

**EVALUATING HABITAT SUITABILITY FOR LESSER
PRAIRIE-CHICKEN REINTRODUCTIONS**

by

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A PROPOSAL

**Submitted in partial fulfillment of the
Requirements for the degree
MASTERS OF SCIENCE**

**Department of Animal and Range Sciences
MONTANA STATE UNIVERISTY—BOZEMAN**

2020

INTRODUCTION and LITERATURE REVIEW

Grasslands are the largest and one of the most important biomes on the planet, covering an estimated 25–45% of Earth's land surface. Grasslands afford substantial economic services such as fertile soil for agriculture production, forage for livestock, and energy resources. Additionally, they provide significant ecosystem services including climate regulation, water retention, soil conservation, and genetic reservoirs for biodiversity (White et al. 2000). However, despite their importance, grasslands are one of the most threatened and least protected biomes in the world (Hoekstra et al. 2005). For example, an estimated 50–70% of grasslands in the Great Plains of North America have been degraded or lost (Samson et al. 2004, Hoekstra et al. 2005). The central mixed-grass prairie region in the southern Great Plains have been particularly vulnerable to grassland loss and fragmentation, with losses exceeding 50% of historical extent (Samson et al. 2004). Degradation and loss of grasslands are attributed to anthropogenic impacts, such as the conversion of grassland to cropland, energy development and the loss of ecological processes important to ecosystem function (Fuhlendorf and Engle 2001, Samson et al. 2004).

Historically, grasslands of the southern Great Plains were shaped by broad-scale variation in geographical characteristics and climatic conditions. Predictable patterns of seasonal precipitation occurred in the spring and early summer, with the amount of rainfall increasing in a west-to-east gradient from the foothills of the Rocky Mountains to the Mississippi River. Additionally, regional temperatures generally increased from north-to-south along latitudinal gradients. These climatic gradients combined with regional differences in abiotic variables (e.g. topography) influenced plant composition across broad spatio-temporal scales, and created distinct ecoregions defined by plant communities: tall-grass, mixed-grass, short-grass, sand-sagebrush, and shinnery-oak ecoregions (Steinauer and Collins 1996, McGranahan et al. 2012). Nested within ecoregions, ecological sites defined by refined differences in soil type, elevation, topography, and hydrology further divided the landscape up into localized vegetative compositions (NRCS 2003).

Most ecological sites in the Great Plains include a characteristic fire regime and type of herbivory (NRCS 2003). Local differences in the frequency and intensity of natural fires, as well as variation in seasons and intensity of grazing contributed to the various fine-scale plant compositions across the landscape. Fire and herbivory also provided structural diversity to prairie landscapes across the southern Great Plains. Fire removed built-up vegetation and promoted the growth of nutrient rich forage. Large ungulates such as the bison (*Bison bison*), were attracted to the high-quality regrowth and grazed intensively on recently burned patches of prairie. Consequently, neighboring patches of unburned prairie were grazed less intensively, allowing for the buildup of plant litter to fuel successive wildfires. Throughout time, this interaction of fire and herbivory, termed 'pyric herbivory', created a shifting mosaic of burned and grazed, and unburned and ungrazed patches of prairie landscape. This structurally diverse setting promoted plant biodiversity and created various habitat types which facilitated higher animal biodiversity (Coppedge et al. 1998, Samson et al. 2004, Hovick et al. 2015).

Grasslands 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 grasslands

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 grasslands further. Natural fires have generally been suppressed leading to the encroachment of woody plants, and large populations of free-roaming bison have been replaced by intensively managed domestic livestock (Engle et al. 2008, Fuhlendorf et al. 2008). Furthermore, erratic weather patterns caused by climate change have had adverse effects on grassland communities. Lower frequency yet larger precipitation events, and overall higher temperatures have led to variable soil water content and increased plant water stress (Easterling et al. 2000, Knapp et al. 2002). As a result, reduced soil water has altered above-ground plant production and shifted vegetative composition to a more xeric plant assemblage (Knapp et al. 2002).

These transformations in land-use and ecosystem function can have deleterious effects on animal populations (Fuhlendorf et al. 2008). For example, significant declines in grassland bird populations in the Great Plains have coincided with increased grassland loss and fragmentation (Coppedge et al. 2001b). In particular, prairie grouse (*Tympanuchus* and *Centrocercus* spp.) have been highly sensitive to changes in their environment, having overall negative trends in population sizes since the 1960s, that have corresponded to the intensification of farming practices, the expansion of trees, exurban and energy development, and persistent droughts (Garton et al. 2011, Hovick et al. 2014, Garton et al. 2016).

One such species of concern is the lesser prairie-chicken (*T. pallidicinctus*). This obligate grassland bird has experienced significant declines in range-wide occurrence and population size. Prior to Euro-American settlement, the range of lesser prairie-chickens extended over 180,000 km² of grasslands across western Kansas and Oklahoma, eastern Colorado and New Mexico, and north-central Texas. However, lesser prairie-chicken populations have declined approximately 90% during the last century and now occupy just 17% of their historical range (Taylor and Guthery 1980, Hagen et al. 2004). Declines in population and range have been attributed to the loss and fragmentation of habitat resulting from human activities. Recent comprehensive population analyses have demonstrated long-term declines during the last century until apparent population stabilization in the mid-1990s (Garton et al. 2016). Regional populations exhibited signs of recovery during the early 2000s, but a range-wide drought during 2011–2013 reduced populations by 50% (McDonald et al. 2014). As a result of this rapid decline, the U.S. Fish and Wildlife Service (USFWS) listed the lesser prairie-chicken as threatened in 2014; however, listing was quickly overturned by the U.S. District Court for the Western District of Texas in 2015 due to “substantial efforts already being made by state wildlife agencies, industries, and private landowners to restore and conserve lesser prairie-chicken habitat” (USFWS 2016). A petition to relist the lesser prairie-chicken is currently under review by the USFWS.

Lesser prairie-chickens occur in disjunct populations in four ecoregions: the short-grass, sand sagebrush, shinnery oak, and mixed-grass ecoregion (Figure 1; McDonald et al. 2014). Populations of lesser prairie-chickens in the mixed-grass ecoregion are located in southcentral Kansas, western Oklahoma and northern Texas. The mixed-grass ecoregion is characterized by a mixture of mid-height native grasses on loamy sandy soils and is primarily used for livestock grazing. Annual precipitation ranges from about 40–75 centimeters per year and is more reliable

than other parts of the lesser prairie-chickens distribution (USDA, Natural Resource Conservation Service, esis.sc.egov.usda.gov/). Historically the mixed-grass ecoregion contained the greatest density of lesser prairie-chickens, however, recent population analyses indicate populations have declined 1–2.3% annually over the past decade. Additionally, estimates of long-term persistence in the mixed-grass ecoregion project declining carry capacities due to habitat loss, and extirpation is expected to occur within the next century if targeted conservation efforts are not initiated (Garton et al. 2016).

The sand sagebrush ecoregion is characterized by sand sagebrush (*Artemisia filifolia*) interspersed with native grasses on sandy-to-loamy soils. Precipitation ranges from about 31–61 centimeters annually (USDA, Natural Resource Conservation Service, esis.sc.egov.usda.gov/). Sand sagebrush populations are located in southeast Colorado, southwest Kansas, and northwest Oklahoma (Figure 1; McDonald et al. 2014). Sand sagebrush populations are at the highest risk for extirpation with the estimated number of leks declining drastically from 357 leks in 2013 to 61 leks in 2014 (McDonald et al. 2014, Garton et al. 2016). Lek loss was coincident with the range-wide droughts in 2011–2013 (McDonald et al. 2014).

The short-grass ecoregion is characterized by short native grasses intertwined with Conservation Reserve Program grasslands and row-crop agriculture on sand-loamy soils (McDonald et al. 2014). Annual precipitation ranges from about 28–51 centimeters (USDA, Natural Resource Conservation Service, esis.sc.egov.usda.gov/). Short-grass populations are located partially outside of the species' historical range in northcentral Kansas and makeup 65% of the current range-wide population (McDonald et al. 2014). Since the early 2000s, lesser prairie-chicken populations have increased annually by about 8% in the short-grass ecoregion, corresponding to efforts made by private landowners and wildlife agencies to restore vital habitat for lesser prairie-chickens (Rodgers and Hoffman 2005, Garton et al. 2016).

The shinnery-oak ecoregion is characterized by the dominance of sand shinnery-oak (*Quercus havardii*) interspersed with sand sagebrush and native mid-height grasses on sandy soils. Precipitation ranges from 38–43 centimeters annually (USDA, Natural Resource Conservation Service, esis.sc.egov.usda.gov/). Shinnery-oak populations are located in eastern New Mexico and western Texas and are geographically disconnected from populations throughout the rest of the species' distribution (Hagen et al. 2010). Despite a regional decline in lesser prairie-chicken populations in the 1980s that coincided with the arrival of row-crop agriculture, recent population analyses show shinnery-oak populations to be relatively stable throughout the past few decades (Garton et al. 2016). However, future changes in regional climatic conditions, such as higher temperatures and decreased number of annual precipitation events, may put lesser-prairie-chicken populations at risk of extirpation (Grisham et al. 2013).

Lesser prairie-chickens, and prairie grouse in general, are often described as an indicator species for grassland ecosystems (Poiani et al. 2001, Rowland et al. 2005, Hanser and Knick 2011). Prairie grouse require large contiguous grasslands that encompass several habitat types to satisfy different life history stages including lekking, nesting, brood-rearing, and winter habitat. Each habitat type includes a diverse array of vegetational resources to fulfill different life stage requirements (Applegate and Riley 1998). Thus, theoretically, identifying tools for conservation

and management of lesser prairie-chickens could benefit a multitude of grassland species that require individual or a subsample of similar habitat types in the southern Great Plains (Poiani et al. 2001, Rowland et al. 2005, Donnelly et al. 2017).

Prairie Grouse Demography

Population declines of lesser prairie-chickens are driven by poor fecundity or survival. Initial studies found fecundity, specifically reproductive success, to have the greatest influence on prairie grouse population dynamics (Wisdom and Mills 1997, Hagen et al. 2009). Moreover, research on adult male and female survival has largely observed higher mortality rates during the breeding season compared to the non-breeding season (Hagen et al. 2005a;2007, Lyons et al. 2011), indicating tradeoffs of reproductive success and survival during the breeding season of critical importance to prairie-grouse. The reproductive season for lesser prairie-chickens occurs from March–September which are divided into distinct periods corresponding to important life-history events including the lekking period (March-May), nesting period (mid-April-June), and the brood-rearing period (June-September), with each period requiring a different subset of habitat conditions (Boal and Haukos 2016). Sensitivity analyses show nest and brood survival to be the most crucial factors influencing population fluctuations in prairie grouse (Hagen et al. 2009, Gillette 2014). However, declining populations with poor reproductive success have also shown sensitivity to changes in adult survival (Johnson and Braun 1999, McNew et al. 2012, Taylor et al. 2012) For example, female prairie grouse demonstrate uniparental care and therefore nest and brood survival are largely dependent on the survival and habitat selection of females (Taylor et al. 2012).

Lekking ecology

A lek is an aggregation of male prairie grouse that congregate to display and compete for females who are surveying for a potential breeding partner. Interactions between the sexes is believed to be limited to lek sites during the breeding season. Another name for the lek sites of lesser prairie-chickens is ‘gobbling grounds’ due to the sound males produce when attempting to court a female. Leks are often located in areas with higher elevation and little vegetation to increase visibility for the detection of potential predators and visiting females, and to increase the distance male mating calls can be heard (Copelin 1963, Taylor and Guthery 1980, Aspbury and Gibson 2004, Boal and Haukos 2016). Male prairie grouse exhibit high lek site fidelity, often returning to the same lek year-after-year. Stable leks, or leks that have had consistent male attendance for five-to-ten consecutive years, are important to prairie grouse demography because females will generally visit established lek locations rather than newly formed lek locations (Haukos and Smith 1999). Females also tend to visit more than one lek, so clusters of leks are important for sustaining populations (Applegate and Riley 1998). Additionally, leks are the focal point of prairie grouse life-histories with most prairie grouse activity occurring within +/-5 kilometers of a lek, including nesting and brood-rearing (Applegate and Riley 1998, Pitman et al. 2005, Patten et al. 2011, Winder et al. 2015a). Thus, the formation and persistence of lek locations are likely concomitant with the highest availability of nesting and brood rearing habitat. In fact, recent research has found that established lek locations have greater proportions of high quality nesting

and brooding habitat than would be expected at random locations (Lautenbach 2015, Gehrt et al. 2020).

