Retrospective assessment of lesser prairie-chicken habitat in the Sand Sagebrush Prairie Ecoregion

by

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Abstract

Populations of lesser prairie-chickens (*Tympanuchus pallidicinctus*) in the Sand Sagebrush Prairie Ecoregion of southwest Kansas and southeast Colorado, USA, have declined sharply since the mid-1980s. Decreased habitat quality and availability are believed to be the main drivers of declines; however, no broad-scale assessment of habitat change has been conducted for the ecoregion. My objectives were to reconstruct landscape-scale change in the ecoregion since 1985, assess changes in vegetation structure and composition relative to management goals, and compare features of Conservation Reserve Program (CRP) grasslands used and apparently unused by lesser prairie-chickens. I assessed change in landcover types and calculated landscape metrics using Land Change Monitoring, Assessment, and Projection (LCMAP) layers, and documented presence of anthropogenic structures including oil wells and transmission lines. I compared historical and contemporary fine-scale vegetation composition and structure survey data from public lands. I also tested for differences in landscape-scale and field-scale characteristics between CRP with tagged bird locations and those without. Landcover type composition and tree occurrence changed little since 1990 across the Sand Sagebrush Prairie Ecoregion. However, anthropogenic structures (i.e., oil/gas wells, cell towers, wind farms, and transmission lines) increased, potentially causing functional habitat loss as a result of avoidance by lesser prairie-chickens. Quality vegetation structure has declined on Comanche National Grassland since 1985. Used CRP fields were closer to release sites of translocated lesser prairie-chickens than apparently unused CRP, with a greater proportion of used fields associated with ≥60% grassland. Increased anthropogenic structures and decrease in vegetation vertical structure appears to have decreased habitat as well as the quality of existing habitat for

lesser prairie-chickens, likely contributing to recent population declines throughout the Sand Sagebrush Prairie Ecoregion. Tracts of CRP associated with ≥60% grassland within 5km may continue to provide habitat for lesser prairie-chickens, but is a precarious option for habitat conservation in a trend of declining CRP enrollment. If lesser prairie-chickens are still considered a management priority by the U.S. Forest Service, the Cimarron and Comanche National Grasslands will need to adjust management practices to promote habitat conditions that support lesser prairie-chicken populations.

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Chapter 1 - Landscape-Scale Changes in Lesser Prairie-Chicken Habitat within the Sand Sagebrush Prairie Ecoregion

Introduction

The lesser prairie-chicken (*Tympanuchus pallidicinctus*) is the iconic prairie grouse of the southwestern Great Plains (Boal and Haukos 2016). The species inhabits distinctive ecological sites within this semi-arid region, usually represented by sandy soils and mid- to tall grasses or native shrubs (e.g., sand sagebrush [*Artemisia filifolia*] and sand shinnery oak [*Quercus havardii*]) within the larger expanse of short-grass or mixed-grass prairie (Haukos and Zavaleta 2016, Spencer et al. 2017). Currently, lesser prairie-chickens occur in a patchy distribution among four distinct ecoregions (Short-Grass Prairie/CRP Mosaic, Mixed-Grass Prairie, Sand Sagebrush Prairie, and Sand Shinnery Oak Prairie) across five states (Kansas, Colorado, Oklahoma, Texas, and New Mexico; McDonald et al. 2014; Figure 1). The lesser prairie-chicken has persisted through numerous negative environmental and anthropogenic events, including the Dust Bowl of the 1930s (Rodgers et al. 2016, Boal and Haukos 2016) and widespread prairie loss with the advent of center-pivot agriculture in the 1960s (Sexson 1980). However, range-wide populations have declined since the mid-1980s, leading to consideration for listing under the federal Endangered Species Act (U.S. Fish and Wildlife Service 2014). Since 2012, the legal and regulatory status of the lesser prairie-chicken underwent frequent changes including a short-term listing as a threatened species and judicial vacation of the listing rule, followed by a complete removal of any federal status (U.S. Fish and Wildlife Service 2014, U.S. Fish and Wildlife Service 2016a). In November 2016, the U.S. Fish and Wildlife Service determined that once again, the status of the lesser prairie-chicken would be reviewed in response to a petition to list the species as threatened or endangered (U.S. Fish and Wildlife

Service 2016b). In May 2021, the U.S. Fish and Wildlife Service proposed listing the Sand Shinnery Oak Ecoregion populations as endangered, and populations of all other ecoregions as threatened (U.S. Fish and Wildlife Service 2021). Regardless of legal and regulatory status, the issues and environmental conditions presumed to threaten lesser-prairie chickens largely remain. By a recent estimate, the total population abundance of lesser prairie-chickens exists at 26, 591 individuals (Nasman et al. 2021).

For much of the past 50 years, the Sand Sagebrush Prairie Ecoregion of southwestern Kansas and southeastern Colorado supported the greatest density of lesser prairie-chickens among range-wide ecoregions (Haukos et al. 2016). This includes the only public lands currently supporting lesser prairie-chickens; the Cimarron and Comanche National Grasslands (CCNG) in Kansas and Colorado, respectively, managed by the U.S. Forest Service (Elmore and Dahlgren 2016). Garton et al. (2016) estimated >86,000 birds across the Sand Sagebrush Prairie Ecoregion (as defined by Van Pelt et al. 2013) during the 1970s. Population estimates were relatively consistent from the mid-1970s through mid-late 1980s (Hagen et al. 2017). However, since the mid-1980s populations decreased dramatically, and the Sand Sagebrush Prairie Ecoregion currently contains the lowest density and abundance of lesser prairie-chickens of any ecoregion. Historically, lesser prairie-chickens in the ecoregion were subjected to extreme environmental events that greatly reduced the population size (e.g., extensive, prolonged drought; blizzard/ice storms), yet typically showed resilience to such events despite population fluctuations (Haukos et al. 2016). In fact, aerial surveys initiated in 2012 estimated that the Sand Sagebrush Prairie Ecoregion population increased from a low of 513 birds in 2014 to 3,083 in 2018. However, this still represents a >96% decline in population abundance since the 1970s and 69% below the population objective of 10,000 for the ecoregion, and a lack of recovery despite favorable weather conditions from 2014–2019. Furthermore, populations declined further between 2018 and 2022, with only 1,713 estimated in 2022 (90% CI = 209, 3,861; Nasman et al. 2022).

The dominant factor(s) contributing to the long-term decline of lesser prairie-chickens in the Sand Sagebrush Prairie Ecoregion is unclear. Conversion of native sand sagebrush prairie to row-crop agricultural fields has been widespread throughout the ecoregion (Haukos and Boal 2016). However, much of the conversion occurred prior to the 1980s, with only local, relatively minor levels of conversion since the 1980s (Spencer et al. 2017). The U.S. Department of Agriculture Conservation Reserve Program was created in 1985, through which highly erodible crop fields could be converted to perennial grassland under contract periods lasting an average of 10 years (Stubbs 2014). Spencer et al. (2017) found that establishment of CRP essentially offset grassland losses from the 1950s through 2014 in the lesser prairie-chicken range in Kansas. The only significant decrease in grassland during this period was in the Sand Sagebrush Prairie Ecoregion due to localized increasing area of center-pivot irrigation beginning in the early 1980s (Spencer et al. 2017). The US Forest Service Cimarron and Comanche National Grasslands within the Sand Sagebrush Prairie Ecoregion did not change in coarse land cover; however, anecdotal reports suggest that cover and density of woody plants increased, mid- and tall grasses decreased, and vertical structure of vegetation declined during the past three decades. Unfortunately, there has not been any broad-scale quantification of hypothesized changes in potential habitat composition and structure for the National Grasslands or across the Sand Sagebrush Prairie Ecoregion.

Both anthropogenic infrastructure and woody encroachment potentially affected the Sand Sagebrush Prairie Ecoregion populations of lesser prairie-chickens. Lek sites (i.e., areas where males gather and display to attract females and breed) across the lesser prairie-chicken range are typically associated with large areas free of anthropogenic features (Bartuszevige and Daniels 2016). Areas with <10 vertical anthropogenic structures per 12.6 km^2 are important for current populations (Sullins et al. 2019). Nest site selection has been shown to depend on distance to features such as transmission and distribution lines, oil and gas wellheads, and center-pivot irrigation structures (Robel et al. 2004, Pitman et al. 2005, Plumb et al. 2019). Similarly, south-central Kansas nest sites were located in areas with lower tree densities (overall, fewer than 2 trees/ha at the 16-km scale) and further from trees than would be expected at random (Lautenbach et al. 2017). These patterns could be expected in the Sand Sagebrush Prairie Ecoregion as well, where oil and gas infrastructure is widespread and woody vegetation such as cottonwood (*Populus deltoides*), elm (*Ulmus* spp.), and salt cedar (*Tamarix* spp.) have become increasingly common. It is hypothesized that both trees and vertical anthropogenic structures provide perches for aerial predators (Reinert 1984, Manzer and Hannon 2005). Thus, areas with vertical structures are typically avoided, essentially creating spaces of functionally unavailable habitat (Hagen et al. 2011, Lautenbach et al. 2017, Plumb et al. 2019, Sullins et al. 2019).

In addition to functional loss of habitat, changes in spatial arrangement of remaining prairie may have occurred. Lesser prairie-chicken habitat is characterized by large contiguous grassland areas (Hagen et al. 2004, Haukos and Zavaleta 2016). It is estimated that lesser prairie-chickens require landscapes with at least 63% grassland for populations to persist (Crawford and Bolen 1976), though peak probability of use has been observed at approximately 77% grassland (Sullins et al. 2019) and resilience to intensive drought is maximized at 90% grassland (Ross et al. 2016). Independent of the amount of total grassland remaining, loss of larger connected prairie regions could have reduced the amount of habitat on the landscape and as a result, reduced the capacity of the ecoregion to support lesser prairie-chickens over time.

Previous research addressed landcover change since 1950 in the Kansas extent of the lesser prairie-chicken occupied range (Spencer 2014, Spencer et al. 2017), and tree cover change in south-central Kansas (Lautenbach 2015, Lautenbach et al. 2017). To complement those efforts, a longitudinal retrospective analysis of landcover change for the entire Sand Sagebrush Prairie Ecoregion is needed. I measured change in landscape composition and configuration in the ecoregion from 1985-2015, as well as the change in frequency of tree occurrence and the number of vertical anthropogenic structures (i.e., oil and gas wells, wind turbines, transmission lines, and cell towers) present in the ecoregion. I hypothesized that anthropogenic structures increased, trees increased in percent cover and frequency, and native prairie has decreased and cropland increased, all of which would negatively affect population abundance and distribution of lesser prairie-chickens in the ecoregion.

