-cost path analyses revealed a low degree of connectivity between areas of occupied and unoccupied habitat indicating a low probability of natural recolonization. Managers should consider focusing conservation efforts on targeting habitat restoration between, within and around areas of identified occupied and unoccupied habitat. Habitat quality under the HEG habitat assessment protocol showed the property had excellent prairie-chicken habitat quality while research-based estimates showed the property only had marginal habitat quality for prairie-chickens. Differences in habitat quality assessments were in areas that had low percent cover of vegetation species preferred by prairie-chickens and in areas that had recently experienced fire. Thus, managers should consider using components of both habitat assessments protocols when quantifying habitat for prairie-chicken conservation to reduce the probability of producing erroneous estimates of habitat quality. Limited sample size within moderate categories of similarity index across ecological sites prevented us from reliably executing further analyses exploring the utility of using a similarity index as a tool for monitoring prairie-chicken habitat.

CHAPTER ONE

INTRODUCTION AND LITERATURE REVIEW

Rangelands of the southern Great Plains have seen significant degradation since the arrival of Euro-American settlers in the early 1800's. Once structurally diverse and contiguous, rangelands have been lost or fragmented into smaller isolated patches within a matrix of croplands, exurban development, and energy development (Fuhlendorf et al. 2002, Samson et al. 2004, Hagen et al. 2011). Additionally, loss of ecological drivers has degraded rangelands further. Fire suppression and the replacement of free-roaming bison with intensively managed livestock has led to woody encroachment and a reduction in heterogeneity in vegetation composition and structure, which in turn has decreased wildlife diversity (Engle et al. 2008, Fuhlendorf et al 2009). For example, significant declines in grass- and shrubland bird populations have coincided with increased rangeland loss and fragmentation (Coppedge et al. 2001). In particular, prairie grouse (*Tympanuchus spp*.) have been highly sensitive to changes in their environment, having overall negative trends in population sizes since the 1960's, which have corresponded to the intensification of farming and ranching practices, the expansion of trees, exurban and energy development, and persistent droughts (Garton et al. 2011, Garton et al. 2016).

The lesser prairie-chicken (*Tympanuchus pallidicinctus*; hereafter "prairie-chicken") is a species of conservation concern in the southern Great Plains. Prior to European settlement, prairie-chickens occupied large swaths of rangeland habitat across western Kansas and Oklahoma, eastern Colorado and New Mexico, and north-central Texas with range-wide populations speculated to be as high as 2 million birds (Hagen et al. 2004, Garton et al. 2016). However, large-scale changes in landscape composition and land use following increased human settlement in the early 1900's greatly reduced the amount and connectivity of available habitat and constrained the species' range to four isolated ecoregions: shinnery oak, sand sagebrush, short-grass/CRP, and the mixed-grass prairie. Droughts have also had a profound effect on prairie-chickens with range-wide populations declining by ~45% following a severe drought between the years 2011–2013 (McDonald et al. 2014, Garton et al. 2016). Concern for the species led the U.S. Fish and Wildlife Service (USFWS) to list the prairie-chicken under the Endangered Species Act (ESA) as threatened in 2014 (USFWS 2014); however, listing was quickly overturned by a judicial decision in September 2015 (USFWS 2016). Estimates in 2021 show range-wide populations have recovered to pre-2011 numbers (Nasman et al. 2021); however, increases in range-wide populations are likely due to increases in populations in the short-grass/CRP ecoregion only. Populations in all other ecoregions, including the mixed-grass prairie, have remained relatively stable or have declined (Nasman et al. 2021). The USFWS has proposed to relist the prairie-chicken under the ESA as endangered in the shinnery oak ecoregion and threatened in all other ecoregions (USFWS 2021).

Conservation and Population Status in the Mixed-grass Prairie Ecoregion

Populations of prairie-chickens in the southern mixed-grass prairie ecoregion are located in southcentral Kansas, western Oklahoma, and northern Texas (McDonald et al. 2014). The mixed-grass prairie ecoregion is at the geographical center of the extant distribution of prairie-chickens and historically is thought to have held the highest density of birds (Van Pelt et al. 2013, McDonald et al. 2014, Wolfe et al. 2016) indicating high potential for population increases in the region. In addition, the southern mixed-grass prairie ecoregion includes a relatively large amount of grassland and potential prairie-chicken habitat (~40,280 km^2; LANDFIRE 2020). Thus, developing methods to identify management actions that increase prairie-chicken populations in the mixed-grass prairie could be critical to the species' overall recovery (Wolfe et al. 2016). Recent population analyses indicate populations in the mixed-grass prairie have declined by 1–2.3% annually during 2005–2020 (Garton et al. 2016, Hagen et al. 2017, Nasman et al. 2021). Local populations showed signs of recovery following the drought in 2011 – 2013; however, recent megafires in 2016 and 2017 burned over 4,100 km^2 (~ 1 million acres) of grass- and shrublands in the mixed-grass prairie, temporarily decreasing the amount of available habitat and reducing local populations (Parker 2021). In addition, projections of long-term persistence in the mixed-grass prairie ecoregion are pessimistic due to ongoing declines in carrying capacity (Garton et al. 2016, Hagen et al. 2017) resulting from continued habitat loss to cultivation (Woodward et al. 2001), energy development (Hagen et al. 2011, Plumb et al. 2019), unsuitable vegetation structure (Knopf 1994, Kraft et al. 2021), and the expansion of eastern red cedar (*Juniperus virginiana*; Lautenbach et al. 2017).

Habitat Relationships

Prairie-chicken populations are constrained by the amount, connectivity, and quality of rangeland habitats. Like all animals, prairie-chickens select habitat at multiple spatial scales including the selection of home ranges at the landscape-scale and selection of nesting and foraging sites at the fine scale (Fuhlendorf et al. 2002). General habitat requirements for prairie-chickens in the mixed-grass ecoregion include relatively large tracts of intact grassland free of tall, vertical features that include a diverse array of vegetation conditions to fulfill different life stages such as lekking, nesting, and brood-rearing (Haukos and Zavaleta 2016).

Landscape-scale Habitat Relationships. At the landscape scale, habitat selection and survival of prairie-chickens are largely driven by vegetation cover and land use (Woodward et al. 2001, Fuhlendorf et al. 2002, Plumb et al. 2019). While prairie-chickens have been reported to use cropland as foraging sites during the nonbreeding period (Ahlborn 1980, Jamison 2000), the conversion of large areas of rangeland habitat to cropland has displaced prairie-chickens and reduced regional populations (Woodward et al. 2001). In addition, recent research reported prairie-chickens were 40 times more likely to select for habitats with tree densities of 0 trees per hectare than habitats with 5 trees per hectare (Lautenbach et al. 2017). The avoidance of trees is likely caused by a perceived increase in predation risk or a general propensity for prairie-chickens to avoid tall landscape features (Hagen et al. 2019).

Infrastructure associated with energy and exurban development has also been shown to impact prairie-chicken habitat use and survival. While one study indicated wind turbines had no negative effects on prairie-chicken habitat selection or demographics (LeBeau et al. 2020), research in the mixed-grass prairie found prairie-chickens avoided pumpjacks, oil wells, highways, buildings, and powerlines (Pitman et al. 2005, Hagen et al. 2011, Plumb et al. 2019). However, the degree to which prairie-chickens avoid these features likely depends on the spatial configuration, density of features, and the level of activity associated with areas of energy and exurban development (Winder et al. 2014, Lloyd et al. 2022).

Fine-scale Habitat Relationships. Sensitivity analyses indicate that variation in nest and brood survival are the most crucial factors influencing fluctuations in established prairie-chicken populations (Hagen et al. 2009); therefore, managing for fine-scale vegetation conditions that increase reproductive success can improve prairie-chicken population viability (Starns et al. 2020, Lautenbach et al. 2021). The reproductive season (breeding through brood independence) for prairie-chickens occurs during March–September and is divided into distinct periods (lekking, nesting, and brood-rearing) with each period requiring a different subset of habitat conditions (Hagen and Giesen 2005).

During the lekking period males congregate at communal display grounds to compete for females who are surveying for a potential breeding partner. Lek locations are often located in areas with higher elevation and increased bare ground to increase visibility for detection of potential predators and visiting females (Haukos and Smith 1999, Boal and Haukos 2016). Stable lek locations (e.g., leks that have had consistent male attendance for three or more years or have ≥10 birds within an individual year) are the focal point of prairie-chicken life-histories with most prairie-chicken activity, including nesting and brood-rearing, occurring within 5 km of a lek (Applegate and Riley 1998, Winder et al. 2014). Thus, the formation and persistence of lek locations are likely concomitant with the availability and quality of nesting and brood-rearing habitat (Aulicky 2020). In fact, recent research has found that established lek locations have greater proportions (≥25%) of fine-scale nesting and brood-rearing habitat than would be expected at random locations (Gehrt et al 2020)

Nest sites are generally located within 3 km of a lek (Hagen et al. 2013) and are often associated with local vegetation conditions that provide concealment from predators and less variable thermal conditions (Giesen 1994, Grisham et al. 2016, Lautenbach et al. 2019). Species composition at nest sites varies depending on regional differences in available vegetation but is typically made up of shrub and grass species that provide the greatest concealment (Larsson et al. 2013, Lautenbach et al. 2019). Vertical obstruction reading (VOR), a measure of nest concealment, is a consistent correlate of nest site selection and survival across studies (Pitman et al. 2005, Pitman et al. 2006, Hagen et al. 2013, Grisham et al. 2014, Lautenbach et al. 2019). Nest-site selection is often optimized at intermediate measures of VOR (1.5 – 3.0 dm; Pitman et al. 2005, Hagen et al. 2013, Grisham et al. 2014, Lautenbach et al. 2019), presumably due to a life-history tradeoff between nest success and female survival. While denser cover enhances nest concealment, it decreases the ability of the female to escape predation (Wiebe and Martin 1998, McNew et al. 2013). In addition, increased grass cover and decreased bare ground (≤10%) positively influences both nest site selection and nest survival (Hagen et al. 2013, Lautenbach et al. 2019).

Quality brood-rearing habitat is characterized by a mixture of vegetation conditions that provide concealment from predators and increased access to critical food sources (Jamison et al. 2002, Hagen et al 2005, Fields et al. 2006, Lautenbach 2015). Brood-rearing often occurs in habitats with moderate grass and shrub cover (VOR: 2.0 – 5.0 dm) and greater percent bare ground, which facilitates chick movement and predator escape (Hagen et al. 2005, Lautenbach 2015, Haukos and Zavaleta 2016). Female selection of brood-rearing areas include a greater forb cover (7% – 35%; Lautenbach 2015) which is positively correlated to increased invertebrate densities (Hagen et al. 2005, Hagen et al. 2013). Forbs and invertebrates are chief food sources for broods and increased densities of both have been linked to higher chick survival (Hagen et al 2005, Sullins et al. 2018a).

Conservation Efforts

Extensive research identifying habitat conditions that support prairie-chicken populations has been critical to developing management solutions for prairie-chicken conservation (Fuhlendorf et al. 2000, Larsson et al. 2013, Hagen et al. 2013, Jarnevich et al. 2016, Lautenbach et al. 2017, Spencer et al. 2017, Sullins et al. 2018b, Gulick 2019, Lautenbach et al. 2019, Plumb et al. 2019, Sullins et al. 2019, Kraft et al. 2021). In addition, because 94% of the prairie-chicken distribution occurs on private lands, conservation initiatives with strong partnerships between private landowners and resource managers has been essential to increasing prairie-chicken habitat. For example, the Lesser Prairie-Chicken Initiative (LPCI) administered by the USDA

Natural Resources Conservation Service (NRCS) uses current Farm Bill conservation programs to provide financial and technical assistance to landowners for implementing conservation practices to improve prairie-chicken habitat (USDA 2016). The five states in the prairie-chickens range (Kansas, Colorado, Oklahoma, Texas, New Mexico) developed the Lesser Prairie-Chicken Range-wide Conservation Plan (RWP) to provide biological goals for the implementation of conservation efforts to improve prairie-chicken habitat (Van Pelt et al. 2013). Additionally, the RWP includes a voluntary mitigation framework for development in the prairie-chicken range; this framework is administered by the Western Association of Fish and Wildlife Agencies (WAFWA). Within the mitigation framework, WAFWA uses mitigation funds paid by oil, gas, wind, electricity and telecommunications industries to incentivize landowners to implement land management practices that improve prairie-chicken habitat (Van Pelt et al. 2013). These practices include cropland to grassland restoration, heterogeneity-based prescribed grazing and fire management, planting native grasses and forbs, and brush management (Van Pelt et al. 2013, NRCS 2020).

Identifying and Evaluating Habitat for Prairie-chicken Conservation

Efforts to identify and quantify habitat for prairie-chicken conservation have generally followed a hierarchical process. As a first step, managers identify existing habitat at a landscape-scale (i.e., within a regional area of interest) using mapping applications and habitat modeling techniques (Niemuth 2011). Once potential habitat has been identified, managers identify targeted management actions to improve and conserve habitat at the local scale (Jarnevich et al. 2016, Sullins et al. 2019). Finally, management strategies are applied, and monitoring efforts track the quality of fine-scale vegetation conditions and the relative success of conservation practices overtime using field-based indices developed by species experts (Van Pelt et al. 2013).

Landscape-scale Evaluation Tools for Measuring Lesser Prairie-Chicken Habitat. Habitat models combined with GIS have played a large role in identifying habitat for species conservation and are commonly used to guide management decisions (Zeigenfuss et al. 2000, Clevenger et al. 2002, Jarnevich et al. 2016). Habitat models relating prairie grouse occurrence to landscape-scale habitat conditions have been based on lek location data as leks play a central role in population monitoring and persistence (Davis et al. 2008, Garton et al. 2011, Doherty et al. 2018). For example, Jarnevich et al. (2016) developed methods to predict prairie-chicken habitat by comparing habitat conditions at leks versus a set of pseudo-random points using machine-learning niche modeling. While cartographical depictions of relative habitat use are a useful tool in wildlife management, they can also provide a false sense of certainty when making management decisions because predictions and the associated error depends on the model structure and the data used. As such, predictions derived across multiple models can be highly variable and depict different delineations of potential habitat which may make it difficult to accurately prescribe management actions for species conservation (Lawler et al. 2006, Pearson et al. 2006). One approach to accommodate this uncertainty are ensemble modeling approaches where multiple discrete and independent models are developed and predictions from each are combined into one averaged prediction; the result is more robust predictions of habitat suitability (Araújo and New 2007, Marmion et al. 2009, Kotu and Deshpande 2014).

Field-based Indices for Monitoring Lesser Prairie-chicken Habitat. In general, field-based indices used to monitor and quantify available prairie-chicken habitat have included a local evaluation of the amount of potential habitat available combined with a fine-scale assessment of vegetation conditions to quantify the quality of reproductive habitat (Morrison et al. 2013, Van Pelt et al. 2013, McNew et al. 2017, Gehrt et al. 2020). For example, as part of the RWP, WAFWA developed the Habitat Evaluation Guide (HEG) to quantify prairie-chicken habitat on potential and enrolled private lands. Landowners who are enrolled in WAFWA's mitigation program are incentivized to manage for prairie-chicken habitat because their annual payment is based on the total acreage enrolled and the HEG scores of the property, where a property's HEG scores are based on four habitat variables known to be associated with prairie-chicken reproductive success: vegetation cover (non-overlapping canopy cover), vegetation composition, percent cover of tall woody plants, and availability of potential habitat in the surrounding area (Van Pelt et al. 2013). Akin to habitat suitability indices, HEG habitat variables and their respective scoring classifications are predetermined and qualitatively developed by prairie-chicken experts. HSI models are commonly used to make conservation decisions and are an efficient and often robust tool used for coarsely quantifying species habitat quality (USFWS 1981).

Concurrent to the development of qualitative HEG criteria, recent field-based research has focused on quantifying conditions that describe prairie-chicken habitat quality in relation to measures of use and demography (Hagen et al. 2013, Larsson et al 2013, Lautenbach 2015, Jarnevich et al. 2016, Spencer et al. 2017, Gulick 2019, Plumb et al. 2019, Sullins et al. 2019, Kraft et al. 2021), which may then be used to identify more resolute habitat targets for prioritizing prairie-chicken conservation. For example, recent research quantifying the abundance of reproductive habitat used criteria developed by Lautenbach (2015) to classify randomly selected points within 5 km of lek locations as having sufficient vegetation conditions for successful nesting and brood-rearing (Lautenbach et al. 2019, Gehrt et al. 2020) and reported areas that support stable populations have ~25% suitable nesting and brood-rearing habitat. Criteria used to classify random points as suitable reproductive habitat included VOR and cover of bare ground during the nesting period and VOR and forb cover during the brood-rearing period (Gehrt et al. 2020).

Another tool potentially beneficial to monitoring prairie-chicken reproductive habitat in the field are ecological site descriptions. Ecological sites, an already commonly used component in rangeland monitoring and management (Herrick et al. 2006), may have application in monitoring reproductive habitat for prairie-chickens. Ecological site descriptions describe the climate, soil, hydrology, vegetative dynamics, and historical plant community of an area (NRCS 2003), all of which can constrain prairie-chicken use and vital rates (Fuhlendorf and Engle 2001, Van Pelt et al. 2013, Grisham et al. 2016, Kraft 2016). While researchers have ranked ecological sites in terms of their capacity to support reproductive habitat for prairie-chickens (Van Pelt et al. 2013), limited research has investigated the associations among ecological sites, relative condition, and important prairie-chicken population measures (but see Anderson et al. 2015, Kraft 2016) and no research has linked these rangeland indicators to measures of the amount, type, and quality of prairie-chicken habitat.

Research Need

Despite gains in range-wide populations, recent efforts aimed at increasing the quantity and quality of prairie-chicken habitat within the mixed-grass prairie ecoregion has resulted in little change in population size. Thus, efforts to expand on current methods for identifying prairie-chicken habitat are needed to further conservation efforts and increase the potential for prairie-chicken persistence. In addition, no research has evaluated whether current field-based indices for quantifying vegetation conditions and monitoring prairie-chicken habitat provide appropriate estimates of habitat quality. As prairie-chicken populations are constrained by the amount and quality of available habitat, ensuring methods for identifying and monitoring habitat at multiple scales are providing the best possible estimates of habitat quality is essential to prairie-chicken conservation.

My goals for this research were to develop robust methods for identifying and prioritizing habitat to improve conservation delivery for prairie-chickens in the mixed-grass prairie ecoregion. In Chapter 2, I developed multiple lek-based relative habitat models within the mixed-grass prairie ecoregion and used ensemble approaches to combine predictions of relative habitat suitability across all models to identify habitat for prairie-chicken conservation at the landscape scale. In Chapter 3, I evaluated current field-based indices for monitoring and quantifying prairie-chicken habitat on private lands.

CHAPTER TWO

LANDSCAPE-BASED EVALUATION OF HABITAT SUITABILITY FOR PRIORITIZING LESSER PRAIRIE-CHICKEN CONSERVATION IN THE MIXED-GRASS PRAIRIE ECOREGION

Introduction

Populations of lesser prairie-chickens (*Tympanuchus pallidicinctus*; hereafter "prairie-chickens") in the southern Great Plains have declined by an estimated 85% (Taylor and Guthery 1980, Garton et al. 2016) and are currently being reconsidered under the federal Endangered Species Act after being removed in response to a judicial decision in September 2015 (USFWS 2016; 2021). Historically, prairie-chickens occurred in large swaths of grass- and shrubland habitat in Texas, New Mexico, Colorado, Kansas and Oklahoma with range-wide populations speculated to be as high as 2 million birds (Taylor and Guthery 1980, Hagen et al. 2004, Garton et al. 2016). However, large-scale changes in landscape composition and land use following increased human settlement in the early 1900s greatly reduced the amount and connectivity of available habitat and constrained the species' range to four disjunct ecoregions: shinnery oak, sand sagebrush, short-grass/CRP, and mixed-grass prairie (Fig. 1; Taylor and Guthery 1980, McDonald et al. 2014). In addition, prolonged and more frequent droughts have had a profound effect on prairie-chickens with range-wide populations declining by ~45% following a severe drought between the years 2011–2013 (McDonald et al. 2014, Garton et al. 2016). Fortunately, recent efforts aimed at restoring prairie-chicken habitat seem to have had a positive effect on the species' overall population. Range-wide population trends suggest an upward trajectory over the past decade (2013–2022); however, despite gains in range-wide populations, efforts to increase prairie-chicken numbers in the mixed-grass prairie ecoregion have resulted in very little change in population size (Nasman et al. 2021).

The mixed-grass prairie ecoregion is at the geographical center of the extant distribution of prairie-chickens and historically is thought to have supported the highest density of birds (Van Pelt et al. 2013, McDonald et al. 2014, Wolfe et al. 2016) indicating high potential for population increases in the region. As such, extensive collaborative efforts have been made by state, federal, and private stakeholders to develop management strategies to increase the number and distribution of prairie-chickens within the area. For example, the Lesser Prairie-chicken Initiative administered by the Natural Resources Conservation Service (NRCS) removed ~ 280 km^2 of conifers encroaching on private lands in the southern Great Plains (USDA 2020). Nevertheless, recent analyses report that prairie-chicken populations in the mixed-grass prairie declined 1–2.3% annually during 2005–2020 (Garton et al. 2016, Hagen et al. 2017, Nasman et al. 2021). Local populations showed signs of recovery following the drought in 2011 – 2013; however, recent megafires in 2016 and 2017 burned over 4,100 km^2 (~ 1 million acres) of grass- and shrublands in the mixed-grass prairie, temporarily decreasing the amount of available habitat and reducing local populations (Parker 2021). Additionally, estimates of long-term persistence in the mixed-grass prairie ecoregion are pessimistic due to projected declines in carrying capacity (Garton et al. 2016, Hagen et al. 2017) resulting from continued habitat loss to cultivation (Woodward et al. 2001), energy development (Hagen et al. 2011, Plumb et al. 2019), and the expansion of eastern red cedar (*Juniperus virginiana*; Lautenbach et al. 2017).

Prairie-chickens are highly sensitive to changes in landscape composition that reduce the amount and connectivity of available habitat (Woodward et al. 2001), and this sensitivity is influenced by several biological traits that affect prairie-chicken space use (Niemuth 2011). For example, prairie-chickens tend to avoid tall features likely due to a perceived increase in predation risk by raptors (Londe et al. 2022). Thus, features such as trees and powerlines can cause barriers to movement and render otherwise suitable habitat unusable and concentrate prairie-chickens into smaller areas (Pitman et al. 2005, Hagen et al. 2011, Plumb et al. 2019, Londe et al. 2022). In addition, prairie-chickens generally disperse no more than ~16 km and mean daily movement averages between 0.5–2 km, making potential areas of high-quality habitat that are separated by non-habitat (e.g., areas of high cropland or tree cover) inaccessible (Boal and Haukos 2016, Earl et al. 2016, Peterson et al. 2020). As such, restoring and conserving large areas of grassland that are free of tall features and connected by patches of habitat that facilitate movement is essential to the long-term sustainability of the prairie-chicken (Samson 1980, DeYoung and Williford 2016, Costanzi et al. 2019).

Developing and expanding methods to identify prairie-chicken habitat and provide information regarding the value of potential habitat restoration activities for increasing connectivity is essential to prairie-chicken recovery. Spatially-explicit habitat models provide a means for relating large-scale habitat conditions to species occurrence and are commonly used to guide management and conservation decisions (Zeigenfuss et al. 2000, Clevenger et al. 2002, Jarnevich et al. 2016). Habitat models relating prairie grouse occurrence to landscape-scale habitat conditions are based on lek location data due to the central role that leks play in population persistence and monitoring (Garton et al. 2011, Jarnevich et al. 2016, Doherty et al. 2018). Because the majority of prairie-chicken habitat use is within 5 km of a lek (Hagen 2003, Pitman et al. 2005, Winder et al. 2017), stable lek locations (e.g., sites with leks that have had consistent male attendance for three or more years or have ≥10 birds within an individual year) are important to prairie-chicken demography because females will generally visit established lek locations rather than newly formed lek locations (Haukos and Smith 1999). Thus, stable leks are likely concomitant with higher quality prairie-chicken habitat than non-stable leks (Gehrt et al. 2020) and can be used to identify landscapes that support stable prairie-chicken populations. For example, Jarnevich et al. (2016) developed methods to predict prairie-chicken habitat by comparing habitat conditions at leks versus a set of pseudo-random points using machine-learning niche models. Results from their study were integrated into the Southern Great Plains Crucial Habitat Assessment Tool (CHAT) and is used to identify potential prairie-chicken habitat (Van Pelt et al. 2013, WAFWA GIS Services 2013). The Southern Great Plains CHAT developed by the Western Association of Fish and Wildlife Agencies (WAFWA) is an online decision support tool that delineates high-priority areas for prairie-chicken habitat conservation and low impact areas for responsible energy development (WAFWA GIS Services 2013; http://sgpchat.org).

Cartographical depictions of relative habitat use are a useful tool in wildlife management; however, they can also provide a false sense of certainty when making management decisions because predictions and the associated error depends on the model structure and the data used. As such, predictions derived across multiple models can be highly variable and depict different delineations of potential habitat which may make it difficult to accurately prescribe management actions for species conservation (Lawler et al. 2006, Pearson et al. 2006). To deal with this uncertainty, researchers have used ensemble approaches where multiple discrete and independent models are developed and predictions from each are combined into one averaged prediction; the result is more robust predictions of habitat suitability (Araújo and New 2007, Marmion et al. 2009, Kotu and Deshpande 2014).

My research intended to expand on previous analyses identifying prairie-chicken habitat by using ensemble approaches and a set of more restrictive criteria to not only identify habitat for prairie-chicken conservation, but to identify habitat conditions associated with stable lek locations. To do this, I developed lek-based relative habitat suitability models within the mixed-grass prairie ecoregion using both resource selection functions (RSF) and Random Forest classification trees and calculated ensembled predictions of relative habitat suitability across all models. For each approach, I developed two predictive models; one based on all known lek locations identified by cooperating state wildlife agencies and a more restrictive model based upon leks classified as ‘stable’. Specifically, my objectives were to 1) use ensemble modeling approaches to develop spatially-explicit predictions of prairie-chicken habitat within the mixed-grass prairie ecoregion, 2) identify areas of potentially suitable, unoccupied habitat for prairie-chicken translocations, 3) use ensemble predictions and least-cost path analyses to assess connectivity of identified unoccupied habitat to current self-sustaining subpopulations, and 4) provide information for prioritizing areas for targeted habitat restoration efforts.