Nesting ecology

Nest success has consistently been identified as the most important vital rate affecting prairie grouse populations (Wisdom and Mills 1997, Hagen et al. 2009, Gillette 2014). Prairie grouse have high reproductive potential with high nest initiation rates, large clutch sizes and high egg viability (Connelly et al. 1998, Connelly et al. 2011, McNew et al. 2011). Nest success for prairie grouse is variable and likely depends on the region, predator densities, weather, and local habitat conditions (Table 1). Depending on the cost and benefit for females, prairie grouse will re-nest if the first clutch is lost (Pitman et al. 2006, McNew et al. 2011). Likelihood of females re-nesting is probably a function of nest age, with a lower probability of initiating a second nest if the first clutch is lost later in the nesting period (Pitman et al. 2006, Kaczor et al. 2011, McNew et al. 2011). Nest and re-nest initiation rates typically do not differ between yearling and adults (Kaczor et al. 2011, McNew et al. 2012, Taylor et al. 2012). Research evaluating differences in nest success rates between adults and yearlings has yielded variable results; some studies show higher success rate in adults (Fields et al. 2006, Connelly et al. 2011, McNew et al. 2012, Taylor et al. 2012), while others found no differences between the two age groups (Pitman et al. 2006, Shepard et al. 2011, Grisham et al. 2014, Lautenbach et al. 2019). Additionally, adverse weather events can negatively affect nest success. For example, years of drought have been associated with increased delays in nest initiation, decreased likelihood of re-nesting, and higher rates of nest abandonment (Grisham et al. 2014, Lautenbach et al. 2019)

The selection of nest sites is directly related to nest success. Nest sites are generally located within 5 kilometers of a lek (Connelly et al. 2011, Hagen et al. 2013) and are often associated with local vegetative conditions that provide concealment from potential predators and less variable thermal conditions (Kirsch 1974, Giesen 1994, Connelly et al. 1998). Predation accounts for most nest failures of prairie grouse with 55–75% of prairie-chicken nests lost to mammalian predators (Pitman et al. 2006, Winder et al. 2016, Lautenbach et al. 2019). Vertical obstruction reading (VOR), a measure of nest concealment, is a consistent correlate of nest survival across studies (Giesen 1994, McKee et al. 1998, Holloran et al. 2005, Pitman et al. 2005, Kaczor et al. 2011, Lautenbach et al. 2019). Nest-site selection often exhibits a quadratic relationship with VOR, presumably due to a life-history tradeoff between nest success and female survival. For example, while denser cover enhances nest concealment, it decreases the ability of the female to escape predation. Thus prairie grouse may select relatively moderate levels of cover to enhance both nest success and the females survival (Wiebe and Martin 1998, McNew et al. 2013, 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). Increased concealment has also been linked to cooler and less variable microclimates which is thought to positively influence nest success (Grisham et al. 2016).

While habitat conditions at the nest site are important, nesting decisions are not solely based on local vegetation that provides the best concealment. Large scale differences in vegetative cover have also been shown to influence nest site selection and impact nest survival. Previous research

shows nest site selection to be positively associated with increased grass cover and negatively associated with increased tree cover (Johnson 2001, McNew et al. 2013, Lautenbach et al. 2017). Additionally, ecological sites with higher potential production and greater VOR measurements have been suggested to maintain higher quality nesting habitat (Van Pelt et al. 2013, Kraft 2016). For example, in the mixed-grass prairie ecoregion, Kraft (2016) observed loamy upland, limy upland, and loamy lowland ESs to have greater vegetative biomass that corresponded to higher VOR readings and overall higher nesting success for lesser prairie-chickens. Although, one study found greater sage grouse nest occurrence and survival was not predicted by ecological sites and the authors cautioned against relying solely on ecological site information to make management decisions (Doherty et al. 2011).

Human development has also been found to negatively affect nest site selection with nests placed farther away from unnatural features even though sufficient vegetative conditions were available (Hagen et al. 2011, Plumb et al. 2019). However, several studies found nest site selection and nest survival were not affected by wind turbines (McNew et al. 2014, LeBeau et al. 2017). In fact, nest site selection and nest success are likely a product of the interaction of several habitat conditions at multiple scales (Pitman et al. 2005, McNew et al. 2013). For example, one study found differences in the interactions between vegetative condition at the nest site and distance to human features to be greatest predictor of lesser prairie-chicken nest success (Pitman et al. 2005). Moreover, since no two points in space have the same arrangement of habitat conditions, nesting decisions by females likely depend on the local habitat conditions across multiple scales and the relative importance of resources needed to increase both nesting success and female survival (McNew et al. 2013).

Brood survival

Brood survival has also been identified as one of the most influential factors driving prairie grouse population dynamics (Hagen et al. 2009). Prairie grouse chicks are precocial and leave the nest with the female soon after hatch to look for food and shelter (Bergerud 1988). Chicks are most vulnerable to predation and exposure during the 2-week pre-fledge stage; more than half of all chick mortalities occur in the first 2-3 weeks post-hatch (Fields et al. 2006, Pitman et al. 2006). Brood success and chick survival in lesser prairie-chickens are less documented in the literature compared to nesting success (Table 2). Causes for brood mortality include predation, starvation, and exposure to inclement weather directly post hatch. Thus, female selection of brood habitat that mitigates these threats is vital to brood survival (Fields et al. 2006, Goddard and Dawson 2009).

The use of habitats by lesser prairie-chicken broods is likely driven by female selection behavior to minimize exposure to predators and maximize access to critical food resources for chicks (Goddard et al. 2009). Brood habitat for lesser prairie-chickens can be characterized as having moderate grass and shrub cover and greater percent bare ground, which facilitates chick movement and predator escape (Riley and Davis 1993, Manzer and Hannon 2008). Selection for areas with moderate grass and shrub cover can also aid in thermoregulation during inclement weather; chicks cannot regulate body temperature for about 14 days post-hatch and thus are vulnerable to exposure to environmental conditions during this period. However, chick survival has been positively associated with increases in pre-hatch precipitation that corresponds to

increased food availability (Fields et al. 2006, Goddard and Dawson 2009). Female selection of brood habitat is also often associated with greater forb cover which is positively correlated to increased invertebrate densities (Jamison et al. 2002, Hagen et al. 2005b, Goddard et al. 2009, Hagen et al. 2013). Forbs and invertebrates are chief food sources for prairie grouse broods and increased densities of both have been linked to higher chick survival in prairie grouse populations (Hagen et al. 2005b, Gregg and Crawford 2009, Sullins et al. 2018a).

Brood habitat selection is influenced by large scale habitat conditions, as well. While more research is needed, it is likely that ecological sites and their relative condition influence the availability of fine-scale brood habitat. For example, in the Sandhills of Nebraska where bare ground is prolific, greater prairie-chickens were found to select brood habitat within ecological sites that provided higher VOR measurements and relatively denser vegetation (Anderson et al. 2015). Additionally, greater sage-grouse broods have been found to select areas farther from exurban development, croplands, and oil and gas development (Aldridge and Boyce 2007) and the relative risk of chick and brood mortality has generally been found to increase in proximity to energy development (Aldridge and Boyce 2007, LeBeau et al. 2014). Although, some found no difference in brood survival between areas with or without energy development (Williamson 2009).

Adult survival

Annual survival rates of adult lesser prairie-chickens has ranged from 0.23 to 0.62 in the mixed-grass prairie ecoregion and varies over space and time with intrinsic and environmental conditions (Wolfe et al. 2016a). Research on age-specific survival has yielded variable results, however some studies have reported higher survival of first-year birds in prairie grouse during the breeding season, presumably due to the increased cost of higher reproductive effort of older birds (Hagen et al. 2005a;2007, Connelly et al. 2011). For example, lower survival rates of adult males compared to yearlings suggests that competition in dominant adult males could involve higher costs for survival as a result of male-to-male competition (Hagen et al. 2005a). Additionally, differences in behavioral responses between female adults and yearlings during the reproductive period could explain the lower survival rates in adults; i.e. adult females may be more tenacious when guarding nests and broods compared to yearlings (Hagen et al. 2007). Differences in survival between sexes are likely a function of seasonal reproductive activities. For example, male mortalities increase during the lekking season when they are more conspicuous to predation, while female mortalities are highest during the nesting period (Patten et al. 2005, Wolfe et al. 2007, Manzer and Hannon 2008, Lyons et al. 2009).

Predation is the leading cause of mortality in prairie grouse populations with the majority of deaths caused by mesocarnivores and raptors (Wolfe et al. 2007, Manzer and Hannon 2008, Boal and Haukos 2016). Like nest and brood survival, adult survival is linked to the selection of local vegetative characteristics believed to decrease detection by predators. Increased cover has been linked to cooler and more humid microclimates that provide reprieve from hotter temperatures and dryness (Patten et al. 2005). However, survival of nesting lesser and greater prairie-chicken females has been found to be greater in only moderate levels of vegetative cover, as denser

vegetation increases nest concealment but also increases the risk of predation for the female (Pitman et al. 2005, McNew et al. 2013).

At a larger scale, differences in vegetative cover and land-use can also influence adult survival rates. For example, lesser and greater prairie-chicken populations declined more rapidly in landscapes with greater rates of change in land-use and loss in shrub and grass cover (Ryan et al. 1998, Woodward et al. 2001). Additionally, adult survival is negatively related to features associated with energy and exurban development such as roads, powerlines, and fences; higher densities of human features has been shown to potentially increase the risk of collisions and reduce prairie grouse survival (Lyon and Anderson 2003, Patten et al. 2005, Stevens et al. 2012). Lower survival rates of adult prairie grouse in areas with increased development may also be a product of interactions between predator communities and human development. Habitat edges associated with human infrastructure and land-use are thought to facilitate mesocarnivore occupancy and enhance foraging by creating easy pathways for travel (Andr n 1995, Kuehl and Clark 2002). Moreover, tall structures provide perches for avian predators, thus higher levels of human development may increase predation risk on adult prairie grouse (Marion and Ryder 1975). However, the significance of human development on the predation risk of prairie grouse likely depends on the spatial configuration, density of associated features, and the level of activity associated with human development. For example, some studies have suggested high levels of activity associated with the early stages of energy development may temporarily reduce predator communities, thus reducing predation risk for prairie grouse (Winder et al. 2014b, Burr et al. 2017).

Prairie grouse spatial ecology

Prairie grouse select habitat at multiple spatial scales including the large-scale selection of home ranges and the fine-scale selection of nest and foraging locations (Fuhlendorf et al. 2002, Doherty et al. 2010). Prairie grouse require large tracts of grassland that allow for the establishment of home ranges and include the fine-scale availability of lekking, nesting, brooding, and nonbreeding habitats (Applegate and Riley 1998, Van Pelt et al. 2013). Like all animals, habitat selection in prairie grouse is often affected by tradeoff decisions that address variable life history requirements such as reproduction and survival. For example, prairie grouse may weigh the cost and benefit of remaining in taller vegetation to avoid predators versus procuring food in open areas. Or while selecting a nest site, females may consider whether to select for denser cover that improves nest concealment or intermediate cover that decreases the risk of predation for the female. For example, some studies have found lower survival rates of female grouse that selected for nest sites with taller and denser vegetation suggesting that although higher residual cover improves nest concealment, mortality rates for the female may increase due to their inability to detect and escape predators (Wiebe and Martin 1998, Hagen et al. 2007, McNew et al. 2012). Thus, habitat selection depends on the local environment and the relative importance of resources needed to increase survival *and* reproductive success (McNew et al. 2013).