Study Area

I classified landcover types within the Sand Sagebrush Prairie Ecoregion, one of four ecoregions delineated by McDonald et al. (2014) and Van Pelt et al. (2013) making up the estimated occupied range of the lesser prairie-chicken in the southern Great Plains (Boal and Haukos 2016). To define my study area, I used the boundary modified by Western Association of Fish and Wildlife Agencies, which added a 10-mi buffer around the 2013 estimated occupied range (Van Pelt et al. 2013; Figure 1). The Sand Sagebrush Prairie Ecoregion spanned 32,516 km^2 in 21 counties of southwest Kansas, southeast Colorado, and northern Oklahoma panhandle, including both public (U.S. Forest Service Cimarron and Comanche National Grasslands) and private lands. The latter were composed primarily of grazed working grasslands, U.S.

Department of Agriculture Conservation Reserve Program (CRP) grasslands, and row-crop agriculture (Berigan 2019). Grazing intensity in the region was typically moderate to high, with grazing pressure on the Cimarron having nearly doubled since the mid-1950s (Berigan et al. 2022.) Bands of sandy soils such as those running parallel to the Cimarron and Arkansas River drainages, characteristic of the sand sagebrush prairie, supported the eponymous sand sagebrush shrubs as well as mid- to tall grasses including big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), sideoats grama (*Bouteloua curtipendula*), and sand dropseed (*Sporobolus cryptandrus*; Haukos et al. 2016). Short-grass prairie species, supported in clayey and loamy soils, included blue grama (*Bouteloua gracilis*) and buffalograss (*B. dactyloides*; Berigan et al. 2022).

Common forbs in the ecoregion included annual buckwheat (*Eriogonum annuum*), blazing star (*Liatris* spp.), western ragweed (*Ambrosia psilostachya*), prairie sunflower (*Helianthus periolaris*), annual sunflower (*H. annuus*), Russian thistle (*Salsola tragus*), camphorweed *(Heterotheca subaxillaris),* Indian blanket flower (*Gaillardia pulchella*), tansy aster (*Machaeranthera tanacetifolia*), buffalo bur (*Solanum rostratum*), buffalo gourd (*Cucurbita foetidissima*), wax goldenweed (*Grindelia papposa*), prickly lettuce (*Lactuca serriola),* and marestail (*Conzya canadensis*; Berigan et al. 2022).

Newly enrolled CRP fields in Kansas and Colorado were initially seeded with a native grass mixture; seed mixtures did not include forbs until later years (Rodgers 2016). Historically, CRP grasses in Kansas were limited to native warm-season species such as little bluestem, sideoats grama, and switchgrass (*Panicum virgatum*). Other grass species in CRP throughout the ecoregion included big bluestem, western wheatgrass (*Pascopyrum* spp.), blue grama, buffalograss, and Indiangrass (*Sorghastrum nutans*). Forbs in CRP fields included alfalfa

(*Medicago sativa*), sweet clover (*Melilotus* spp.), Maximillian sunflower (*Helianthus maximiliani*), prairie bundleflower (*Desmanthus illinonensis*), purple prairie-clover (*Dalea purpurea*), and upright prairie coneflower (*Ratibida columnifera*; Berigan at al. 2022).

Average annual precipitation for Elkhart, Kansas, in Morton County from 1985-2020 was 43.8 cm, varying from 24.2 cm to 67.6 cm (data from High Plains Regional Climate Center 2021). Drought in this region occurred on a roughly 5-year cycle (Haukos et al. 2016).

Methods

I quantified several landscape-scale factors in the Sand Sagebrush Prairie Ecoregion (SSPE) to determine whether changes in any of those factors could have affected the region's capacity to support lesser-prairie chickens, potentially contributing to the downward population trends observed in recent decades. I quantified the total number of wind turbines, cellular/radio towers, transmission lines, and oil and gas wells in 5-year intervals from 1985 (the approximate point of contemporary peak lesser-prairie-chicken population) through 2015 to determine the change in the frequency of those structures over time.

To measure changes in the dominant vegetation landcover types during this period, I quantified the area of cropland, native prairie (cover class "grass/shrub" with CRP omitted), and CRP in 5-year time steps. Change in the total area for each cover type between each time step reflects conversion among cover types. I estimated change in tree cover over time from landcover layers, and estimated change in tree occurrence by recording presence/absence of trees in aerial photographs. I also estimated change in shrub cover, as this could affect the availability of drought-hardy cover for lesser prairie-chickens in the semi-arid SSPE (Haukos et al. 2016). Finally, I estimated change in the area and number of large prairie patches, as well as proportion of prairie in the ecoregion at the 5-km scale, to identify changes in the ecoregion's potential to support lesser prairie-chickens.

Data Sources

Wind turbines:

Wind turbine locations were available from the US Geological Survey's US Wind Turbine Database for 1985-2019.

Cellular towers:

I obtained Kansas cellular/radio tower locations from Kansas Data Access and Support Center, a dataset based on Federal Communications Commission (FCC) Antenna Structure Registration. For Oklahoma and Colorado, I downloaded tower locations from the Homeland Infrastructure Foundation Level Data (HIFLD) open data site from the Department of Homeland Security as provided through arcgis.com. I used "Location Buildout Deadline", "Location Buildout Notification Date" or "Five Year Buildout Date", in that order of availability as found in the cell tower FCC registry (wireless2.fcc.gov) to identify when towers were built.

Transmission lines:

I obtained data for transmission lines (defined on the HIFLD site as ranging from 69kV-765 kV) for Oklahoma, Kansas, and Colorado from the HIFLD open data site as provided through arcgis.com. I referred to Google Earth imagery and its historical imagery tool to estimate the time step at which each transmission line segment appeared on the landscape. Lines that I estimated as having been "constructed" during the late 1980s through mid-1990s in Oklahoma and Colorado are likely even older, but Google Earth imagery is blurry at such a fine scale prior to 1991. For Kansas lines, I cross-referenced my estimated dates with a shapefile from Sunflower Energy, and as a result was able to categorize many of the older Kansas lines as "pre-1985".

Where structures appeared to have been updated/replaced but the path of the line itself stayed the same, I listed the time period when the original structure appeared. I quantified change in transmission based on the number of new individual pieces added to that interval, which were designated as separate objects in the shapefile (presumably representing an individual construction project). I also calculated the change in total length (km) of transmission lines.

Oil and gas:

I obtained Colorado oil and gas well data from the Colorado Oil and Gas Conservation Commission (COGCC) WELLS shapefile in the COCGCC COGIS Database. I referenced the COGCC WELLS shapefile metadata and the Orphan Well Program page for attribute definitions. From the clipped SSPE set of 2,365 wells, I deleted 165 entries classified as “Abandoned Location” (AL; meaning the operator permitted the project, but no construction took place [Colorado Department of Natural Resources, 2021]), as well as 130 wells marked as being “Dry and Abandoned” before 1985 (DA facility status, with a status date before 1985). I also removed 42 records lacking spud dates because of the lack of start date; nearly all were classified by Location Qualifier “Planned Footage” or “Planned Lat/Long”; the label “Planned” indicating that a well was never drilled. After removing these entries from the set, there remained 2,006 wells in the Colorado part of the ecoregion as of 2014.

I sourced Kansas well locations from a Kansas Geological Survey data set available through the Kansas Data Access and Support Center. From my original SSPE-only set of 27,010 records, I deleted 931 entries with the Well Class “Expired Intent to Drill”, and entries lacking *both* a Spud and Completion date (2,123 records), as at least one was needed for an approximate start date. I also deleted wells plugged prior to 1985 to stay consistent with the deletion decisions I made in the Colorado record. I also omitted wells from 2015 onward from the final output to stay consistent with the temporal range of my landcover analysis. After removing these wells from the set, there remained 20,864 wells in the Kansas component of the SSPE as of 2014.

Oklahoma well records came from the Oklahoma Corporation Commission web site. I was unable to filter out wells Oklahoma wells plugged before 1985 as I had in Colorado and Kansas. Plug dates were not included in the Oklahoma set, and there were inconsistencies in classifiers (e.g., some AC [active] wells also being labeled "dry"), so I was conservative when deciding whether to remove wells from this set. I removed duplicate points, leaving 302 well records for the Oklahoma portion of the SSPE.

I removed "dry and abandoned" wells, those labeled "plugged" prior to 1985, wherever possible, as well as planned/pending wells without an apparent construction date, and "abandoned location" wells. It should be noted that my reported well locations do not necessarily mean a currently producing well. Given lesser prairie-chicken avoidance of vertical structures (Bartuszevige and Daniels 2016, Robel et al. 2004, Pitman et al. 2005, Plumb et al. 2019) my aim was to quantify number of structures on the ground, regardless of producing status. Partially due to identifier inconsistencies among data sets, and in many cases because neither plug dates nor any date indicating the end of an "active/producing" period was listed, I considered all wells as having a more or less permanent structure once established. It is possible that this approach errs on the side of overestimation, as some wells (including Kansas wells with actual plug dates; around 4,700 wells) may not have the associated hardware present. According to the COGCC, "plugged and abandoned" means the well is not only plugged with cement or metal cap at the surface, but "production equipment has been decommissioned and removed" and "location has been remediated and reclaimed". It is unclear whether all plugged wells are fully reclaimed sites, and in some cases well identifiers were incomplete or conflicting. As of 2014, I estimated 23,167 extant wells present in the SSPE.

I classified time steps for all anthropogenic features in intervals beginning with “pre-1985”, and then in 5-year intervals from “1985-1989” (that is, January 1, 1985 through December 31, 1989) through “2010-2014” to assign features to one of the time intervals. I referred to the recorded spud, completion, on-line, or construction dates (dependent on feature type) to determine when features appeared on the landscape. I then summed up the total number of each feature by time step to determine the increase in structures over time.

Landcover on private lands

I selected 3 land cover types of interest: combined non-CRP grassland and shrubland (hereafter, “prairie” or “native prairie”); cropland; and CRP. These were the dominant cover types in the ecoregion. I used these data to identify changes in total area of each cover type as well as changes in configuration of patches of each cover type.

I first identified “Cropland” and “Grass/Shrub” cover types from Primary Land Cover rasters for 5-year intervals from 1985-2015 from the USGS’s Land Change Monitoring, Assessment, and Projection (LCMAP) Collection 1.0. This collection was derived from Landsat surface data spanning 1985-2017, and characterized by a 30-m resolution.

The “Grass/Shrub” category in LCMAP did not differentiate between CRP and non-CRP grassland (prairie). It was necessary to identify CRP as a separate cover type, given its role in lesser prairie-chicken conservation (Sullins et al. 2018), as well as its impermanent status on the landscape. The first plantings of CRP fields occurred in 1986, and took at least 4 years to mature (Rodgers 2016). I assumed that by 1990, CRP plantings had become established enough to be considered a separate cover class. Accurate maps of CRP fields for the SSPE were unavailable prior to 2014, and shapefiles of CRP enrollment were inconsistent among years, states, and counties. I therefore estimated CRP cover across the ecoregion from 1990 through 2015 based on changes in LCMAP cropland and "grass/shrub" cover type between years, and separated the "grass/shrub" into "native prairie" and "CRP" cover classes (Appendix A). For this reason, I refer to change in "prairie" or "native prairie", a separate cover type from CRP grasslands.