Methods

Study Area

My study area included the mixed-grass prairie ecoregion within southcentral Kansas, northwestern Oklahoma, and the northeastern portion of the Texas panhandle. I buffered the study area out to 16 km which is the average dispersal distance for a prairie-chicken (Fig.1; Van Pelt et al. 2013, Earl et al. 2016, Peterson et al. 2020). Total area within the extent of my analyses was ~66,000 km^2 (~16.3 million acres); ~40,280 km^2 (~10 million acres) of the study area is in grass- or shrubland cover (LANDFIRE 2020). Vegetation in the mixed-grass prairie ecoregion is a mixture of sand sagebrush *(Artemisia filifolia)* and mid-height perennial grasses.

Woody vegetation includes sand plum (*Prunus spp.)*, cottonwood *(Populus deltoides*), and eastern red cedar. Upland soils are typically deep, loamy sands and precipitation ranges between 40–75 cm annually. The primary land use for the area is livestock grazing (USDA, Natural Resource Conservation Service, esis.sc.egov.usda. gov/). In 2021, prairie-chicken populations in the mixed-grass prairie were estimated at ~3,200 birds (Nasman et al. 2021).

<u>Lek Data</u>

I obtained prairie-chicken lek location and survey data for the years 2010–2019 for the mixed-grass prairie ecoregion collected by the Kansas Department of Wildlife and Parks and the Oklahoma Department of Wildlife Conservation (Fig. 2). Lek data from Texas was unavailable due to landowner privacy policies. Landscapes surrounding lek locations (i.e., within 5 km) were assumed to represent habitat conditions that support populations of prairie-chickens. I classified leks into two categories: 1) leks where birds were counted in at least one year during 2015–2019 and 2) leks where birds were only counted in years 2010–2014 and not 2015–2019. I then developed a set of criteria to classify a subset of known lek locations as being ‘stable’, where stable lek locations were those that had persisted (i.e., detected 3 out of 5 consecutive years) or had ≥10 birds within an individual year and were within 2 km of other stable lek locations which is the recommended maximum distance between leks in a complex (Applegate and Riley 1998, Haukos and Zavaleta 2016, Wolfe et al. 2016). In addition, select lek locations were added to the stable lek data after confirming their stability with local researchers (N. Parker and D. Sullins, Kansas State University, personal communication). Leks classified as stable were used to identify habitat conditions associated with lek locations that have persisted. I then generated a set of 20 random points for every lek location (Northrup et al. 2013) within terrestrial areas ≥ 5 km from towns (Hagen et al. 2011, Plumb et al. 2019).

Habitat Data

I collated 25 layers representing habitat conditions known to affect prairie-chicken recruitment, survival, and lek persistence (Table 1). All geospatial layers were imported into ArcGIS Pro (ESRI, Redlands, CA), resampled to a ~30-m resolution, and clipped to the study area (Van Pelt et al. 2013, Earl et al. 2016, Peterson et al. 2020). Because the majority of prairie-chicken activity occurs within 5 km of a lek location (Hagen 2003, Pitman et al. 2005, Winder et al. 2015), I conducted a circular moving-window analyses using tools in ArcGIS Pro to quantify habitat conditions within a 5-km radius of each 30-m cell (ESRI, Redlands, CA). I then used the 'raster' and 'rgdal' package in program R to import spatial files and extract values of habitat covariates for each lek location and random point (Bivand et al. 2013, Hijmans and Etten 2013). To ensure habitat conditions were temporally appropriate, I extracted values of habitat covariates for leks that occurred in only years 2010–2014 and their associated random points from geospatial layers describing habitat conditions in the mixed-grass prairie in 2014; values of habitat covariates for leks that occurred at least once in 2015–2019 were extracted from geospatial layers describing habitat conditions in 2019.

Model Development

Prior to developing my models, I tested for potential spatial autocorrelation among lek locations with a Moran's I test available in the 'dharma' package in program R (Moran 1950, Hartig and Hartig 2017). I then developed two habitat suitability models using resource selection functions (RSF) and Random Forest classification trees to model the effects of habitat conditions within a 5-km radius and predict the relative probability of prairie-chicken lek occurrence across the entire mixed-grass prairie ecoregion (Breiman 2001, Boyce et al. 2002, Manly et al. 2002, Evans et al. 2011). I then developed a second set of models using only leks classified as 'stable'.

<u>Resource Selection Functions.</u> To evaluate potential nonlinear responses for each habitat covariate, I used generalized additive models (GAM) with lek locations and random points as binary responses (Crawley 2007, Wood 2008). I evaluated linear, quadratic, and natural log (i.e., ln[x+0.001]) threshold responses for each habitat covariate used in my analysis by examining plots of predicted relationships and the partial residuals (Fig. 3). Following patterns observed in my GAM analyses, I fit a fully parameterized RSF and estimated mean coefficients of each habitat covariate using a generalized linear model (GLM) with a binomial error structure and a logistic link function (Boyce and McDonald 1999, Manly et al. 2002). To prevent my model from overfitting my data and limiting predictability to potential new areas, I used backwards stepwise selection and Akaike's Information Criterion (AIC) to iteratively remove covariates that contributed little to explaining the variation in my data (p-value ≥ 0.05; Akaike 1987, Burnham and Anderson 2002, Arnold et al. 2010).

<u>Random Forest Classification Trees.</u> Random Forest classification trees are highly sensitive to imbalanced datasets in which the minority class (i.e., represents only a small percentage of the entire dataset) has a much lower number of observations than the majority class (i.e., represents a large percentage of the dataset; Evans and Cushman 2009). Therefore, I reduced observations of random points (majority class) to equal the number of observations of leks (minority class; Table 5) because this provided the lowest error rate for my lek locations without drastically compromising the model's ability to correctly classify random points (Evans and Cushman 2009). Next, I developed Random Forest models where I again compared habitat conditions at lek locations to those at random points (Breiman 2001, Evans et al. 2011). I conducted Random Forest analyses using the 'caret' package in program R to increase model performance by refining model training parameters such as the number of branches that will

grow at each split (Fig. 4) and the number of randomly sampled habitat covariates at each node in a classification tree (Fig. 5; Kuhn et al 2008). After fitting my Random Forest models, I evaluated the importance each habitat covariate had in predicting lek occurrence using standard measures of variable importance (Breiman et al. 2001, Evans et al. 2011). In essence, variable importance is estimated by quantifying the difference in predictive accuracy when a variable (i.e., habitat covariate) is included in a test dataset versus if it was not included for each tree in a Random Forest. The differences are then averaged across all trees to represent a relative importance value for each habitat covariate (Evans et al. 2011). Predictions from all models were rescaled and scored between 0 and 1, where 0 indicated very low habitat suitability and 1 indicated very high habitat suitability.

Model Validation

I validated each model using a Receiver Operating Characteristic combined with an Area Under the Curve (ROC-AUC) analysis where I withheld 20% of the original data from model development to use in evaluating my RSF and Random Forest models predictive performances. Area under the curve scores were then averaged to provide an overall estimate of predictive performance over 500 iterations (Fielding and Bell 1997, Boyce et al. 2002). A predictive model will have an ROC-AUC score of 0.70 or higher, where a score of 0.50 indicates the model does no better at predicting leks from random points than random chance alone and a score of 1.0 indicates the model classified leks and random points perfectly (DeLeo 1993, Fielding and Bell 1997).

Ensembled Predictions

I calculated the average habitat suitability scores across my ensemble of models to reduce the uncertainty associated with each model and to make more accurate and unbiased predictions of potential prairie-chicken habitat (Araújo and New 2007, Hao et al. 2020). Averaged predictions were then used to create my final map depicting the relative probability of a prairie-chicken lek occurring across the mixed-grass prairie ecoregion. Performance of my ensembled predictions were evaluated using a ROC-AUC analysis.

Identifying Potential Prairie-chicken Habitat

To quantify the total number of squared kilometers classified as habitat potentially suitable for prairie-chickens, I extracted raster cells from my ensembled map that were greater than or equal to the mean habitat suitability score for lek locations and estimated the total area potentially suitable for prairie-chickens ($\widehat{K}$) as:

$$\widehat{K} = 10^{-6} (S \times 30 \text{ m}^2),$$

where S is the total number of 30-m^2 cells that had a habitat suitability score ≥ the mean value for lek locations, ~30 m^2 is the resolution of my raster, and 10^{-6} converts squared meters to squared kilometers. I removed all large water bodies, major cities and towns because areas within water or that are associated with exurban development are unavailable for prairie-chicken use (i.e., non-habitat; Hagen et al. 2011, Plumb et al. 2019). While it is difficult to define the minimum area of habitat needed to support a viable prairie-chicken population due to differences in the availability and configuration of habitat conditions, recommended space needs have ranged between approximately 25–200 km^2 (Bidwell et al. 2003, Davis 2005, Haufler et al. 2012). Thus, we defined identified areas of unoccupied habitat that were ≥25 km^2 as potentially suitable for translocation. Areas of high uncertainty with our RSF models (Fig. 10) were masked out and subtracted from the total amount of squared kilometers identified as potentially suitable habitat for prairie-chickens.

Evaluating Habitat Connectivity

I identified potential corridors and evaluated the relative connectivity of identified habitat patches using a least-cost path analysis. I used the inverse of predictions from my ensembled map as a cost path function and assumed the cost for movement is negatively related to the habitat suitability score derived from my ensembled predictions (Chetkiewicz and Boyce 2009). I then used the Least Cost Path analysis tool in ArcGIS Pro 2.9 (ESRI 2020) to find the path of least resistance between identified patches of occupied habitat and potentially suitable, but unoccupied habitat. Current goals for prairie-chicken connectivity zones between identified focal areas of prairie-chicken habitat are as follows: 1) at least 40% good-high quality habitat, 2) no greater than 3 km between focal areas and 3) a minimum of 8 km in width (Van Pelt et al. 2013). I buffered paths of least resistance on each side to 4 km to identify potential areas for targeted habitat restoration efforts to improve habitat connectivity and evaluated the potential for natural recolonization and the relative connectivity of contiguous areas of suitable prairie-chicken habitat ≥ 25 km^2 based on the distance to current occupied areas of prairie-chicken habitat and the relative quality of habitat between identified areas of potential prairie-chicken habitat (Van Pelt et al. 2013).

Results

We identified 272 lek locations in the mixed-grass prairie ecoregion (Fig. 2) and classified 88 of those lek locations as stable during 2010–2019. I found no evidence for spatial autocorrelation among lek locations (*Morans* $I \leq 0.01$, $P = 0.66$). The majority of habitat covaritates differed substantially between lek locations and random points (Table 2). All models exhibited a strong positive relationship with average perennial forb and grass cover and distance to highway and transmission lines, and a strong negative relationship with average cropland cover, average annual forb and grass cover, average tree cover and density of oil wells (Table 3; Fig. 6 & 7). All RSF and Random Forest models exhibited high predictive accuracy with an average cross-validated ROC-AUC score of 0.88 (Table 5). Habitat covariates of less importance included variation in bare ground, litter, and shrub cover, distance to oil wells, wind turbines, roadways, and density of wind turbines.

Mean habitat suitability scores for lek locations in all models was 0.77, except for the Random Forest analysis using only stable lek data which had a mean score of 0.88 (Table 5; Fig. 8). Total area identified as potentially suitable prairie-chicken habitat ranged between 4,526 and 7,728 km^2 (6,900 – 18,000 acres) across models, with a mean of ~6,107 km^2 (1.5 million acres; Table 5; Fig. 9); however, in general, areas identified as having high suitability remained relatively consistent across all individual models (Fig. 9). The Random Forest model developed using all lek location data had the most conservative estimates of prairie-chicken habitat (4,526 km^2) while the Random Forest model developed with stable lek only data had the most liberal predictions of prairie-chicken habitat (7,728 km^2; Table 5; Fig. 9).

Mean habitat suitability score for my ensembled predictions was 0.78 (CI: 0.75 – 0.89; Fig. 12) and total number of squared kilometers identified as potentially suitable prairie-chicken habitat was ~4,576 km^2 (1.1 million acres) both occupied and unoccupied (Table 6; Fig. 13). Area under the curve score was 0.91 indicating my ensemble predictions classified leks versus random points with high accuracy (Table 6 & Fig. 14). Primary areas identified as suitable habitat in Kansas and Oklahoma are already occupied by prairie-chickens (i.e., areas are in close proximity to currently active lek locations). However, I identified three areas between ~28 and ~73 km^2 (6,900 – 18,000 acres) of suitable but potentially unoccupied habitat within Seward county in Kansas and Beaver, Ellis and Woodward counties in Oklahoma (Table 7 & Fig. 15). I also identified large areas of potentially suitable habitat for prairie-chickens in Texas (Table 7 & Fig. 15), but as I was unable to obtain lek survey data from Texas state agencies, it is unknown whether identified areas are currently occupied by prairie-chickens at this time.

I found low-to- moderate potential for natural recolonization between areas identified as potentially unoccupied and suitable for prairie-chickens and areas currently occupied by prairie-chickens. Euclidean distances between areas of occupied habitat and potentially unoccupied habitat were greater than 3 km and <1% of raster cells classified as potentially suitable for praire-chickens in between (Table 7; Fig. 16 & 17). Areas between occupied and potentially unoccupied habitat had a habitat suitability score of 0.65 or lower.

Discussion

Effective conservation requires accurate landscape-level assessments of habitat quality and connectivity. My study expands on previous research (Jarnevich et al. 2016) by combining predictions across multiple modeling techniques and response types (all leks vs stable leks) to provide rigourous estimates of habitat suitability for prairie-chicken conservation. Habitat conditions defining predicted prairie-chicken habitat suitabilty within models are consistent with previous research and include low proportions cropland, annual grass, and tree cover, lower density of oil wells, greater proportions of perennial grass and forb cover, and greater relative distances to highways and transmission lines (Woodward et al. 2001, Fuhlendorf et al. 2002, Hagen et al. 2011, Lautenbach et al. 2017, Plumb et al. 2019). While I identified three smaller areas of potentially suitable and unoccupied habitat for potential translocations, my results indicate conservation efforts may be better directed towards increasing the quality and connectivity of available habitat by strategically implementing habitat restoration projects within and adjacent to the species’ current distribution. In addition, least-cost path analyses revealed a low degree of connectivity between areas of occupied and unoccupied habitat, highlighting the importance of implementing habitat improvement projects to increase connectivity for prairie-chicken persistence. My results provide information that professionals may use to prioritize conservation delivery for prairie-chickens in the mixed-grass prairie ecoregion.

Model Evaluations

All four RSF and Random Forest classification tree models had high predictive accuracy. Core areas identified as potential prairie-chicken habitat remained relatively consistent across all four models, supporting my interpretation of high habitat suitability within these areas. High agreement of predictions across models is not unexpected given the restrictive habitat requirements of prairie-chickens. Habitat models for habitat specialists, like prairie-chickens, are typified by higher predictive accuracy than those of habitat generalists (Segurado and Araseujo 2004, McPherson and Jetz 2007, Grenouillet et al. 2011). Similarities in performance and predictions between all four of my models could also be a product of consistent behavioral traits of prairie-chickens (e.g., avoidance of human altered landscapes). In addition, some research indicates models developed for residential species have higher accuracy than models developed for migratory or nomadic species whose occurrence depends on seasonal or annual resource availability (McPherson and Jetz 2007).

Nevertheless, I did find subtle differences in habitat predictions among models. Predicted total potential habitat varied from 4,526 to 7,728 km^2 (1.1 – 1.9 million acres) across models, highlighting the uncertainty that exists among modeled predictions of habitat suitability (Lawler et al. 2006, Pearson et al. 2006). To address this uncertainty, I employed an ensemble approach to improve certainty in my estimates of habitat suitability and provide more rigorous inferences regarding prairie-chicken conservation (Marmion et al. 2009). Generally, ensemble approaches reduce the probability of wrongfully predicting potentially suitable habitat in areas of non-habitat or vice versa (i.e., lowers the risk of a Type I/II error; Araújo and New 2007, Marmion et al. 2009, Hao et al. 2020, Ramirez-Reyes et al. 2021).

My ensembled predictions provided the second most conservative predictions of habitat suitability identifying a total of ~4,575 km^2 (1.1 million acres) of potential prairie-chicken habitat in Kansas, Oklahoma, and Texas and were very similar to predictions made with my Random Forest model developed using all lek data which identified 4,526 km^2 (1.1 million acres) of potentially suitable habitat. High similarity could be because Random Forest models are in a sense an ensemble and thus better capture data complexities relative to many regression-based or decision tree models (Evans et al. 2011, Ali et al. 2012). Additionally, some research suggests that models that allow you to control model parameters (e.g., the number of branches that grow in each tree), such as my Random Forest models, perform just as well as ensembled predictions from multiple models (Hao et al. 2020).

I developed stable lek models with the intention of identifying habitat conditions that support viable populations throughout time. However, in contrast to the Random Forest model developed using all lek data, the stable lek model developed using Random Forest classification trees had substantially more liberal predictions of habitat suitability, predicting the largest area of suitable habitat for prairie-chickens of all my models. Random Forest models developed with small datasets, such as the stable lek model, may have a higher degree of uncertainty due to the low number of observations within the dataset (Ali et al. 2012, Han et al. 2021). Therefore, the stable lek model developed using Random Forest classification trees may not perform as well as other models when generalizing to new areas. Nonetheless, this result further supports the use of ensemble approaches for not only reducing the risk of committing a Type I/II error, but making more accurate inferences and thus, sound management decisions.

Identifying and Prioritizing Conservation Actions

My ensembled predictions of habitat suitability can be used to prioritize areas for habitat restoration efforts and to identify areas for potential translocations. I identified three smaller contiguous areas of potentially unoccupied habitat in Kansas and Oklahoma ranging in size from 28–73 km^2 (6,900 – 18,000 acres) that are potentially suitable for prairie-chickens. Prairie grouse have been among the most difficult species to translocate (McNew et al. 2017), partially because low survival and reproductive success of translocated birds require a large founder size to establish a viable population (Milligan et al. 2018). In addition, recent research evaluating the success of prairie-chicken translocations in the sand sagebrush ecoregion found that prairie-chickens have an innate tendency to disperse remarkable distances following their release with average dispersal distances immediately post-translocation estimated to be ~145 km (Berigan 2019). Thus, identified areas of habitat need to be of high enough quality to encourage establishment and prevent the diffusion of individuals into degraded habitats. Consequently, rather than translocating birds outside of the species current range, efforts may be better focused on increasing the size and connectivity of habitat by targeting habitat restoration within and adjacent to identified areas of unoccupied habitat.

Increasing connectivity of habitat to facilitate movement between subpopulations is likely the best way to ensure the long-term persistence of the prairie-chicken population in the mixed-grass prairie ecoregion (DeYoung and Williford 2016, Garton et al. 2016). My least-cost path analyses provide a starting point for identifying appropriate conservation actions to increase connectivity. I identified least-cost paths between occupied and unoccupied, but potentially suitable habitat and found low-to-moderate potential for natural recolonization. Long-distance dispersal has been reported as being an infrequent occurrence in established populations of prairie-chickens, with one study reporting only 28% of females and 9% of males attempted long-distance movements (i.e., ≥ 5 km net displacement from within an individuals established home range) within a summer (Earl et al. 2016). In addition, current goals are that prairie-chicken connectivity zones 1) have 40% good-high quality habitat, 2) within 3 km of one another, and 3) be 8-km wide (Van Pelt et al. 2013). While identified areas of unoccupied habitat were within 16 km (average dispersal distance of a prairie-chicken) of occupied habitat, none are within 3 km of one another and there is relatively little habitat in between that could provide stopover sites and facilitate movement of individuals from nearby subpopulations. Nevertheless, areas surrounding least-cost paths that have lower habitat suitability scores due to high tree cover or cropland cover for example, may be appropriate areas to focus habitat restoration efforts. Future research should explore more advanced measures of connectivity and movement patterns to better evaluate the effects of landscape composition on dispersal patterns and the colonization of new habitats. For example, the UNICOR program applies a modification of Dijkstra shortest path algorithm (Dijkstra 1959) to find all shortest paths between two points of interest (e.g., occupied and unoccupied habitat) where the combination of all paths creates a path density map and can be used to identify areas with the greatest potential for species movement (Landguth et al. 2012).

Conclusions

Spatially explicit habitat models combined with ensemble approaches can be an effective way to reduce uncertainty and accurately identify areas to prioritize for species conservation. My ensembled predictions provide new delineations of core habitat areas and can be used to strategically implement management practices for prairie-chicken conservation. In general, areas identified as potentially suitable prairie-chicken habitat are or were in areas previously occupied suggesting efforts would be better focused on targeting habitat improvement projects within or adjacent to current areas harboring established populations to increase the quantity and quality of habitat. By spatially clustering conservation projects around areas of already occupied habitat and avoiding spread-out and short-lived projects, we can sustainably reduce the risk of extirpation. In addition, initiating habitat improvement projects to increase connectivity between disjunct areas of suitable prairie-chicken habitat will be an important component to ensure prairie-chicken persistence.

Table 1. Description of potential habitat covariates used in the development of my habitat suitability models predicting lesser prairie-chicken (LPCH) lek occurrence across the mixed-grass prairie ecoregion (MGP) in Kansas, Oklahoma, and Texas including covariate name, source of the original data, description of covariate, and justification for potentially including in my models. All models were developed using resource selection functions and Random Forest classification trees.

Name	**Source**	**Description**	**Justification**
Average perennial grass and forb cover	Rangeland Analysis Platform (2014 & 2019)	Average perennial cover within 5 km buffer of each raster cell in the MGP	Identified positive relationship between increased grassland cover and LPCH abundance (Woodward et al. 2001)
Average annual grass and forb cover	Rangeland Analysis Platform (2014 & 2019)	Average annual cover within 5 km buffer of each raster cell in the MGP	Potential relationship between increased cover of annual forbs and grass and LPCH occurrence (Lautenbach 2015, Lautenbach et al. 2019)
Average shrub cover	Rangeland Analysis Platform (2014 & 2019)	Average shrub cover within 5 km buffer of each raster cell in the MGP lek	Identified positive relationship between shrub cover and LPCH nest and brood survival in MGP (Lautenbach 2015, Lautenbach et al. 2019)
Average bare ground	Rangeland Analysis Platform (2014 & 2019)	Average cover of bare ground within 5 km buffer of each raster cell in the MGP	Identified negative relationship between increased bare ground and nesting success, but positive w/ brood survival (Lautenbach 2015, Lautenbach et al. 2019)
Average litter cover	Rangeland Analysis Platform (2014 & 2019)	Average cover of litter within 5 km buffer of each raster cell in the MGP	Identified positive relationship between increased litter and nest success (Lautenbach et al. 2019).

Table 1. continued.

Average tree cover	Rangeland Analysis Platform (2014 & 2019)	Average tree cover within 5 km buffer of each raster cell in the MGP	Identified negative relationship between increased tree cover and LPCH habitat use (Lautenbach et al. 2017)
Average cropland cover	National Agriculture Statistics Service (2014 & 2019)	Proportion of raster cells classified as tilled agriculture within 5 km buffer	Identified negative relationship between increased cropland cover and LPCH abundance (Woodward et al. 2001)
Average Enhanced Vegetation Index (EVI)	NASA Earthdata: MODerate resolution Imaging Spectroradiometer (MODIS) Vegetation Indices, MOD13Q1 (2014 & 2019)	Average of average annual EVI within 5 km buffer of each raster cell in the MGP	Potential quadratic relationship with average EVI and lek occurrence (Jarnevich et al. 2016)
Oil well density	Kansas GIS Data & Support Center Oklahoma Corporation Commission Texas Railroad Commission (2014 & 2019)	Density of oil and gas well locations within 5 km buffer of each raster cell in the MGP	Identified negative relationship between greater oil well densities and LPCH habitat use (Hagen et al. 2011, Plumb et al. 2019)
Distance to oil well	Kansas GIS Data & Support Center Oklahoma Corporation Commission Texas Railroad Commission (2014 & 2019)	Distance of raster cells to nearest oil well	Potential negative relationship between decreased distance to oil well and LPCH habitat use (Plumb et al. 2019)

Table 1 continued.

Distance to transmission line	Homeland Infrastructure Foundation-level Data (2020)	Distance of raster cells to nearest transmission line	Identified negative relationship between distance to transmission/powerlines lines and LPCH habitat use (Hagen et al. 2011, Plumb et al. 2019)
Distance to highway	Kansas GIS Data & Support Center, Oklahoma Corporation Commission, Texas Department of Transportation (2017, 2015, 2019)	Distance of raster cells to nearest highway	Identified negative relationship between distance to highways and LPCH habitat use (Hagen et al. 2011, Plumb et al. 2019)
Distance to roadway	Kansas GIS Data & Support Center, Oklahoma Corporation Commission, Texas Department of Transportation (2017, 2015, 2019)	Distance of raster cells to nearest roadway (i.e., county and town roads)	Identified negative relationship between density of and distance to roads and LPCH habitat selection (Hagen et al. 2011, Plumb et al. 2019)
Density of roadways	Kansas GIS Data & Support Center, (2017) Oklahoma Corporation Commission, (2017) Texas Department of Transportation (2020)	Density of roadways within 5-km buffer of each raster cell in the MGP (i.e., county and town roads)	Potential negative relationship between density of and distance to roads and LPCH habitat selection (Hagen et al. 2011, Plumb et al. 2019)
Density of wind turbines	US Energy Information Administration (2019)	Density of wind turbines within 5-km buffer of each raster cell in the MGP	Potential positive relationship between wind turbine density LPCH survival (LeBeau et al. 2020).

Table 1 continued.

Distance to wind turbines	US Energy Information Administration (2019)	Distance of raster cells to nearest wind turbine	Potential relationship between distance to wind turbine and LPCH space use (LeBeau et al. 2020).
Variation in average perennial grass and forb cover	Rangeland Analysis Platform (2014 & 2019)	Standard deviation of average perennial cover within 5 km of each raster cell in the MGP	Identified positive relationship between increased heterogeneity in grassland landscapes and LPCH habitat use and survival (Kraft et al. 2021, Lautenbach et al. 2021)
Variation in average annual grass and forb cover	Rangeland Analysis Platform (2014 & 2019)	Standard deviation of average annual cover within 5 km of each raster cell in the MGP	Identified positive relationship between increased heterogeneity in grassland landscapes and LPCH habitat use and survival (Kraft et al. 2021, Lautenbach et al. 2021)
Variation in average bare ground	Rangeland Analysis Platform (2014 & 2019)	Standard deviation of average bare ground within 5 km of each raster cell in the MGP	Identified positive relationship between increased heterogeneity in grassland landscapes and LPCH habitat use and survival (Kraft et al. 2021, Lautenbach et al. 2021)
Variation in average litter cover	Rangeland Analysis Platform (2014 & 2019)	Standard deviation of average litter cover within 5 km of each raster cell in the MGP	Identified positive relationship between increased heterogeneity in grassland landscapes and LPCH habitat use and survival (Kraft et al. 2021, Lautenbach et al. 2021)
Variation in average shrub cover	Rangeland Analysis Platform (2014 & 2019)	Standard deviation of average litter cover within 5 km of each raster cell in the MGP	Identified positive relationship between increased heterogeneity in grassland landscapes and LPCH habitat use and survival (Kraft et al. 2021, Lautenbach et al. 2021)

Table 1 continued.