Extensive research has shown prairie grouse to be highly sensitive to the amount and configuration of grassland habitat. As a resident species, the majority of prairie grouse activity occurs around a breeding lek, with only a few individuals participating in long-distance

movements (Applegate and Riley 1998, Pitman et al. 2005, Patten et al. 2011, Winder et al. 2015a). High site fidelity in prairie grouse makes them highly sensitive to landscape change, especially in isolated populations where gene flow is low and impacts of demographic and local environmental stochasticity is high (DeYoung and Williford 2016, McNew et al. 2017). Sensitivity to landscape composition is influenced by life history traits that affect prairie grouse behavioral decisions. For example, prairie chickens have a neophobic tendency to avoid areas with unnatural disturbances and tall structures (Woodward et al. 2001, Baruch-Mordo et al. 2013, Robinson 2018). As a result of this avoidance behavior, changes in landscape composition can provide barriers to movement and fragment otherwise suitable habitat (Pitman et al. 2005). Although prairie grouse have been shown to exhibit exploratory behaviors (Earl et al. 2016), continual increases in landscape change has reduced the probability of connecting isolated populations (DeYoung and Williford 2016). Changes to landscape that have been shown to affect prairie grouse demographics include the conversion of grassland to cropland, the development of exurban communities and energy, and the expansion of trees into grassland landscapes (Woodward et al. 2001, Gregory and Beck 2014, Lautenbach et al. 2017).

Anthropogenic development

Research evaluating the effects of human development on prairie grouse populations have produced variable results; although most studies focused on lesser prairie-chickens have reported negative impacts on habitat use and survival (Pitman et al. 2005, Hagen et al. 2011, Gregory and Beck 2014, Burr et al. 2017, Plumb et al. 2019). Infrastructure associated with human development include, oil wells and pumpjacks, wind turbines, powerlines, buildings, fences, and roads. Human development can impact prairie grouse demography directly, such as through the collision of individuals with fences or cars, or indirectly, through the effect on behavioral decisions that result in the displacement of populations from suitable habitat (Pitman et al. 2005, Wolfe et al. 2007, Hagen et al. 2011, Gregory and Beck 2014).

In general, oil and gas development have been implicated in the displacement of prairie grouse populations (Walker et al. 2007, Hagen et al. 2011, Hess and Beck 2012, Gregory and Beck 2014). For example, greater sage-grouse males avoided leks closest to oil well fields when selecting breeding territories (Walker et al. 2007, Timmer et al. 2014) and female lesser prairie-chickens selected nest sites farther from oil wells than would be expected (Pitman et al. 2005, Plumb et al. 2019). Displacement is due to behavioral avoidance of tall structures and possibly the unnatural noise emanating from oil wells (Blickley et al. 2012, Plumb et al. 2019). Avoidance distances vary across different spatial scales and oil well densities, suggesting landscape composition influences the effects of energy development on displacement (Hess and Beck 2012, Gregory and Beck 2014, Plumb et al. 2019). For instance, increasing oil well densities decreased male greater sage-grouse presence at lek sites (Hess and Beck 2012, Gregory and Beck 2014).

Effects of oil and gas development on prairie grouse survival are less pronounced; for example, while greater sage-grouse brood and yearling survival were found to be lower in oil fields than those outside of oil fields (Aldridge and Boyce 2007, Holloran et al. 2010, LeBeau et al. 2014), recent research found no effect on sharp tailed grouse habitat use or survival (Burr et al. 2017, Milligan 2019). In fact, one study reported a two-fold increase in nest survival in areas with greater densities of oil well than those with lower oil well densities (Burr et al. 2017). Although,

the authors cautioned that decreased disturbances associated with later stages of energy development could present a very different predator community.

Future expansion of wind energy development is expected to occur in the Great Plains due to the regions high wind resource (National Renewable Energy Laboratory 2015, <https://windexchange.energy.gov/maps-data>). Recent research on the effects of wind development on prairie grouse populations is highly variable, but largely suggests moderate effects on habitat use and little-to-no effect on survival. For example, male greater prairie-chicken lek persistence declined as distance to wind infrastructure decreased (Winder et al. 2015b) and while female greater prairie-chickens home range centers were not displaced, overall sizes of home ranges doubled following the construction of a wind facility (Winder et al. 2014a, Winder et al. 2014b). Conversely, other studies found distance to the nearest wind turbine did not affect greater prairie-chicken and greater sage-grouse nest site selection (McNew et al. 2014, Harrison et al. 2017, LeBeau et al. 2017). Most research suggests adult survival in females is not affected by wind turbines, with even one study reporting an increase in greater prairie-chicken survival post wind development (LeBeau et al. 2014, Winder et al. 2014b, Smith et al. 2017). Additionally, construction of wind energy facilities had no impact on nest survival of greater prairie-chickens or greater sage-grouse (McNew et al. 2014, LeBeau et al. 2017).

Powerlines are the one structure that has consistently exhibited negative impacts on lesser prairie-chicken habitat use (Plumb et al. 2019). Lesser prairie-chickens use habitat more intensely away from powerlines than areas with powerlines suggesting they act as barriers to movement (Hagen et al. 2007, Hagen et al. 2011, Plumb et al. 2019). Lesser prairie-chickens also select nest sites farther from powerlines than would be expected even though suitable habitat was present (Pitman et al. 2005, Plumb et al. 2019). Avoidance of powerlines is likely caused by the behavioral propensity of prairie grouse to avoid structures that provide perches for avian predators. Additionally, given that prairie grouse are a low flying bird, mortality from collision with powerlines is of little concern (Hagen et al. 2007).

Livestock fences are prominent features on prairie landscapes and have been identified as a potential risk to prairie grouse survival (Wolfe et al. 2007, Stevens et al. 2012). Fences can alter predator communities by providing perches for avian predators and corridors for mammalian predators (Marion and Ryder 1975, Andr n 1995, Kuehl et al. 2002). Little is known on the effect of fences on prairie grouse habitat selection, and research assessing the effects of fence collisions on prairie grouse mortality have produced little evidence that fences directly affect population viability (Wolfe et al. 2007, Robinson et al. 2016). However, in studies where fence collisions were thought to be the cause of mortality, higher collision rates corresponded with higher fence densities, proximity of fences to lekking grounds and high-use areas, and lower topography (Wolfe et al. 2007, Steven et al. 2012). Thus, the impacts of fences on prairie grouse survival may depend on the local landscape composition.

Roads have been implicated in the displacement and mortality of prairie grouse populations (Lyon and Anderson 2003, Walker et al. 2007, Hagen et al. 2011). Roads inherently fragment grassland habitat by dividing contiguous habitat patches into smaller patches. Without roads, the Great Plains would retain contiguous grassland patches of over 10,000 km². However, with roads, most grassland patches are reduced to areas between 100 and 1000 km² (White et al.

2000). Increased human activity, and possibly the noise associated with roads results in prairie grouse avoidance and thus can effectively inhibit movement and displace populations (Lyons and Anderson 2003, Plumb et al. 2019). For example, lesser prairie-chicken and greater sage-grouse have been reported to select nest sites farther away from roads than would be expected at random (Lyons and Anderson 2003, Pitman et al. 2005). Areas with higher densities of roads have been shown to decrease lek densities (Timmer et al. 2014) and impede nest initiation (Lyons and Anderson 2003). However, the degree to which roads affect prairie grouse habitat selection depends on the intensity of use; prairie grouse generally select habitat farther from paved roads than they would from two-track dirt roads that rarely see traffic (Lyons and Anderson 2003, Pitman et al. 2005, Walker et al. 2007).

Finally, buildings have similar negative effects on lesser prairie-chicken space use. Pitman et al. (2005) observed that lesser prairie-chickens preferred nest sites that were 1000 meters from buildings. Seasonal habitat use decreased with distance to buildings, as well, indicating higher densities of exurban development has likely displaced populations (Pitman et al. 2005, Hagen et al. 2011). Displacement of habitat use is caused by lesser prairie-chickens general tendency to avoid tall structures that provide perches for avian predators and their general neophobic dislike for anything new.

Tree Encroachment

Within the southern Great Plains, encroachment by trees is a major cause of grassland fragmentation and has been linked to the recent decline of grassland birds (Coppedge et al. 2001a, Fuhlendorf et al. 2002, Engle et al. 2008, Fuhlendorf et al. 2008, Lautenbach et al. 2017). Early conservation programs endorsed and facilitated the planting of trees for windbreaks to prevent soil erosion and provide habitat for wildlife (Capel 1988, USDA, The Prairie States Forestry Program). Suppression of natural fire and the replacement of free-roaming bison with intensively managed domestic livestock has assisted in the expansion of woody plants, including eastern red cedar (*Juniperus virginiana*) (Fuhlendorf et al. 2002, Engle et al. 2008).

Eastern red cedar is a highly adaptable species thriving in multiple soil types and climatic conditions which allows for high rates of expansion (Schmidt and Leatherberry 1995). Evidence from aerial photography demonstrates eastern red cedar can transform highly productive tall-grass prairies into a closed canopy forest in just 40 years (Briggs et al. 2002). Trees can decrease biodiversity and alter plant composition by constraining the growth of shrubs and forbs and shifting grass composition from warm season grasses to cool season grasses (Gehring and Bragg 1992, Coppedge et al. 2001a). Increases in tree density alter animal communities, as well (Coppedge et al. 2001a, Horncastle et al. 2005, Alford et al. 2012). For example, native grassland birds are replaced with exotic or generalist species that prefer higher tree densities (Coppedge et al. 2001a, Chapman et al. 2004).

Woody encroachment has been implicated in the displacement and decline of prairie grouse populations (Fuhlendorf et al. 2002, Coates et al. 2017, Lautenbach et al. 2017). The spatial distribution of woody plants on the landscape negatively affects the occupancy, reproductive performance, and survival of prairie grouse (McKee et al. 1998, Lautenbach et al. 2017, Sandford et al. 2017). Avoidance of trees is likely caused by a perceived increase in predation risk or a general propensity for prairie grouse to avoid tall structures (Gregory et al. 2011, Lautenbach et al. 2017, Sandford et al. 2017). For example, Lautenbach et al. (2017) found

female lesser prairie-chickens rarely used areas with more than eight trees per hectare and never selected nest sites with more than two trees per hectare. Additionally, greater prairie-chickens exhibited increased nest survival as distance from forest cover increased (McKee et al. 1998). Other studies on greater prairie-chickens and greater sage-grouse found negative associations with lek presences and persistence as forest cover increased, as well (Niemuth 2003, Gregory et al. 2011).

Prairie grouse conservation

More than 70% of the land surface in the United States is privately owned which includes 95% of land within the current distribution of lesser prairie-chickens (Ciuzio et al. 2013, Elmore and Dahlgren 2016). Thus, land-use decisions by private landowners have direct effects on lesser prairie-chicken habitat and important consequences for lesser prairie-chicken populations. Consequently, conservation initiatives with strong partnerships between private landowners and resource managers is essential for lesser prairie-chicken recovery. Three programs that have played an integral role in bringing together federal, state, nongovernmental organizations, and private landowners to conserve lesser prairie-chicken habitat are 1) the Lesser Prairie-Chicken Initiative (LPCI), administered under the Natural Resource Conservation Service's (NRCS) Working Lands for Wildlife Program (WLFW), 2) the Conservation Reserve Program (CRP) administered by the Farm Service Agency (FSA), and 3) the Western Association of Fish and Wildlife Agencies' (WAFWA) Lesser Prairie-Chicken Range-wide Conservation Plan (RWP).

FSA & Conservation Reserve Program

Conversion of grassland to cropland is implicated as a major cause for habitat fragmentation and loss (Fuhlendorf et al. 2002). The CRP is thought to play a substantial roll in lesser prairie-chicken recovery by paying farmers a yearly rental payment to remove environmentally sensitive lands from agriculture production and plant species that would improve lesser prairie-chicken habitat through their State Acres for Wildlife Enhancement (SAFE) initiative. Indeed, since the program's establishment in 1985, the CRP has increased the abundance of grassland by 12% in the southern Great Plains, most of which has occurred in the short-grass ecoregion. However, range-wide lesser prairie-chicken populations have continually declined since the 1980s, suggesting the quality of grassland habitat may be more important to population expansion than the amount of available grasslands (Spencer et al. 2017).

Depending on local landscape composition, CRP grasslands have had differing effects on grassland bird habitat use and demography (Coppedge et al. 2001b, Ripper et al. 2008, Sullins et al. 2018b). For example, CRP grasslands with increased lesser and greater prairie-chicken populations were associated with fine-scale qualities proven to be beneficial to prairie-chicken nest survival such as, higher species diversity, increased vegetation height, higher average forb cover, and higher densities of native plant species (Rodgers and Hoffman 2005, Ripper et al. 2008). However, CRP grasslands may only provide habitat for certain life stages. Sullins et al. (2018b) found that post-hatch lesser prairie-chickens moved their brood from CRP grasslands to other cover types, proposing that although CRP grasslands provided nesting habitat, they lack favorable brood habitat. Furthermore, CRP grasslands with monoculture plantings have been implicated as ecological traps. For example, grassland songbirds only utilized CRP grasslands in areas with increasing woody encroachment suggesting that grasslands with less trees had higher habitat quality than neighboring CRP monocultures (Coppedge et al. 2001b).