Using Raster Calculator in ArcGIS Pro (version 2.7), I identified changes in cell values from cropland to grassland, and vice versa, between consecutive time steps and used those to build a putative CRP cover class. During this process, I also added or removed cells in that cover type in succeeding steps (e.g., remove cells where grass/shrub became cropland, as well as when CRP fields expired and were converted to cropland). I then replaced all "non-CRP" cells in the newly created CRP rasters with the rest of the LCMAP landcover cells; this resulted in a new series of LCMAP maps with CRP included as an additional cover class.

I attempted to reduce some of the isolated pixels inherent to the LCMAP raster classification by applying the Majority Filter tool in ArcGIS Pro (using the 8 cell and majority settings) to first the original raster and then to the output raster of that run, repeating the process until pixels no longer appeared to change in a significant way; in total, 10 runs of the tool. This incorporated isolated pixels of one cover type to match the surrounding cover in a given parcel, thereby reducing the number of stray "patches" of a single pixel in an otherwise homogeneous crop or prairie patch, each of which could inflate the estimated number of patches on the landscape as quantified later in FRAGSTATS.

I separated private lands from public to the extent that data were available, as coarse vegetation cover class (grassland vs. cropland) was not expected to change over time on public lands. To identify public lands, I downloaded a shapefile of Colorado Parks and Wildlife public access properties. I also obtained a shapefile of Bureau of Land Management pasture lands in Colorado, though ultimately only one pasture fell within the ecoregion boundary. I manually digitized a polygon reflecting the area of Kansas Parks and Wildlife, Sand Sage Bison Range (Finney County Game Reserve) south of Garden City, Kansas. I combined all public lands that I identified, including a shapefile of the Cimarron and Comanche National Grasslands. Using the merged public lands shapefile, I created a "private lands only" LCMAP raster to avoid incorporating public lands into landcover change calculations.

Landscape metrics

I used program FRAGSTATS (McGarigal et al. 2012) to generate landcover metrics from LCMAP rasters, with public lands removed, for both composition (cover types present, and amount) and configuration (spatial shape or arrangement) of each dominant landcover type for each time step. I used the 8-cell neighborhood rule as a search radius for adjacent cells in my calculations (Spencer 2014).

Following Spencer et al. (2017), I calculated several landscape composition and configuration metrics for the private lands LCMAP rasters. Contagion (CONTAG) measures connectivity incorporating both the proportional abundance of each patch type on the landscape as well as the number of cell adjacencies for that patch type or another patch type as an index to landscape-scale fragmentation (McGarigal et al. 2012). Percentage of Landscape (PLAND), a composition metric, reports the proportion of the study area composed of each cover type of interest. Total Edge (TE) is a measure of the total length of patch edges for a cover type. Mean Patch Area (AREA_MN) reports the mean patch size in the landscape. Number of Patches is the total number of patches for each cover type. Clumpiness Index (CLUMPY), an aggregation index, incorporates like adjacencies and reflects the degree of clustering among patches for a given cover type (McGarigal et al. 2012).

All landcover metrics for CRP represent change since 1990 rather than 1985, as it would have taken some time for the first CRP plantings to become established. Additionally, though landcover data were available for 1985, I provided only change since 1990 for configuration metrics, which incorporate the spatial arrangement of cover types (McGarigal et al. 2012). Such metrics would be affected by the introduction of the CRP cover type. PLAND was based on cover totals, independent of spatial arrangement (McGarigal et al. 2012); therefore, I included 1985 (without the added CRP cover type) when calculating that metric.

Estimating change in large prairie areas

Estimates for ideal lesser prairie-chicken habitat patch area range from 4,900 ha to >20,000 ha (Haukos and Zavaleta 2016). The U.S. Fish and Wildlife Service (2012) recommends a minimum of 10,118 ha of high-quality habitat. From the patch-level FRAGSTATS output, I identified the largest patches of prairie and grouped them into three categories: 5,000-9,999 ha, 10,000-19,999 ha, and ≥ 20,000 ha.

Percent prairie within 5-km scale

The greatest probability of use by lesser prairie-chickens is estimated to occur at 77% prairie at the 5-km scale (Sullins et al. 2019). I conducted a moving window analysis to determine the proportion of prairie within 5 km at any given point (Appendix A). I used raster data to measure the change in total number of large prairie patches, as these represented patches with the greatest potential for supporting lesser prairie-chickens in the ecoregion. I created a set of binary rasters (1 = grass/shrub; 0 = non-grass/shrub) from the LCMAP cover class rasters and used Focal Statistics in ArcGIS Pro to generate a raster image of 5-km windows. I excluded public lands, as well as windows bordering public lands from the analysis. From these outputs, I then created a series of maps identifying areas as <30%, >30%, and >60% prairie within 5 km (the latter being close to the minimum proportion of grassland needed to sustain lesser prairie-chickens long-term [Crawford and Bolen 1976]). I also calculated the change in cell counts between time steps.

Rangeland Analysis Platform

I downloaded layers for tree and shrub cover from the Rangeland Analysis Platform (RAP) via Google Earth Engine, at 10-year intervals (1985, 1995, 2005, 2015; Jones et al. 2018; https://rangelands.app/). I clipped RAP tree and shrub layers to the ecoregion boundary, then recorded summaries of average pixel values for the rasters at each interval from the average pixel value given in ArcGIS Pro. I followed the same procedure to estimate shrub cover change. My estimate of tree and shrub cover change is based on the summaries in the RAP Analysis Panel feature, which charts the average percent cover for a study area for a given time span. The estimate for a polygon such as the SSPE study area is based on the average of 30 x 30-m pixel values in the polygon (https://rangelands.app/about/rapUserGuide.pdf), with each pixel value representing the estimated percent cover for that particular cell.

Estimating change in tree frequency

I estimated frequency of tree occurrence in the ecoregion to determine whether tree presence has increased. I accessed Digital Orthophoto Quadrangle (DOQ) aerial photographs from 1991-1993 (the earliest dates available for that data set) for “all months” from USGS EarthExplorer. Imagery at the desired resolution was not available for the 1980s. I generated a random subset of 10-ha grid cells across the ecoregion, omitting most incomplete cells along the SSPE border. I reduced the original random subset of 1000 x 100 m grid cells from 250 to 146, as that was the number of cells covered by the image catalog in the starting subset. Images were not available for most of the Colorado portion of the SSPE for the early time steps, aside from the border with Kansas and the northwest extent of the SSPE. For consistency, I limited the grid sample area for each time step to reflect the area available in 1991.

For the early 2000s, I was able to include images from the National Agriculture Imagery Program (NAIP, initiated 2003) to fill in gaps in DOQ imagery availability while still adhering to the same reduced SSPE region from the first timestep. I generated more random grid cells as necessary until the sample size of grid cells was met; generating one round of random cells often included areas for which there was no image coverage available. I used only NAIP imagery for 2010-2011 and 2019 images. No 2020 images were available for the study area.

Results

Anthropogenic structures

Numbers of each type of anthropogenic feature in the Sand Sagebrush Prairie Ecoregion increased since 1989 (Table 1, Figure 3). The greatest increases in structures occurred mainly during the 1990s through mid-2000s.

No wind farms existed in the SSPE until 2003. Wind turbines increased by 166 structures (148.2%) from 2003 to 2014 (Table 1, Figure 2). There were 278 turbines in the ecoregion as of 2014, with 40.3% in Colorado and 59.7% in Kansas. By the end of 2014, approximately 38.9% of turbines built since 1989 were located on grassland (either CRP or native prairie). The largest wind farm projects prior to 2015 were constructed in 2003 and 2013.

Cell towers increased by 131 (291.11%) from 1989 to 2014 (Table 1, Figure 2). There were 176 cell towers as of 2014. The greatest increase in cell towers occurred in the early- to mid-2000s. An estimated 34.7% of towers built since 1989 were located in grassland.

The total length of transmission lines increased by 1,447.79 km (127.6%) from 1989 to 2014 (Table 1, Figure 2). Most transmission line segments in the ecoregion were constructed during or prior to the early 2000s, with this period including the majority of larger segments constructed. At least 28.10% of transmission lines cross through, or are immediately adjacent to, grassland cover.

Oil and gas wells increased by 10,840 (87.9%) from 1989 to 2014 (Table 1, Figure 2). By a liberal estimate, there may be as many as 23,826 extant oil wells (not all of which are active) currently in the ecoregion. The actual number of pump jacks on the ground may be lower, as plug dates (when wells were plugged with cement) were absent from part of the Kansas data and all of the Oklahoma data. Therefore, all wells and associated hardware were assumed to be present on the landscape through 2014, with the exception of wells recorded as "plugged" prior to 1985. The greatest increase in oil and gas wells occurred in the 1990s. As of 2015, around 31.1% of wells built since 1989 were located in grassland.

Landcover

Landcover totals changed little from 1990-2015, with the exception of introducing an additional cover type (CRP) by 1990 (Figure 4). Area and arrangement of prairie patches changed over time. Total prairie on private lands decreased by 11.7%, CRP increased by 134.6%, and cropland decreased by 7.3% since 1985 (Figure 4). Although total cropland decreased since 1990, this was concurrent with CRP increasing (which by definition is the product of converting croplands to grass) as enrollment became more widespread in the SSPE after the mid-1980s. If prairie and CRP were combined, total grassland increased by 1.1% from 1990-2015.

Percentage of Landscape consisting of prairie decreased by 4.3 percentage points from 1985 through 2015, while percent consisting of cropland decreased by 4.6 percentage points (Figure 5). From 1990 to 2015, the percent of landscape consisting of CRP increased by 5.1 units.

From 1990 (by which time CRP fields had become established on the landscape) to 2015, Mean Patch Area on private lands decreased by 26.4% for prairie and 23.8% for cropland, but increased by 79% for CRP (Tables 2-4). Since 1990, Total Edge increased by 6.6% for prairie and 12.2% for cropland, while TE of CRP increased by 62.8% since 1990 (Tables 2-4). Number of Patches in the ecoregion increased by 18.7% for prairie, 30.7% for cropland, and 30.7% for CRP since 1990 (Tables 2-4). Clumpiness decreased by only 0.7% and 0.6% for prairie and cropland, respectively, since 1990 while CRP CLUMPY increased by 6.5% (Tables 2-4). Contagion decreased by only about 3.4% from 1990-2015 (Table 5).