Variation in Enhanced Vegetation Index (EVI)	NASA Earthdata: MODerate resolution Imaging Spectroradiometer (MODIS) Vegetation Indices, MOD13Q1 (2014 & 2019)	Standard deviation of average annual EVI within 5 km of each raster cell in the MGP	Potential relationship between increased heterogeneity in EVI and LPCH use and survival (Jarnevich et al. 2016).
Annual summer temperature	USFS (1961-1990)	Average annual summer temperature within 5 km of each raster cell in the MGP (June – Sept.)	Identified negative relationship between greater mean annual temperatures and LPCH reproductive success and survival (Grisham et al. 2016)
Annual precipitation	USFS (1961-1990)	Average annual precipitation within 5 km of each raster cell in the MGP	Large scale ecological driver influencing grassland production. Carry over effects for LPCH survival (Fields et al. 2006, Grisham et al. 2013)
Ruggedness	National Elevation Data (2013)	Standard deviation of elevation within 5 km of each raster cell in the MGP	Identified negative relationship b/w LPCH habitat use and rough terrain (Hagen et al. 2004, Hagen and Giesen 2005)

Table 2. Mean and standard deviation (SD) of habitat covariates associated with lek locations and available locations (random points) used in the development of my habitat suitability models for lesser prairie-chicken lek occurrence in the southern mixed-grass prairie ecoregion

	Lek locations		Available locations	
Covariate	**Mean**	**SD**	**Mean**	**SD**
Tree Cover (%)	3.0	1.2	5.2	3.7
PFG Cover (%) *	64.4	10.2	55.2	11.0
AFG Cover (%) **	11.9	4.9	15.0	6.4
Bare ground (%)	7.5	4.2	9.3	4.3
Shrub Cover (%)	3.6	2.2	4.1	2.9
Litter (%)	6.7	2.5	7.7	2.6
Cropland Cover (%)	20.0	20.0	30.0	30.0
Density of Oil Wells	8.5	9.3	24.0	34.9
Density of Roadway	1668.9	757.8	2024.5	977.0
Density of Wind Turbines	0.5	3.8	2.9	11.4
Dist. to Highway (km)	7.05	4.02	5.13	4.26
Dist. to Oil Wells (km)	2.63	1.79	2.45	2.85
Dist. to Roadway (km)	0.90	0.95	0.69	0.74
Dist. to Windmills (km)	24.72	12.24	24.07	15.97
Ruggedness	15.4	4.9	15.2	7.0
Variation in AFG **	10.0	3.8	11.6	4.9
Variation PFG *	18.5	4.3	19.7	3.8
Variation Bare ground	7.5	3.0	8.9	3.2
Variation Tree	4.5	1.8	7.6	4.3
Variation Litter	4.0	1.7	4.6	1.8
Variation Shrub	3.2	2.2	4.0	3.0
Ave. annual precipitation (cm)	574.9	44.1	603.6	67.2
Ave. summer temp. (ºC)	26.2	0.4	26.3	0.6

*PFG = perennial grass and forb cover
** AFG = annual forb and grass cover

Table 3. Coefficients and standard errors for my two resource selection function (RSF) models predicting the relative probability of a lek occurring in the mixed-grass prairie ecoregion of Kansas, Oklahoma, and Texas. Bold values are significant at $p \geq 0.05$.

RSF – all leks			**RSF – stable leks only**		
Coefficient	***B***	***SE***	**Coefficient**	***B***	***SE***
Ave. tree	-0.335	0.064	Ave. tree	-0.18	0.13
Ave. PFG*	**-0.074**	**0.077**	Ave. PFG	**-0.36**	**0.11**
Ave. PFG^2*	0.00029	0.0007	Ave PFG^2	**0.005**	**0.001**
Ave. AFG**	**-0.119**	**0.033**	log (Ave. AFG)	**4.49**	**1.35**
Ave. shrub	**0.649**	**0.177**	log (Ave. Litter)	**3.96**	**1.53**
Ave. shrub^2	**0.016**	**0.005**	Ave. cropland	-0.90	3.47
Ave. cropland	**-2.518**	**1.755**	Ave. cropland^2	-4.87	4.21
Ave. cropland^2	**-4.990**	**2.339**	Ave. summer temp	70.80	49.86
Ave. summer temp.	11.835	24.784	Ave. summer temp^2	-1.38	0.95
Ave. summer temp^2	-0.254	0.470	Ruggedness	0.08	0.13
Ruggedness	0.006	0.067	Ruggedness^2	-0.007	0.004
Ruggedness^2	-0.003	0.002	Distance to highway	**0.06**	**0.03**
Distance to trans. lines	**0.1034**	**0.036**	Density of roads	0.009	0.0008
Distance to trans. lines^2	**-0.003**	**0.001**	Density of roads^2	-0.00046	0.00002
Distance to highway	**0.213**	**0.057**	Density oil wells	**-0.05**	**0.01**
Distance to highway^2	**-0.009**	**0.004**	Density wind turbines	-0.09	0.07
Distance to windmill	**0.053**	**0.024**	Variation in AFG**	**-0.28**	**0.10**
Distance to windmill^2	**-0.0001**	**0.00**	Variation in PFG*	**0.16**	**0.06**
Density of oil wells	**-0.039**	**0.007**	Constant	-924.95	653.66
Density of windmills	-0.028	0.019			
Variation in PFG*	0.076	0.143			
Variation in PFG^2*	0.0013	0.003			
Varriation in BG***	**-0.16**	**0.059**			
Log (Variation in shrub)	**0.878**	**0.326**			
Constant	-131.2	326.4			

*PFG = perennial forb and grass; **AFG = annual forb and grass; ***BG = bare ground

Table 4. Top 10 variables selected for Random Forest classification models predicting lesser prairie-chicken lek occurrence across the mixed-grass prairie ecoregion in Kansas, Oklahoma and Texas.

RF model – all leks		**RF model -stable leks only**	
Variable	**Importance**	**Variable**	**Importance**
Ave. tree	100	Ave. tree	100
Ave. PFG*	84.87	Ave. PFG*	80.09
Ave. annual precipitation	75.31	Distance to highway	76.92
Density of oil wells	69.50	Density of oil wells	68.89
Ave. cropland	68.94	Ave. cropland	65.11
Distance to highway	65.98	Distance to trans. lines	59.79
Ave. AFG**	57.48	Density of roadways	59.70
Distance to trans. lines	51.28	Ave. annual precipitation	55.17
Ave. summer temperatures	50.42	Ave. shrub	55.12
Ruggedness	47.61	Ruggedness	52.07

*PFG = perennial forb and grass
**AFG = annual forb and grass

Table 5. Cross-validated area under the curve scores and 95% confidence intervals, mean and 95% confidence intervals for habitat suitability scores for lek locations, and total area predicted as potentially suitable habitat for lesser prairie-chickens for each resource selection function (RSF) and Random Forest (RF) classification tree model developed to predict the relative probability of a lek occurring.

Modeling technique	**Num. lek locations used**	**Num. random points used**	**AUC-ROC score**	**AUC-ROC 95% CI**	**Mean Habitat Suitability Score**	**95% Confidence Intervals**	**Area predicted as suitable prairie chicken habitat (km^2)**
RSF	272	5460	0.89	0.84 – 0.91	0.74	0.69 – 0.87	6,180
RSF	88	1760	0.90	0.82 – 0.95	0.78	0.46 – 0.95	4,914
RF	272	272	0.90	0.84 – 0.95	0.76	0.37 – 0.94	4,526
RF	88	88	0.86	0.74 – 0.94	0.87	0.43 – 0.98	7,728

Table 6. Mean and 95% confidence intervals for habitat suitability scores for lek locations, area under the curve score. and total area predicted as potentially suitable habitat for lesser prairie-chickens for my ensembled predictions that were developed by averaging predictions across the four different models: 1) resources selection function (RSF) model developed using all lek location data, 2) RSF model developed stable lek only data, 3) Random Forest classification tree model developed using all lek location data, 4) Random Forest classification tree model developed using stable lek only data.

Modeling technique	**Mean Habitat Suitability Score**	**95% Confidence Intervals**	**AUC Score**	**Area predicted as suitable prairie chicken habitat (km^2)**
Ensembled predictions	0.77	0.75 – 0.89	0.91	4,576

Table 7. Counties and state, the number of contiguous squared kilometers, and distance to the nearest subpopulations of lesser prairie-chickens for each area identified as potentially suitable but unoccupied lesser prairie-chicken habitat in the mixed-grass prairie ecoregion in Kansas, Oklahoma, and Texas.

Identified Area/Color on Map	**Counties and State**	**Area predicted as suitable prairie chicken habitat (km^2)**	**Distance to nearest subpopulation (km)**
1 / purple	Ellis and Woodward, OK	74	15.0
2 / yellow	Seward, KS & Beaver, OK	46	5.0
3 / orange	Ellis, OK	28	4.0
4* / blue	Ochiltree & Lipscomb, TX	40	NA
5* / green	Roberts &Gray, TX	97	NA

*Unknown whether areas identified in Texas are occupied or unoccupied as I was unable to obtain lek survey data from Texas state agencies.

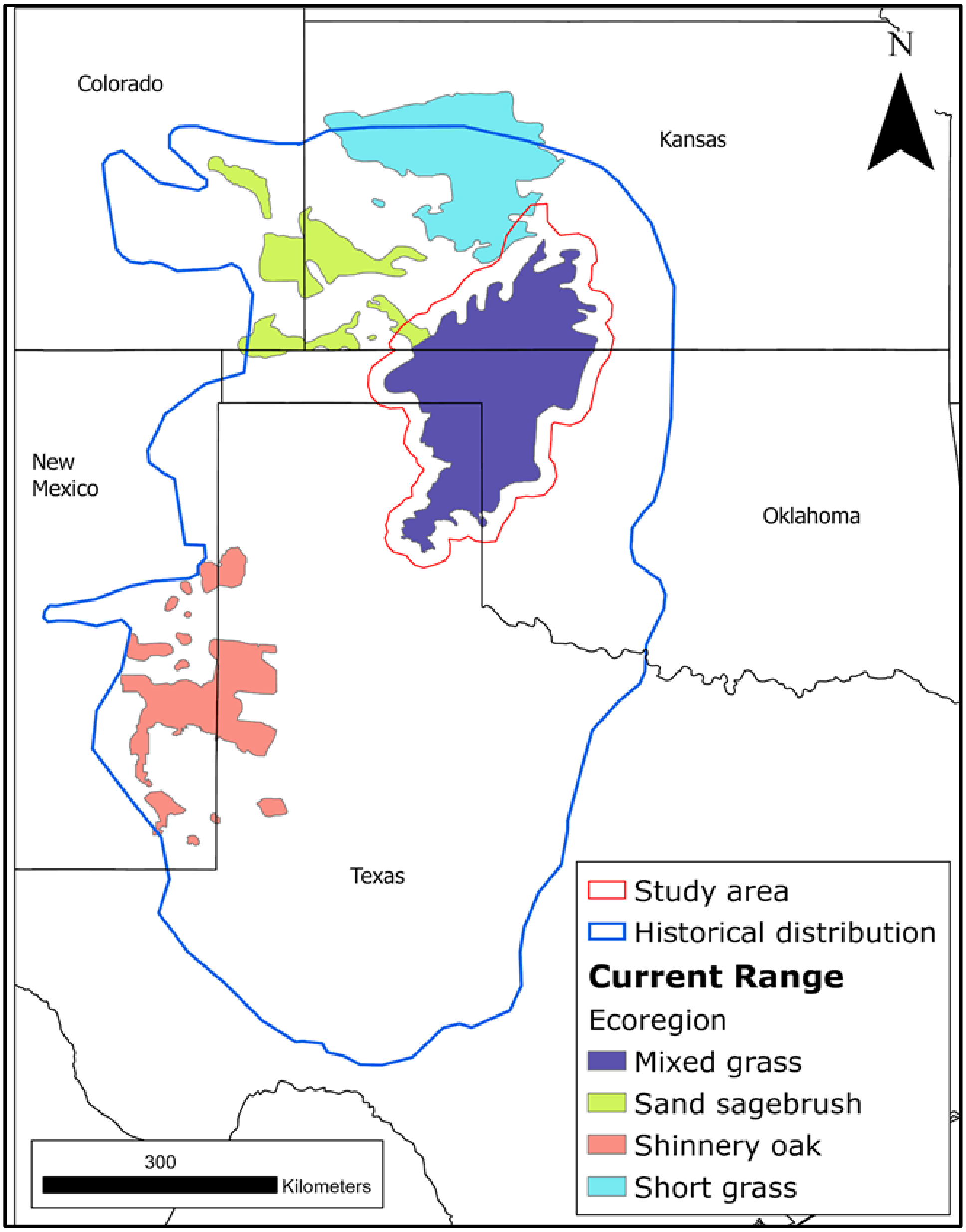

Figure 1. Map detailing the extent of my study area and the estimated historical and current distribution of lesser prairie-chickens in the southern Great Plains across four ecoregions: the mixed-grass prairie, short-grass/CRP, the sand sagebrush, and the shinnery oak. Study area is the mixed-grass prairie ecoregion outlined in red.

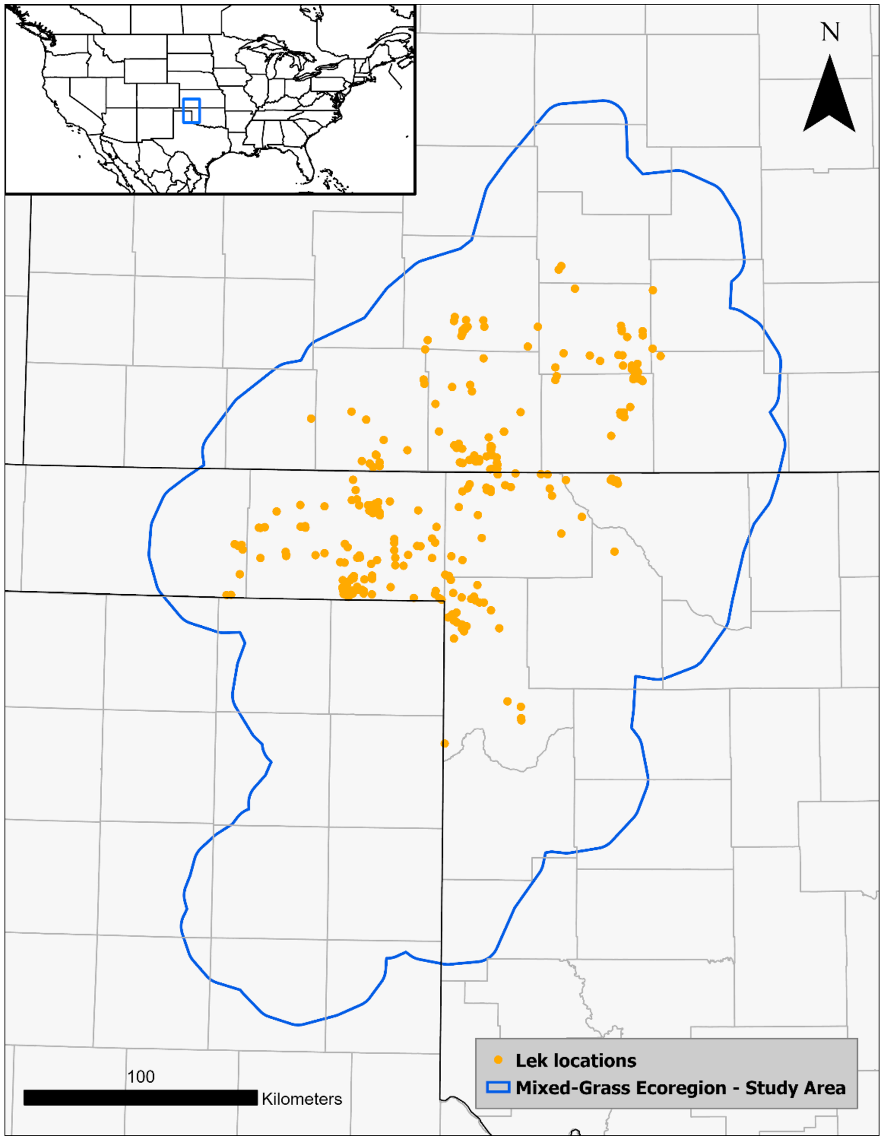

Figure 2. Map detailing the extent of my study area in the mixed-grass prairie ecoregion in Kansas, Oklahoma, and Texas and the location of all leks used in developing habitat suitability models predicting lek occurrence.

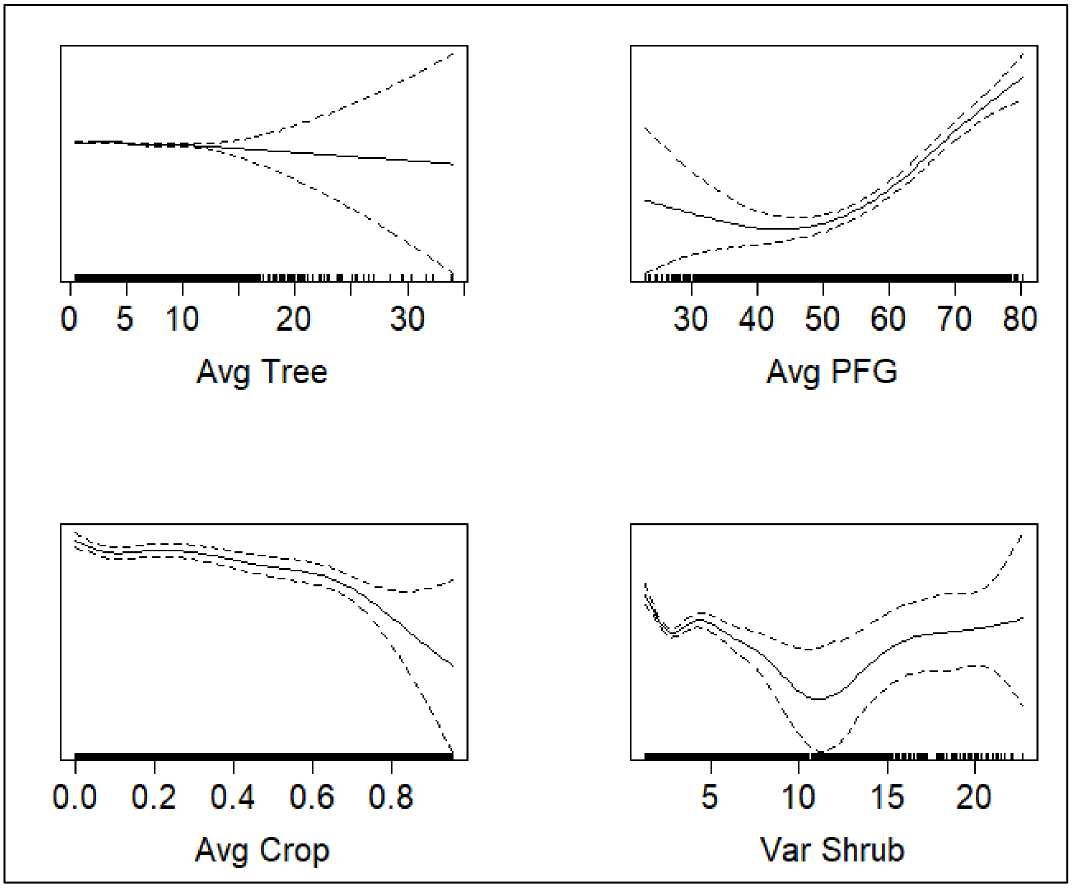

Figure 3. Predicted linear and nonlinear relationships between relative lek occurrence in the southern mixed-grass prairie ecoregion and habitat predictors for select habitat covariates used in the development of my resource selection function (RSF) models. Average tree cover exhibited a decreasing linear response, average perennial forb and grass cover an increasing quadratic response, average cropland a decreasing quadratic response, and variation in shrub had a pseudolinear response.

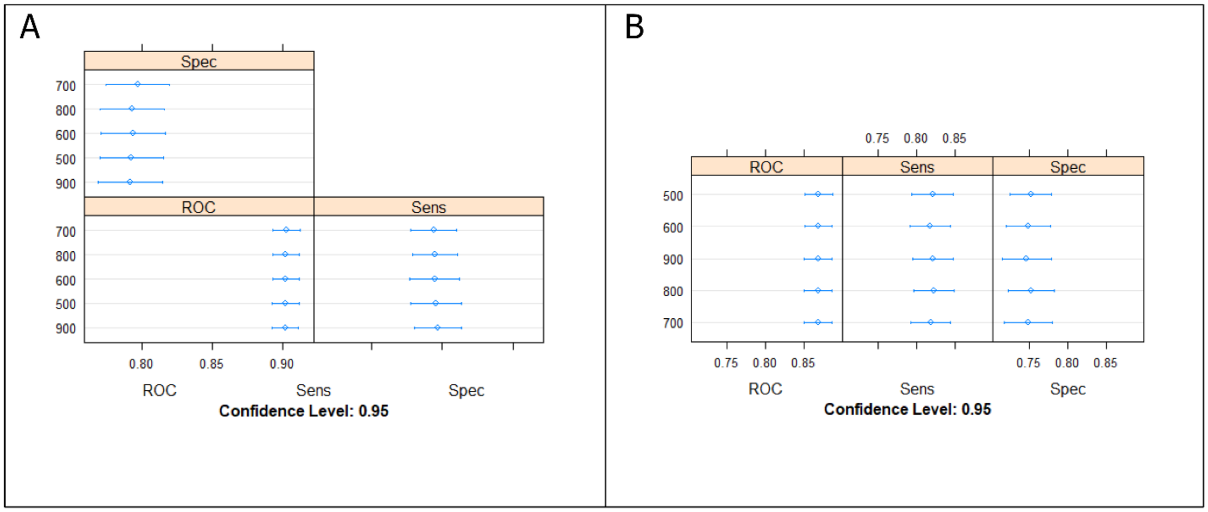

Figure 4. Predictive performance associated with using 500, 600, 700, 800, and 900 trees in my Random Forest classification trees for the model using (A) all lek data and (B) stable lek only data as measured with a Receiver Operating Characteristic (ROC) curve. Sensitivity (Sens) indicates the model's ability to correctly classify lek locations and specificity (Spec) indicates the model's ability to correctly classify a random point. The model with the highest performance included (A) 700 trees and (B) 500 trees.

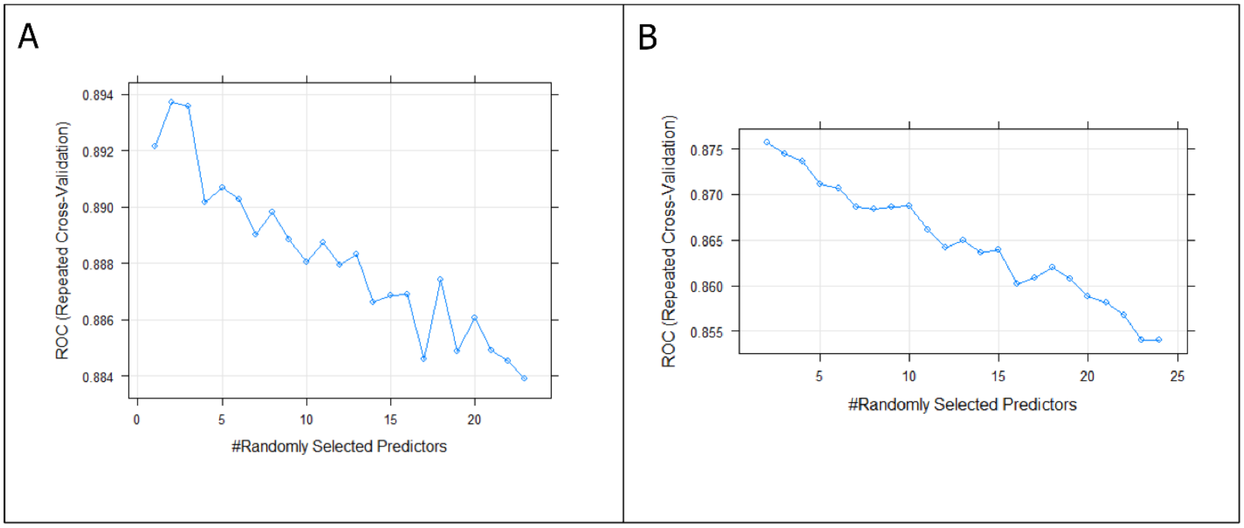

Figure 5. Predictive performance associated with various numbers of randomly selected predictor variables used in the development of my Random Forest model using (A) all lek data and (B) stable lek only data as measured with a Receiver Operating Characteristic (ROC) curve. The model with the highest performance included using two randomly-selected predictor variables at each node for both models.