LPCI & Tree removal

In the southern Great Plains, woody encroachment has replaced the conversion of grassland to cropland as the largest threat to grassland habitat fragmentation and loss (Coppedge et al. 2001a, Engle et al. 2008). The LPCI uses current Farm Bill conservation programs such as the Environmental Quality Incentive Program (EQIP) and the Wildlife Habitat Incentive Program (WHIP) to assist landowners in implementing conservation practices to improve lesser prairie-chicken habitat including the removal of trees. Specifically, conservation practices 314 (Brush Management) and 645 (Upland Wildlife Habitat Management) under the LPCI target the removal of woody plants, both invasive (e.g., eastern red cedar) and noxious (e.g., salt cedar). Methods used for tree removal include prescribed burning, individual tree ignition, mechanical removal with large machinery or chainsaws, and herbicide (Ortmann et al. 1998).

Within just three-to-four years, tree removal can return grassland communities to their original composition (Pierce and Reich 2010, Severson et al. 2017). Cool season grasses are reverted to warm season, and plant height and density increase (Alford et al. 2012, Severson et al. 2017). Additionally, small mammalian communities shift to a more grassland assemblage and invertebrate densities increase (Alford et al. 2012). Although little research has directly evaluated the effects of tree removal on grassland birds in the Great Plains, focused tree removal effectively improved habitat for obligate sage-steppe communities of birds in the intermountain West (Donnelly et al. 2017, Sandford et al. 2017, Severson et al. 2017).

Tree removal can improve nesting habitat by increasing important vegetative characteristics known to contribute to nest survival such as plant height and density (Severson et al. 2017). For example, one study found that the majority of greater sage-grouse females selected nest sites closer to conifer removal areas and nest and brood success decreased as nest sites got closer to areas with higher tree densities (Sanford et al. 2016). Furthermore, increased greater sage-grouse lek distribution and songbird abundance simultaneously occurred in areas with recent conifer removal, suggesting management actions that promote prairie grouse conservation could benefit other native species who have similar habitat requirements (Donnelly et al. 2017).

WAFWA: Lesser Prairie-Chicken Range-wide Conservation Plan

Another program that targets conservation for lesser prairie-chickens on private land is WAFWA's Range-wide Conservation Plan (RWP). Under the RWP, WAFWA's Mitigation Framework uses dollars paid into conservation banks by oil, gas, wind, electricity and telecommunications industries to pay landowners annually to minimize and offset impacts to lesser prairie-chicken habitat in areas critical to the species persistence. Recommended management plans developed in collaboration with property owners use the same framework, conservation practices, and standards as NRCS' LPCI. Prescribed conservation actions under WAFWA management plans include prescribed grazing and burning, tree removal, and planting of native grasses and forbs to increase vegetative cover. Landowners who are enrolled in WAFWA's RWP are incentivized to manage for quality habitat because their annual payment is based on the total acreage enrolled and the Habitat Evaluation Guide (HEG) score of the property; if the property owner does not follow WAFWA's recommended management plan and their property declines in habitat quality, annual payments are reduced. HEG scores are based on four habitat variables: vegetative cover, vegetative composition, percent cover of tall woody plants, and availability of potential habitat in the surrounding area. Annual vegetation monitoring

for enrolled properties in WAFWAs RWP generally follows the same protocol that NRCS uses to monitor the effectiveness of their LPCI program (Van Pelt et al. 2013).

WAFWAs RWP has generally been credited with restoring and conserving large amounts of land for lesser prairie-chicken recovery. In fact, following the drought of 2011 – 2013 and the establishment of WAFWA’s RWP, range-wide lesser prairie-chicken numbers increased about 25% by 2016 (Wolfe et al. 2016b). However, little research has evaluated whether current working guidelines are accurate and effective in identifying potential areas for lesser prairie-chicken recovery and habitat conservation. WAFWAs HEG is consistent with habitat suitability index (HSI) models that are used by many federal and state agencies to standardize and evaluate habitat quality for a variety of species. HSI models are developed using the best available information from the literature and expert opinions, incorporating several limiting factors at multiple spatial scales into an overall rating of habitat quality (USFWS 1981). HSI models are often accused of being too subjective and ultimately unreliable. Despite this concern, however, HSI models continue to be used by several agencies to monitor habitat impacts and evaluate habitat quality for a wide range of species without ever validating their effectiveness—WAFWA’s HEG included (Roloff and Kernohan 1999). Because WAFWA’s RWP makes up a large portion of lesser prairie-chicken conservation efforts, research is needed to evaluate whether protocols for identifying potential habitat for species conservation are sufficient enough to set the lesser prairie-chicken on a path for long-term recovery.

Ecological Sites: a tool for conservation management

In the mid-1990s the USDA, Natural Resource Conservation Service developed an ecological site system that divided the landscape up into discrete geographic areas of differing ecological potential and processes for the purpose of providing resource managers with a tool to manage and conserve rangeland and forestland. An ecological site (ES) is defined as “a distinctive kind of land based on recurring soil, landform, geological, and climate characteristics that differs from other kinds of land in its ability to produce distinctive kinds and amounts of vegetation and in its ability to respond similarly to management actions and natural disturbances” (USDA 2006). Ecological sites are part of a nested hierarchical land classification and management system of broad scale major land resource areas (MLRA) which are nested within even broader scaled land resource regions (LRR); MLRAs divide up land areas with similar elevation, topography, climate, water, soils, biological sources, and land uses and LRRs divide the landscape up into areas of similar agronomic potential. This hierarchical framework breaks the landscape up into manageable categorical classes of differing plant communities and are regularly used by resource managers as a tool for managing vegetation, restoration, and monitoring rangeland condition (USDA 2006).

Ecological site descriptions (ESD) provide full reports on the climate, soil, hydrology, ecological dynamics, and the historical plant community for each distinct ES. Historical plant communities are described by the amount, types, and proportions of different plant species that represent the natural vegetative condition at the time of European settlement. Many range activities, or lack thereof, can influence changes in historical plant communities and thus existing plant communities are often different from what was established as the historical plant community for a specific ES. For example, plant communities that are subject to continued disturbance (e.g., overgrazing), or alternatively, lack of natural disturbances (e.g., natural fires), will rarely resemble the natural vegetative community. This comparison has been termed the similarity

index (SI) and is often used by rangeland managers to monitor the extent and direction of changes in vegetative composition over time in response to management activities such as prescribed fire and grazing. The SI can provide baseline information for predicting how new management strategies can change range conditions within a given ES. SI values are generally expressed as a percentage by weight (biomass) of the historical climax plant community presently at the site (NRCS 2003).

Researchers have ranked ecological sites in terms of their capacity to support reproductive habitat for lesser prairie-chickens (Van Pelt et al. 2013). However, little research has investigated relationships between prairie grouse habitat use and ESs, and what research that has been done has provided variable conclusions (Doherty et al. 2011, Anderson et al. 2015, Kraft 2016). Furthermore, no studies have examined relationships between ecological site condition (similarity index), related fine-scale vegetative conditions, and prairie grouse habitat use. While it is probable that ecological sites have the potential to infer habitat quality for lesser prairie-chickens, it is difficult to describe habitat quality solely using ESDs. Yearly variation in rangeland management activities (e.g. prescribed fire and grazing) and local environmental conditions (e.g. precipitation and tree invasion) can influence vegetative composition and structure, producing differing habitat conditions from one year to the next. As ecological sites are commonly used to make land management decisions and delineating areas for lesser prairie-chicken conservation, more research is needed to assess how best to use ESs and their relative condition as a habitat metric.

Technology in Conservation

Recent technology advancements have afforded the ability to empirically quantify and statistically evaluate habitat quality for a multitude of species. Research on the relationships between landscape characteristics and prairie grouse have provided extensive evidence that prairie grouse require large blocks of habitat and are highly sensitive to landscape composition (Woodward et al. 2001). However, it remains unclear how coinciding landscape characteristics geographically constrain prairie grouse distributions. Spatially-explicit models can be used to fill those knowledge gaps and predict the presence of prairie grouse populations. Moreover, model estimations can be used to assess the best place to allocate not only restoration efforts, but the practicality of translocations (Niemuth 2003, Sullins et al. 2019). As lesser prairie-chickens are an area-sensitive species and respond sharply to changes in landscape composition, research is needed to objectively identify the best possible areas for habitat restoration and reintroduction.

JUSTIFICATION

Populations of lesser prairie-chickens in the mixed-grass prairie could potentially go extinct in the next 100 years unless targeted conservation efforts are put into place to improve habitat quantity, quality, and connectivity (Garton et al. 2016). Habitat loss and fragmentation resulting from the expansion of eastern red cedar remains a key threat to population persistence and continues to reduce and isolate lesser prairie-chicken populations. Although significant efforts made by federal, state, non-governmental organizations, and private landowners have slowed the rate of tree expansion, many areas have seen limited success; some landowners are unaware or indifferent to the effects of trees on native biota, and some agencies still promote the use of eastern red cedar for wind protection and erosion control (Wolfe et al. 2016a). Additionally,

energy development has likely fragmented habitat further and exacerbated declines in many lesser prairie-chicken populations in the mixed-grass prairie (Jarnevich and Laubhan 2011).

Isolation of smaller populations sizes can have severe negative consequences on the genetic structure and persistence of lesser prairie-chicken populations. Because lesser prairie-chickens have limited dispersal capabilities, broad-scale habitat fragmentation can reduce connectivity between subpopulations and thus the potential for demographic rescue and maintenance of genetic diversity through the interchange of individuals or natural recolonization. This makes smaller, isolated populations exceedingly vulnerable to inbreeding depression, loss of genetic diversity, demographic stochasticity, impacts of severe local environmental conditions, and ultimately extirpation (Bouzat et al. 1998, Brook et al. 2002, DeYoung and Williford 2016).

Lesser prairie-chickens are largely a residential species; thus, some researchers have questioned the significance of inbreeding depression and loss of genetic diversity on population performance. While current lesser prairie-chicken populations in the mixed-grass ecoregion have maintained high levels of genetic diversity (Hagen et al. 2010), shrinking populations increase the probability of closer relatives inbreeding, resulting in declines in individual fitness and poor reproductive performance. For example, small, isolated populations of greater prairie-chickens in Illinois have experienced severe declines in reproductive success due to inbreeding depression that resulted in low fertility and hatching rates (Keller and Waller 2002). Thus, increasing the size and connectivity of sub-populations is a priority for lesser prairie-chicken conservation (Allendorf and Ryman 2002). Additionally, habitat restoration in areas previously occupied and adjacent to the species' current restricted distribution is critical for species recovery through either natural recolonization or managed reintroductions (Bouzat et al. 2009, Ross et al. 2016).

The development of quantitative methods for 1) identifying and prioritizing areas for lesser prairie-chicken reintroductions and 2) evaluating the relative value of habitat restoration activities would provide critical information for lesser prairie-chicken conservation. Currently, the relative suitability of areas for lesser prairie-chickens are based upon habitat suitability indices (HSIs). Although easy to develop and apply, HSIs are subjective and often lack scientific rigor (Brooks 1997, Roloff and Kernohan 1999). Validation of HSIs requires models to accurately predict species presence across an entire range of habitat conditions, where scores of habitat quality are calibrated between 0 and 1 (e.g. scores of 0–0.2 indicate poor habitat quality and scores of 0.8–1.0 indicate optimal habitat quality). Although WAFWAs HEG, an HSI, has been used extensively to evaluate habitat quality and predict lesser prairie-chicken distribution, it has yet to be validated. Failure to validate WAFWAs HEG could potentially produce unreliable predictions of habitat quality and ultimately waste time and resources in allocating conservation efforts to areas of unsuitable habitat.