The number of largest prairie patches, as well as the amount of SSPE covered by prairie at the 5-km scale, changed over time. All three size classes of large prairie patches (from 5,000 ha through ≥ 20,000 ha) fluctuated from 1990-2015, but there was a 57% decrease in patches greater than or equal to 20,000 ha on private lands in the SSPE from 1990-2015 (Figure 6). In 1990, approximately 25% of the ecoregion consisted of ">60%" prairie at the 5-km scale, and as of 2015 this decreased to 20% of the ecoregion; most of the area occurring in Colorado. There was an 11.5% decrease in ">30% prairie" cells from 1990-2015 in prairie on private lands (Figure 7). Landscapes having ">60% prairie" decreased by 18.4% from 1990-2015. The time step with the greatest loss in ">30% prairie" and ">60% prairie" was from 2010 through 2015 (6.5% decrease and 9.9% decrease, respectively, from the 1990 cell counts; Table 6).

I estimated that tree cover increased from 1.4% to 2.2% (± mean absolute error of 4.7%) in the ecoregion based on the RAP data results (Table 7). Mean percent shrub cover decreased by 22.3% (± mean absolute error of 6.9%) on private land, but only by 3.9% on public lands (Table 8).

Tree frequency

My analysis showed little increase in tree occurrence since 1990 (Table 9). For images from 1991-1993, only 16 of 146 grid cells contained trees (11.0%). For 2000-2003, there were 14 cells with trees (9.6%). For 2010, 15 cells had trees (10.3%). Of the 2019/2020 cells, 17 had trees (11.6%). Overall, the estimated tree frequency increased in the SSPE by 0.6 percentage points.

Discussion

Establishment of CRP minimized the loss of total grassland cover (prairie and CRP combined) in the ecoregion over time, increasing this combined cover type slightly from 1990 through the start of 2015. My finding of little decrease in total grassland cover in the SSPE is consistent with findings from Spencer et al. (2017) that CRP offset grassland losses in the lesser prairie-chicken current range in Kansas. However, although the total area of prairie in the SSPE declined only slightly since 1985, losses occurred in prairie areas with the greatest potential for supporting lesser prairie-chicken populations. Additionally, the number of vertical anthropogenic structures in the ecoregion significantly increased, which likely reduced the amount of remaining habitat available to lesser prairie-chickens over time (Hagen et al 2011, Plumb et al. 2019, Sullins et al. 2019).

A slight decrease in total prairie cover and percent of landscape was consistent with the trend of prairie conversion to cropland in the ecoregion. It is unlikely that change in landcover type alone was a primary contributor to population declines; however, these changes (e.g., increase in CRP) may have contributed to shaping distribution of lesser prairie-chickens in the SSPE (Sullins et al. 2019, Berigan et al. 2022).

A relative increase in total edge for all cover types across time steps would suggest a shift to a more fragmented landscape. However, there was only a slight decrease in Contagion of prairie, with the patch arrangement for prairie remaining fairly clumped as of 2015, indicating a lack of increasing fragmentation over time. Each of the CLUMPY results for the three cover types was close to approaching a value of 1 in 2015 (0.9454, 0.8547, and 0.9476 for prairie, CRP, and cropland, respectively) indicating cells in each cover type were closer to being maximally clustered; a CLUMPY value of exactly 1 would mean the landscape belongs to a single patch (McGarigal et al. 2012). The values being closer to 1, as opposed to a random arrangement with value approaching 0, indicates patches of similar types were connected in arrangement on the landscape (McGarigal et al. 2012). Lesser prairie-chickens need contiguous grassland area for continued persistence (Haukos et al. 2016); if remaining prairie patches are in fact fairly clustered, this would be a positive for restoration potential.

The increase in vertical anthropogenic structures could affect the availability and, potentially, connectivity of habitat for lesser prairie-chickens. Given what is known about avoidance of anthropogenic structures by lesser prairie-chickens during nesting (Robel et al. 2004, Pitman et al. 2005), in home range placement (Plumb et al. 2019), and lekking (Bartuszevige and Daniels 2016), increased presence of vertical structures may have essentially created zones of “non-habitat” or inaccessible habitat in the ecoregion, even in cover types otherwise conducive to lesser prairie-chickens such as native prairie (Hagen et al. 2011, Plumb et al. 2019, Sullins et al. 2019). In fact, Pruett et al. (2009) found that lesser prairie-chickens crossed transmission lines less frequently than would be expected at random, suggesting vertical structures may also contribute to population isolation by limiting dispersal**.** Additionally, recent research found that utility pole density has a negative effect on lesser prairie-chicken survival in New Mexico (Lawrence et al. 2021). My results show that parts of the ecoregion once free of major transmission lines and other features during population peak are now populated by new structures, each associated with its own avoidance zone (Robel et al. 2004). Increased structure density could likely reduce available habitat, connectivity among life-stage specific habitat, limited dispersal potential among prairie patches, and ultimately affecting gene flow among local populations (Boal and Haukos 2016). Sullins et al. (2019) estimated that only 9% of the SSPE remained as potential habitat for lesser prairie-chickens, based on anthropogenic structures and cover composition. Continued development of oil and gas wells and other features would further decrease the amount of available habitat throughout the SSPE, effectively rendering the remaining prairie unable to support lesser prairie-chickens.

The implications of functional habitat loss are even more concerning given the continued physical loss of grassland dominated landscapes relevant for lesser prairie-chickens. My results suggest that although total grassland loss is fairly minimal, the loss of ≥20,000 ha prairie patches and isolation of areas with >60% prairie was extensive in my study area. Grassland-dominated landscapes are recommended for promoting stable lesser prairie-chicken populations (Hagen et al. 2011, Van Pelt et al. 2013, Haukos et al. 2016). Smaller patches of prairie, or even isolated CRP fields not bounded by native prairie, are less likely to provide the heterogeneity of cover needed for each phase of the lesser prairie-chicken’s life history including nesting and brooding, as well as sheltering during extreme weather (Sullins et al. 2018, Kraft et al. 2021). My findings on large habitat patches alone, even without considering the effects of anthropogenic structure presence, indicate that the SSPE has lost significant capacity to support population persistence since 1985. In fact, only an estimated 9% of the SSPE remains as potential habitat for lesser prairie-chickens (Sullins et al. 2019). A drop in lesser prairie-chicken numbers such as that observed in the SSPE could be expected to result from such fragmentation of grassland and further loss of habitat within grasslands where anthropogenic features increased.

Mean tree cover likely increased from 1985 – 2015, but low densities of trees in the study area were not readily detected and impeded estimation of spatially explicit trends. Increased tree cover can reduce available habitat for lesser prairie-chickens in native prairie (Lautenbach et al. 2017), additive to the effect of anthropogenic features. However, for this region of the Great Plains, predictive layers from RAP mistakenly identified some land cover (primarily green center-pivot fields) as “forested” and failed to identify known wooded areas as having trees (i.e., the cottonwood galleries along the Cimarron and Arkansas River riparian zones). Tree cover estimates for this sparsely-forested area of the country may not so much reflect a change in actual tree coverage, but instead an increase in center-pivot croplands. It is also possible that a drastic change in apparent tree cover between 2005-2015 (seen as a visibly lighter grey appearance across the whole study area image) may be a result of satellite or other technical anomalies in images used by RAP. The overall appearance of these images change with an abruptness not attributable to sudden, widespread change in vegetation growth in this semi-arid region. A similar visual pattern was present in the 1990s shrub cover data, with what appears to be a darker region and a clear line of demarcation as if following a satellite path. In fact, RAP’s web site explains that from 1984 through the 1990s, inconsistencies in data were more common due to only one Landsat satellite in use during that period (https://rangelands.app/). Both my tree and shrub cover results, therefore, should be interpreted with some caution.

I found that the landscape of the SSPE has in fact changed since the mid- to late-1990s, most notably due to the increased number of anthropogenic structures as well as a decrease in the number of prairie-dominated areas. My objective in this broad-scale assessment was to determine the extent of landscape-scale changes in lesser prairie-chicken habitat in the Sand Sagebrush Prairie Ecoregion since the contemporary population peak, specifically regarding cover type and configuration of patches, as well as a quantification of anthropogenic structures. This assessment was an essential initial step to identify the extent of broad-scale changes that may have reduced the capacity for supporting lesser prairie-chickens in the SSPE. Substantial habitat change likely occurred in terms of both loss and unavailability of needed grassland. Prairie was lost and fragmented by conversion to cropland, and remaining prairie was made unavailable increasing anthropogenic features. Both elements would alter the capacity of these landscapes to support lesser prairie-chicken populations, potentially contributing to precipitous declines. Although the change in proportion of ">60%" prairie appears trivial, it is possible that a threshold effect is at play in the SSPE, with a baseline level of remaining supportive habitat having been incrementally lost in the time since population peak, eventually resulting in a pronounced population response (Fahrig 2001, With and King 1999). Given the effect of transmission lines acting as movement barriers (Pruett et al. 2009), even a few additional structures could further limit dispersal ability among remaining lek sites by both sexes and access to nesting and brood habitats by females.

My findings show a loss of prairie area with the patch size and arrangement needed to promote lesser prairie-chicken populations. This, compounded with inaccessibility of habitat due to new and widespread anthropogenic structures, reveals the declining potential for private lands in the Sand Sagebrush Prairie Ecoregion to support a lesser prairie-chicken population over the last few decades. The availability of CRP shows promise to increase reproductive habitat and population persistence for lesser prairie-chickens (Sullins et al. 2018). In fact, establishment of native grass plantings in Kansas CRP may have promoted lesser prairie-chicken expansion and population increase north of the Arkansas River, to a greater extent than has been observed historically (Chanell 2010, Dahlgren et al. 2016). Extensive use of CRP has also been recently documented in the SSPE use particularly in eastern Colorado (Haukos et al. 2016). Lesser prairie-chickens in the ecoregion have shown strong selection for CRP over native prairie for nesting habitat (Berigan et al. 2022). However, given the uncertain future of CRP contracts as a long-term source of grassland cover, maintaining existing areas of connected prairie with low anthropogenic structure densities will continue to be essential for avoiding extirpation of lesser prairie-chickens in the Sand Sagebrush Prairie Ecoregion (Hagen et al. 2017).

Pronounced losses of habitat due to anthropogenic structures, together with the loss of large, connected prairie at relevant scales, indicate a reduced capacity of the SSPE to support lesser prairie-chickens over time. Landscape-scale changes may be in part responsible for declines observed in recent decades; however, it is unlikely they are the only contributing factor. Therefore, it is necessary to explore changes in lesser prairie-chicken habitat at finer scales. Grassland vegetation composition and structure on public land managed for lesser prairie-chickens, particularly the U.S. Forest Service Cimarron and Comanche National Grasslands, should reflect characteristics indicative of quality lesser prairie-chicken habitat. An assessment of changes in these finer-scale attributes will help to determine whether additional factors are at play in driving lesser prairie-chicken population trends in the SSPE.

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Table 1.1.Total number or length of vertical anthropogenic structures present in the lesser prairie-chicken Sand Sagebrush Prairie Ecoregion (640, 763 ha focal area plus 16 km buffer [Van Pelt et al. 2013]) of southwest Kansas, southeast Colorado, and northwestern Oklahoma, USA, at 5-year intervals from 1989-2014.