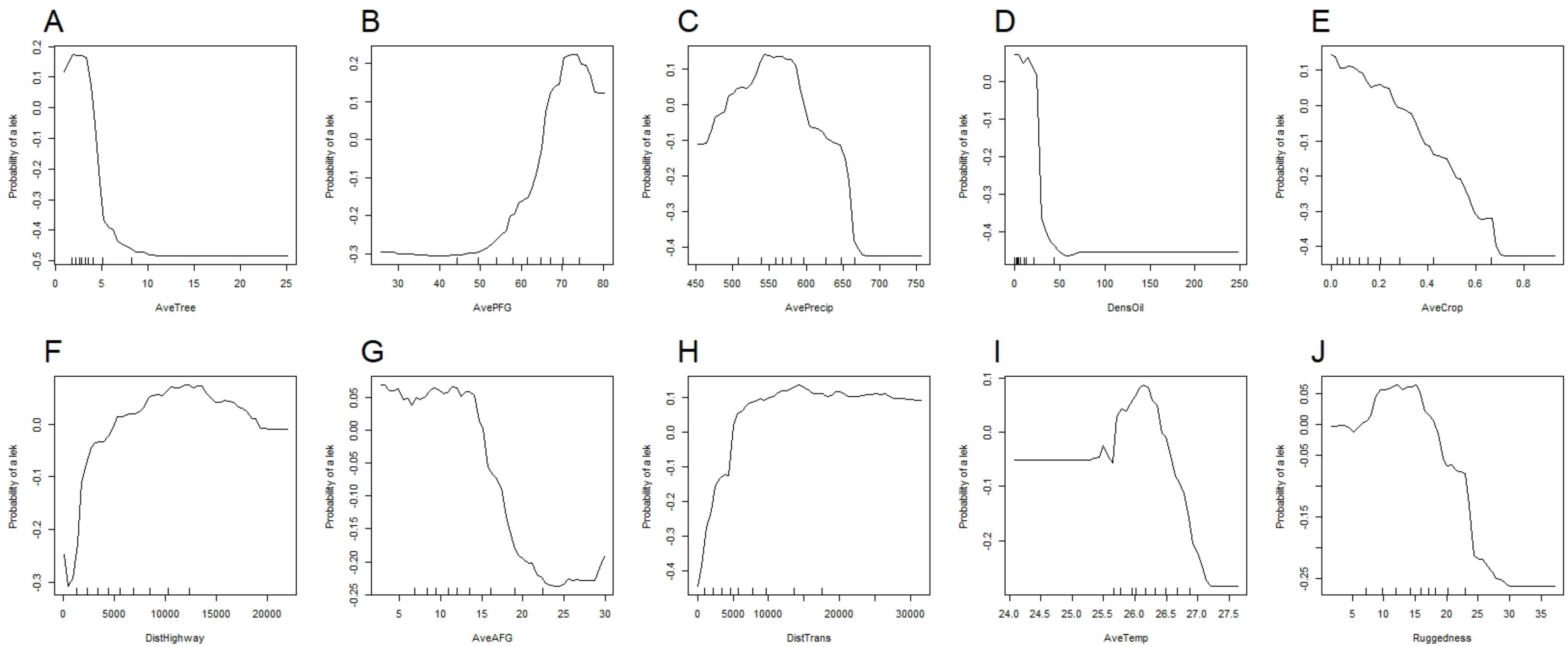

Figure 6. Partial dependency plots showing the top ten ranked predictor variables from the variable importance measure of the Random Forest classification tree model developed using all lek location data. (A) Average tree cover, (B) average perennial grass and forb cover, (C) average precipitation, (D) density of oil wells, (E) average cropland cover, (F) distance to highway, (G) average annual forb and grass cover, (H) distance to transmission lines, (I) average temperature, (J) topographic ruggedness.

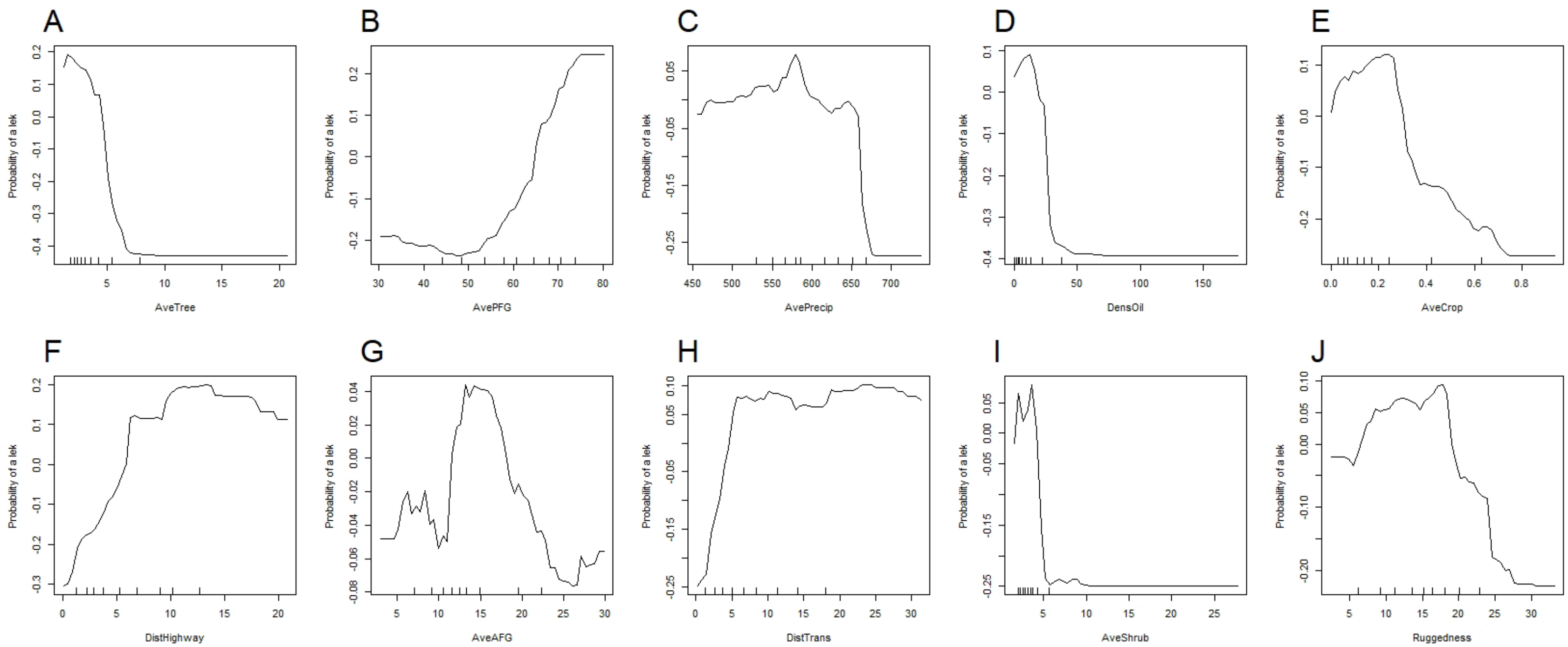

Figure 7. Partial dependency plots showing the top ten ranked predictor variables from the variable importance measure of the Random Forest classification tree model developed using stable lek only data. (A) Average tree cover, (B) average perennial grass and forb cover, (C) average precipitation, (D) density of oil wells, (E) average cropland cover, (F) distance to highway, (G) average annual forb and grass cover, (H) distance to transmission lines, (I) average shrub cover, (J) topographic ruggedness.

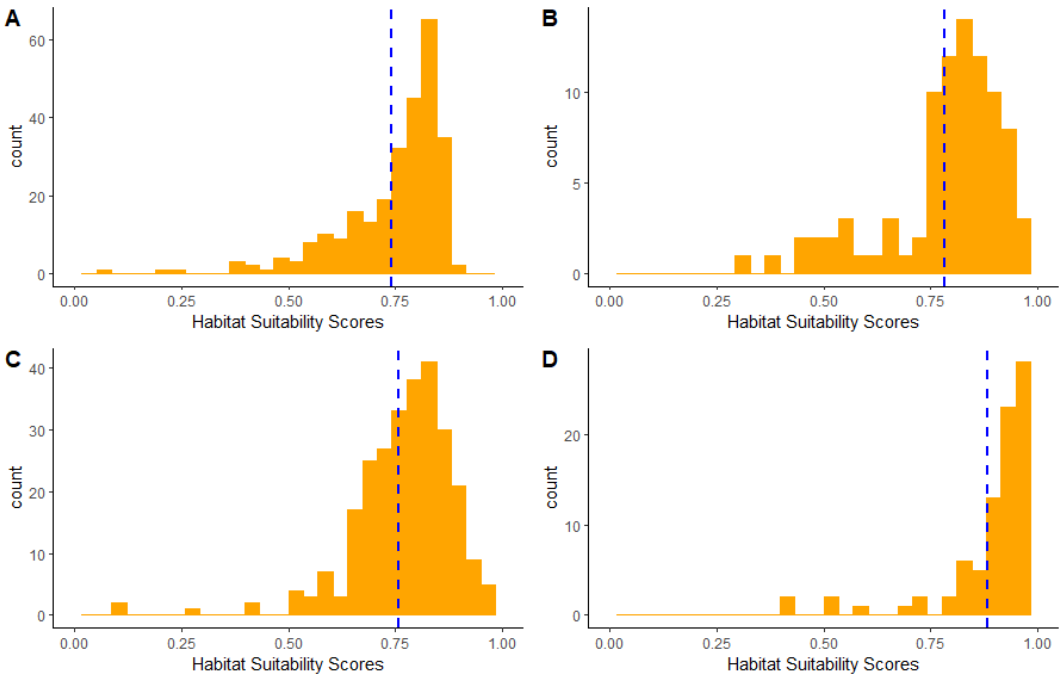

Figure 8. Distribution of habitat suitability scores for all lesser prairie-chicken lek locations for (A) the resource selection function (RSF) model developed using all lek locations, (B) RSF model developed using stable lek only data, (C) Random Forest model developed using all lek locations and, (D) Random Forest model using stable lek only data. Blue dotted line indicates mean habitat suitability score extracted at lek locations for each model.

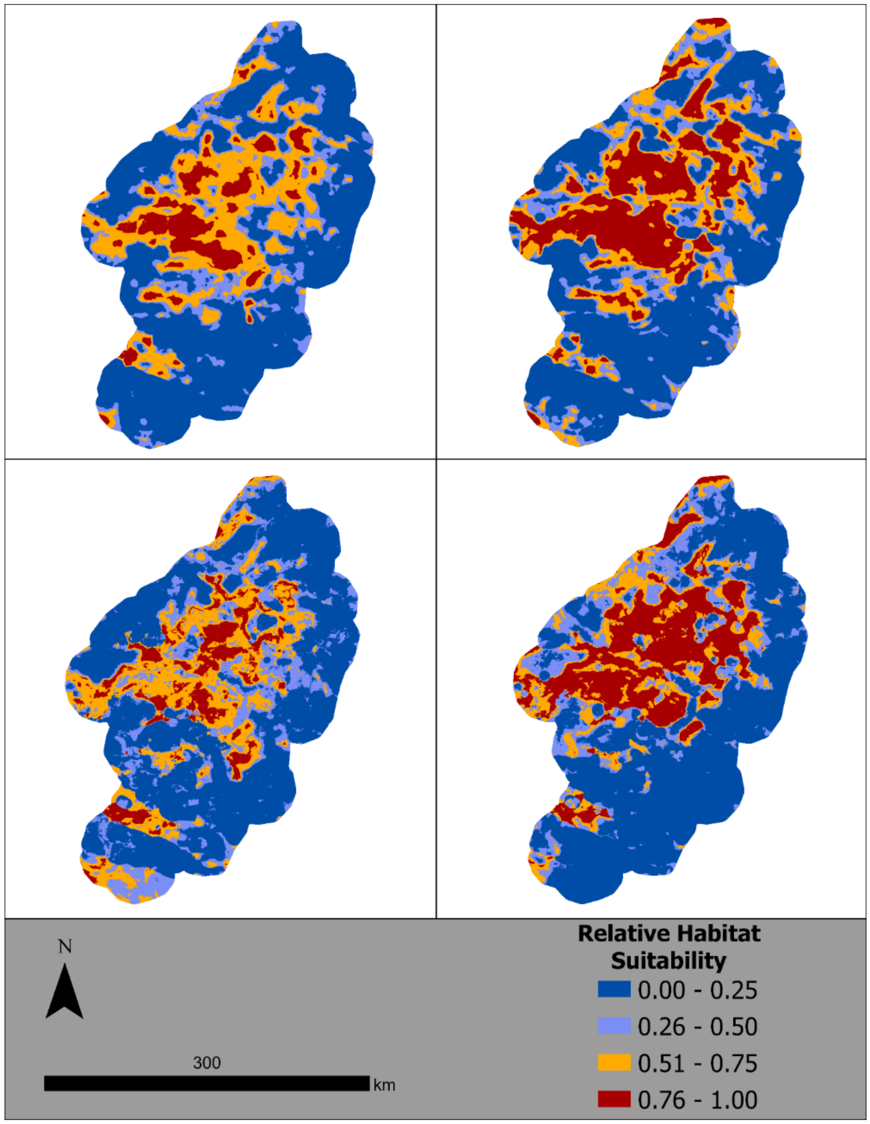

Figure 9. Predicted probability of lesser prairie-chicken lek occurrence in the southern mixed-grass prairie ecoregion for (A) the resource selection function (RSF) model developed using all lek locations, (B) RSF model developed using stable lek only data, (C) Random Forest model developed using all lek locations and, (D) Random Forest model developed using stable lek only data. Predictions from all four models were combined to accurately identify habitat for lesser prairie-chicken conservation.

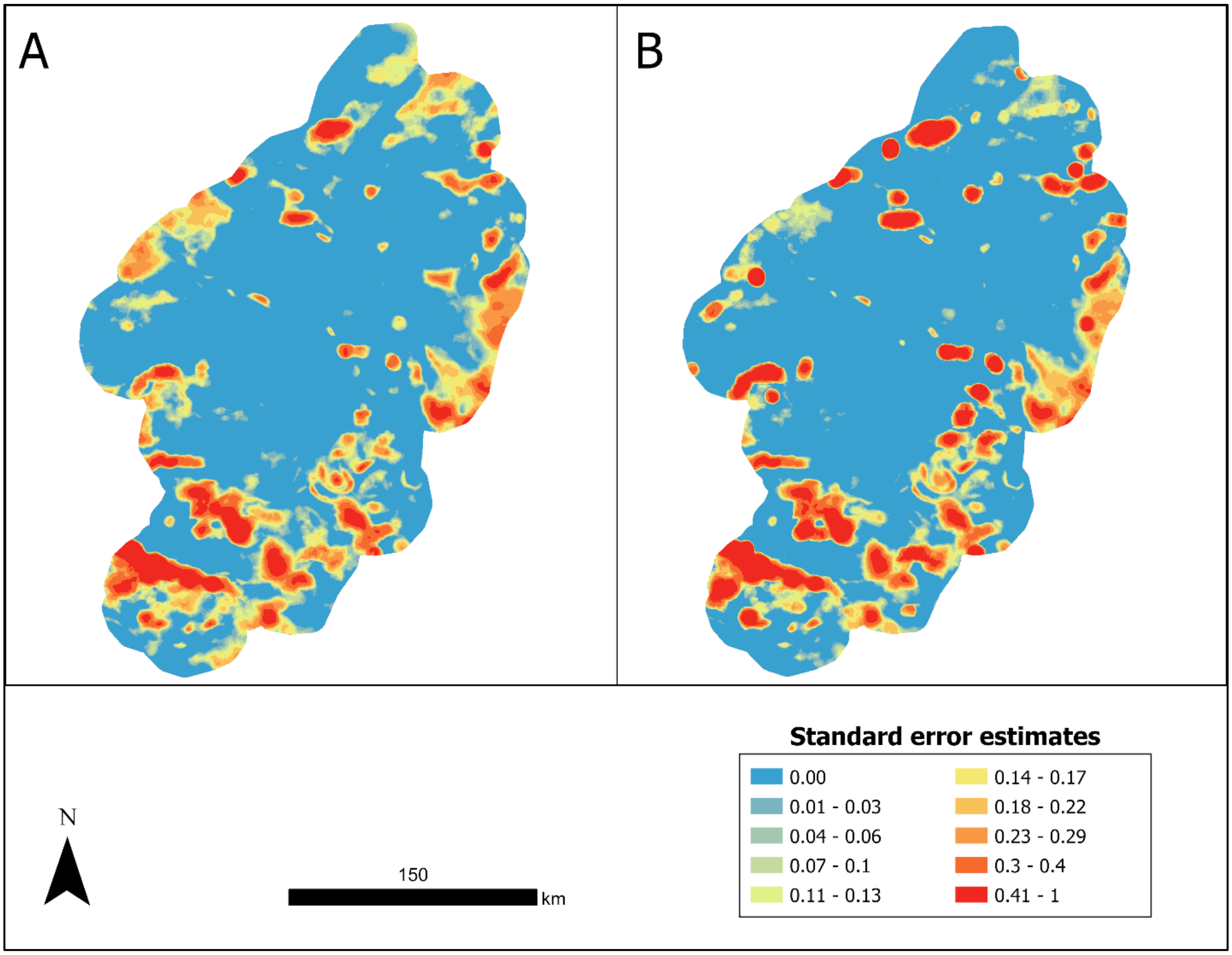

Figure 10. Standard error estimates of habitat suitability for predictions of lesser prairie-chicken lek occurrence in the southern mixed-grass prairie ecoregion made with my resource selection function (RSF) models developed using (A) all lek location data and (B) stable lek only data.

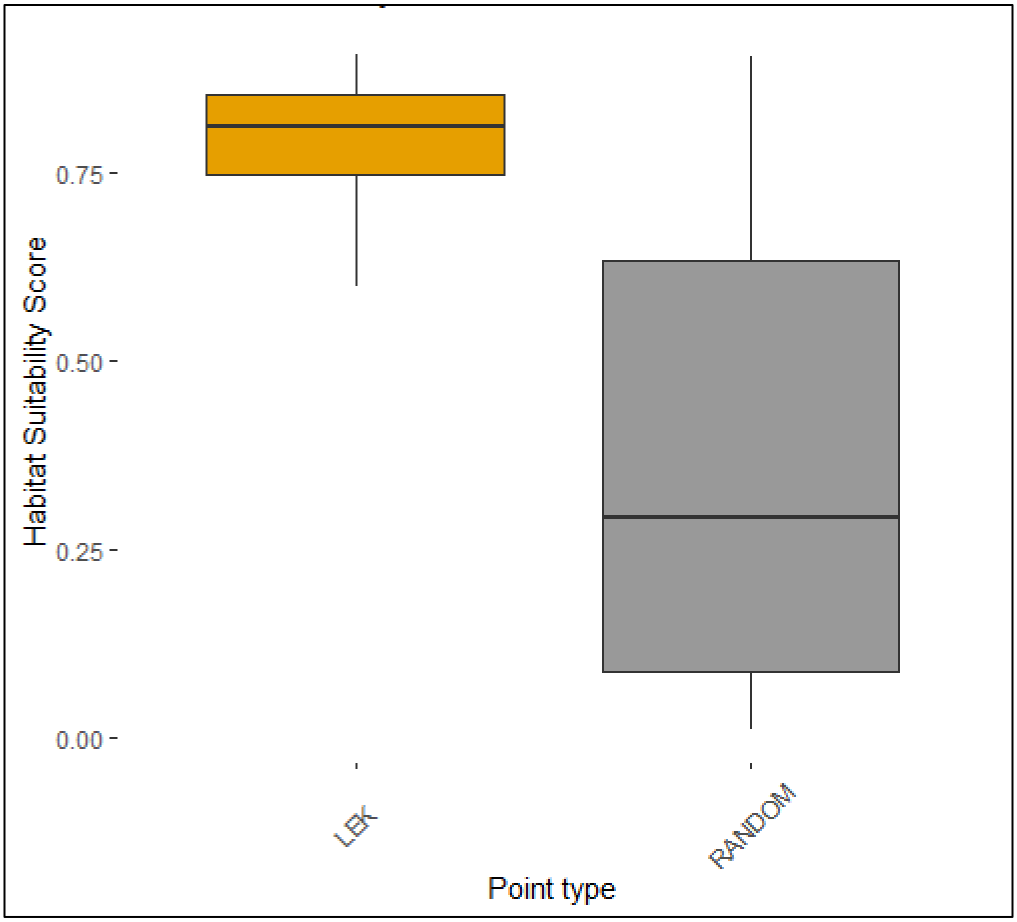

Figure 11. Distribution of habitat suitability scores for leks versus all random points for my ensembled model predicting lesser prairie-chicken lek occurrence in the southern mixed-grass prairie ecoregion that was developed by averaging predictions across four different models: 1) resources selection function (RSF) model developed using all lek location data, 2) RSF model developed stable lek only data, 3) Random Forest classification tree model developed using all lek location data, 4) Random Forest classification tree model developed using stable lek only data.

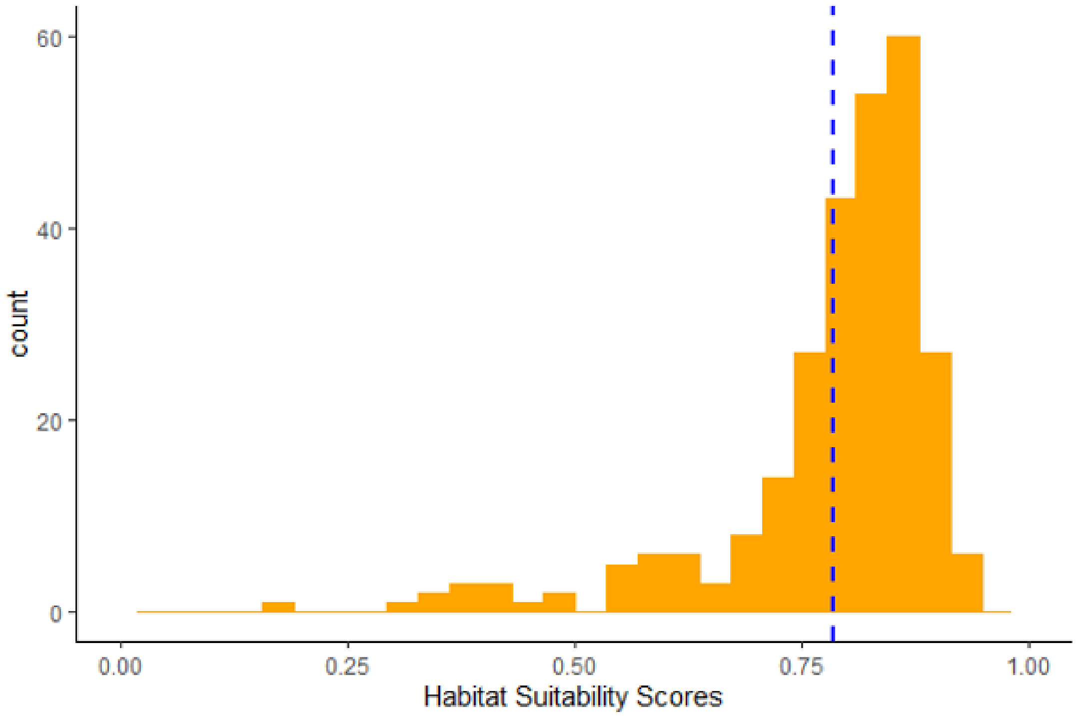

Figure 12. Distribution of habitat suitability scores for all leks for my ensembled model predicting lesser prairie-chicken lek occurrence in the southern mixed-grass prairie ecoregion that was developed by averaging predictions across four different models: 1) resources selection function (RSF) model developed using all lek location data, 2) RSF model developed stable lek only data, 3) Random Forest classification tree model developed using all lek location data, 4) Random Forest classification tree model developed using stable lek only data.

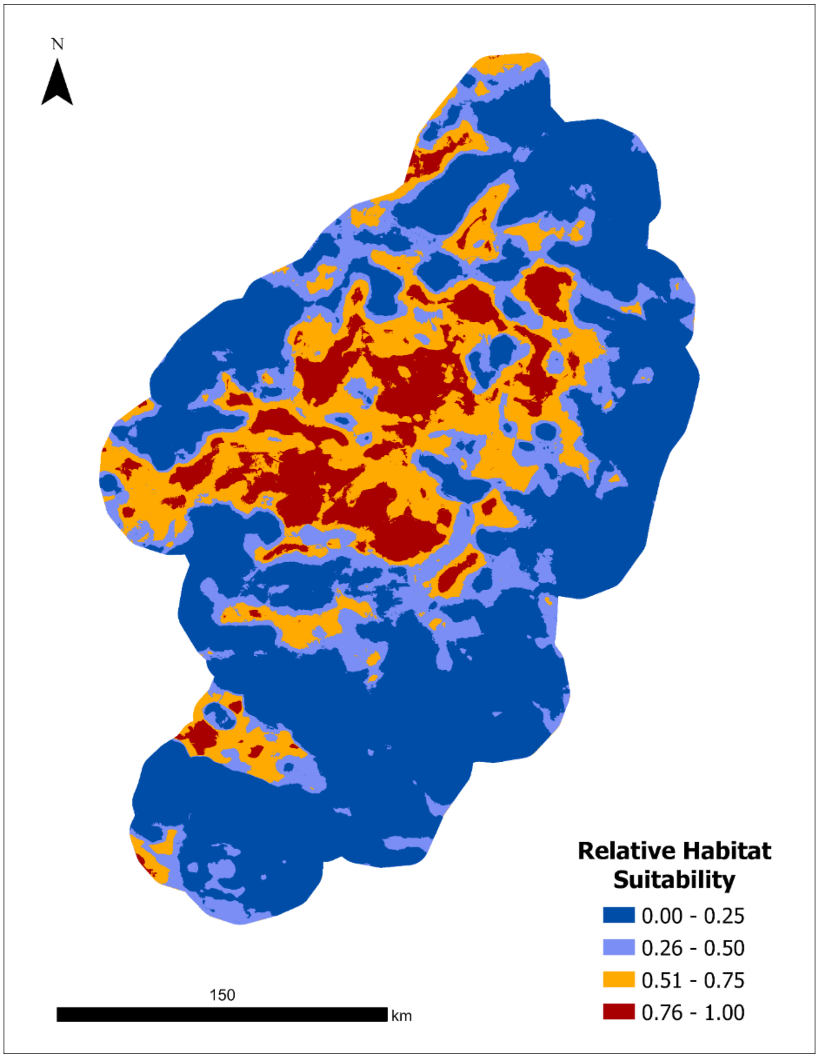

Figure 13. Relative habitat suitability for lesser prairie-chickens in the mixed-grass prairie ecoregion from ensemble predictions. Ensembled predictions were developed by averaging predictions across four different models: 1) resources selection function (RSF) model developed using all lek location data, 2) RSF model developed stable lek only data, 3) Random Forest classification tree model developed using all lek location data, 4) Random Forest classification tree model developed using stable lek only data.