As 95% of grasslands in both Kansas and Oklahoma are in private ownership (USDT 2006), it is likely that the majority of potential areas for lesser prairie-chicken reintroduction and habitat restoration occurs on private land, as well. Consequently, land management decisions by private landowners can and will influence the quantity and quality of fine-scale reproductive habitat vital to lesser prairie-chicken viability. For example, prescriptive management actions such as prescribed fire, grazing, and tree removal can positively or negatively affect the structure and

composition of vegetation important to lesser prairie-chicken nest and brood survival. Thus, because population dynamics in lesser prairie-chickens are largely driven by nest and brood survival, easy-to-use tools should be developed to assist landowners in monitoring and maintaining habitat to improve lesser prairie-chicken reproductive success. Ecological site condition is one common tool used by landowners to monitor the overall health and potential for an area to respond to specific land-use practices (NRCS 2003). In addition, several conservation entities, including those listed above, commonly reference ESs and their respective historical climax communities to evaluate landscapes for the relative suitability and potential to produce nesting and brooding habitat. However, little research has investigated the effectiveness of using ESs as a habitat metric, let alone examined the relationships between the relative condition (similarity index) of ecological sites and the amount, type, and quality of lesser prairie-chicken habitat. Nonetheless, changes in the SI are likely to have direct effects on the availability and quality of reproductive habitat. For example, following severe disturbance events such as fire or concentrated grazing, land areas will typically have a lower SI with greater percentage of bare ground and higher cover of forbs which may result in higher quality foraging and brood habitat (Collins 1987, Riley and Davis 1993, Manzer and Hannon 2008). Alternatively, areas that have experienced low-to-moderate levels of disturbances over long periods of time may have a relatively higher SI, with higher cover of native grasses and shrubs that are preferred by lesser prairie-chickens when selecting nesting habitat (Gulick 2019, Lautenbach et al. 2019). Thus, quantitatively evaluating fine-scale habitat conditions in relation to ecological site condition could provide insights relevant to land management considerations when managing grassland habitat for nesting and brooding habitat.

OBJECTIVES

This study will quantify potential habitat and evaluate common methods for assessing habitat availability and quality for lesser prairie-chicken recovery in the mixed-grass ecoregion of Kansas and Oklahoma. My objectives are:

Objective 1. Delineate potential suitable habitat for lesser prairie-chicken reintroduction and evaluate the effectiveness of habitat restoration. I will accomplish this by addressing specific sub-objectives:

- 1.1. Develop and validate predictive landscape-level habitat suitability models to identify and prioritize suitable habitat for lesser prairie-chickens,
- 1.2. Estimate the population sizes that currently unoccupied habitat patches may potentially support,
- 1.3. Evaluate habitat patch connectivity and identify potential corridors for natural recolonization of unoccupied areas,
- 1.4. Quantify changes in available habitat resulting from tree removal throughout the mixed-grass ecoregion.

Objective 2. Quantify the abundance of fine-scale reproductive habitats for areas participating in incentive-based conservation programs.

- 2.1. Estimate the relative abundance of nesting and brood-rearing habitat available for participating in incentive-based programs.
- 2.2. Examine associations between common fine-scale vegetation measurements that describe nesting and brood-rearing habitat and the relative condition of ecological sites.

Objective 3. Assess the accuracy of existing habitat assessments in predicting lesser prairie-chicken distribution and habitat suitability by comparing results of our landscape-level habitat model and fine-scale habitat inventory to WAFWAs' commonly applied HEG.

EXPECTED BENEFITS

This research will assist future reintroduction and habitat restoration plans for the mixed-grass prairie ecoregion by identifying areas to prioritize for lesser prairie-chicken reintroductions and habitat restoration. This research will also provide empirical information on the potential increase in lesser prairie-chicken habitat following wildfire events that reduced the coverage of invasive trees. Evaluations of fine-scale habitat, as well as identified relationships between ecological site conditions and lesser prairie-chicken habitat conditions will be used to develop specific management recommendations for areas participating in incentive-based conservation programs. Additionally, this research will examine habitat assessment frameworks used by conservation agencies to manage lesser prairie-chicken habitats in the mixed-grass prairie ecoregion. Specific products from this research project will include:

- 1) Spatially-explicit predictions of potential lesser prairie-chicken habitat use and maps delineating potential habitat patches for reintroduction and potential corridors for lesser prairie-chicken movement for the entire mixed-grass prairie ecoregion in southern Kansas and northern Oklahoma.
- 2) Empirical information on the change in available lesser prairie-chicken habitat resulting from tree removal.
- 3) Empirical information on the amount of fine-scale reproductive habitat available for lesser prairie-chickens for areas participating in incentive-base programs, accompanied by recommendations to promote lesser prairie-chicken habitat.
- 4) Information on how to better use ecological site descriptions and the similarity index to monitor lesser prairie-chicken habitat.
- 5) An evaluation of current working guidelines being used to assess the suitability of lesser prairie-chicken habitat supplemented with recommendations to improve current methods.
- 6) A finished Master's thesis summarizing completed research.
- 7) At least two scientific publications related to the mixed-grass ecoregion lesser prairie-chicken population coauthored by project investigators at Montana State University.

STUDY AREA

This study will identify and prioritize habitat for lesser prairie-chicken reintroduction across the entire mixed-grass prairie ecoregion within southcentral Kansas and northern Oklahoma (Figure 2). Vegetation in the mixed-grass prairie ecoregion can be characterized as a mixture of sand sagebrush (*Artemisia filifolia*) and mid-height perennial grasses. Woody vegetation includes sand

plum (*Prunus spp.*) and yucca (*Yucca glauca*), as well as eastern red cedar and the cottonwood (*Populus deltoides*) (Table 3). Upland soils are typically deep, loamy sands and precipitation in the area generally ranges 40–75 centimeters annually from east-to-west. The primary land-use for the area is livestock grazing (USDA, Natural Resource Conservation Service). The mixed-grass prairie ecoregion contains the second largest extant of lesser prairie-chicken populations, with populations estimated at just under 4,000 birds in 2013 (McDonald et al. 2014, Garton et al. 2016)

Part of this study will be located on the Z Bar Ranch, a 43,000-acre bison production ranch owned and managed by Turner Enterprises, Inc. (TEI) in the Gypsum Hills of Kansas. The population of lesser prairie-chickens at the Z Bar has steadily declined during the last 25 years, with the last known lek disappearing about 15 years ago. The cause of the local population extirpation is unknown but is likely due to the loss and fragmentation of grasslands resulting from the expansion of eastern red cedar and energy development. Only an occasional lesser prairie-chicken is now observed. However, in 2014 TEI joined the WAFWA incentive program in an effort to restore a portion of the ranch to conditions considered suitable for lesser prairie-chickens. This cooperation included the mechanical removal of eastern red cedar and other woody species across ~25,000 acres of land, changes in grazing utilization and pasture rotation, and a prescribed fire plan that, in theory, would produce a mosaic of vegetative structure able to support a viable population of lesser prairie-chickens (WAFWA 2014). Additionally, a wildfire burned 90% of the Z Bar in 2016, removing woody debris from tree cutting work and rejuvenating native grasses and forbs. Nevertheless, lek surveys conducted in 2017 and 2018 have detected no sign that a breeding population of lesser prairie-chickens have recolonized the Z Bar.

METHODS

Objective 1. Delineate potential suitable habitat for lesser prairie-chicken reintroduction and evaluate the effectiveness of habitat restoration.

Data Collection

I will obtain lek location and survey data for the mixed-grass prairie ecoregion collected by the Kansas Department of Wildlife, Parks and Tourism and the Oklahoma Department of Wildlife Conservation. Landscapes associated with stable leks will be used to represent habitat conditions that support stable sub-populations of lesser prairie-chickens. Stable leks will be defined as leks that consistently have had 10 birds counted for five of the past 10 years and occurred in the same geographic location from one year to the next. Leks that do not meet this criterion or have been observed to be recently declining will be removed.

Because the majority of lesser prairie-chicken activity occurs within 5 km of a lek, I will conduct geospatial analyses to measure habitat conditions within a 5 km radius of both stable lek locations and random points. Habitat use is often inherently autocorrelated in space and time (e.g., lek locations), thus I will use a Moran's I test in the DHARMA R package to test for non-independence between lek locations that are in close proximity to one another (Moran 1950). If independence between lek locations is found to be violated, a minimum convex polygon with a 5

km buffer will be drawn around each identified lek cluster. I will then conduct geospatial analyses to measure habitat conditions within a 5-km radius of both stable leks or lek clusters and random points. Random points will be randomly sampled at a ratio of 20 random points to 1 lek within a study area defined by a minimum convex polygon placed around all stable lek or lek cluster locations (Smith et al. 2005, Milligan 2019). I will measure habitat conditions by identifying and evaluating abiotic and biotic habitat characteristics that have been found to be important to lesser prairie-chicken ecology using remotely sensed environmental geodata sets and a geographic information system (GIS). Potential habitat characteristics will include grass cover, tree canopy cover, distance to nearest tree, cropland cover, density of oil wells, roads and powerlines, mean annual precipitation and temperature, and landscape roughness (Table 4).

Development of Habitat Suitability Model

To identify suitable patches of potential habitat for reintroduction, I will develop habitat suitability models using resource selection functions (RSFs) and machine learning tools, such as Random Forests, to evaluate first-order habitat selection as a function of several habitat characteristics in a used (1) versus available (0) study design (Johnson 1980, Boyce and McDonald 1999, Manly et al. 2002). Resource selection functions use measurements of used versus available habitat variables to quantify the relative probability of use for any given unit in space (Boyce et al. 2002, Manly et al. 2002). Measurements of potential habitat variables (e.g., Table 4) obtained from geospatial analyses will be used as covariates in models comparing habitat conditions at stable lek locations and random points (Boyce and McDonald 1999, Manly et al. 2002). Any given model will have a minimum of 10 leks per covariate (Hosmer 2000). To develop our RSF model, I will use a generalized linear model (GLM) with a binomial error structure to estimate mean coefficients of each covariate and calibrate a fully parameterized RSF model using program R (R Core Team 2014). Depending on the results of our previously conducted Moran's I test, spatial autocorrelation between lek locations will be considered for model development by identifying and including a random effect of lek clusters. Additionally, prior to fitting our RSF model, I will screen for multi-collinearity between each pair of covariates using Pearson's correlation coefficient (r). If two covariates are found to be highly correlated, I will identify which covariate to include in our model by using logistic regression to measure which one of the correlated pair accounted for the most variation in the data. Finally, to evaluate the reliability of our final RSF model I will withhold data from our original sample of leks and evaluate predictive performance using a Receiver Operating Characteristic (ROC) combined with an Area Under the Curve (AUC) analysis (Fielding and Bell 1997, Hosmer et al. 2000, Boyce et al. 2002).

Alternately, I will upload measurements of potential habitat variables (Table 4) obtained from previous geospatial analyses into the Random Forest package using program R. Random forest models (RF) are developed from a classification system that combines bootstrap aggregation with regression trees and can be used to predict species distribution. Random forest models function by allowing each individual tree to randomly sample a sub-set of covariates, resulting in the development of several weak sub-models that cumulatively converge to make one stable predictive model. Random forests are desirable because they are fast and they account for several concerns associated with spatial data such as overfitting, robustness to noise, and autocorrelation. Additionally, because RFs are developed from several sub-models that were similarly created from a subset of covariates, they negate the need to inspect covariates for multicollinearity (Breiman 2001, Evans et al. 2011). To evaluate the predictive performance of our RF model, I

will use ROC and AUC on the same data set that was withheld from model development to validate our RSF model. I will evaluate and compare the relative accuracy of our best RSF model and RF model by comparing the sum of the proportion of stable lek locations correctly predicted as suitable habitat (sensitivity) and the proportion of random points predicted as unsuitable habitat (specificity) for each model (Fielding and Bell 1997).

Given that both models will likely be highly accurate yet vary slightly in performance and spatial predictions of stable lek locations, I will consider using ensemble methods to combine our RSF model and our RF model into one robust habitat suitability model. Ensemble methods simultaneously apply multiple models into a consensus style framework, with the idea that if you combine spatial predictions from more than one model, you reduce the predictive uncertainty that would come from just using one model. Thus, I will combine our RSF model with our RF model using a Mean(all) approach and a Weighted Average approach. Mean(all) takes the average value of the final outputs from all single models and Weighted Average pre-evaluates single models and weights them according to which model has a higher predictive accuracy (Araújo and New 2007, Marmion et al. 2009). Our combined RSF-RF habitat suitability model will be validated on the same withheld data set and the same methods (ROC and AUC) that our RSF and RF models were previously validated with (Fielding and Bell 1997).