Year	Oil/Gas Wells	Cell Towers	Wind Turbines	Transmission Lines (km)
1989	12,327	45	0	1134.27
1994	15,427	58	0	1552.05
1999	18,301	76	0	1819.00
2004	20,288	109	112	2316.04
2009	21,930	145	112	2443.46
2014	23,167	176	278	2582.06

Table 1.2. Class-level FRAGSTATS metrics (McGarigal et al. 2012) for prairie cover on private lands in the lesser prairie-chicken Sand Sagebrush Prairie Ecoregion (640, 763 ha focal area plus 16 km buffer [Van Pelt et al. 2013]) of southwestern Kansas, southeastern Colorado, and northwestern Oklahoma, USA, from 1985 (pre-CRP establishment) through 2015.

Year	Total Area [1]	Mean Patch Area [2]	Percentage of Landscape [3]	Total Edge [4]	Number of Patches [5]	Clumpiness [6,7]
1985	1137390.21	NA	36.46	NA	NA	NA
1990	1150138.35	35.89	36.87	45300.63	32045	0.9517
1995	1140711.66	35.69	36.57	44545.20	31965	0.9524
2000	1117750.41	34.91	35.83	43390.62	32022	0.9532
2005	1100248.11	32.34	35.27	44068.02	34019	0.9522
2010	1077971.94	29.34	34.56	45385.02	36744	0.9503
2015	1004313.42	26.40	32.20	48291.99	38037	0.9454

1. Total area, in hectares, of all the patches combined for a given cover class in the landscape.
2. Average size, in ha, of patches within a given cover class in the landscape.
3. Proportion of the landscape consisting of a given cover class
4. Total length of edge, in km, of all patches in a given cover class in the landscape
5. Number of patches (groups of connected cells of the same cover class) for a given cover class on the landscape
6. Degree of aggregation of cells in a given cover class in the landscape
7. For consistency in comparisons following Conservation Reserve Program establishment in 1986, Mean Patch Area, Number of Patches, and Clumpiness were excluded from 1985 time step.

Table 1.3. Class-level FRAGSTATS metrics (McGarigal et al. 2012) for cropland cover on private lands in the Sand Sagebrush Prairie Ecoregion (640, 763 ha focal area plus 16 km buffer [Van Pelt et al. 2013]) of southwestern Kansas, southeastern Colorado, and northwestern Oklahoma, USA, from 1985 (pre-CRP establishment) through 2015.

Year	Total Area [1]	Mean Patch Area[2]	Percentage of Landscape[3]	Total Edge [4]	Number of Patches[5]	Clumpiness[6,7]
1985	1938710.52	NA	62.15	NA	NA	NA
1990	1803324.51	68.56	57.81	46448.13	26301	0.9531
1995	1776380.22	74.90	56.95	43720.23	23717	0.9561
2000	1767507.93	77.37	56.66	41892.81	22844	0.9580
2005	1713267.9	67.72	54.92	43025.94	25299	0.9573
2010	1712343.87	56.75	54.89	46399.77	30171	0.9540
2015	1796846.04	52.28	57.60	52092.48	34367	0.9476
2015	1796846.04	52.28	57.60	52092.48	34367	0.9476

1. Total area, in hectares, of all the patches combined for a given cover class in the landscape.
2. Average size, in ha, of patches within a given cover class in the landscape.
3. Proportion of the landscape consisting of a given cover class
4. Total length of edge, in km, of all patches in a given cover class in the landscape
5. Number of patches (groups of connected cells of the same cover class) for a given cover class on the landscape
6. Degree of aggregation of cells in a given cover class in the landscape
7. For consistency in comparisons following Conservation Reserve Program establishment in 1986, Mean Patch Area, Number of Patches, and Clumpiness were excluded from 1985 time step.

Table 1.4. Class-level FRAGSTATS metrics (McGarigal et al. 2012) for Conservation Reserve Program cover class in the Sand Sagebrush Prairie Ecoregion (640, 763 ha focal area plus 16 km buffer [Van Pelt et al. 2013]) of southwestern Kansas, southeastern Colorado, and northwestern Oklahoma, USA, from 1990 through 2015. Metrics for CRP prior to 1990 were not available, as CRP enrollment began in 1986 and fields were not immediately established.

Year	Total Area [1]	Mean Patch Area[2]	Percentage of Landscape[3]	Total Edge [4]	Number of Patches[5]	Clumpiness[6]
1990	118588.59	2.68	3.80	29999.40	44283	0.8023
1995	154098.27	3.60	4.94	32320.20	42852	0.8341
2000	185531.94	4.42	5.95	34597.17	41967	0.8508
2005	260312.76	5.46	8.34	43571.58	47637	0.8624
2010	284180.22	5.31	9.11	49183.47	53560	0.8566
2015	278266.32	4.81	8.92	48853.47	57875	0.8547

1. Total area, in hectares, of all the patches combined for a given cover class in the landscape.
2. Average size, in ha, of patches within a given cover class in the landscape.
3. Proportion of the landscape consisting of a given cover class
4. Total length of edge, in km, of all patches in a given cover class in the landscape
5. Number of patches (groups of connected cells of the same cover class) for a given cover class on the landscape
6. Degree of aggregation of cells in a given cover class in the landscape

Table 1.5. Landscape-scale contagion results for the Sand Sagebrush Prairie Ecoregion (640, 764 ha focal area plus 16 km buffer [Van Pelt et al. 2013]) of southwestern Kansas, southeastern Colorado, and northwestern Oklahoma, USA, from 1990-2015. Contagion reflects both dispersion and interspersion of all patch types in the landscape (McGarigal et al. 2012) and is used here as a reflection of landscape-scale fragmentation.

Year	**Contagion**
1990	74.7506
1995	74.0602
2000	73.555
2005	72.3159
2010	71.8579
2015	72.2412

Table 1.6. Number of cells from Focal Statistics (moving window analysis) output rasters representing windows with greater than 30% prairie and greater than 60% prairie cover in the Sand Sagebrush Prairie Ecoregion of southwestern Kansas, southeastern Colorado, and northwestern Oklahoma, USA

	Proportion Prairie Cover	
Year	**>0.30**	**>0.60**
1990	10404722	5653832
1995	10334095	5603681
2000	10175254	5475878
2005	10041991	5312348
2010	9885602	5186681
2015	9206383	4625561

Table 1.7. Average percent tree cover in the Sand Sagebrush Prairie Ecoregion (640, 763 ha focal area plus 16 km buffer [Van Pelt et al. 2013]) of southwest Kansas, southeast Colorado, and northwest Oklahoma, USA, based on the Rangeland Analysis Platform landcover rasters. Average cover represents the mean pixel value of the raster for that year.

Year	Average Percent Cover (± SD)
1985	1.36 (2.78)
1995	1.14 (2.62)
2005	1.35 (3.72)
2015	2.22 (4.36)

Table 1.8. Average percent shrub cover in the Sand Sagebrush Prairie Ecoregion (SSPE) of southwest Kansas, southeast Colorado, and northwest Oklahoma, USA, based on the Rangeland Analysis Platform landcover rasters. Average cover represents the mean pixel value of cells in the shrub cover raster for that year.

	Average Percent Shrub Cover (± SD)		
Year	**All SSPE**	**Public Only**	**Private Only**
1985	6.02 (4.13)	7.38 (4.91)	5.96 (4.09)
1995	4.51 (3.42)	6.23 (3.27)	4.44 (3.41)
2005	4.46 (4.00)	6.64 (4.55)	4.37 (3.95)
2015	4.73 (4.11)	7.09 (4.46)	4.63 (4.06)

Table 1.9. Estimated frequency of occurrence of trees in the Sand Sagebrush Prairie Ecoregion of southwest Kansas, southeast Colorado, and northwest Oklahoma, USA. Grid cells represented a 10-ha area (1000 m x 100 m).

Time Period	**Percent of Grid Cells with Trees**
Early 1990s	11.0%
Early 2000s	9.6%
2010s	10.3%
2019	11.6%

Figure 1.1.Presumed historical and current distribution, as well as current ecoregion boundaries, for lesser prairie-chickens (LPC) in the Southern Great Plains, USA. Figure adapted from Boal and Haukos (2016).