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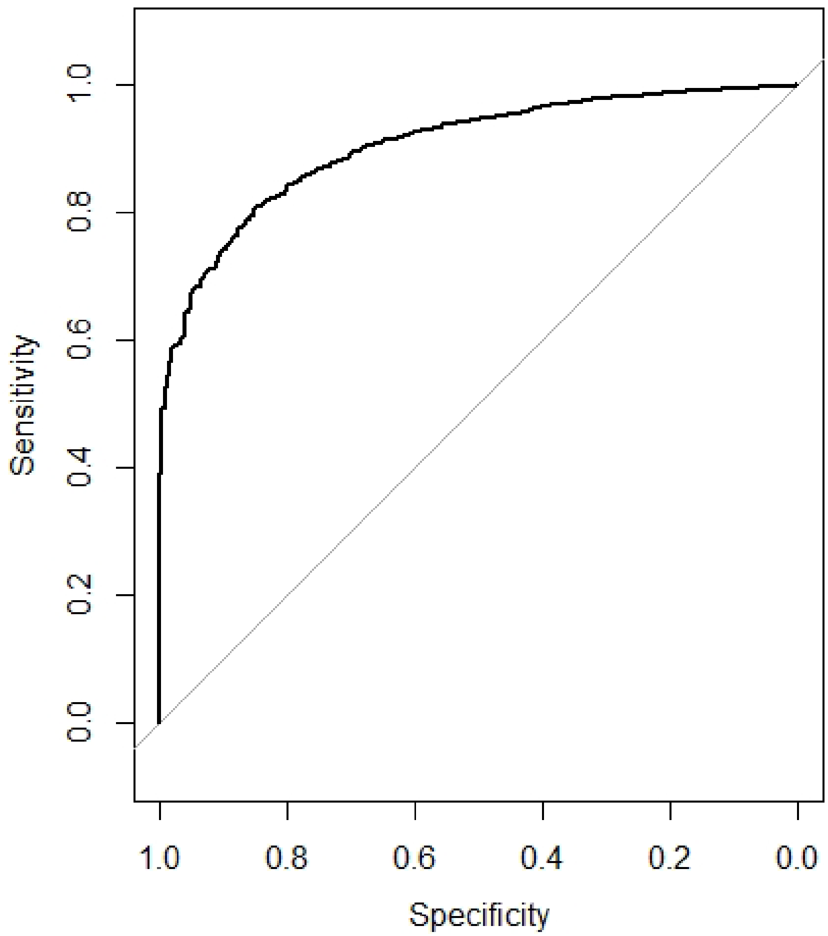

Figure 14. Receiver Operator Characteristic and Area Under the Curve (ROC-AUC) output depicting the ability of my ensembled predictions to accurately classify leks (sensitivity) and random points (specificity) from one another. Area under the curve score was 0.91 indicating my ensembled predictions have excellent predictive performance.

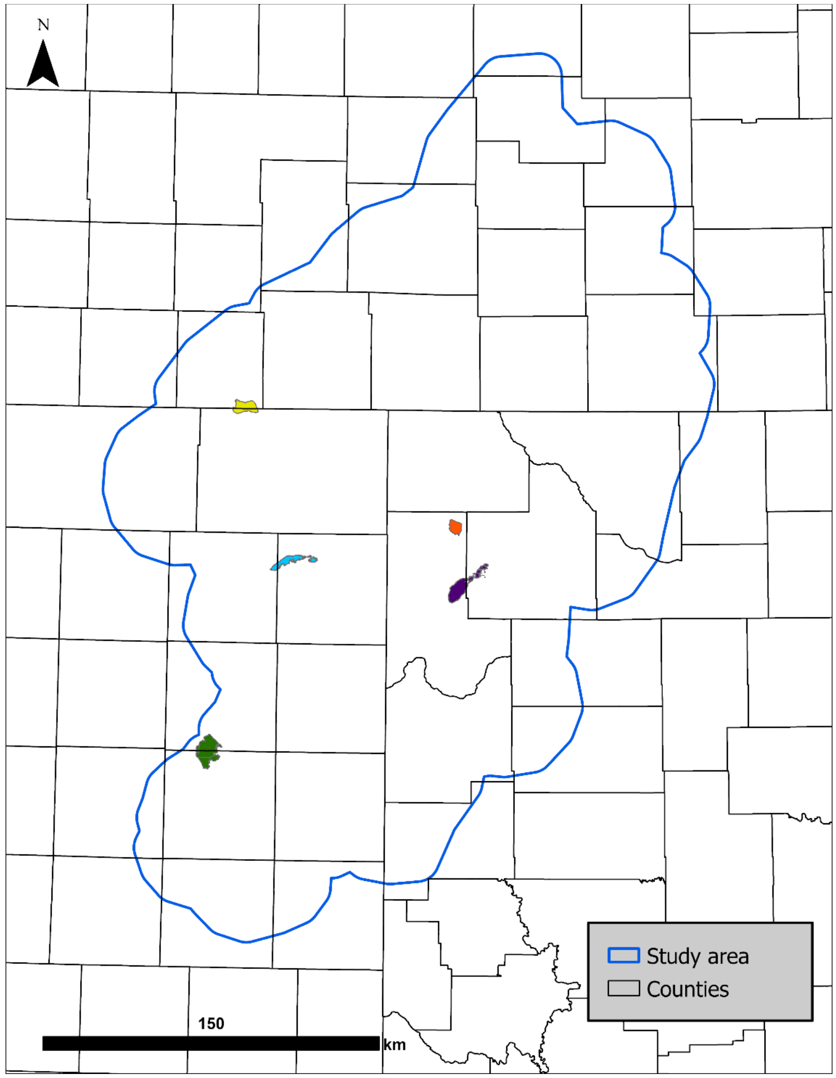

Figure 15. Map delineating identified areas of potentially suitable, but unoccupied lesser prairie-chicken habitat in the southern mixed-grass prairie ecoregion using my ensembled predictions: 1) Woodward and Ellis counties in Oklahoma (purple), 2) Seward county in Kansas and Beaver county in Oklahoma (yellow), 3) in Ellis county in Oklahoma (orange), 4) Roberts and Gray county in Texas (green), and 5) Ochiltree and Lipscomb county in Texas (blue).

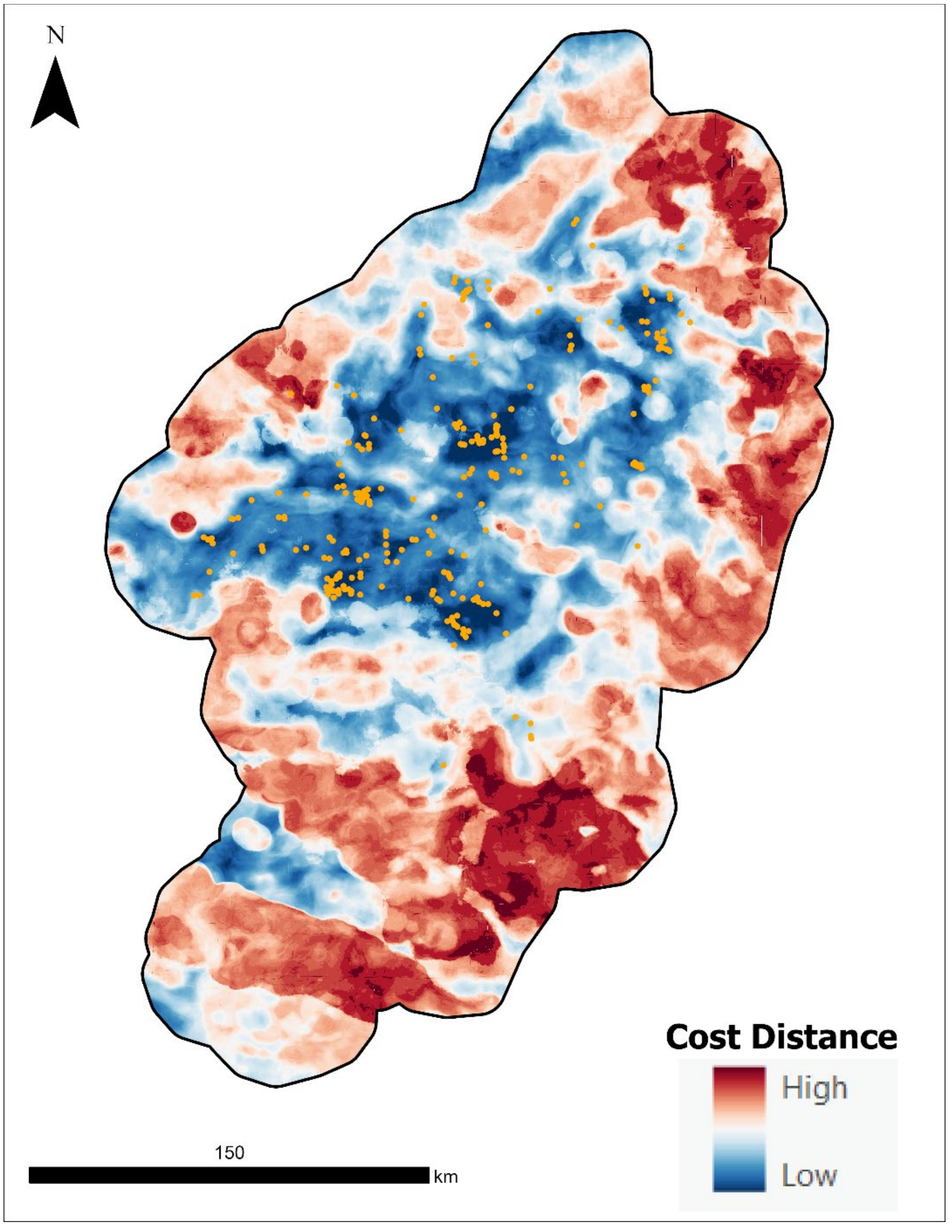

Figure 16. Cost raster developed to calculate least-cost path between identified areas of potentially suitable, but unoccupied habitat and nearest lek locations (orange) in the southern mixed-grass prairie ecoregion, where I calculated the cost for movement by taking the inverse of my habitat suitability scores for my ensembled predictions. Blue indicates areas of relatively low cost for movement and red indicates areas where there is a high cost for movement.

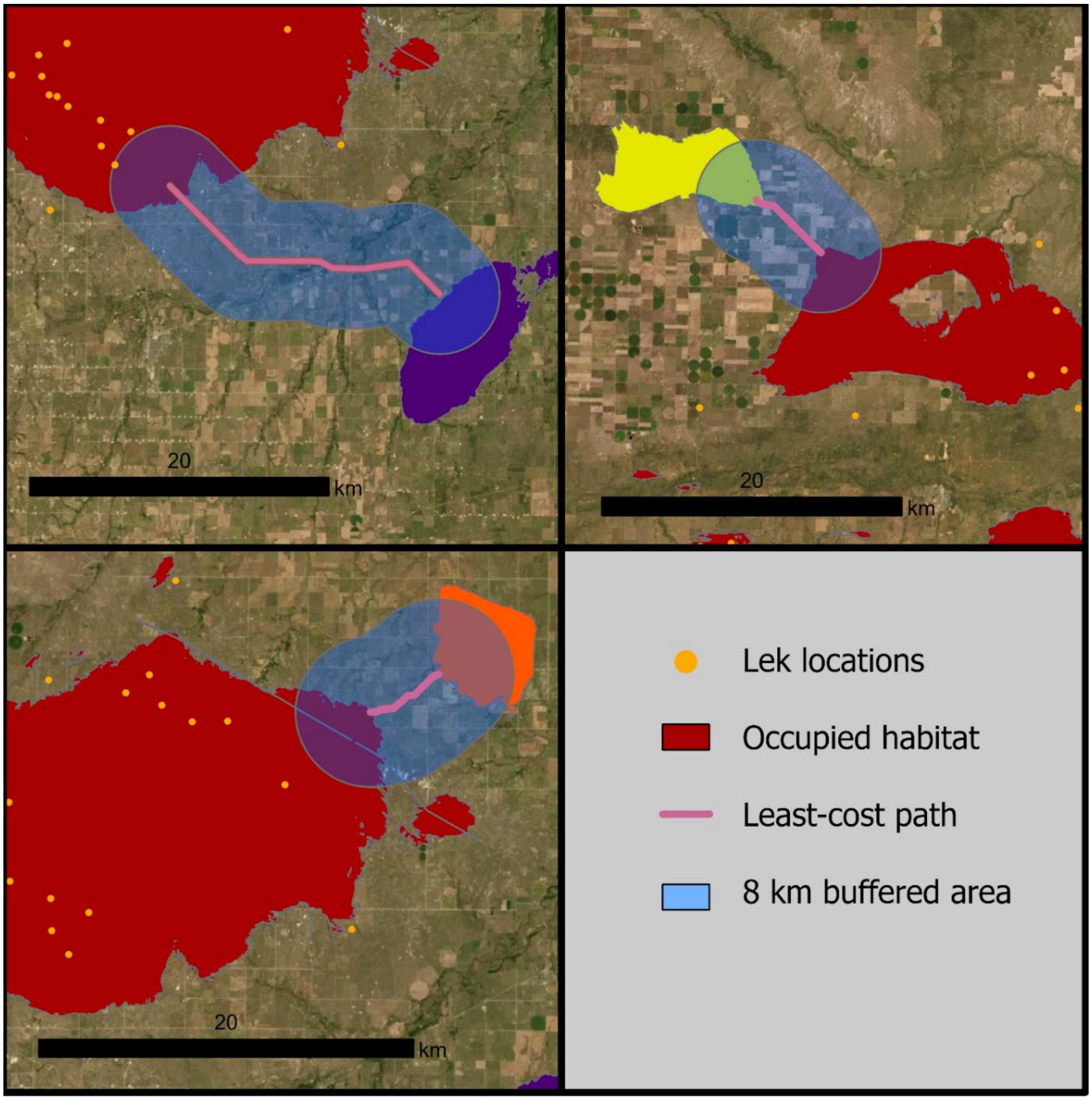

Figure 17. Map delineating identified areas of potentially suitable, but unoccupied habitat in the southern mixed-grass prairie ecoregion, least-cost path between identified areas and the nearest occupied habitat, and the 8-km wide buffered areas recommended for target habitat restoration efforts in 1) Woodward and Ellis counties in Oklahoma (purple), 2) Seward county in Kansas and Beaver county in Oklahoma (yellow), and 3) in Ellis county in Oklahoma (orange).

CHAPTER THREE

A COMPARISON OF FIELD-BASED HABITAT ASSESSMENT TOOLS FOR LESSER PRAIRIE-CHICKEN CONSERVATION

Introduction

The lesser prairie-chicken (*Tympanuchus pallidicinctus*; hereafter "prairie-chicken") is a prairie grouse species endemic to the southern Great Plains and is found in Colorado, Kansas, Oklahoma, Texas, and New Mexico (Taylor and Guthery 1980, Hagen et al. 2004, Garton et al. 2016). Once numbering in the millions, prairie-chicken populations have declined by ~85% during the last century and now only occur in isolated populations across four disjunct ecoregions: the mixed-grass, short-grass/CRP, sand sagebrush, and shinnery oak (McDonald et al. 2014, Garton et al. 2016; Fig. 1). Range-wide population declines have resulted in recent reconsideration for listing under the Endangered Species Act after being removed in response to a judicial decision in September 2015 (USFWS 2016; 2020). Declines in prairie-chicken populations have been attributed to habitat fragmentation and loss resulting from cultivation (Woodward et al. 2001), energy development (Hagen et al. 2011, Plumb et al. 2019), droughts and increased temperatures (Grisham et al. 2013, McDonald et al. 2014), and mismanagement of rangelands resulting in unsuitable vegetation structure and tree encroachment (Fuhlendorf et al. 2002, Lautenbach et al. 2017, Kraft et al. 2021).

Approximately 94% of the prairie-chickens' distribution occurs on private land; thus, conservation programs with strong partnerships between private landowners and resource managers has been essential to increasing the amount and quality of available prairie-chicken habitat. For example, the Lesser Prairie-Chicken Initiative (LPCI) administered by the USDA

Natural Resources Conservation Service (NRCS) uses current Farm Bill conservation programs to provide financial and technical assistance to landowners for implementing conservation practices to improve prairie-chicken habitat (USDA 2016). The five states in the prairie-chicken range (Kansas, Colorado, Oklahoma, Texas, New Mexico) developed the Lesser Prairie-Chicken Range-wide Conservation Plan (RWP) to provide biological goals for the implementation of conservation efforts to improve prairie-chicken habitat (Van Pelt et al. 2013). Additionally, the RWP includes a voluntary mitigation framework for development in the prairie-chicken range; this framework is administered by the Western Association of Fish and Wildlife Agencies (WAFWA). Within the mitigation framework, WAFWA uses mitigation funds paid by energy and telecommunications industries to incentivize landowners to implement land management practices that improve prairie-chicken habitat (Van Pelt et al. 2013). These practices include cropland to grassland restoration, heterogeneity-based prescribed grazing and fire management, planting native grasses and forbs, and brush management (Van Pelt et al. 2013, USDA 2020).

Targeted conservation programs for prairie-chickens require efficient and practical methods for accurately quantifying and monitoring habitat. Field-based indices used to quantify prairie-chicken habitat often include landscape-scale evaluations of the amount of potential habitat available combined with fine-scale assessments of vegetation conditions to quantify reproductive habitat quality (Morrison et al. 2013, Van Pelt et al. 2013, McNew et al. 2017, Gehrt et al. 2020). For example, as part of the RWP, WAFWA developed the Habitat Evaluation Guide (HEG) to quantify and monitor prairie-chicken habitat quality on potential and enrolled private lands. Landowners who are enrolled in WAFWA's RWP are incentivized to increase and maintain prairie-chicken habitat because their annual payment is based on the total habitat provided (i.e., quantity) and a measure of habitat quality (i.e., HEG score). The HEG scores of a property are based on four habitat variables known to be associated with prairie-chicken reproductive success: vegetation cover (non-overlapping canopy cover), vegetation composition, percent cover of tall woody plants, and availability of potential habitat in the surrounding area (Van Pelt et al. 2013). Akin to habitat suitability indices, HEG habitat variables and their respective scoring classifications are predetermined and qualitatively developed by prairie-chicken experts. While some researchers have questioned the validity of using HSI models due to issues related to subjectivity (Roloff and Kernohan 1999), they are an efficient and often robust tool used for quantifying species habitat and are commonly used to make conservation decisions (USFWS 1981). As such, field-based assessments like the HEG have been valuable for prairie-chicken conservation and developing targeted habitat management plans on conservation properties.

Concurrent to the development of qualitative HEG criteria, recent field-based research has focused on quantifying conditions that describe prairie-chicken habitat quality in relation to measures of habitat use and demography (Hagen et al. 2013, Larsson et al 2013, Lautenbach et al. 2015, Jarnevich et al. 2016, Spencer et al. 2017, Gulick 2019, Plumb et al. 2019, Sullins et al. 2019, Kraft et al. 2021). For example, recent research quantifying the abundance of reproductive habitat used criteria developed by Lautenbach (2015) to classify randomly selected points within 5 km of lek locations as having sufficient vegetation conditions for successful nesting and brood-rearing (Gehrt et al. 2020). Research-based estimates of habitat quality are thought to provide the best information as they often relate species vital rates and occurrence to specific habitat conditions, such as vegetation cover, which may then be used to identify more resolute habitat targets for prioritizing prairie-chicken conservation. However, research-based methods of field data collection are often data-hungry, labor-intensive, and logistically unrealistic to conduct at large scales. Nonetheless, a critical comparison of both WAFWA's HEG and research-based assessments could elucidate better ways to quantify prairie-chicken habitat.

Due to limited time and labor resources, habitat assessments, including those of WAFWA, are typically conducted only once a year during the nesting period (Van Pelt et al. 2013). As seasonal changes in precipitation and management practices influence range condition (e.g., species composition and vegetation structure), providing private landowners and conservation organizations with easy-to-use tools to roughly monitor changes in vegetation conditions could potentially benefit rangeland managers in maintaining habitat for prairie-chickens year-round. Ecological site descriptions, an already commonly used component in rangeland monitoring and management (Herrick et al. 2006), may have application in monitoring reproductive habitat for prairie-chickens. Ecological site descriptions describe the climate, soil, hydrology, vegetative dynamics, and the historical plant community of an area (NRCS 2003), all of which can constrain prairie-chicken use and vital rates (Fuhlendorf and Engle 2001, Van Pelt et al. 2013, Grisham et al. 2016, Kraft 2016). Thus, ecological sites and their relative condition (formerly 'range condition') may have broad applicability in identifying and ranking prairie-chicken habitat quality. While researchers have classified ecological sites in terms of their capacity to support reproductive habitat for prairie-chickens (Van Pelt et al. 2013), limited research has investigated the associations among ecological sites, relative condition, and important prairie-chicken population measures (but see Anderson et al. 2015, Kraft 2016) and no research has linked these rangeland indicators to measures of the amount, type, and quality of prairie-chicken habitat.

A main challenge of wildlife conservation is identifying and quantifying the quality of habitats that are essential to the long-term persistence or recovery of an at-risk species (Morrison et al. 2013). To further conservation efforts, I compared field-based assessment tools for quantifying prairie-chicken habitat in order to provide information that will assist in delivering more appropriate qualifications of available prairie-chicken habitat. Specifically, my objectives were to 1) summarize and compare the amount and quality of available prairie-chicken habitat resulting from research-based habitat assessments and the HEG, 2) examine associations between common fine-scale vegetation measurements that describe nesting and brood-rearing habitat and the relative condition of ecological sites, and 3) provide information for potentially integrating field-based assessments to improve qualifications of available habitat for prairie-chicken conservation.

Study Area

My field study was located on a 174 km^2 (~43,000 acres) private bison production ranch in the Gypsum Hills of southcentral Kansas and northern Oklahoma (Fig. 18). The property and surrounding area once supported a modest prairie-chicken population; however, populations were extirpated in the early 2000's, likely due to the loss and fragmentation of grasslands resulting from the expansion of eastern red cedar (*Juniperus virginiana*) and energy development (C. Kruse, Turner Enterprises, Inc.; personal communication). In 2014, the property enrolled in WAFWA's mitigation program as a prairie-chicken conservation property and implemented management practices designed to improve prairie-chicken habitat, including the mechanical removal of eastern red cedar and other woody species across ~100 km^2 (24,000 acres) of land, and the implementation of a prescribed fire plan (WAFWA 2014). Additionally, the Anderson Creek wildfire burned 90% of the property in 2016, removing woody debris from tree cutting work and rejuvenating native grasses and forbs. Vegetation on the study site can be characterized as a mixture of sand sagebrush *(Artemisia filifolia)* and mid-height perennial grasses and forbs such as big bluestem (*Andropogon gerardii*) and switchgrass (*Panicum virgatum*). Woody vegetation includes sand plum (*Prunus spp.)*, cottonwood *(Populus deltoides*), and eastern red cedar. Upland soils are typically clayey and precipitation in the area ranges between 60–70 cm annually (USDA, Natural Resource Conservation Service, esis.sc.egov.usda. gov/).

Methods

I conducted field surveys of potential habitat within the study site using methods described in the Lesser Prairie-Chicken Range-wide Conservation Plan (Van Pelt et al. 2013, WAFWA 2015) and similar methods outlined in previous research identifying habitat conditions that promote prairie-chicken survival and habitat use (Lautenbach 2015, Gehrt et al. 2020) and compared empirical estimates of habitat quality. In addition, I examined the potential effects of fire on my compared estimates of habitat quality by examining potential differences in vegetation measurements used to describe prairie-chicken habitat. Finally, to explore the potential for using ecological sites and relative condition to monitor reproductive habitat, I identified the ecological site and similarity index at each research-based habitat survey (USDA 2003).

Habitat Surveys

HEG Scores. Consistent with protocols in the HEG, I defined evaluation units (sampling strata) as similarly managed areas of homogenous vegetation (Van Pelt et al 2013, Appendix I); thus, I stratified evaluation units first by pasture, and then by ecological sites based on similar soil type (i.e., loamy, clayey, sandy; Fig. 18). I defined non-habitat as areas that fell within rough terrain, wetlands, and areas that are within 300 m of dense tree cover (i.e., river corridors, eastern red cedar ravines; Lautenbach et al. 2017). Four habitat variables make up the overall HEG score for an evaluation unit: 1) average vegetation cover (non-overlapping canopy cover), 2) vegetation composition, 3) percent cover of tall woody plants >3 ft (1 m) in upland sites, and 4) the proportion of grassland cover with <1% canopy cover of trees that is within a 1.6-km radius (~1 mile) of the geometric center of each evaluation unit (Van Pelt et al. 2013, WAFWA 2015; Table 8). Consistent with Van Pelt et al. (2013), I measured habitat variables 1–3 corresponding to nesting conditions during mid-May–mid June in 2020 and 2021. Transects were placed in areas found to be representative of the current plant community and structure throughout the entire evaluation unit. I collected vegetation measurements using line-point intercept sampling along a ~45-m (150-ft.) transect oriented northeast–southwest originating at the northeast end of the transect. Standing on the south side of the tape, I measured the tallest plant height within a ~15 cm (6-inch) radius at every 3-m (10-ft.) interval. I then estimated non-overlapping canopy cover (HEG habitat variable 1) and species composition (HEG habitat variable 2) at every 1-m (3-ft.) interval by lowering a wire-flag in a vertical descent through the foliar canopy directly on the north side of the tape and recorded the growth forms in the order that they touched the wire for up to six individual hits. I recorded big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), sideoats grama (*Bouteloua curtipendula*), Indiangrass (*Sorghastrum nutans*), and switchgrass (*Panicum virgatum*), as preferred grasses (PG) of prairie-chickens while all other grasses were recorded as either tufted grass (TG) or sod grass (SG; Van Pelt et al. 2013). I then recorded ocular estimates of tree cover in upland portions for each evaluation unit as having 0%, <1%, 1–5%, >5%, or being tilled (cropland) using methods described in Kansas Range Technical Note KS-8, (HEG habitat variable 3, WAFWA 2015). In addition, I measured visual obstructions readings (VOR) at every 6-m (20-ft.) interval by placing a Robel pole on the north side of the tape and recording the number of completely obstructed bands from a perpendicular distance of 2 m and a height of 0.5 m (Robel et al. 1970). Visual obstruction readings and plant height are not incorporated into the final HEG score of the property but are likely measured to provide baseline information for management recommendations (e.g., prescribed grazing).