Delineating potential habitat patches for reintroduction

To identify contiguous patches of potential lesser prairie-chicken habitat I will first use GIS to create a hexagonal grid of 30 m resolution across the entire mixed-grass prairie ecoregion of southcentral Kansas and northern Oklahoma. I will then extrapolate our habitat suitability model and apply it to the historical range of lesser prairie-chickens to quantify habitat conditions within each grid cell. Each habitat variable will be individually quantified using a 5 km buffered moving window analysis at each 30 m grid cell. Predictions for each habitat variable will be limited to the range of values (inference space) measured at both stable lek locations and random points, where I will assign a N/A to any habitat variable that lies outside the measured range used to develop our habitat suitability model. RSF values for each grid cell will be rescaled to value ranging between zero to 1, with a score of zero being the least suitable, and a score of 1 being ideal lesser prairie-chicken habitat. Habitat patch sizes will be measured by summing up the area of adjacent 30 m grid cells that have reached a relative probability ≥ 0.80 of a stable lek occurring. Identified habitat patches will be classified based on their size and the proportion of 30 m cells ranked as high quality lesser prairie-chicken habitat, keeping in mind the recommended area (21,000 – 25,000 acres) needed to support multiple leks and a sustainable lesser prairie-chicken population (Applegate and Riley 1998, Bidwell et al. 2003, Van Pelt et al. 2013).

Evaluating patch connectivity

To evaluate the relative probability for natural re-colonization to potential habitat patches, I will use our habitat suitability model combined with a least-cost path (LCP) analysis to identify potential patches of connectivity for lesser prairie-chicken movement. Least-cost path analysis assumes that the cost for an animal to move between habitat patches is inversely related to the relative probability of use, or in our research the relative probability of a stable lek occurring. Our habitat suitability model will be used to generate source habitat patches of existing populations and potential suitable habitat for lesser prairie-chickens. I will then use our habitat suitability model values as a least-cost path function in GIS to identify corridors for lesser prairie-chicken

movement between existing populations and potential suitable habitat (Chetkiewicz and Boyce 2009).

Estimating population sizes

To predict the number of lesser prairie-chickens that the Z Bar Ranch could potentially support I will link our habitat suitability model to estimated population sizes for existing lesser prairie-chicken populations. Provided we know the estimated number of lesser prairie-chickens within the current range in the mixed-grass ecoregion, I will calculate the total predicted area required to support the current population using our habitat suitability model. I will then be able to use the ratio of the existing habitat patch size and the associated current population to predict the potential numbers of lesser prairie-chicken in any given habitat patch by using the following equation:

$$N_{potential} = X_{potential} * \frac{N_{current}}{\sum X_{current}}$$

where $N_{current}$ is the lesser prairie-chicken population estimate for the existent population in the mixed-grass ecoregion, $X_{current}$ is the sum of relative probabilities of use for the habitat suitability model for the existent population, and $X_{potential}$ is the sum of relative probabilities of use for a potential lesser prairie-chicken habitat patch. To estimate the error in predicted populations sizes for the Z Bar Ranch, I will calculate the mean and standard deviation of the current population of lesser prairie-chickens across years 2009–2018 to construct a 95% confidence interval for a subset of stable lek locations. I will then use the confidence interval constructed for existing populations and the relative suitability of habitat at the Z Bar to develop a 95% confidence interval for the potential lesser prairie-chicken population size at the Z Bar Ranch (Boyce and McDonald 1999, Boyce and Waller 2003).

Estimating changes in available habitat provided by recent tree removal

In March 2016 and 2017 southcentral Kansas and northern Oklahoma experienced some of the largest and hottest wildfires in their history. Notably, the Anderson Creek wildfire (2016) and Starbuck wildfire (2017) burned over one million acres within the current distribution of lesser prairie-chickens in the mixed-grass prairie. As a result, several thousand acres of trees were killed and partially removed from the landscape, likely shifting vegetation from a more woodland plant assemblage to a grassland plant assemblage and potentially increasing the quantity and quality of lesser prairie-chicken habitat.

To quantify the relative change in available lesser prairie-chicken habitat, I will use our habitat suitability model to compare current habitat conditions to habitat conditions prior to the Anderson Creek and Starbuck wildfires for each entire burn area. I will extrapolate our habitat suitability model and apply it to the entire area affected by both wildfires to quantify the amount of lesser prairie-chicken habitat currently available. I will then upload shapefiles of landcover data for the entire Anderson Creek and Starbuck wildfire areas prior to each wildfire (years 2015 and 2016, respectively) into a GIS database and use our habitat suitability model to sum up the total amount of available habitat prior to the wildfires. Specifically, I will examine distance to nearest trees and tree density using the Euclidean distance tool in ArcGIS to create rasters depicting the distance to each tree and a moving window analysis to estimate the density of trees

within 0.8 kilometer radii of each 30 m grid cell (Lautenbach et al. 2017). I will then calculate the change in total available habitat (C) using this equation:

$$C = \sum X_{\text{current}} - \sum X_{\text{pre-fire}}$$

where $X_{\text{pre-fire}}$ and X_{current} are the summation of grid cells that have a relative probability of use of ≥ 0.80 for before and after the Anderson Creek and Starbuck wildfire, respectively.

Objective 2. Quantify the abundance of fine-scale reproductive habitats for areas participating in incentive-based conservation programs.

Animals select habitat at multiple hierarchical levels across differing spatial and temporal scales. In objective 1 I used coarse scale habitat information to predict the relative probability of a stable lek occurring across the mixed-grass ecoregion. However, local vegetative conditions such as vegetative structure and cover are also known to impact important vital rates for lesser prairie-chickens, including nest and brood survival (Hagen et al. 2009, Lautenbach 2015, Lautenbach et al. 2019). Thus, I will measure vegetative characteristics at the Z Bar Ranch to quantify the relative abundance of fine-scale reproductive habitat potentially available for lesser prairie-chickens. I will also investigate relationships between the relative availability of lesser prairie-chicken nesting and brooding habitat in relation to ecological sites and their relative condition by visually estimating similarity index values across upland portions of the ranch.

Field sampling

Quantifying available nesting and brooding habitat

Because reproductive success and survival during the breeding season has been shown to be the most important factors effecting lesser prairie-chicken population dynamics, fine-scale habitat conditions will be measured during months that coincide with lesser prairie-chicken nesting (May – June) and brooding rearing (July – August) periods. Fine-scale habitat conditions include, VOR, percent canopy cover of grass, forbs, shrubs, litter, and bareground, and litter depth (Riley and Davis 1993, Hagen et al. 2005b, Lautenbach 2015, Lautenbach et al. 2019). Sampling plots for measuring both nesting and brood-rearing habitats will be stratified first by pasture, and then by ecological sites that have similar potential to produce lesser prairie-chicken habitat. Ecological sites with similar production capabilities and climax communities that are adjacent to one another will be grouped into the same study plot. I will randomly generate fifteen transect locations within each sampling plot using the Create Random Points tool in ArcGIS. Transect locations that fall within areas of unsuitable habitat (i.e. forested areas, riparian areas, two-track and county roads) will be replaced.

Once in the field, I will determine the necessary number of transect locations needed to accurately account for variation in vegetation within each study plot. First, I will conduct vegetative sampling at 5 of the randomly generated points for each sampling plot. I will then calculate the mean and standard deviation for vegetative covariates found to be the most important to reproductive success (e.g., VOR and percent bareground for the nesting period and VOR and % forb cover during the brooding period) using this 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, s is the standard deviation, and B is the mean multiplied by the desired precision (Elzinga et al. 1998; Appendix 7). I will continue to collect vegetative measurements at additional random points until an adequate sampling size is reached to account for in-plot variation in vegetative conditions, where sampling plots with higher levels of variation within vegetation measurements will have more random points sampled and vice versa.

At each transect location, I will estimate percent canopy cover of shrub, grass, forbs, bareground, and litter using a 60 x 60 Daubenmire frame directly at random points and at 4-m from the random point in each cardinal direction (five estimates/random point). Visual obstruction will be recorded at the transects point center from each cardinal direction at a distance of 4 meters and a height of 1 meter. I will measure litter depth (cm) and plant height at 0.5 meter increments along each 4-m stretch from the random points center (Figure 4). I will also estimate percent canopy cover of preferred grasses of lesser prairie-chickens during the nesting period (e.g., little bluestem, sideoats grama, big bluestem, Indian grass, sand bluestem, and switchgrass). Likewise, during the brood rearing period, I will estimate percent cover of forbs found to be important components of lesser prairie-chicken diets during the brooding and winter periods (Sullins et al. 2018a). Correlations found between forb cover and invertebrate densities will also allow us to infer the relative availability of invertebrates as food for lesser prairie-chickens during the brooding period (Table 6; Jamison et al. 2002, Hagen et al. 2005b).

Examining availability of reproductive habitat in relation to ecological site condition

At each fine-scale sampling location, ecological site condition will be described using a similarity index by walking a 4-meter radius circle around random points where I will conduct ocular surveys to visually compare the current plant community to its historical climax community. Ecological site conditions will be estimated during both the nesting and brood-rearing periods.

Prior to sampling each day, I will calibrate my ability to characterize ecological site condition by visually estimating the proportion of decreaser plant species within a 20 x 50 m Daubenmire frame by first clipping and weighing the overall vegetative biomass. I will then separate and weigh the decreaser plant forms independently from the invader and increaser plant forms and calculate the overall proportion of decreaseers, or the similarity index (SI), using this equation:

$$SI = (D \div X) * 100\%$$

where D is the weight of the decreaser plant forms and X is the total vegetative weight. SI values will be divided up into 5 classes based on overall similarity to the historical climax community where, for example, 0 – 20% would indicate low similarity and 80 – 100% would indicate high similarity.

Statistical analysis

Quantifying available nesting and brooding habitat

Based on evidence from the literature, habitat variables measured above (e.g., VOR, grass cover, forb cover, bare ground) will be sorted into classes of suitable and unsuitable lesser prairie-chicken habitat (Table 5). Transect locations with an average VOR between 1.5 – 3.5 dm and less than 10% bareground will be classified as optimal nesting habitat. Similarly, points with 10%

– 35% forb cover and VORs between 2 – 5 dm will be classified as optimal brood habitat (Lautenbach 2015, Lautenbach et al. 2019, Gehrt et al. 2020). Vegetative measurements below or above optimal habitat conditions will be classified as unsuitable nesting and brooding habitat.

To quantify the proportion of reproductive habitat (R_{total}) potentially available for lesser prairie-chickens within each sampling plot, I will calculate the amount of available nesting and brooding habitat using the following equation:

$$R_{total} = ((N + B) \div X) * A$$

where N and B are the proportion of transect locations classified as optimal nesting and brooding habitat, respectively, X is the total amount of transect locations and A is the number of acres within each sampling plot. I will then quantify the total amount of available reproductive habitat available at the Z Bar Ranch by summing up the total amount of acres from each sampling plot.

To identify areas that meet the criteria for optimal habitat at both large- and fine-scale levels on the Z Bar Ranch, I will first use our habitat suitability model to identify habitat patches with a radius of 5 km or more where a stable lek could potentially occur on the Z Bar Ranch. I will then quantify the amount of optimal nesting and brood rearing habitat available at each identified area by taking the proportion of random points within that area that meet the criteria for optimal reproductive habitat and divide it by the total amount of random points within that given area. Areas identified by our habitat suitability model will be sorted by increasing potential to sustain a lesser prairie-chicken population, with habitat patches that have $\geq 50\%$ of random points classified as suitable reproductive habitat (25% nesting and 25% brood rearing) having the greatest potential. Evaluations of fine-scale nesting and brooding habitat combined with large-scale predictions of potential stable lek occurrence will provide detailed estimations of the total amount of available lesser prairie-chicken habitat at the Z Bar Ranch. Conclusions from these analyses will be used to make future management recommendations to improve lesser prairie-chicken habitat at the Z Bar Ranch.