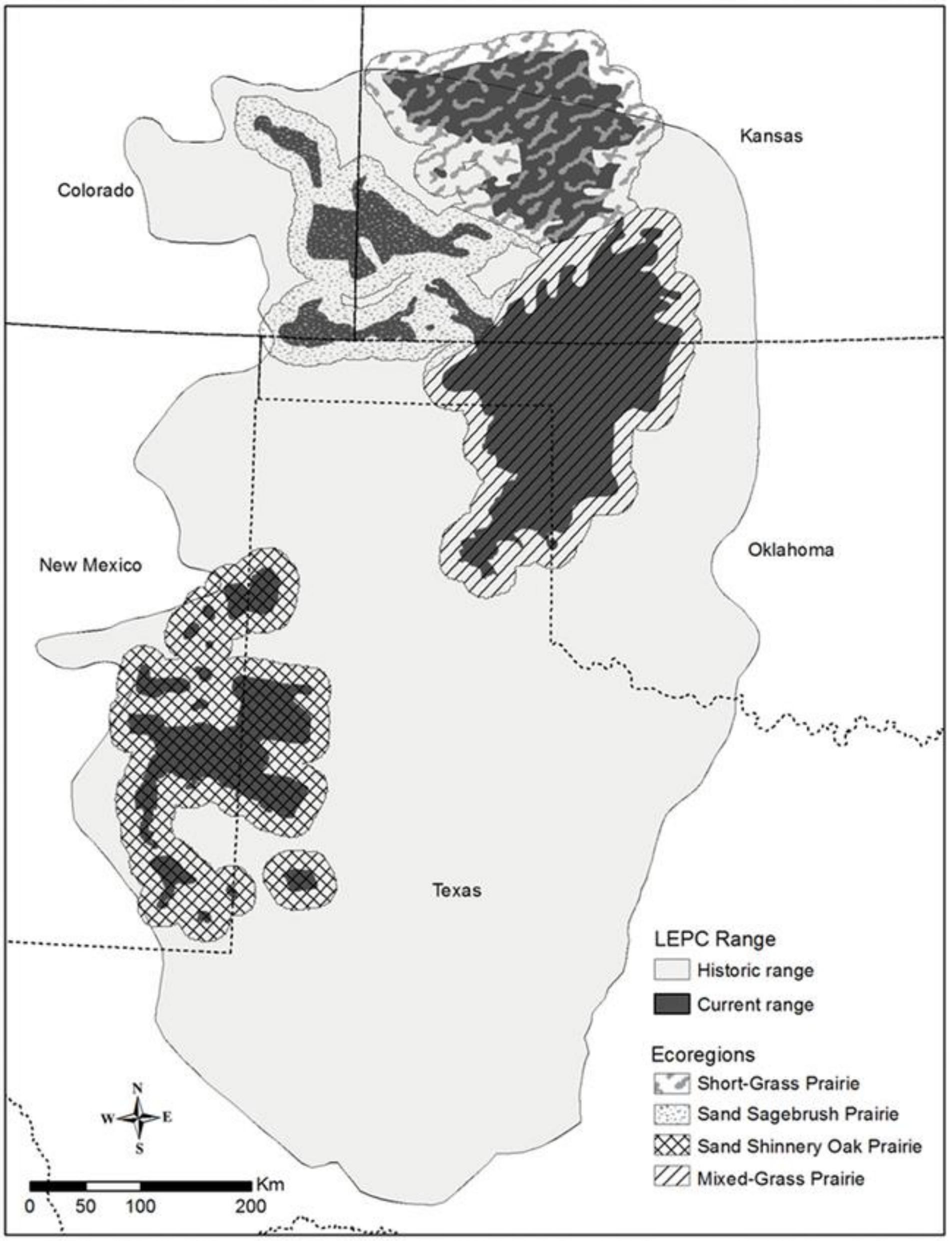


Figure 1.2. The Sand Sagebrush Prairie Ecoregion (SSPE) within the estimated occupied range of the lesser prairie-chicken, including U.S. Forest Service Cimarron (Kansas) and Comanche (Colorado) -National Grasslands, USA. Shown are both the SSPE area as delineated by McDonald et al. (2014) and the boundary updated with a 10-mile buffer by the Western Association of Fish and Wildlife Agencies (WAFWA; Van Pelt et al. 2013).

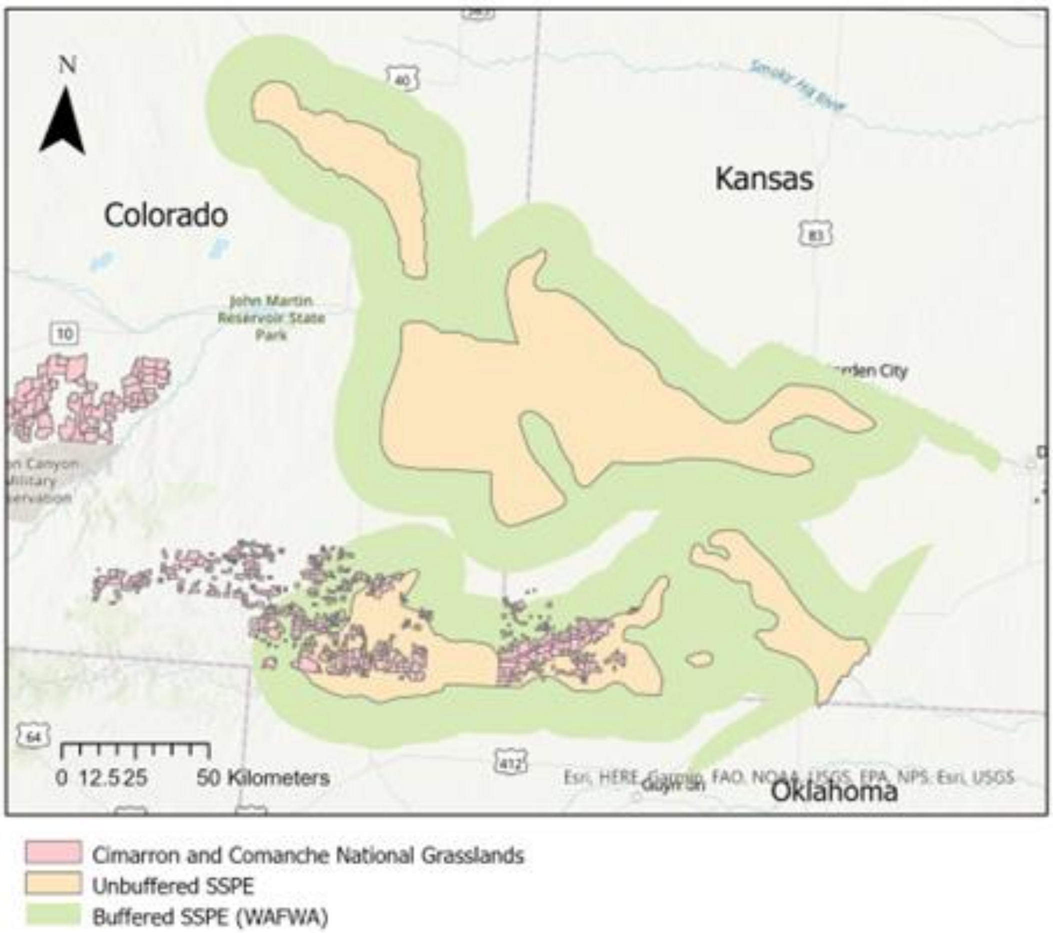


Figure 1.3. Number of anthropogenic structures (transmission lines, cell/radio towers, wind turbines, and oil/gas wells) present in the Sand Sagebrush Prairie Ecoregion of Kansas, Colorado, and Oklahoma, USA, in 1989 (approximate contemporary lesser prairie-chicken population peak) through the end of 2014.

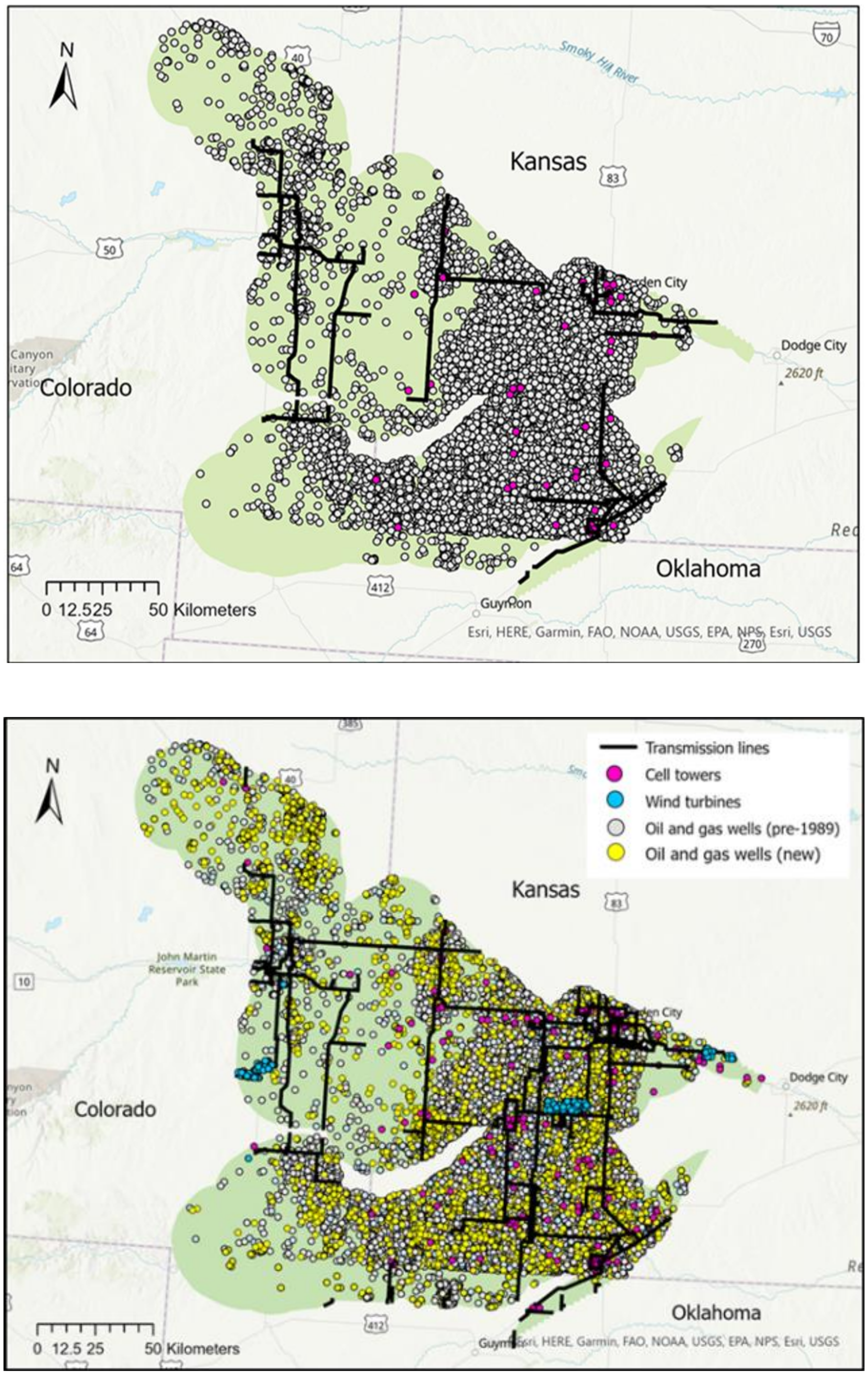


Figure 1.4. Total area of each cover type (prairie, cropland, and Conservation Reserve Program [CRP] grassland) in the Sand Sagebrush Prairie Ecoregion of southwestern Kansas, southeastern Colorado, and northwestern Oklahoma, USA, in 5-year intervals from 1985-2015. The Conservation Reserve Program was initiated in 1986, and therefore excluded from my analysis as an established cover type before 1990.

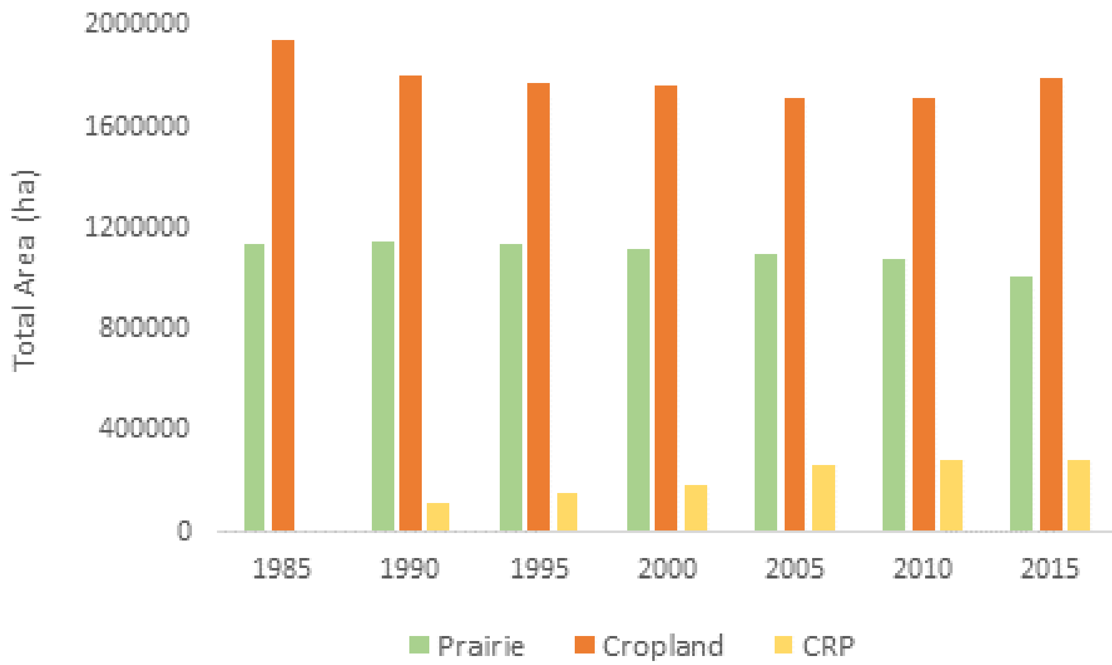


Figure 1.5. Proportions of the three dominant cover types (cropland, Conservation Reserve Program [CRP] grassland, and prairie) in the Sand Sagebrush Prairie Ecoregion of southwestern Kansas, southeastern Colorado, and northwestern Oklahoma, USA, from 1985-2015. The Conservation Reserve Program was initiated in 1986, and therefore I excluded CRP from my analysis as an established cover type before 1990.