To measure the fragmentation of an area (HEG habitat variable 4), I used modified protocols developed by WAFWA, (M. Houts, personal communication). I obtained landcover classifications at a 30-m^2 resolution from the 2016 National Landcover Data database (NLCD 2016). Landcover data were reclassified to obtain values of 'potential habitat' (1) and 'non-habitat' (0), where 'potential habitat' included any areas classified as grassland and shrubland, and 'non-habitat' included cultivated cropland, deciduous tree cover, and emergent herbaceous wetland. Shapefiles of roads were obtained from the Kansas GIS Data and Support Center (http://www.kansasgis.org) and the Oklahoma GIS Data Clearinghouse (http://www.okmaps.org) where highways were buffered to 50 m and county roads were buffered to 15 m (M. Houts, personal communication). I then merged all shapefiles of roads into one layer and converted it to raster dataset with a 30-m^2 resolution using the Feature to Raster tool in ArcGIS Pro (ESRI 2020). I reclassified all buffered highways and county roads as non-habitat. I then used cell statistics to obtain the minimum value for each cell from the reclassified NLCD habitat layer and the reclassified road buffers layers for my final Potential Habitat layer. Finally, I used the focal statistics tool in ArcGIS Pro (ESRI 2020) to quantify the percent 'potential habitat' (1) within a 1.6-km (1-mile) radius of the geometric center of each evaluation unit (M. Houts, personal communication

<u>Research-based Habitat Assessments.</u> I divided sampling periods into the nesting (mid-May–June) and brood-rearing (late-June–mid-July) periods (Hagen and Giesen 2005) which are thought to be the primary limiting periods for prairie-chickens (Hagen et al. 2009) Evaluation

units remained the same as those used to conduct habitat surveys under HEG protocol (Van Pelt et al. 2013). I generated 30 random points within each evaluation unit using the Create Random Points tool in ArcGIS Pro (ESRI 2020). I conducted vegetation surveys at an initial 5 of these randomly generated points and calculated the mean and standard deviation of proportional grass cover to identify adequate sample sizes needed to accurately reflect variation in vegetation measurements within each evaluation unit using the following equation:

$$n = (Z_\alpha)^2 (s)^2 \div (B)^2$$

where n is the uncorrected sample size needed, Z_α is the standard normal coefficient calculated for a confidence interval (1.28), s is the standard deviation, and B is the mean multiplied by a precision level of 0.15. I then compared the uncorrected sample size (n) to a table in Elzinga et al. (1998; Appendix 7) to evaluate whether my sampling effort was sufficient to describe within-evaluation unit variation. Logistical constraints dictated a maximum of ~20 random points per evaluation unit.

At each random point, I evaluated nesting and brood-rearing habitat conditions using methods adopted from previous studies of prairie-chickens in the mixed-grass prairie ecoregion (Lautenbach 2015, Lautenbach et al. 2019). I estimated non-overlapping canopy cover for all vegetation types (percent shrub, grass, forb) as well as litter and bare ground using a 60 × 60-cm quadrat located directly at the random point center and at 4 additional subsampling points located 4 m from the random point center in each cardinal direction. I also recorded VOR at the random points center from each cardinal direction at a distance of 4 m and a height of 1 m. Random points that fell within nonhabitat (e.g., rough terrain, water, etc.) were not used and I moved on to the next randomly selected point.

Ecological Sites and the Similarity Index. I identified the ecological site intersecting each random point and characterized the relative range condition within and immediately surrounding each random point (~15-meter radius) using a similarity index (USDA 2003, Herrick et al. 2006). I estimated similarity index values by visually estimating the current proportion of tallgrass production present, specifically big bluestem, little bluestem, Indiangrass, and switchgrass, and compared this proportional coverage to the amount of tallgrass production under each respective ecological sites reference climax plant community, where tallgrass production is expressed as a percentage by weight (biomass). Similarity index values were classified into 5 categories where 0–20% indicated low similarity, and 81–100% indicated high similarity.

Quantifying Available Prairie-chicken Habitat

HEG Scores. Consistent with the HEG protocol, I classified HEG scores for habitat variables 1–3 into 5 different categories ranging from 0.05 = low quality to 1.0 = high-quality prairie-chicken habitat (Van Pelt et al. 2013; Table 8). For HEG habitat variable 4, I split HEG scores into 10 classes, where areas that have 90 – 100% grass cover = 1.0 (i.e., 90 – 100% of the surrounding 1.6-km radius is classified as 'potential habitat') and areas that have 0–10% grass cover = 0.1 (Table 8). I calculated non-overlapping canopy cover (HEG habitat variable 1) by taking the total number of growth forms that first hit the wire flag divided by the total possible number of first hits (51). I defined scores for species composition (HEG habitat variable 2) by the relative cover of grasses and shrubs preferred by prairie-chickens, which I found by dividing the number of hits classified as PG or SS (sand sagebrush) and dividing it by the total number of grass and shrub hits (PG, TG, SG, SS, SH). To score habitat variable 4, I used my final Potential Habitat layer and extracted raster cell values from the cell at the geometric center of each evaluation unit (Fig. 19). Finally, I calculated final HEG scores for each evaluation unit by multiplying the score for HEG habitat variable 4 by the minimum value for HEG habitat variables 1–3 (Van Pelt et al. 2013). To calculate the number of squared kilometers (Van Pelt et al. 2013) of prairie-chicken habitat, I multiplied the final HEG score by the number of squared kilometers within each evaluation unit (Van Pelt et al. 2013).

Research-based Habitat Assessments. I classified random points into classes of habitat versus non-habitat using criteria from previous literature, where random points locations that had an average VOR between 1.5 – 3.5 dm and less than 10% bare ground were classifed as nesting habitat (Lautenbach 2015, Lautenbach et al. 2019, Gehrt et al. 2020). Similarly, points that had an average of 7% – 35% forb cover and VOR between 2.0 – 5.0 dm were classified as brood-rearing habitat (Table 9; Lautenbach 2015, Gehrt et al. 2020). Vegetation measurements below or above these ranges of conditions were classified as non-habitat for prairie-chicken nesting or brood-rearing.

Next, I calculated the proportion of nesting and brood-rearing habitat (T) potentially available for prairie-chickens within each evaluation unit for the years 2020 and 2021, using the following equations:

$$T_{nest} = \sum_{i=1}^{k} (\mathrm{N} \div X) \times \mathrm{K}$$

$$T_{brood} = \sum_{i=1}^{k} (B \div X) \times K$$

where *k* is the number of evaluation units sampled that year, N and B are the proportion of random points classified as optimal nesting and brood-rearing habitat, respectively, X is the total amount of random points, and K is the number of squared kilometers within each evaluation unit.

I then quantified the total amount of nesting and brood-rearing habitat available by summing up the number of squared kilometers classifed as suitable for nesting or brood-rearing from each evaluation unit (Gehrt et al. 2020). Previous research in the mixed-grass prairie has reported areas surrounding established lek locations to have ~ 25% suitable nesting and brood-rearing habitat (Gehrt et al. 2020); thus, I considered evaluation units to have sufficent nesting or brood-rearing habitat if ≥25% of random points surveyed were classified as suitable for prairie-chicken reproduction.

I examined potential effects of fire on the quality of prairie-chicken nesting and brood-rearing habitat by testing for differences in the mean and standard deviation (heterogeneity; Londe et al. 2020) of VOR and cover of bare ground (nesting), and VOR and forb cover (brood-rearing) between evaluation units that had experienced fire between the years 2019 – 2021, both paritally (i.e, only burned a portion of the evaluation unit) and throughout the evaluation units entire extent, to evaluation units that had not experienced fire using a Wilcoxon-Mann-Whitney U test. I set $\alpha = 0.05$ (Wilcoxon 1945, Mann and Whitney 1947).

Ecological Sites and the Similarity Index. I summarized the distriubtion of cover and VOR values for each ecological site sampled using simple box plots. I was unable to examine relationships between relative range condition and vegetation measurements known to affect prairie-chicken reproductive success, as there were few random points that were classified within moderate categories on my similarity index scale (i.e., 21 – 40%, 41 – 60%, 61 – 80% similarity). Thus, I could not reliably execute further analysis exploring the utility of using a similarity index as a tool for monitoring prairie-chicken habitat (Ramsey and Schafer 2012).

Results

HEG Scores. I completed 28 habitat surveys across 15 evaluation units during the nesting sampling period (mid-May – June). Final HEG scores for all evaluation units did not change between years 2020 and 2021 (Table 10). Scores for HEG habitat variable 4 contributed to overall high HEG scores with all but one unit receiving a score of 1.0, indicating >90% of the surrounding area of each evaluation unit was classified as potential prairie-chicken habitat (i.e., grassland and not cropland, forest or urban areas; Table 10; Fig. 19). Approximately 67 % of evaluation units received a final HEG score of 0.85–1.0, indicating that under the HEG habitat monitoring protocol the study area had good-to-excellent prairie-chicken habitat (Van Pelt et al. 2013; Fig. 20). Evaluation units with a final HEG score <1.0 generally had lower HEG scores due to low relative cover of preferred grasses and shrub species (HEG habitat variable 2). In total, I quantified 64 km^2 (15,814 acres) of available habitat for prairie-chickens using the HEG within surveyed areas (Table 10).

Research-based Habitat Assessments. During 2020 and 2021, I sampled 738 random points across 15 evaluation units during the nesting (324) and brood-rearing (414) sampling periods (Tables 11 & 12). Sample sizes within some evaluations units for the year 2020 were relatively low due to complications with identifying soil types prior to and during the 2020 field sampling season (Fig. 21; http://www.wildlifehabitatecologylab.com/8203evaluating-habitat-suitability-for-lesser-prairie-chick-reintroductions.html). Thus, I report results only from samples collected in 2021. Approximately 60% of all evaluation units sampled in 2021 were found to have ≥25% suitable nesting habitat (Table 11; Fig. 22) whereas only 25% of the evaluation units sampled in 2021 had ≥ 25% brood-rearing habitat (Table 12; Fig. 22). In total, I quantified 28.0 km^2 (6,920 acres) of potentially suitable nesting habitat and and 17.0 km^2 (4,224 acres) of potential brood-rearing habitat (Table 11 & 12). In general, there was a high amount of variability in all vegetation measurements except for average shrub cover and forb cover (Table 13 & 14). Wilcoxon-Mann-Whitney U-test revealed strong evidence for differences in median values of VOR (P = <0.001) and bare ground (P = <0.001) between evaluation units that had recently experienced fire (2019 – 2021) and those that had not during the nesting period (Table 15; Fig. 23). Similarly, there was strong evidence for a difference in median values of VOR (P = <0.001) and forb cover (P = <0.001) between evaluation units that had recently experienced fire (2019 – 2021) and those that had not during the brood-rearing sampling period (Table 16; Fig. 24). I found no difference in median values of standard deviation for average cover of bare ground (P = > 0.05) and average VOR (nesting; P = > 0.05) or average forb cover (P = > 0.05) and average VOR (brood-rearing; P = > 0.05) between evaluation units that recently experienced fire (<2 year post-fire) and evaluation units that had not (>6 year post-fire).

Ecological Sites and the Similarity Index. During 2020 and 2021, I evaluated ecological site conditions at 719 random points across 10 ecological sites during the nesting and brood-rearing sampling periods (Table 17). Most random points sampled were classified as either having a high similarity index value (42% of random points) or a low similarity index value (32% of random points; Table 18 & 19; Fig. 25). Only 26% of all random points were classified in the three remaining middle categories (i.e., 21 – 40%, 41 – 60%, 61 – 80% similarity). In general, ecological sites had similar distribution of values for shrub cover, grass cover, forb cover, bare ground, and VOR (Table 20 & 21; Fig. 26 & 27). However, ecological site R078CY114TX had lower VOR during both the nesting (6.08 ± 7.35 SD) and brood-rearing period (4.03 ± 3.97 SD; Table 20 & 21).

Discussion

Prairie-chicken populations are currently constrained by the amount and quality of available habitat (Haukos and Zavaleta 2016). Thus, developing and improving field-based methods to monitor and quantify habitat at local scales is essential to developing management plans to increase prairie-chicken numbers range-wide. In general, I found estimates of habitat quality under the HEG indicated the property had excellent habitat for prairie-chickens while estimates under my research-based assessments showed the property only had marginal habitat quality. Specifically, evaluation units that had recently experienced fire (<1 years post-fire) and evaluation units that had lower percent composition of vegetation preferred by prairie-chickens (e.g., little bluestem or sand sagebrush) had very different estimates of habitat quality between the two habitat assessment methods. Because these two methods do not use the same habitat variables to classify habitat, nor do they have the same habitat classification scheme (i.e., HEG scores vs. proportional number of random points classified as suitable nesting/brood-rearing habitat), it is impossible to make direct comparisons in the amount and quality of habitat. Nonetheless, each method provides insight into how we can better develop field surveys to measure habitat quality for prairie-chickens and managers should consider using components of both methods to assess habitat quality for prairie-chicken conservation efforts. In addition, managers should continue to use ecological sites to identify areas that may inherently provide habitat conditions suitable for prairie-chicken survival and reproductive success.

Evaluation units where HEG assessments indicated lower levels of habitat quality than research-based assessments were in areas with lower cover of native grass and shrub species preferred by prairie-chickens (HEG habitat variable 2). For example, although evaluation unit FL had high proportions of both nesting and brood-rearing habitat under the research-based assessments, it also had the lowest HEG score (0.25) on the property due to low percent cover of preferred grass and shrub species. In contrast, evaluation units that had recently experienced fire (<1 yr.) throughout their entire extent (e.g., NCC and SC) had high HEG scores but lower estimates of habitat quality with the research-based assessments due to lower VOR values and increased bare ground. Differences in estimates of relative habitat quality meant the total area identified as suitable habitat between the two methods were not correlated. Disparity in estimates of habitat quality between habitat assessment methods indicates managers should use both species composition and a measure of concealment (e.g., VOR) to accurately quantify habitat for prairie-chickens.

It is our understanding that habitat variables under the HEG were selected because they correlate with prairie-chicken demography but are not affected by annual variation in weather patterns out of the control of landowners (e.g., drought; Van Pelt et al. 2013, J. Pitman, former WAFWA conservation coordinator, personal communication). For example, species composition (i.e., HEG habitat variable 2) is unaffected by annual fluctuations in temperature and precipitation and, relative to non-native or exotic species, native species may provide suitable vegetation structure and food resources needed for prairie-chicken survival and reproductive success (J. Pitman, personal communication). Relative to exotic and non-native vegetation, native vegetation may provide increased resources needed for prairie-chicken survival and reproductive success (Rodgers and Hoffman 2005), including increased access to food resources (Hickman et al. 2006. Fulbright et al. 2013), improved cover for evading predators (Litt and Pearson 2013, Fulbright et al. 2013), and more heterogeneous vegetation conditions needed to support different life-history stages (Rodgers and Hoffman 2005, Haukos and Boal 2016, Sullins et al. 2018b). In contrast, VOR changes from year-to-year depending on differences in annual precipitation and management (e.g., grazing and fire; Starns et al. 2020, Lautenbach et al. 2021), but is also an important habitat variable to consider when developing targeted management strategies for prairie-chicken conservation due to its documented effects on habitat selection and reproductive success (Hagen et al. 2013, Lautenbach et al. 2019). While VOR is measured when conducting habitat surveys under the WAFWA HEG protocol, none of the four habitat variables that determine the final HEG score for an evaluation unit include this index. The omission of VOR from the calculation of HEG scores may give landowners a false indication of the amount and quality of reproductive habitat on their property. At our assessment site, for example, evaluation unit SC had an HEG score of 0.85 (i.e., excellent habitat quality), but no random points were classified as having adequate nesting or brood-rearing cover according to our research-based assessment. Managers should consider including important seasonally variable measures of habitat quality (e.g., VOR), as well as seasonally invariable measures (e.g., species composition), in habitat assessments for prairie-chickens.

Prairie-chickens require a variety of vegetation conditions to support various life-history stages such as nesting and brood-rearing (Haukos and Zavaleta 2016). As such, heterogeneity in vegetation conditions at the patch-level (i.e., within or among pastures) is important for providing the necessary arrangement and distribution of habitat needed to support successful reproduction in prairie-chicken populations (McNew et al. 2015, Haukos and Zavaleta 2016, Sullins et al. 2018, Winder et al 2017, Londe et al. 2020, Lautenbach et al. 2021). Apart from evaluation units that had experienced fire throughout their entire extent just prior to the 2021 sampling period (e.g., SC, NCC), I found a large amount of variability in vegetation measurements within evaluation units when using my research-based habitat assessments. Variability in vegetation measurements within evaluation units likely resulted from 1) inherent differences in vegetation composition and structure across different ecological sites (Herrick et al. 2006) and 2) the use of patch-burn grazing management techniques (Fuhlendorf et al. 2009). While I found no evidence for differences in the variability in vegetation measurements between evaluation units that had partially experienced fire and those that had not, evaluation units that had both suffiicient nesting and brood-rearing habitat under my research-based habitat assessments were in areas that had experienced recent small-scale fires (<2 years) and increased grazing pressure (e.g., SPL, SRC, YC) which could have provided increased heterogeneity on the landscape. My results are consistent with previous research demonstrating postive effects of patch-burn grazing ($\leq$5 km^2) on reproductive habitat for prairie-chickens (Fuhlendorf and Engle 2001, Fuhlendorf et al. 2009, Gulick 2019, Starns et al. 2020). For example, Lautenbach et al. (2021) found prairie-chickens selected for areas with greater time since fire ($\geq$4 year post-fire) during the nesting period and 1-year post-fire during the brood-rearing period suggesting it is important to maintain availability of an array of time-since-fire habitat patches on the landscape for prairie-chicken reproduction.

Ecological site descriptions may have many advantages to delineating habitat and identifying conservation actions for wildlife species. However, while it is probable that ecological sites have potential application in identifying course classes of habitat suitability for prairie-chickens, they are unlikely to describe habitat quality (Doherty et al. 2011, Kraft 2016). Yearly variation in rangeland management activities (e.g., prescribed fire and grazing) and local environmental conditions (e.g., precipitation and tree invasion) interact to influence vegetation composition and structure, producing differing range (similarity index) and habitat conditions. While a few studies have investigated relationships among ecological sites and prairie grouse habitat availability and use (Anderson et al. 2015, Kraft 2016), none have examined relationships between potential differences in range condition (similarity index) among ecological sites and fine-scale vegetation conditions that describe prairie grouse nesting and brood-rearing habitat. I attempted to examine associations between the relative condition (similarity index) of various ecological sites and fine-scale vegetation measurements that describe nesting and brood-rearing habitat. However, I was unable to reliably complete further analyses exploring the utility of using similarity index as a tool for monitoring reproductive habitat for prairie-chickens because there were few random points that were classified within the moderate categories of similarity index (e.g., 21 – 40%, 41 – 60%, 61 – 80% similarity). Nonetheless, as ecological sites differ from one another in vegetation composition and production, it is likely some ecological sites have a greater abundance of reproductive habitat for prairie-chickens than others (Van Pelt et al. 2013, Anderson et al. 2015, Kraft 2016). In addition, landscapes that have a combination of ecological sites that differ in production and vegetation composition may provide the necessary resources needed to support both nest and brood survival (Van Pelt et al. 2013). For example, on my study site, ecological site R078CY065OK (soil type = clayey) which has increased production of tallgrass species suitable for nesting was often associated with ecological site R078CY114TX (soil type = clayey) which has higher forb cover and increased bare ground to potentially support brood survival.

Management Recommendations

Based on the discrepancy in my estimates of habitat quality using research-based methods and the HEG, I recommend incorporating estimates of both species composition and a measure of concealment (VOR) into future habitat assessments to avoid producing erroneous estimates of available prairie-chicken habitat. Habitat assessments that include a habitat variable that takes into account species composition (e.g., similar to HEG habitat variable 2) and a range of VOR values (e.g., 1.5 – 4 dm) that are indicative of vegetation structure selected for by prairie-chickens would assist in providing more accurate delineations of the amount and quality of potential prairie-chicken habitat and further incentivize landowners to manage for vegetation structure that improves annual prairie-chicken survival. In addition, as heterogeneity in vegetation conditions at the patch-level has consistently been identified as being important to prairie-chicken survival and habitat use, I recommend developing a measure of heterogeneity when assesssing habitat quality within or among evaluation units. For example, similar to habitat variable 3 in WAFWAs HEG, there is potential to include ocular estimates of heterogeneity based on vegetation structure within each evaluation unit (Godina-Alvarez et al. 2009). In addition, continuing advancements in GIS technology may allow for the use of LiDAR and/or NDVI data to measure heterogeneiety within vegetation strucutre. For example, researchers have developed surface roughness maps based on vegetation heights using LiDAR data to distinuish between burned and unbruned areas in the sagebrush steppe of Idaho (Streuker and Glenn 2006).

Table 8. Habitat Evaluation Guide (HEG) classification scores for habitat variables 1 – 4 used to quantify the amount and quality of available lesser prairie-chicken habitat when conducting habitat assessments using protocols outline in the Western Association of Fish and Wildlife Agencies Range-wide Conservation Plan (Van Pelt et al. 2013, WAFWA 2015).

Habitat variable 1 – 3				**Habitat variable 4**	
Score	**Vegetation cover (canopy cover)**	**Vegetation composition**	**Percent cover of tall woody plants > 3 ft. tall**	**Score**	**Proportion of area within a 1.6-km radius in grass cover**
1.0	>45%	>75%	0	1.0	>90%
0.85	31 – 45%	51 – 75%	<1%	0.9	80 – 89%
0.60	15 – 30%	25 – 50%	1 – 5%	0.8	70 – 79%
0.25	<15%	<25%	>5%	0.7	60 – 69%
0.05	Tilled	Tilled	Tilled	0.6	50 – 59%
				0.5	40 – 49%
				0.4	30 – 39%
				0.3	20 – 29%
				0.2	10 – 19%
				0.1	1 – 9%
				0.0	<1%

Adapted from Van Pelt et al. 2013.

Table 9: Fine-scale habitat quality criteria for classifying nesting and brooding habitat when using research-based habitat assessment protocols for quantifying the amount and quality of prairie-chicken habitat in the mixed-grass prairie of southcentral Kansas (Lautenbach 2015, Lautenbach et al. 2019).

	Nesting		**Brooding**	
Habitat variable	**Unsuitable**	**Suitable**	**Unsuitable**	**Suitable**
% Forb cover	-	-	0 –10%; >35%	10 – 35%
% Bare ground	>10%	≤10%	-	-
VOR* (dm)	0 – 1.5; 5+	1.5 – 3.5	0 – 2.0; >5	2.0 – 5.0

*Visual obstruction reading

Table 10**.** Habitat Evaluation Guide (HEG) scores for all evaluation units sampled during the nesting sampling period in the mixed-grass prairie of southcentral Kansas in 2020 and 2021 using methods outlined in the Western Association of Fish and Wildlife Agencies Range-wide Conservation Plan (Van Pelt et al. 2013, WAFWA 2015).

Evaluation Unit	**Pasture**	**Min. HEG score for habitat variables 1-3**	**HEG score for habitat variable 4**	**Final HEG score**	**Available habitat (km^2)**
DL	Deadman	0.85	0.9	0.765	1.8
DC	Deadman	0.6	1.0	0.6	4.5
FL	Fuller	0.25	1.0	0.25	0.6
JL	Johnson	1.0	1.0	1.0	9.5
JC	Johnson	1.0	1.0	1.0	7.1
NCC	N. Cottage Creek	1.0	1.0	1.0	8.3
SC	Sandy Compressor	0.85	1.0	0.85	7.0
SCL	S. Cottage Creek	0.85	1.0	0.85	1.8
SCC	S. Cottage Creek	0.85	1.0	0.85	2.3
SPL	Swan Pond	0.85	1.0	0.85	5.1
SPC	Swan Pond	0.6	1.0	0.6	2.2
SRL	S. River	0.85	1.0	0.85	3.0
SRC	S. River	0.85	1.0	0.85	8.1
YL	Yellowstone	0.25	1.0	0.25	0.3
YC	Yellowstone	0.6	1.0	0.6	2.7

Table 11. Summarizing the number of points sampled per evaluation unit during the nesting sampling period, the proportion of random points classified as suitable nesting habitat, and the estimated number of squared kilometers classified as suitable nesting habitat for prairie-chickens during years 2020 and 2021 in the mixed-grass prairie of southcentral Kansas using research-based habitat assessments.

2020					2021				
Eval. unit	**Pasture**	**Total points sampled**	**Proportion suitable**	**Squared km of nesting habitat (acres)**	**Eval. Unit**	**Pasture**	**Total points sampled**	**Proportion suitable**	**Squared km of nesting habitat (acres)**
DC	Deadman	12	0.25	1.3 (326.9)	DC	Deadman	15	0.40	2.15 (523.1)
DL	Deadman	6	0.67	2.0 (488.0)	FL	Fuller	18	0.55	3.6 (892.2)
FL	Fuller	9	0.45	2.9 (713.8)	JC	Johnson	16	0.63	4.5 (1097.8)
JC	Johnson	9	0.67	4.7 (1171.1)	JL	Johnson	15	0.40	0.9 (225.2)
JL	Johnson	4	0.50	1.1 (281.4)	NCC	N. Cottage	13	0.15	1.3 (315.6)
NCC	N. Cottage	12	0.33	2.8 (683.8)	SC	Sandy C.	12	0.00	0.0 (0.0)
NCL	N. Cottage	6	0.33	0.7 (169.5)	SCC	S. Cottage	11	0.09	0.2 (59.6)
SC	Sandy C.	12	0.0	0.0 (0.0)	SCL	S. Cottage	8	0.87	1.9 (458.7)
SL	Sandy C.	6	0.0	0.0 (0.0)	SRC	S. River	15	0.47	4.4 (1097.1)
SCC	S. Cottage	4	0.75	2.0 (492.0)	SRL	S. River	6	0.50	1.8 (438.9)
SCL	S. Cottage	5	0.80	1.7 (419.4)	SPC	Swan Pond	10	1.00	3.7 (914.8)
SRC	S. River	20	0.30	2.8 (705.2)	SPL	Swan Pond	10	0.20	1.2 (294.9)
SRL	S. River	7	0.28	1.0 (250.8)	YC	Yellowstone	10	0.50	2.2 (552.8)
SPC	Swan Pond	4	0.0	0.0 (0.0)	YL	Yellowstone	10	0.20	0.2 (49.6)
SPL	Swan Pond	9	0.10	0.7 (163.8)					
TSL	Two Sign	10	0.20	1.4 (345.1)					
YC	Yellowstone	17	0.58	2.6 (650.4)					
YL	Yellowstone	3	0.67	0.6 (165.4)					

Table 12. Summarizing the number of points sampled per evaluation unit during the brood-rearing sampling period, the proportion of random points classified as suitable brood-rearing habitat, and the estimated number of squared kilometers classified as suitable brood-rearing habitat for lesser prairie-chickens during years 2020 and 2021 in the mixed-grass prairie of southcentral Kansas using research-based habitat assessments.