Examining availability of reproductive habitat in relation to ecological site condition

To examine relationships between ecological site condition at specific ESs and the amount of nesting and brood rearing habitat available, I will compare SI values to percent cover of native grasses, shrubs, forbs, litter, bare ground, and VOR. I will use logistic regression to compare SI classes to classes of nesting and brooding habitat quality based on vegetative measurements (e.g., percent cover of grass, forbs, bare ground, VOR). Additionally, I will use regression to compare ecological site condition to landscape level covariates including, year since fire, current stocking rates, whether the point was grazed or ungrazed at the time of sampling, slope, and elevation. Covariates will be uploaded using the ‘Extract Multi Values to Points’ tool in ArcGIS. Results from this objective will be used to develop a better understanding on how to use ecological sites and their relative condition to assess, monitor, and manage for lesser prairie-chicken nesting and brooding habitat.

Objective 3: Assess the accuracy of existing habitat assessments in predicting lesser prairie-chicken distribution and habitat suitability by comparing results of our landscape-level habitat model and fine-scale habitat inventory to WAFWAs’ commonly applied HEG.

To ensure current working guidelines are providing the best possible evaluations of potential lesser prairie-chicken habitat, I will assess WAFWAs Habitat Evaluation Guide (HEG) and Mitigation Framework to identify any gaps in assessment protocols and provide recommendations to improve future programs. I will use WAFWAs HEG protocol (Appendix I, Van Pelt et al. 2013) to calculate the amount of available lesser prairie-chicken habitat at the Z Bar Ranch and compare it to the amount of suitable habitat found using our habitat suitability model in objective 1. Evaluation units will be defined as similarly managed areas that exhibit the same plant community based on ecological site descriptions. Habitat variables scored and evaluated under the HEG include 1) vegetative cover, 2) vegetative composition, 3) percent cover of tall woody plants >3 ft in upland sites, and 4) the proportion of grassland cover with <1% canopy cover of trees that is within a 1-mile radius (~1.6 km) of the geometric center of each evaluation unit (Table 7). Habitat variables 1 – 3 will be scored using the vegetative sampling protocol laid out by WAFWA to monitor mitigation sites (Van Pelt et al. 2013).

Field methods

Vegetative sampling under the WAFWA protocol will be conducted concurrently with sampling from objective 2. Transects will be placed in areas found to be representative of the current plant community and structure throughout the entire evaluation unit. Vegetative measurements will be collected along a 150 foot transect that runs northeast to southwest. The zero mark will be at the northeast end with the tape measure laid as close to the ground as possible. I will collect data from the south side of the tape as to not disturb vegetative structure on the north side. I will measure the tallest plant height, living or dead, within a 6-inch radius at every 10 foot interval (including seed heads). Additionally, I will measure VOR using a Robel pole at every 10 foot interval, placing the Robel pole on the north side of the tape and recording the number of completely obstructed bands from a perpendicular distance of 2 meters across a 0.5 meter tall sighting pole. Finally, at every 3 foot interval, I will lower a wire in a vertical descent through the foliar canopy directly on the north side of the tape, recording growth forms in the order that they touch the wire.

Statistical analysis

To evaluate WAFWAs HEG and Mitigation Framework I will first calculate HEG scores for each evaluation unit by taking the minimum value from habitat variables 1 – 3 and multiply it by the score for habitat variable 4, where HEG scores will range from a low score of 0.0 to a high score of 1.0. I will calculate habitat patch sizes at the Z Bar Ranch by taking the final HEG scores multiplied by the total acreage for each evaluation unit. The total amount of lesser prairie-chicken habitat available within those evaluation units will be found by subtracting the acreage impacted by landscape characteristics that are not considered potential habitat for lesser prairie-chickens (Table 8).

An HSI model can be justified only if it accurately predicts species presence, or better yet reproductive success. Given that our habitat suitability model will be developed and validated on landscapes with habitat characteristics that have continually supported stable leks for 5–10 years, we can assume that it accurately predicts locations of stable populations and reproductive success. Thus, I will indirectly validate WAFWAs HEG by comparing HEG scores to the

probability of stable lek occurring from our habitat suitability model for the entire Z Bar Ranch. I will also calibrate and assess whether WAFWAS HEG is robust across the entire range of lesser prairie-chicken habitat quality, systematically comparing HEG scores to all possible scores of suitability from our habitat suitability model. Additionally, I will visually examine differences between habitat patch sizes and the total amount of lesser prairie-chicken habitat within those patches for areas predicted by both models using GIS.

Determining effective sampling sizes

Habitat Suitability Index models, such as WAFWAs HEG, are often accused of not accounting for variation attributed to sampling error or inherent heterogeneity within habitat measurements of evaluation units. Moreover, one of the weakest components of most HSI validation studies is that they do not consider variation within input data (Bender et al. 1996, Roloff and Kernohan 1999). Thus, to clarify whether WAFWAs HEG is able to capture the variation within their HSI model I will generate a set of random points for a subset of evaluation units on the Z Bar Ranch. I will collect vegetative measurements using WAFWAs HEG protocol from a portion of randomly generated transect locations and calculate the mean and standard deviation for habitat variables 1–3. I will then use the exact same methods in objective 2 (Elzinga et al. 1998) to determine an effective sample size. Next, I will examine HEG scores for habitat variables 1–3 at various sampling efforts; for example, I will compare evaluation units with HEG scores that had only 2 sampling transects to HEG scores that had 5, 10, or 15 transects. Results from this analysis will be used to demonstrate how variation within evaluation units affects overall HEG scores and could potentially provide false indications of habitat suitability or unsuitability.

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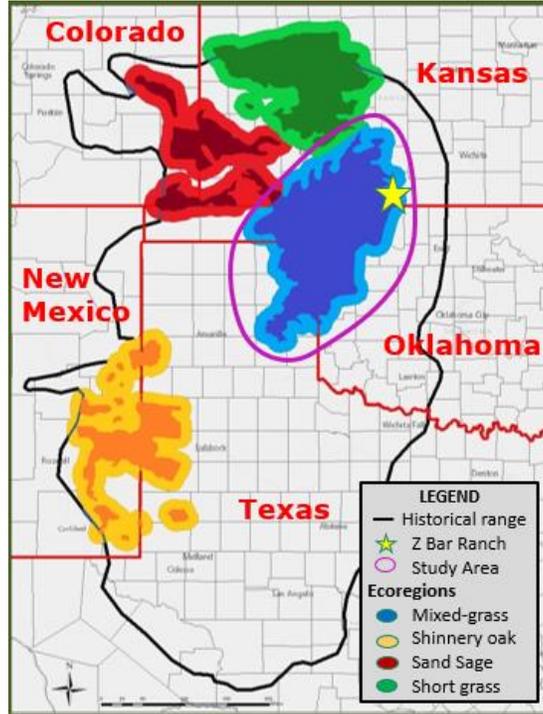


Figure 1. Estimated historical range and current occupied range for lesser prairie-chickens.

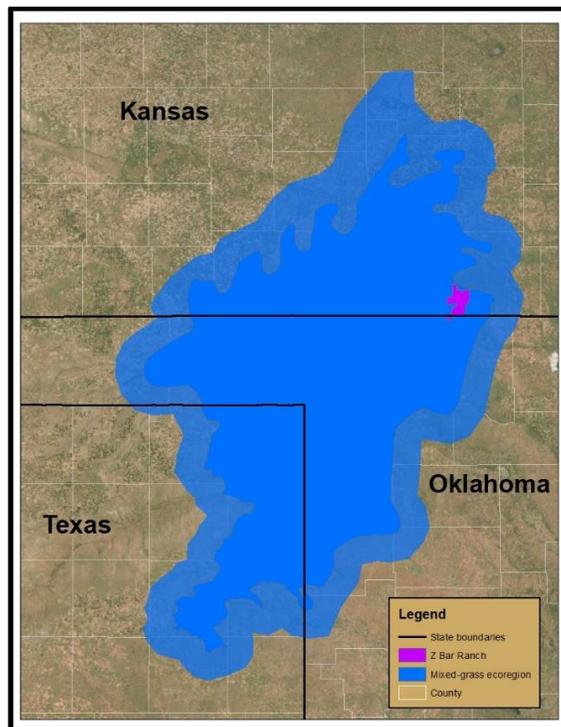


Figure 2: Extent of our study area in the mixed-grass ecoregion. Purple polygon indicates the Z Bar Ranch.

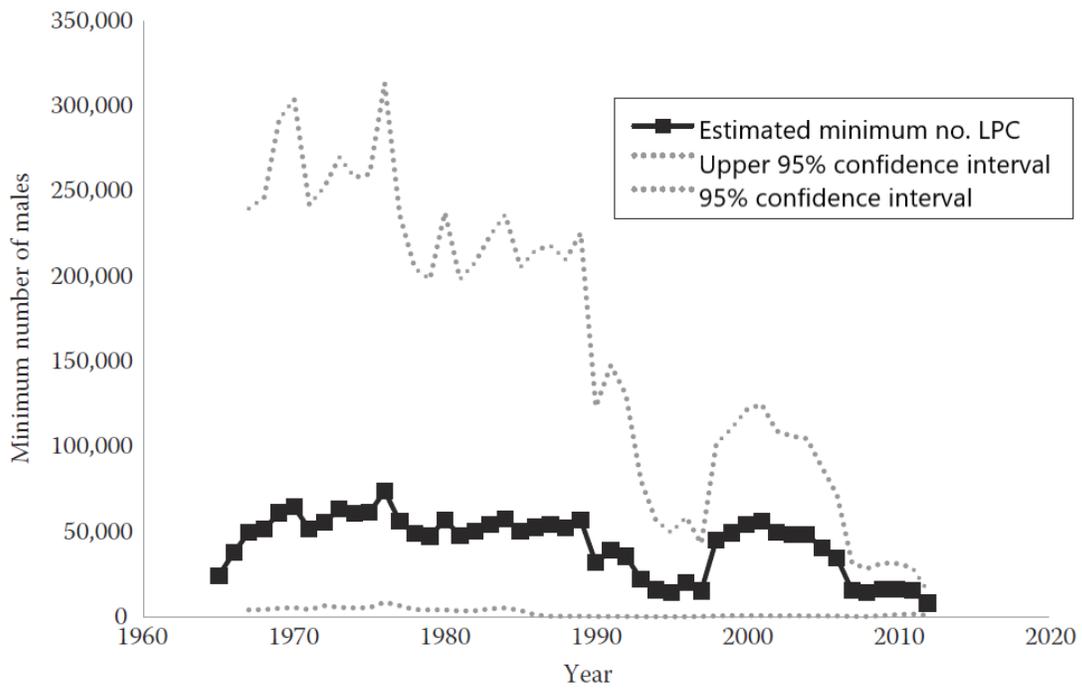


Figure 3: Estimated minimum number of lesser prairie-chickens attending leks 1965–2012 (95% CI) in the Mixed-Grass Prairie Ecoregion based on population reconstructions. Source: Garton et al. 2016.

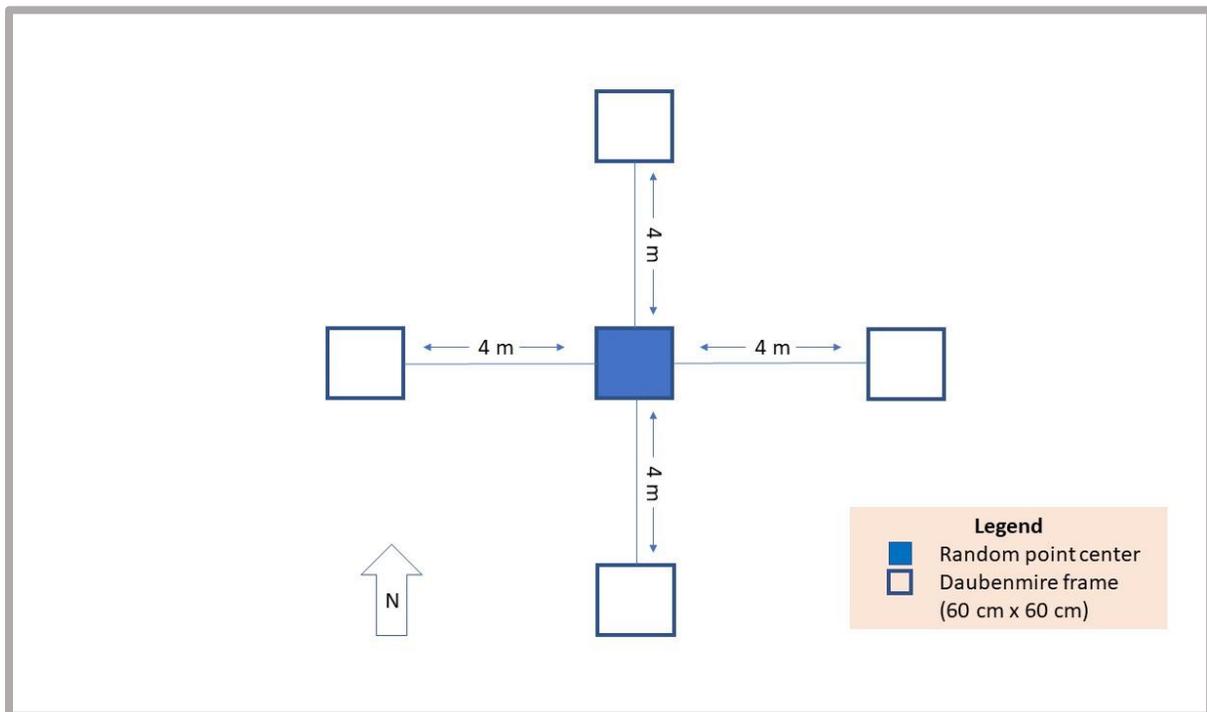


Figure 4: Transect design for vegetative sampling (objective 2).