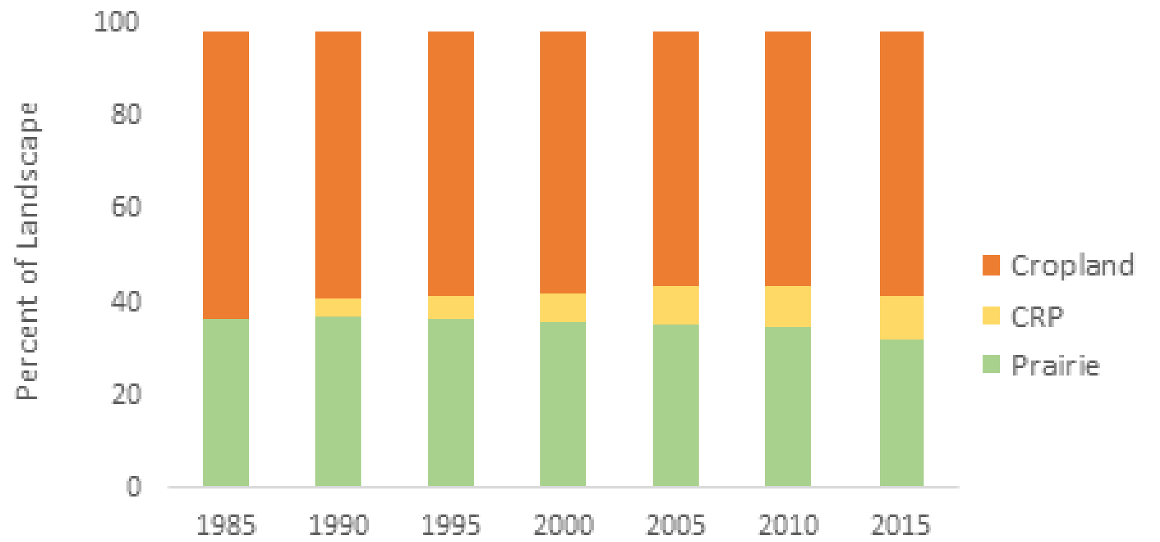


Figure 1.6. The number of large prairie patches (ranging from 5,000 to ≥20,000 ha) conducive to supporting lesser prairie-chicken populations in the Sand Sagebrush Prairie Ecoregion of southwestern Kansas, southeastern Colorado, and northwestern Oklahoma, USA, from 1990-2015.

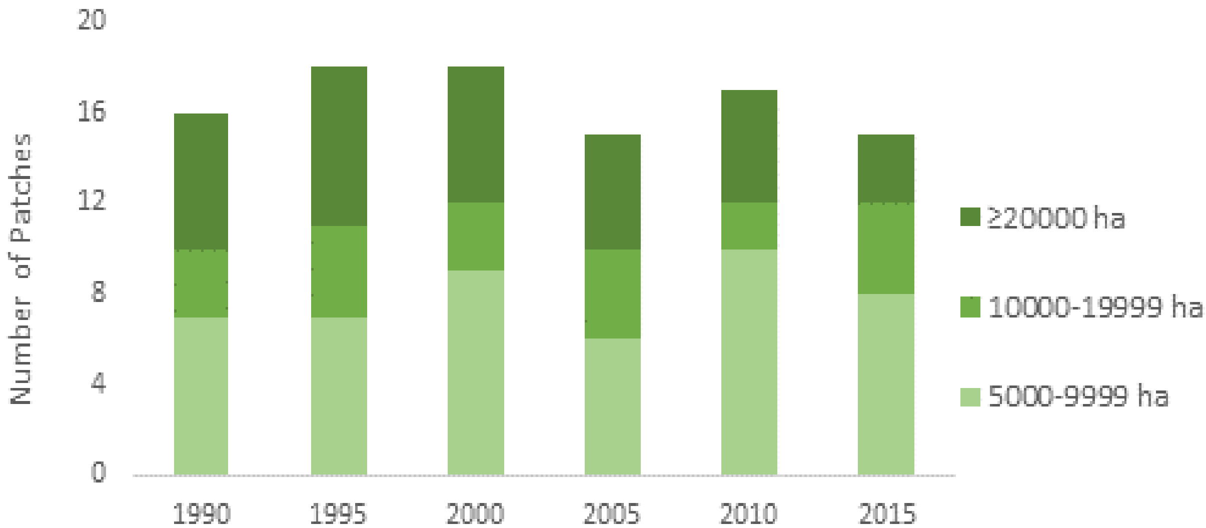
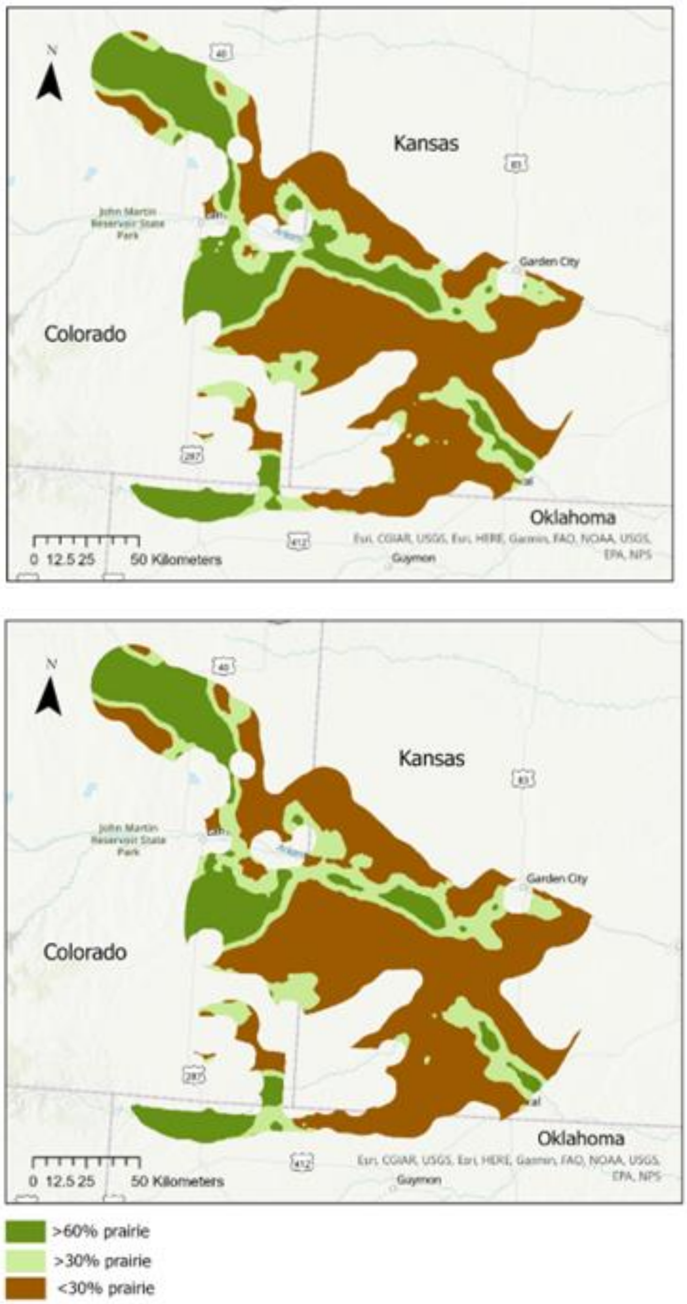


Figure 1.7. Comparison of private lands areas in the Sand Sagebrush Prairie Ecoregion in Kansas, Colorado, and Oklahoma, USA, composed of <30%, >30% and >60% prairie within 5 km, respectively, during 1990 (top) and by 2015 (bottom). Areas with >60% prairie represent the recommended minimum amount of prairie for supporting lesser prairie-chicken populations (Crawford and Bolen 1976).



Chapter 2 - Vegetation Composition and Structure Transitions on the Cimarron and Comanche National Grasslands

Introduction

The lesser prairie-chicken (*Tympanuchus pallidicinctus*) is the iconic prairie grouse of the southwestern Great Plains (Boal and Haukos 2016). The species inhabits distinctive ecological sites within this semi-arid region, usually represented by sandy soils and mid- to tall grasses or native shrubs (e.g., sand sagebrush [*Artemisia filifolia*] and sand shinnery oak [*Quercus havardii*]) within the larger landscapes of short-grass or mid-grass prairie (Haukos and Zavaleta 2016, Spencer et al. 2017). Currently, lesser prairie-chickens occur in a patchy distribution among four distinct ecoregions (Short-Grass Prairie/CRP Mosaic, Mixed-Grass Prairie, Sand Sagebrush Prairie, and Sand Shinnery Oak Prairie) across five states (Kansas, Colorado, Oklahoma, Texas, and New Mexico, USA; McDonald et al. 2014; Figure 1.1).

For much of the past 50 years, the Sand Sagebrush Prairie Ecoregion of southwestern Kansas and southeastern Colorado supported the greatest density of lesser prairie-chickens among the range-wide ecoregions (Haukos et al. 2016). Garton et al. (2016) estimated >86,000 birds in the ecoregion during the 1970s; population estimates were relatively consistent from the mid-1970s through mid-late 1980s. However, populations have dramatically decreased since then, and the Sand Sagebrush Prairie Ecoregion currently has the lowest density and total abundance of lesser prairie-chickens of any ecoregion. The estimated population of 3,083 in 2018 represented a >96% decline since the 1970s and 69% below the population objective of 10,000 for the ecoregion. Furthermore, populations declined further between 2018 and 2022, with only 1,713 estimated in 2022 (90% CI = 209, 3,861; Nasman et al. 2022).