2020					2021				
Eval. unit	**Pasture**	**Total points sampled**	**Proportion suitable**	**Squared km of brood-rearing habitat (acres)**	**Eval. unit**	**Pasture**	**Total points Sampled**	**Proportion suitable**	**Squared km of brood-rearing habitat (acres)**
DC	Deadman	18	0.50	2.6 (653.9)	DC	Deadman	11	0.18	1.0 (237.8)
DL	Deadman	9	0.67	0.99 (244.0)	DL	Deadman	13	0.08	0.2 (56.3)
FC	Fuller	9	0.55	0.5 (129.1)	FL	Fuller	20	0.25	1.6 (401.5)
FL	Fuller	9	0.88	5.8 (1427.6)	JC	Johnson	18	0.22	1.6 (390.4)
JC	Johnson	9	0.22	1.6 (390.4)	JL	Johnson	13	0.23	0.5 (129.9)
JL	Johnson	5	0.0	0.0 (0.0)	NCC	N. Cottage	22	0.05	0.4 (93.3)
NCC	N. Cottage	16	0.15	1.3 (323.9)	SC	Sandy C.	12	0.0	0.0 (0.0)
SC	Sandy C.	12	0.0	0.0 (0.0)	SRC	S. River	13	0.38	3.7 (904.2)
SL	Sandy C.	7	0.0	0.0 (0.0)	SRL	S. River	9	0.44	1.6 (390.1)
SCC	S. Cottage	5	0.20	0.5 (131.2)	SPC	Swan Pond	11	0.0	0.0 (0.0)
SCL	S. Cottage	6	0.33	0.7 (174.7)	SPL	Swan Pond	11	0.72	4.4 (1072.3)
SRC	S. River	12	0.33	3.2 (783.6)	YC	Yellowstone	13	0.31	1.4 (340.2)
SRL	S. River	6	0.17	0.6 (146.3)	YL	Yellowstone	9	0.44	0.5 (110.3)
SPC	Swan Pond	5	0.0	0.0 (0.0)					
SPL	Swan Pond	13	0.23	1.4 (340.3)					
YC	Yellowstone	15	0.53	2.4 (589.6)					
YL	Yellowstone	5	0.40	0.40 (99.2)					

Table 13. Mean and standard deviation of vegetation measurements collected across evaluation units at each random point sampled with research-based habitat assessments during the nesting sampling period in 2021 in the mixed-grass prairie of southcentral Kansas.

Evaluation Unit	Mean Shrub	Mean Grass	Mean Forb	Mean Litter	Mean BG*	Mean VOR**
DC	0.0 ± 0.1	31.7 ± 22.6	9.8 ± 7.9	31.5 ± 20.7	26.2 ± 26.4	13.8 ± 15.2
FL	3.2 ± 6.0	51.6 ± 21.6	6.8 ± 4.9	27.9 ± 14.0	10.7 ± 11.9	20.5 ± 9.1
JC	0.0 ± 0.0	45.6 ± 23.4	10.0 ± 6.3	25.2 ± 20.0	18.3 ± 28.4	15.4 ± 10.0
JL	1.0 ± 2.3	32.5 ± 19.3	12.1 ± 9.0	50.1 ± 25.2	4.3 ± 5.6	17.6 ± 12.3
NCC	0.0 ± 0.0	44.2 ± 13.3	17.6 ± 11.8	12.2 ± 10.1	26.1 ± 19.8	13.6 ± 7.2
SC	0.0 ± 0.0	34.9 ± 21.3	7.5 ± 4.9	14.1 ± 14.1	43.9 ± 25.5	8.5 ± 5.9
SCC	0.0 ± 0.0	16.5 ± 14.3	6.6 ± 5.7	41.7 ± 23.7	33.6 ± 23.8	4.8 ± 6.3
SCL	1.6 ± 2.2	59.4 ± 11.6	5.4 ± 3.8	29.4 ± 9.8	4.6 ± 6.5	23.3 ± 3.7
SPC	0.0 ± 0.0	45.8 ± 24.4	6.2 ± 2.7	3.6 ± 4.5	44.2 ± 12.2	6.5 ± 2.1
SPL	1.2 ± 1.7	65.5 ± 12.5	5.7 ± 8.1	10.6 ± 11.6	16.8 ± 17.5	24.1 ± 14.7
SRC	0.0 ± 0.25	46.0 ± 24.4	7.9 ± 4.4	24.7 ± 18.5	22.0 ± 26.1	17.2 ± 9.8
SRL	0.9 ± 2.2	69.6 ± 13.2	6.7 ± 7.3	19.0 ± 3.1	3.7 ± 4.8	30.0 ± 11.3
YC	0.0 ± 0.1	34.8 ± 16.8	20.5 ± 8.4	34.8 ± 17.8	9.6 ± 15.1	12.7 ± 6.4
YL	0.0 ± 0.0	30.0 ± 21.2	21.3 ± 10.2	45.6 ± 17.8	4.3 ± 5.3	9.5 ± 4.8

*BG = Bare ground

**Visual obstruction reading

Table 14. Mean and standard deviation of vegetation measurements collected across evaluation units at each random point sampled with research-based assessments during the brood-rearing sampling period in 2021 in the mixed-grass prairie of southcentral Kansas.

Evaluation Unit	**Mean Shrub**	**Mean Grass**	**Mean Forb**	**Mean Litter**	**Mean BG***	**Mean VOR****
DC	0.3 ± 0.8	46.6 ± 12.6	7.1 ± 4.8	25.6 ± 18.2	21.5 ± 21.8	15.2 ± 7.8
FL	5.8 ± 8.1	42.2 ± 21.4	11.34 ± 9.4	29.9 ± 13.1	10.9 ± 12.8	30.2 ± 14.8
JC	0.0 ± 0.0	52.1 ± 19.5	13.8 ± 8.1	16.8 ± 9.4	17.1 ± 20.6	22.3 ± 16.3
JL	1.0 ± 2.6	53.6 ± 14.1	11.1 ± 7.6	29.7 ± 9.3	5.0 ± 5.7	20.8 ± 8.7
NCC	0.0 ± 0.2	32.6 ± 20.2	9.6 ± 11.5	16.1 ± 17.4	41.5 ± 30.8	9.0 ± 10.7
SC	0.0 ± 0.0	47.3 ± 24.4	7.4 ± 8.6	9.5 ± 7.4	35.5 ± 30.0	12.7 ± 12.9
SCC	0.0 ± 0.0	22.4 ± 8.1	5.9 ± 2.6	8.5 ± 4.6	63.25 ± 11.4	4.8 ± 3.9
SCL	1.4 ± 3.6	61.4 ± 10.4	5.7 ± 4.5	24.4 ± 9.5	7.2 ± 7.0	24.2 ± 10.2
SPC	0.7 ± 2.4	48.7 ± 19.2	7.4 ± 5.0	4.8 ± 4.9	37.6 ± 18.2	13.5 ± 5.6
SPL	0.1 ± 0.3	69.8 ± 8.3	10.3 ± 5.2	11.7 ± 5.2	8.1 ± 7.2	31.3 ± 10.3
SRC	0.8 ± 2.2	54.6 ± 9.6	8.5 ± 4.2	20.2 ± 7.6	15.5 ± 12.0	29.1 ± 14.1
SRL	0.16 ± 0.47	15.5 ± 12.4	12.4 ± 12.4	23.8 ± 12.8	1.6 ± 1.6	40.2 ± 11.5
YC	0.0 ± 0.0	11.0 ± 8.4	8.4 ± 8.9	25.2 ± 9.1	7.0 ± 14.3	25.8 ± 9.8
YL	0.0 ± 0.0	10.6 ± 11.9	27.8 ± 11.9	20.3 ± 9.6	0.8 ± 1.6	31.1 ± 9.7

*BG = Bare ground

**Visual obstruction reading

Table 15. Mean and standard deviation of vegetation measurements collected across evaluation units that had recently experienced fire (2019 – 2021) and those that had not when conducting research-based habitat assessments during the nesting sampling period in 2020 and 2021 in the mixed-grass prairie of southcentral Kansas.

Plot	Year	Mean Shrub	Mean Grass	Mean Forb	Mean Litter	Mean BG*	Mean VOR**
Fire	2020	0.3 ± 0.9	38.7 ± 24.5	16.3 ± 17.3	22.7 ± 19.2	22.4 ± 24.3	17.2 ± 16.6
Non-fire	2020	0.5 ± 2.9	54.98 ± 22.2	14.2 ± 11.4	19.8 ± 14.4	10.9 ± 18.3	21.7 ± 12.8
Fire	2021	0.1 ± 0.6	38.7 ± 21.8	11.7 ± 9.4	24.3 ± 20.9	25.6 ± 23.8	12.5 ± 10.5
Non-fire	2021	1.4 ± 3.7	48.2 ± 22.3	8.7 ± 6.8	31.8 ± 20.5	9.7 ± 16.8	19.8 ± 10.5

*BG = Bare ground

**Visual obstruction reading

Table 16. Mean and standard deviation of vegetation measurements collected across evaluation units that had recently experienced fire (2019 – 2021) and those that had not when conducting research-based habitat assessments during the brood-rearing sampling period in 2020 and 2021 in the mixed-grass prairie of southcentral Kansas.

Plot	Year	Mean Shrub	Mean Grass	Mean Forb	Mean Litter	Mean BG*	Mean VOR**
Fire plots	2020	0.1 ± 1.4	37.4 ± 11.9	10.7 ± 8.9	5.8 ± 10.3	46.0 ± 17.7	7.5 ± 4.7
Non-fire plots	2020	0.2 ± 1.5	60.4 ± 21.5	11.3 ± 8.2	16.3 ± 14.6	11.6 ± 20.5	27.7 ± 14.7
Fire plots	2021	0.1 ± 1.0	40.4 ± 22.4	9.7 ± 11.0	9.7 ± 10.8	40.1 ± 27.6	11.9 ± 10.5
Non-fire plots	2021	1.3 ± 3.9	53.2 ± 17.2	10.2 ± 8.1	23.4 ± 11.8	11.9 ± 17.3	25.5 ± 14.1

*BG = Bare ground

** Visual obstruction reading

Table 17. Summary of ecological sites sampled to assess potential relationships between the ecological sites relative condition and common fine-scale vegetation condition used to describe lesser prairie-chicken habitat during both the nesting and brood-rearing sampling periods in 2020 and 2021 in the mixed-grass prairie of southcentral Kansas.

Ecological Site Sampled	**Family Particle Size**	**Num. Random Points (nesting)**	**Num. Random Points (brood-rearing)**
Red Clay; R078CY065OK	Clayey	120	166
Sandy Loam; R079XY122KS	Loamy	54	63
Loamy Upland; R080AY056OK	Loamy	33	70
Red Shale; R078CY114TX	Clayey	36	34
Sand Hills; R079XY107KS	Sandy	16	31
Clayey Bottomland; R078CY094TX	Clayey	19	-
Loamy Bottomland; R078CY103TX	Loamy	10	-
Red Shale; R078CY083OK	Loamy	7	-
Rolling Sands; R080AY022OK	Sandy	-	13
Dune; R078CY014OK	Sandy	-	6

Table 18. Number of random points classified in each similarity index value when assessing potential relationships between the ecological sites relative condition and common fine-scale vegetation condition used to describe lesser prairie-chicken habitat using research-based habitat assessments in the mixed grass prairie of southcentral Kansas during the nesting sampling period for the years 2020 and 2021 combined.

Ecological site	**Number of Random Points Classified within Each Similarity Index Value**				
	0 – 20%	**21 – 40%**	**41- 60%**	**61 – 80%**	**81 – 100%**
R078CY056OK	4	4	2	3	18
R078CY065OK	34	9	10	10	54
R078CY083OK	3	0	0	0	4
R078CY094TX	8	0	0	0	5
R078CY103TX	4	1	0	2	3
R078CY114TX	11	5	6	3	12
R079XY107KS	9	4	1	1	2
R079XY122KS	9	6	5	8	26
R080AY056OK	16	7	1	2	6

Table 19. Number of random points classified in each similarity index value when assessing potential relationships between the ecological sites relative condition and common fine-scale vegetation conditions used to describe lesser prairie-chicken habitat using research-based habitat assessments in the mixed grass prairie of southcentral Kansas during the brood-rearing sampling period for the years 2020 and 2021 combined.

Ecological site	**Number of Random Points Classified within Each Similarity Index Value**				
	0 – 20%	**21 – 40%**	**41- 60%**	**61 – 80%**	**81 – 100%**
R078CY056OK	8	2	3	3	18
R078CY065OK	39	19	4	19	84
R078CY114TX	7	6	0	3	18
R079XY107KS	26	2	1	0	2
R079XY122KS	12	5	4	5	37
R080AY056OK	38	7	10	4	11
R078CY014OK	0	3	1	0	1

Table 20. Mean and standard deviation of vegetation measurements collected across ecological sites at each random point sampled in the mixed-grass prairie of southcentral Kansas with research-based assessments during the nesting sampling period in 2020 and 2021. Bold values indicate measurements that meet criteria for suitable nesting habitat under my research-based habitat assessments.

Ecological Site	**Mean shrub**	**Mean grass**	**Mean forb**	**Mean litter**	**Mean BG**	**Mean VOR (cm)**
R078CY056OK	1.14 ± 2.85	60.93 ± 19.71	8.96 ± 8.06	16.90 ± 9.43	12.03 ± 18.64	**28.57 ± 19.25**
R078CY065OK	0.10 ± 0.59	44.96 ± 21.47	11.20 ± 10.31	22.78 ± 18.76	20.88 ± 21.91	14.34 ± 10.30
R078CY083OK	3.77 ± 8.96	49.31 ± 21.36	14.06 ± 12.76	18.43 ± 13.81	13.85 ± 19.86	**17.28 ± 11.89**
R078CY094TX	0.14 ± 0.36	46.00 ± 30.67	15.95 ± 17.31	18.97 ± 22.08	19.14 ± 23.23	**17.04 ± 18.26**
R078CY103TX	0.34 ± 0.94	40.63 ± 23.22	12.83 ± 10.20	30.54 ± 14.96	14.52 ± 18.79	**18.35 ± 7.35**
R078CY114TX	0.00 ± 0.00	17.27 ± 14.89	15.49 ± 18.13	14.88 ± 16.09	54.84 ± 23.64	6.08 ± 7.35
R079XY107KS	0.00 ± 0.00	44.80 ± 24.36	19.44 ± 14.04	28.92 ± 24.64	**8.53 ± 14.43**	**20.22 ± 12.13**
R079XY122KS	2.62 ± 4.90	51.09 ± 19.51	11.10 ± 8.69	25.16 ± 14.75	**10.11 ± 11.95**	**21.50 ± 12.70**
R080AY056OK	0.07 ± 0.35	37.87 ± 23.46	15.26 ± 13.10	38.18 ± 24.85	**8.93 ± 15.65**	**16.18 ± 11.41**

Table 21. Mean and standard deviation of vegetation measurements collected across ecological sites at each random point sampled in the mixed-grass prairie of southcentral Kansas with research-based assessments during the brood-rearing sampling period in 2020 and 2021. Bold values indicate measurements that meet criteria for suitable brood-rearing habitat under my research-based habitat assessments.

Ecological Site	Mean shrub	Mean grass	Mean forb	Mean litter	Mean BG	Mean VOR (cm)
R078CY056OK	0.21 ± 0.74	61.81 ± 17.44	**10.17 ± 8.68**	17.28 ± 15.68	10.51 ± 15.20	**34.59 ± 16.36**
R078CY065OK	0.181 ± 1.02	51.70 ± 19.54	**10.08 ± 8.69**	15.11 ± 13.13	22.88 ± 23.06	**19.84 ± 14.27**
R078CY014OK	7.40 ± 10.21	38.77 ± 13.08	**9.23 ± 3.66**	31.00 ± 16.53	14.10 ± 14.15	**33.63 ± 16.99**
R080AY022OK	0.00 ± 0.00	49.65 ± 14.89	**11.96 ± 7.17**	29.38 ± 13.56	9.35 ± 9.59	**29.06 ± 6.49**
R078CY114TX	0.03 ± 0.17	21.31 ± 14.75	6.63 ± 4.12	5.91 ± 7.23	66.43 ± 20.66	4.03 ± 3.97
R079XY107KS	0.03 ± 0.18	56.05 ± 15.23	**14.10 ± 9.82**	16.99 ± 13.97	13.47 ± 14.72	17.82 ± 9.52
R079XY122KS	4.68 ± 9.19	56.75 ± 18.01	**9.42 ± 7.32**	19.45 ± 10.37	9.87 ± 13.27	**31.44 ± 14.55**
R080AY056OK	0.21 ± 1.18	52.63 ± 18.43	**13.41 ± 7.17**	22.95 ± 14.62	10.88 ± 16.75	**21.05 ± 11.22**

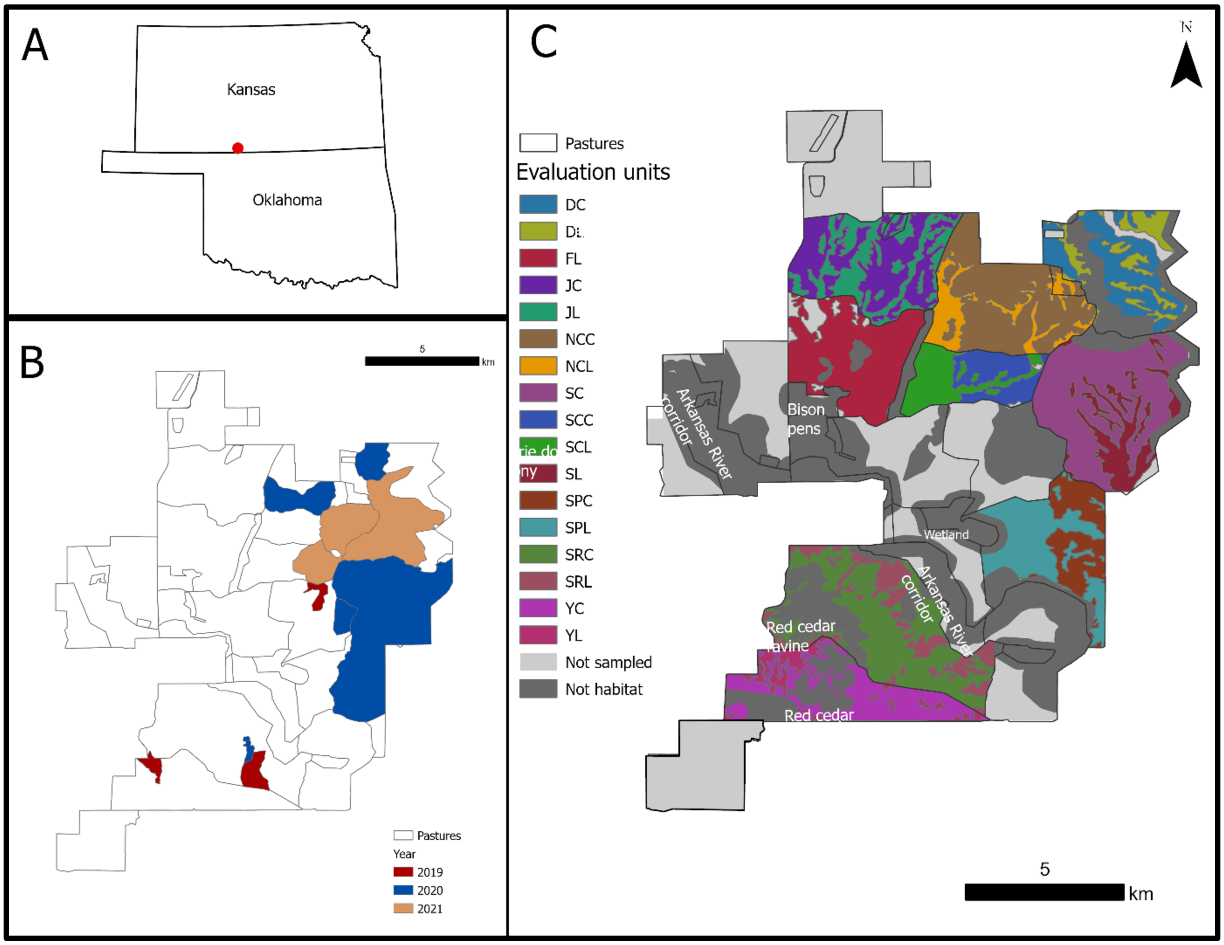

Figure 18. Maps delineating the (A) study site location, (B) the extent of recent fires (2019 – 2021) on the study site, and the (C) stratified evaluation units sampled during 2020 and 2021 in Barber and Comanche county in Kansas and Woods county in Oklahoma. Habitat surveys were conducted at each evaluation unit using methods similar to those outlined in previous research (Lautenbach 2015) and methods outlined in the Western Association of Fish & Wildlife Agencies Habitat Evaluation Guide (HEG).

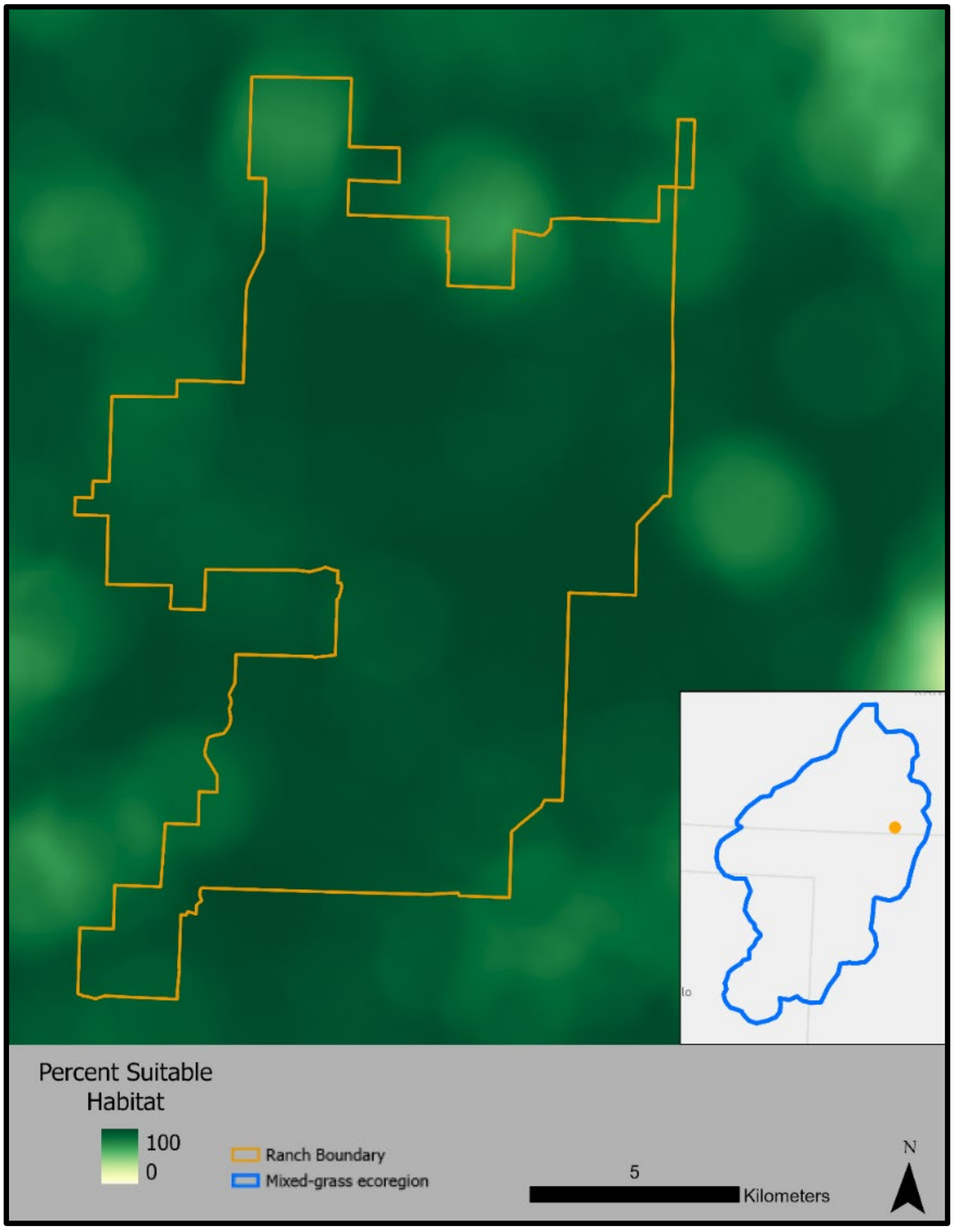

Figure 19. Relative availability of potential prairie-chicken habitat at the study site in the mixed-grassed prairie of southcentral Kansas and northwest Oklahoma as described by habitat variable 4 in the Western Association of Fish and Wildlife Agencies' Habitat Evaluation Guide. Availability of habitat was calculated as proportion of area within a 1-mile radius of each 30-m x 30-m cell in complete grass cover (Van Pelt et al. 2013).

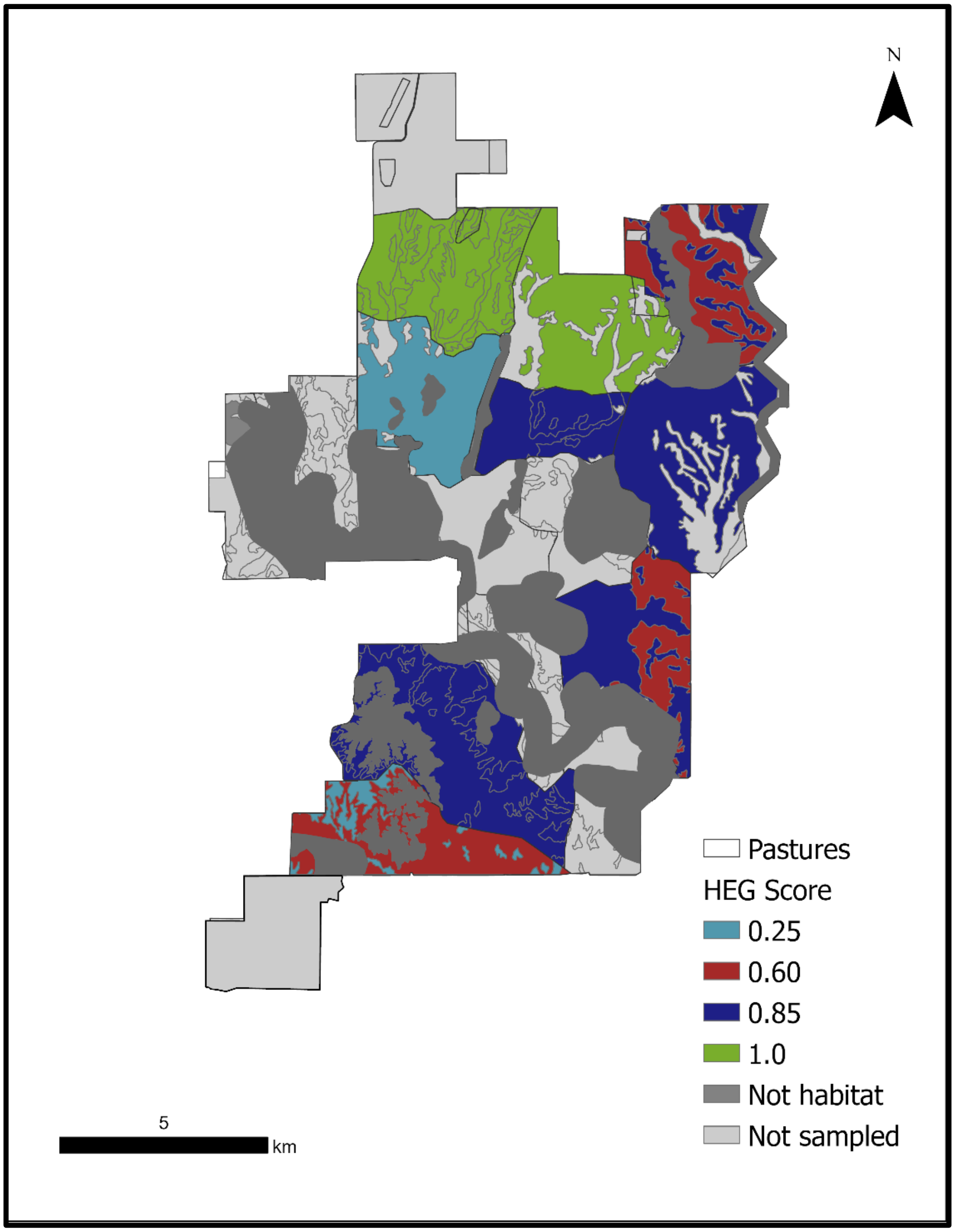

Figure 20. Habitat Evaluation Guide (HEG) scores of habitat quality for prairie-chickens for each evaluation unit sampled in 2020 and 2021 using protocols outlined in the Western Association of Fish and Wildlife Agencies (WAFWA) Habitat Evaluation Guide (Van Pelt et al. 2013, WAFWA 2015).