Table 1: Estimates of Nest Survival and Nest Success for Lesser Prairie-Chickens (CI 95%)

State	Ecoregion	Year	Number of Nests Monitored	Apparrent Nest Success	Estimated Nest Survival	Sources
NM	SSO	1976 – 1978	36	0.27		Wisdom 1980
NM	SSO	1979	13	0.54		Merchant 1982
NM	SSO	1980	11	0.0		Merchant 1982
NM	SSO	1979	15	0.47		Wilson 1982
NM	SSO	1980	11	0.0		Wilson 1982
TX	SSO	1987 – 1988	13	0.15	0.16	Haukos 1988
NM	SSO	1976—1978	36	0.27		Riley 1992
KS	SGP	2001 – 2003	59	0.48	0.72	Fields 2006
KS	SSB	1997—2002	196	0.26 +/- 0.03		Pitman 2006
NM	SSO	2004 – 2005	23	0.76		Davis 2009
TX	SSO, SSB	2001 – 2003	21	0.67		Jones 2009
TX	SSO, SSB	2001 – 2003	22	0.64	0.54	Lyons 2011
TX	SSO	2003 – 2005	35	0.37	0.29	Lyons 2011
NM	SSO	2001 – 2011	182		0.24	Grisham 2012
TX	SSO	2008 – 2010	24	0.37	0.35 +/- 0.05	Holt 2012
TX, NM	SSO	2001 – 2011	229		0.57	Grisham 2013
TX	SSO	2008 – 2011	36		0.43	Grisham 2014
KS	SGP, MGP	2013 – 2014	185	0.26	0.39	Lautenbach 2019

Table 2: Estimated brood and chick survival for Lesser Prairie-Chickens (95% CI)

State	Ecoregion	Year	Number Observed	Type	Duration	Survival	Sources
KS	SGP	2001—2003	27	Brood	60 days	0.28 apparent	Fields 2006
				Brood	60 days	0.49 reared by adults	
				Brood	60 days	0.05 reared by yearlings	
KS	SSB	1997—2003	19	Brood	60	0.41 apparent	Pitman 2006
TX	SSO	2008—2010	9	Chick	63 days	0.88 daily	Holt 2012
KS	SGP, MGP	2013—2014	43	Brood	56 days	0.31	Lautenbach 2015

Table 3: Common plant forms in the mixed-grass prairie ecoregion

Grasses		Forbs		Shrubs		Tree	
Common name	Scientific name	Common name	Scientific name	Common name	Scientific name	Common name	Scientific name
Alkali sacaton	<i>Sporobolus airoides</i>	Cuman ragweed**	<i>Ambrosia psilostachya</i>	Broom snakeweed	<i>Gutierrezia sarothrae</i>	Cotton-wood	<i>Populus deltoides</i>
Cheat grass*	<i>Bromus tectorum</i>	White sagebrush	<i>Artemisia ludoviciana</i>				
Hellers rosette grass	<i>Dichantheium oligosanthes</i>	Annual buckwheat	<i>Eriogonum annuum</i>	Leadplant	<i>Amorpha canescens</i>	Eastern red cedar	<i>Juniperus virginiana</i>
		Oakleaf goosefoot**	<i>Chenopodium glauca</i>				

Hairy grama	<i>Bouteloua hirsute</i>	Prairie sandmat	<i>Chamaesyce missurica</i>	Sand sagebrush	<i>Artemisia filifolia</i>	Roughleaf dogwood	<i>Cornus drummondii</i>
Prairie sandreed	<i>Calamovilfa longifolia</i>	Canadian horseweed	<i>Conyza Canadensis</i>				
Buffalo grass	<i>Bouteloua dactyloides</i>	Purple prairie clover	<i>Dalea purpurea</i>				
Blue grama	<i>Bouteloua gracillas</i>	Curly-cup gumweed	<i>Grindella squarrosa</i>				
Little bluestem	<i>Schizachyrium scoparium</i>	Prickly lettuce**	<i>Lactuca ludoviciana</i>				
Johnson grass*	<i>Sorghum halepense</i>	Alfalfa**	<i>Medicago sativa</i>				
Indian grass	<i>Sorghastrum nutans</i>	Many-flowered scurfpea	<i>Psoraleidum tenuiflorum</i>				
Big bluestem	<i>Andropogon gerardii</i>	Scarlet globemallow	<i>Sphaeralcea coccinea</i>				
Purple threeawn	<i>Aristida purpurea</i>	Silver-leaf nightshade	<i>Solanum elaeagnifolium</i>				
Sand dropseed	<i>Sporobolus cryptandrus</i>	White heathaster	<i>Symphiotrichum ericoides</i>				
Sand lovegrass	<i>Eragrostis trichoides</i>	Common dandelion**	<i>Taraxacum officinale</i>				
Sideoats grama	<i>Bouteloua curtipendula</i>						
Silverbeard grass	<i>Bothriochloa laguroides</i>						
Sixweeks fescue	<i>Vulpia octavia</i>						
Switchgrass	<i>Panicum virgatum</i>						
Tall dropseed	<i>Sporobolus compositus</i>						
Yellow bluestem*	<i>Bothriochloa ischaemum</i>						

*Non-native plant forms

**Important lesser prairie-chicken food

Table 4: Description of potential habitat characteristics to use for predicting lesser prairie-chicken habitat use in the mixed-grass prairie ecoregion

Name	Potential sources	Description	Justification/Reference
Ecological Site	NRCS	ES classification for each grid cell	Potential relationship b/w ecological site description and LPC habitat suitability (Van Pelt et al. 2013, Kraft 2016)
Grassland cover	LANDFIRE, NLCD, Rangeland Analysis Platform	Proportion of grid cells classified as grass/herbaceous	Identified positive relationship b/w increased grassland cover and LPC abundance (Woodward et al. 2001)
Tree cover and distance to nearest tree	Falkowski et al. 2017, Rangeland Analysis Platform	Proportion of grid cells classified as tree cover and distance of grid cells to nearest tree	Identified negative relationship b/w increased tree cover and LPC abundance (Lautenbach et al. 2017)
Gross primary production	LANDFIRE, NLCD	Indicator of early brood habitat.	Identified positive relationship b/w increased forb cover and brood habitat selection (Hagen et al. 2005)
Agriculture cover	NASS	Proportion of grid cells classified as tilled agriculture	Identified negative relationship b/w increased cropland cover and LPC abundance (Woodward et al. 2001)
Oil and gas wells	IHS Oil & Well Database	Density of oil and gas well locations	Identified negative relationship b/w greater oil well densities and LPC habitat selection (Hagen et al. 2011, Plumb et al. 2019)
Transmission lines	Kansas Corporation Commission, Oklahoma Corporation Commission	Distance to/density of transmission lines	Identified negative relationship b/w density of and distance to transmission lines and LPC habitat selection (Hagen et al. 2011, Plumb et al. 2019)
Roads (major and county)	Kansas GIS & Data & Support Center	Distance to/density of roads	Identified negative relationship b/w density of and distance to roads and LPC habitat selection (Hagen et al. 2011, Plumb et al. 2019)

Buildings	Kansas GIS & Data & Support Center	Distance to/density of buildings	Identified negative relationship b/w density of and distance to buildings and LPC habitat selection (Pitman et al. 2005, Hagen et al. 2011)
Mean annual temperature	USDA (https://www.wcc.nrcs.usda.gov/gis/precip.html)	—	Identified negative relationship between greater mean annual temperatures and LPC reproductive success and survival (Grisham et al. 2014)
Mean annual precipitation	USDA (https://www.wcc.nrcs.usda.gov/gis/precip.html)	—	Large scale ecological driver influencing grassland production. Carry over effects for LPC survival (Geisen 2000)
Roughness	National Elevation Data (NED)	Standard deviation of elevation w/in a grid cell	Identified negative relationship b/w lesser prairie-chicken habitat use and rough terrain

Table 5: Fine-scale habitat quality classifications for suitable nesting and brooding habitat

Habitat variable	Nesting		Brooding	
	Unsuitable	Suitable	Unsuitable	Suitable
% Grass cover	0 – 25%	>40%	<45%	45 – 50%
% Forb cover	0 – 19%;	20 – 25%	0 – 10%; >35% †	10 – 35% †
% Bare ground	>10% †	≤10% †	0 – 9%; >25%	10 – 25%
% Litter	0 – 10%	20 – 25%	0 – 9 %; >15%	10 – 15%
Litter depth (cm)	0 – 0.5	1 – 2	>1	0 – 0.5
VOR (dm)	0 – 1.5; 5+ †	1.5 – 3.5 †	0 – 2.0; >5 †	2.0 – 5.0 †

†Fine-scale habitat variables found to be the most important to lesser prairie-chicken nest and brood habitat selection (Lautenbach et al. 2019).

Table 6: Invertebrates found to be the most important to lesser prairie-chicken diets during the brooding and winter periods for the mixed-grass ecoregion.

Family	Genus
Acrididae	<i>Melanoplus</i>
Noctuidae	<i>Dargida</i>
Pentatomidae	<i>Thyanta</i>
Pieridae	<i>Pieris</i>
Acrididae	<i>Arphia</i>
Agaonidae	<i>Valisia</i>
Aphididae	<i>Aphis</i>
Araneidae	<i>Arigiope</i>
Caeciliusidae	<i>Valenzuela</i>
Cicadidae	<i>Tibicen</i>
Coreidae	<i>Leptogloss</i>
Cynipidae	<i>Andricus</i>
Delphacidae	<i>Muirodelpha</i>
Diplopoda	<i>Brachyiulus</i>
Entomobryid	<i>Entomobrya</i>
Gryllidae	<i>Allonemobius</i>
Gryllidae	<i>Gryllus</i>
Muscidae	<i>Musca</i>
Noctuidae	<i>Athetis</i>
Noctuidae	<i>Euxoa</i>
Noctuidae	<i>Halysidota</i>
Noctuidae	<i>Noctua</i>
Noctuidae	<i>Sunira</i>
Notodontidae	<i>Dunama</i>
Philodromid	<i>Ponometia</i>
Philosciidae	<i>Burmoniscus</i>
Ptinidae	<i>Stegobium</i>
Salticidae	<i>Phidippus</i>
Tenthredinidae	<i>Dolerus</i>
Tetragnathidae	<i>Leucauge</i>
Theridiidae	<i>Latrodectus</i>
Theridiidae	<i>Parasteatoda</i>
Thomisidae	<i>Xysticus</i>
Tineidae	<i>Tinea</i>

Adapted from Sullins et al. 2018a.

Table 7: WAFWA HEG Scores for Habitat Variable 1 – 4.

Habitat variable 1 – 3				Habitat variable 4	
Score	Vegetative cover	Vegetative composition	Percent cover of tall woody plants > 3 ft. tall	Score	Proportion of area within a 1-mile radius in grass cover with <1% canopy cover of trees
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 8: WAFWA habitat impact buffers

Type of Impact	Buffer distance meters
Oil, gas pads, small compressors	200
Wind turbines	667
Transmission lines	400
Distribution lines	10
Tall vertical structures	667
Secondary roads	67
Primary roads	500
Industrial buildings	667
Residential buildings	133
Private roads (driveways, ranch roads, etc.)	10

Adapted from Van Pelt et al. 2013.