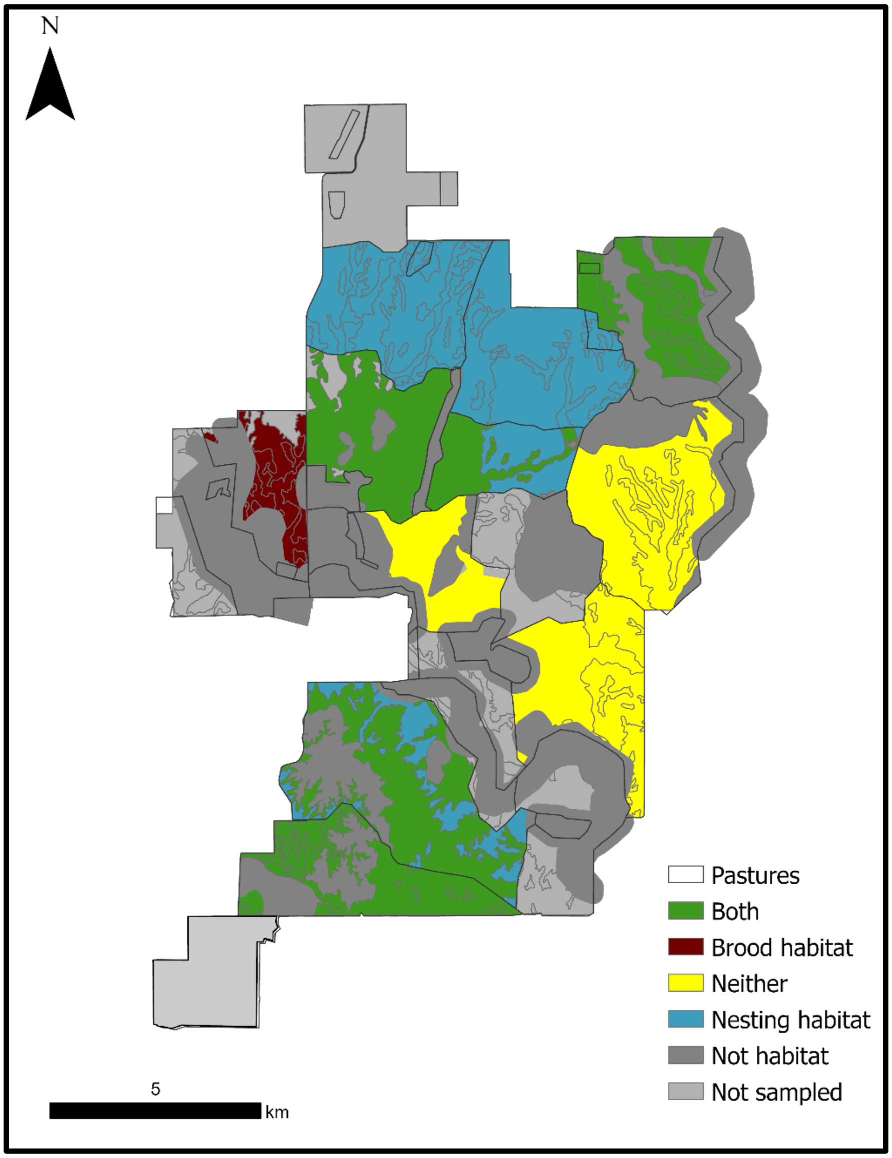

Figure 21. Relative quality of reproductive habitat calculated for each evaluation unit using research-based habitat survey methods during the 2020 sampling season. Evaluation units classified as having suitable reproductive habitat are areas where over 25% of the random points sampled were classified as having suitable nesting (blue), brood-rearing (red) or both types of habitats (green) for the year 2020. Most areas not sampled were classified as non-habitat for prairie-chickens (dark grey) due to increased tree or cropland cover or eastern red cedar ravines.

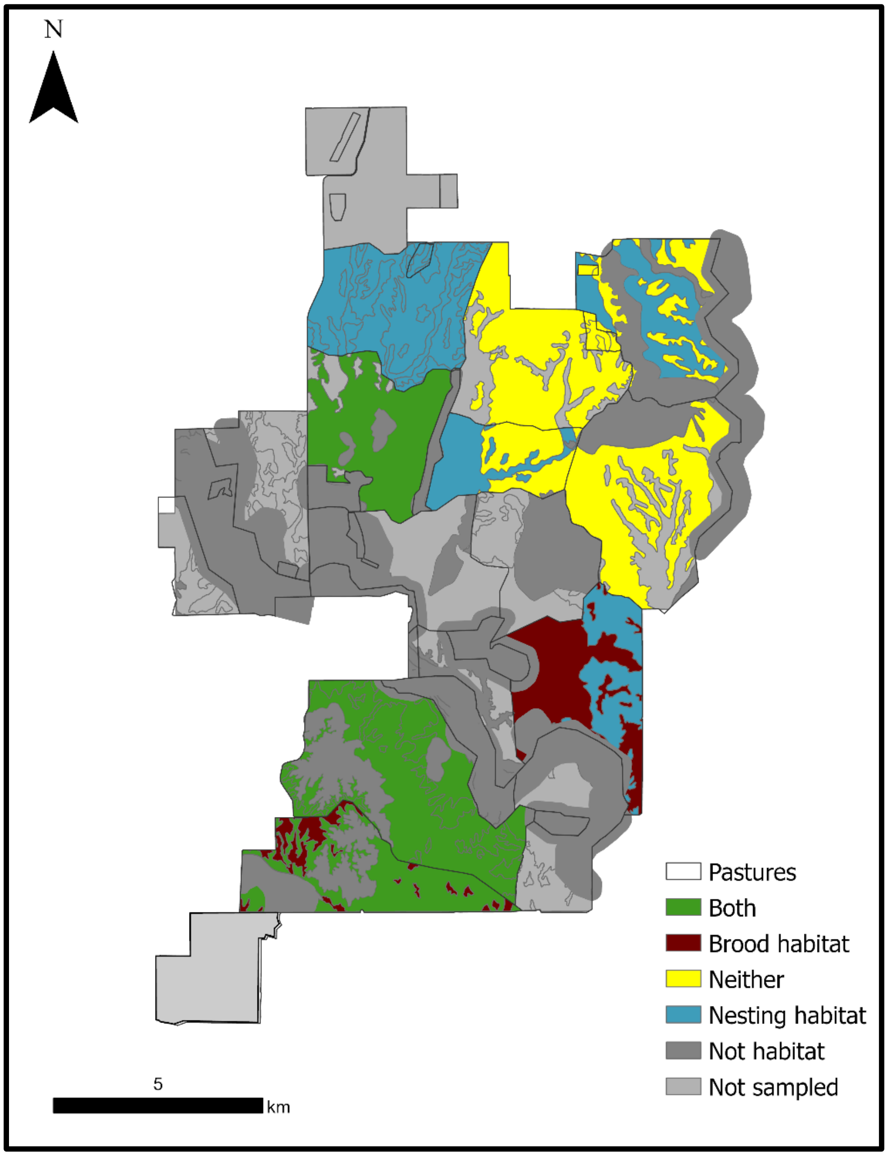

Figure 22. Relative quality of reproductive habitat calculated for each evaluation unit using research-based habitat survey methods during the 2021 sampling season. Evaluation units classified as having suitable reproductive habitat are areas where over 25% of the random points sampled were classified as having suitable nesting (blue), brood-rearing (red) or both types of habitats (both) for the year 2021. Most areas not sampled were classified as non-habitat for prairie-chickens (dark grey) due to increased tree or cropland cover or eastern red cedar ravines.

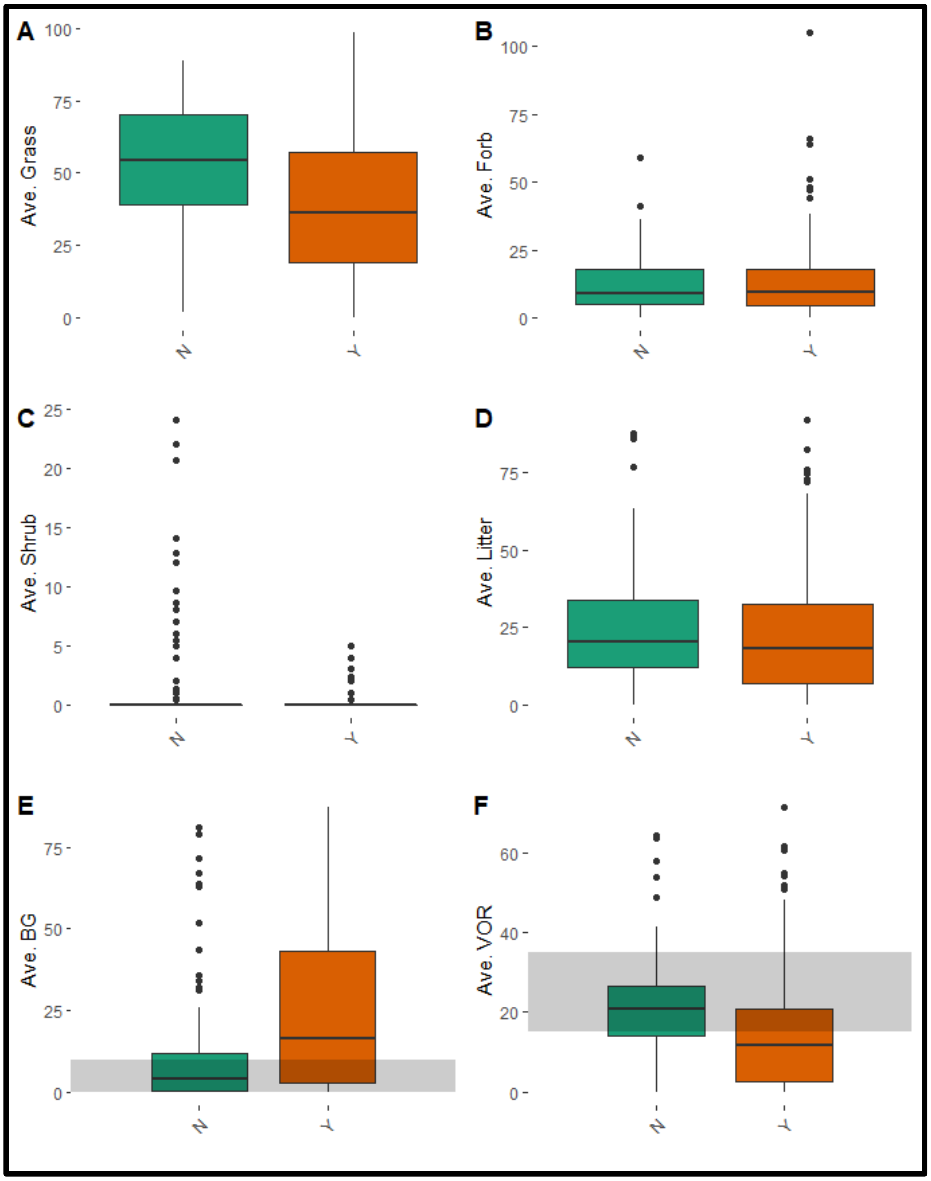

Figure 23. Distribution of values for (A) grass, (B) forb, (C) shrub, (D) litter, (E) bare ground, and (F) visual obstruction readings (VOR) measured using research-level habitat assessment methods for random points sampled during the nesting period in 2021 that were within evaluation units that had experienced fire within the past two years ("Y"; orange) and those that had not ("N"; green) on my study site in the mixed-grass prairie in southcentral Kansas. Areas shaded in grey indicate values of VOR (1.5 – 3.5 dm) and bare ground (≤ 10%) used to classify random points as suitable for nesting using my research-based habitat assessments.

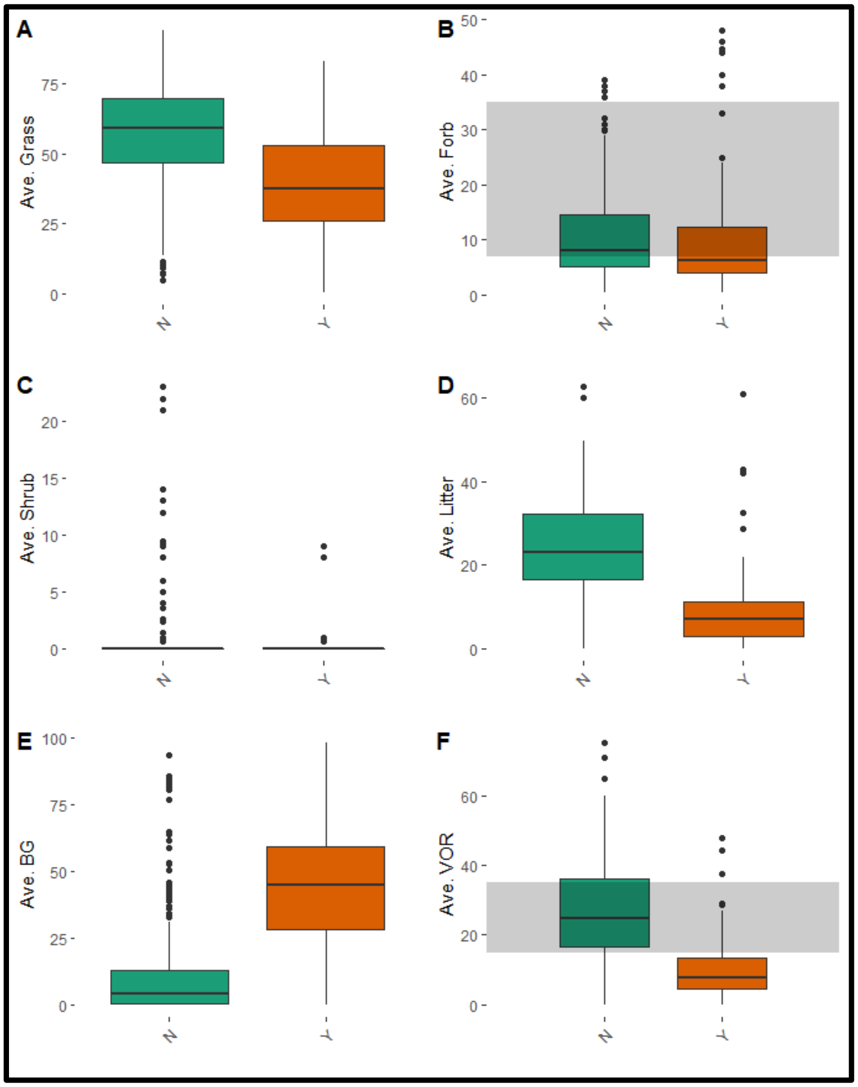

Figure 24. Distribution of values for (A) grass, (B) forb, (C) shrub, (D) litter, (E) bare ground, and (F) visual obstruction readings (VOR) measure using research-level habitat assessment methods for random points sampled during the brood-rearing period in 2021 that were within evaluation units that had experienced fire within the past two years ("Y"; orange) and those that had not ("N"; green) on my study site in the mixed-grass prairie in southcentral Kansas. Areas shaded in grey indicate values of VOR (2 – 5 dm) and forb cover (7 – 35%) used to classify random points as suitable for brood-rearing using my research-based habitat assessments.

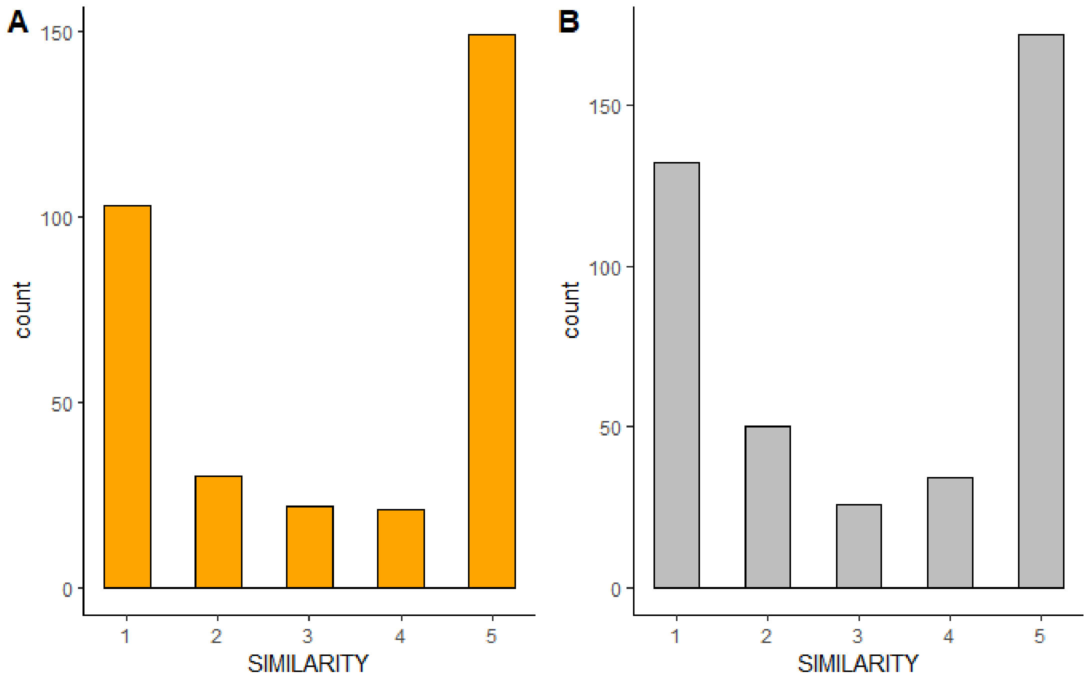

Figure 25. Similarity index values recorded during the nesting (A) and brood-rearing (B) sampling periods to assess ecological site condition at each random point sampled during research-based habitat assessments that measured the amount and quality of reproductive habitat within evaluation units at my study site in the mixed-grass prairie of southcentral Kansas for the years 2020 and 2021 combined.

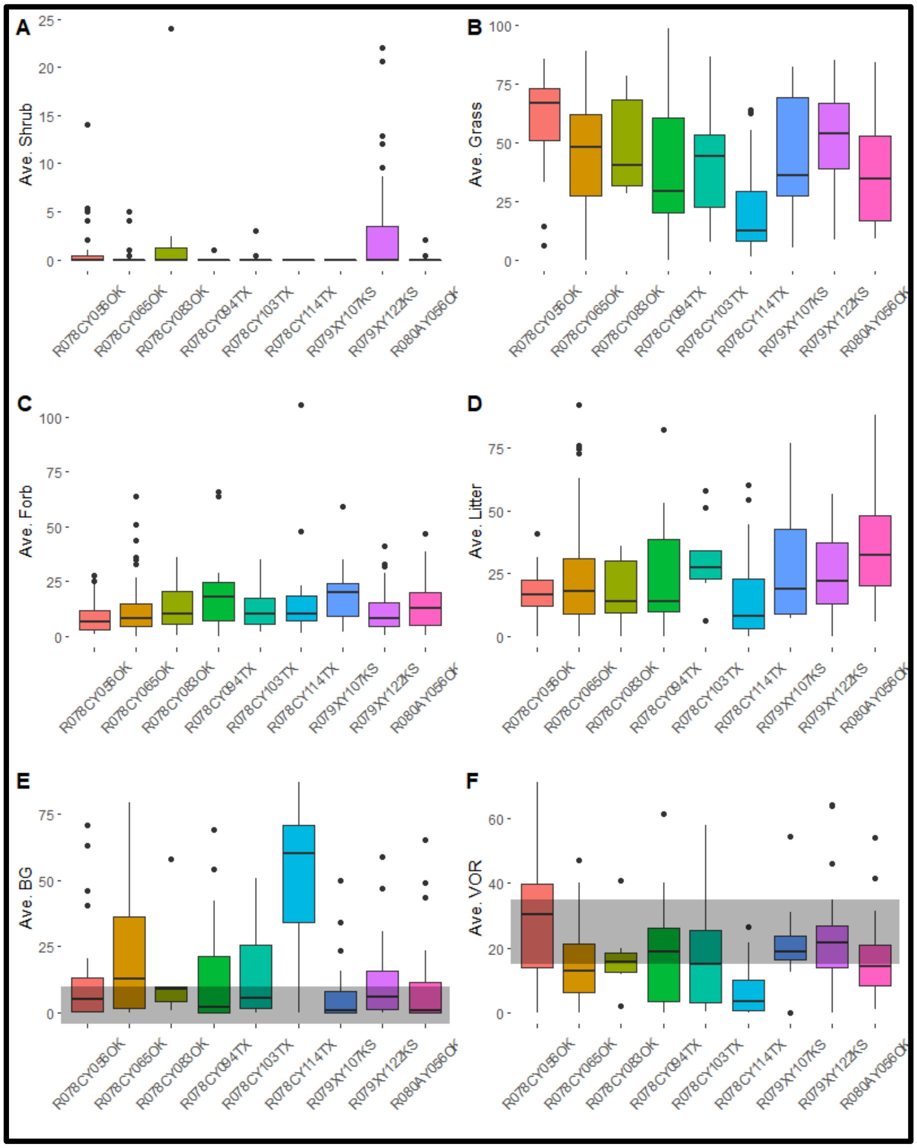

Figure 26. Distribution of cover values for (A) shrub, (B)grass, (C) forb, (D) litter, (E) bare ground and (F) VOR for all ecological sites sampled across the 2020 and 2021 nesting sampling period. Areas shaded in grey indicate values of VOR (1.5 – 3.5 dm) and bare ground (≤ 10%) used to classify random points as suitable nesting habitat for prairie-chickens using research-based habitat assessments.

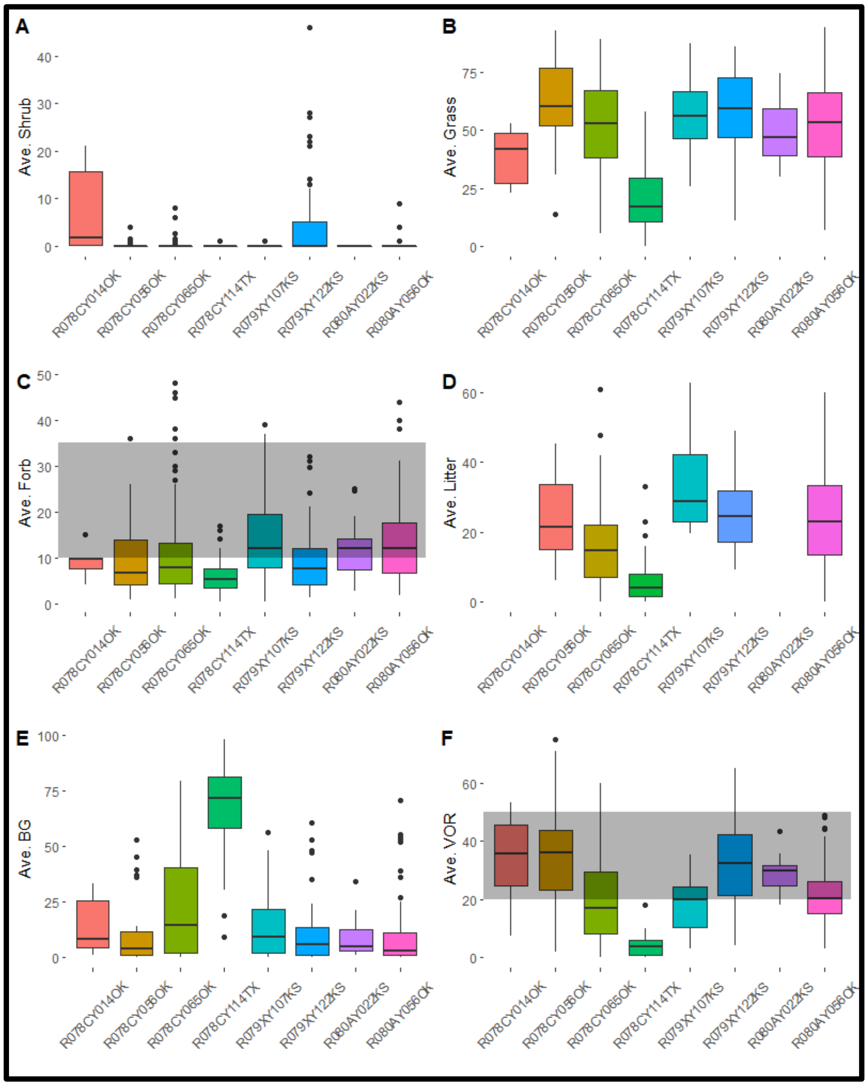

Figure 27. Distribution of cover values for (A) shrub, (B) grass, (C) forb, (D) litter, (E) bare ground, and (F) VOR for all ecological sites sampled across the 2020 and 2021 brood-rearing sampling period. Areas shaded in grey indicate values of VOR (2 – 5 dm) and forb cover (7 – 35%) used to classify random points as suitable brood-rearing habitat for prairie-chickens using research-based habitat assessments.

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