The Sand Sagebrush Prairie Ecoregion is unique in the lesser prairie-chicken range because of the significant area of public lands in the form of National Grasslands administered by the U.S. Forest Service (Elmore and Dahlgren 2016). The Cimarron (Morton and Stevens counties, Kansas; nearly 44,000 ha) and Comanche (Baca, Otero, and Las Animas counties, Colorado; nearly 180,000 ha) National Grasslands comprise approximately 224,000 ha in the ecoregion and represents the bulk of public land in the lesser prairie-chicken range (Elmore and Dahlgren 2016). Once "submarginal lands" deemed unprofitable for farming, the grasslands re-emerged from efforts to conserve soil following the catastrophic Dust Bowl (Fagin et al. 2012?). Since their establishment, the Cimarron and Comanche National Grasslands have been managed under a "multiple-use" framework including cattle grazing and oil and gas production (Fagin et al. 2012). While the U.S. Forest Service has historically managed surface lands, subsurface mineral rights remained tied to previous landowners until the late 1980s, after which time the agency sold leases for oil and gas exploration (Woutat 1987). The National Grasslands are also managed for wildlife, including threatened and endangered grassland species (Fagin et al. 2012).

Following the proposed listing of the lesser prairie-chicken under the 1973 Endangered Species Act in 2014, the U.S. Forest Service developed a management plan for the National Grasslands that had goals for vegetation composition and structure to benefit lesser prairie-chicken populations, including baselines for visual obstruction and percent vegetation cover (USFS 2014). Strategic use of grazing, including deferred grazing following drought, is also recommended to promote desired vegetation structure for breeding lesser prairie-chickens (USFS 2014). The plan also cited the Forest Service Manual 2600 which recommended removal or increased visibility of fences near areas used by prairie grouse, and for limiting development of vertical structures within 3 km of grouse habitat (USFS 2014, Teige et al. 2022). Prescribed fire guidelines included promoting “mosaic burn” patterns to promote heterogeneity and discourage buildup of woody material (USFS 2014).

Although large areas of the National Grasslands are designated as managed for lesser prairie-chicken habitat (USFS 2014), an intensive drought during 2011-2013 reduced the number of extant birds on the National Grasslands to <10 by fall 2016, despite above-average precipitation during 2015 and 2016 (Berigan et al. 2022). Of additional concern is the documentation that between 95%-100% of the 411 lesser prairie-chickens translocated to the National Grasslands in 2018 and 2019 dispersed >5 km from their initial release sites within days of release (Berigan et al. 2022). Approximately 69% of birds eventually settled ≥5 km from the site (Berigan 2019). By the end of the study in February 2021, only 23% of the 148,166 GPS bird locations and nests occurred on the National Grasslands (L. Berigan, personal communication; Aulicky 2020). Nests from translocated birds were primarily located in CRP (Berigan 2019). It is necessary to identify changes in habitat availability, particularly during the breeding season, which may explain the lack of settling at translocation sites as well as observed population trends of translocated lesser prairie-chickens on the National Grasslands (Aulicky 2020, Berigan et al. 2022). While translocation initially created new leks and bolstered existing ones, male high counts at all leks decreased by approximately 43% by 2021 (Teige 2021). Lack of success to increase lesser prairie-chicken abundance via translocation on the National Grasslands and throughout the Sand Sagebrush Prairie Ecoregion, despite favorable environmental conditions prior to translocation (Teige 2021), emphasized the need to identify factors driving lesser prairie-chicken population declines, including potential changes in breeding habitat on private and public lands.

. Previous research found that lesser prairie-chicken populations exhibit a life-history strategy based on episodic high productivity, with population growth rates largely dependent on nest success and chick survival (Hagen et al. 2009, Ross et al. 2018). In fact, nesting and brooding habitat are considered the most important limiting factors for lesser prairie-chicken populations and therefore critical to lesser prairie-chicken demography (Pitman et al. 2006, Hagen et al. 2009, Van Pelt et al. 2013, Ross et al. 2018).

It is likely that landscape-scale change, including increasing vertical structures (e.g., anthropogenic and trees) and decrease of large areas of contiguous prairie, may have contributed to lesser prairie-chicken declines across the Sand Sagebrush Prairie Ecoregion (Chapter 1). Although broad-scale land cover on the National Grasslands have changed little since the 1980s, anecdotal reports suggest that vegetation composition and structure may have changed. Recent research suggests that many of the U.S. Forest Service habitat goals established in 2014 are not currently met on the National Grasslands (Berigan et al. 2022). Unfortunately, there has not been any quantification of these perceived changes on a broader time scale. A retrospective assessment is needed to determine whether additional changes in breeding habitat on finer scales (vegetation composition and structure) occurred on the National Grasslands to an extent that could explain declining lesser prairie-chicken abundance since 1985.

High-quality nesting habitat selected by lesser prairie-chickens is characterized by taller, dense vegetation, which provides thermal cover in addition to visual screening from predators (Haukos and Zavaleta 2016, Lautenbach et al. 2019). Loss of visual cover or a shift to vegetation composition other than high-value mid- and tallgrasses and ~20% sand sagebrush could reduce availability of high-quality nesting habitat (USFS 2014). Within brooding habitat, greater herbaceous forb cover provides invertebrates on which chicks feed, potentially improving growth and survival rates (Jones 1963, Jamison et al. 2002, Hagen et al. 2005, Pitman et al. 2006). Comparatively greater bare ground cover in brooding habitat helps facilitate chick mobility (Jamison et al. 2002, Hagen et al. 2005). Deteriorating nesting habitat quantity and quality could lead to female dispersal to other sites, a shift potentially explaining the absence or collapse of leks, which often occur within 3.2 km of a lek (USFS 2014, Aulicky 2020) Overall, changes in fine-scale vegetation composition and structure may explain lesser prairie-chicken population trends in the Cimarron and Comanche National Grasslands since the late 1980s.

My objective was to assess change in lesser prairie-chicken breeding habitat by comparing mean values of vegetation survey metrics including visual obstruction, percent canopy cover, sand sagebrush density, and occurrence of both sand sagebrush and high-value grass species during a time series from the lesser prairie-chicken population peak (roughly 1985) through recent surveys (2022). I hypothesized that the decline in lesser prairie-chickens in the Sand Sagebrush Prairie Ecoregion since the mid-1980s is related to the reduction of nesting habitat quality (i.e., selected vegetation composition and structure) on the National Grasslands and throughout the ecoregion. I hypothesized that visual obstruction decreased, occurrence of grass species associated with lesser prairie-chicken nesting habitat decreased, and sand sagebrush occurrence and density increased on the Cimarron and Comanche National Grasslands.

Study Area

The U.S. Forest Service Cimarron and Comanche National Grasslands (CCNG) consisted of former pasture and croplands largely restored to sand sagebrush prairie after the prolonged drought and desolation of the 1930s Dust Bowl (Guest 1968, Olson 1997, Berigan et al. 2022). The CCNG encompassed 45,300 ha of the Sand Sagebrush Prairie Ecoregion in Kansas and Colorado and comprised the largest parcel of public lands in the lesser prairie-chicken range

(Elmore and Dahlgren 2016, Berigan 2019). The CCNG was managed under a multi-use strategy including livestock grazing, recreation, and energy development. Grazing intensity in the region was typically moderate to high, with annual grazing pressure on the Cimarron having nearly doubled since the mid-1950s (Guest 1968, Berigan et al. 2022). For allotments with lesser prairie-chicken management goals, CCNG grazing management plans include contingency plans for extended drought (USFS 2014). However, there is risk of overstocking in the Sand Sagebrush Prairie Ecoregion, as the large variation in annual precipitation and resultant grass productivity creates a challenge in establishing stocking rates (Elmore and Dahlgren 2016). Estimated stocking rates ranged among pastures from approximately 0.31 AUM/ha to 1.53 AUM/ha on Cimarron National Grassland in 2018 (USFS, unpublished report). There were at least 15 different pastures under lesser prairie-chicken management in Comanche National Grassland, with stocking rates on cow-calf units ranging from 0.36 AUM/ha to 0.81 AUM/ha (D. Augustine, unpublished data). Prescribed fire is used to promote heterogeneous vegetation cover for lesser prairie-chickens on the CCNG, though resources are often limited (Elmore and Dahlgren 2016). Historical use of herbicides to manage sand sagebrush has been reported on the CCNG, but frequency and extent was not documented.

Vegetation composition was primarily dependent on soil type and grazing intensity (Berigan 2019). Bands of sandy soils such as those running parallel to the Cimarron and Arkansas River drainages, characteristic of the sand sagebrush prairie, supported the eponymous sand sagebrush shrubs as well as mid- to tall grasses including big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), sideoats grama (*Bouteloua curtipendula*), and sand dropseed (*Sporobolus cryptandrus*; Haukos et al. 2016). Short-grass prairie species, supported in clayey and loamy soils, included blue grama (*Bouteloua gracilis*) and buffalograss (*B. dactyloides*; Berigan et al. 2022).

Common forbs in the ecoregion included annual buckwheat (*Eriogonum annuum*), blazing star (*Liatris* spp.), western ragweed (*Ambrosia psilostachya*), prairie sunflower (*Helianthus periolaris*), annual sunflower (*H. annuus*), Russian thistle (*Salsola tragus*), camphorweed *(Heterotheca subaxillaris),* Indian blanket flower (*Gaillardia pulchella*), tansy aster (*Machaeranthera tanacetifolia*), buffalo bur (*Solanum rostratum*), buffalo gourd (*Cucurbita foetidissima*), wax goldenweed (*Grindelia papposa*), prickly lettuce (*Lactuca serriola*), and marestail (*Conzya canadensis*; Berigan et al. 2022).

Average annual precipitation for Elkhart, Kansas, in Morton County from 1985-2020 was 43.8 cm, varying from 24.2 cm to 67.6 cm (data from High Plains Regional Climate Center 2021). Drought in this region occurred on a roughly 5-year cycle (Haukos et al. 2016).

Methods

I obtained available historical CCNG vegetation data sets for the period 1985 to current. Colorado Wildlife and Parks (CPW), U.S. Forest Service (USFS), CCNG offices, nongovernmental organizations, known researchers, and former CPW and USFS biologists were contacted to inquire on the availability of vegetation survey data. I grouped Comanche/Colorado data by time period and site (Table 2.1, Appendix B). I calculated trends by means for each pasture. In my analysis, a “pasture” refers to CCNG lands with the same Allotment or Unit name. “Pastures” as defined this way are not necessarily physically connected, but likely managed with the same grazing practices and management goals. Where necessary, I converted visual obstruction readings (VOR) from centimeters or inches to decimeters. I used only vegetation data collected during approximate lesser prairie-chicken nesting and brooding season (end of March or April through the end of September). Omitting winter vegetation also reduced the risk of reporting differences in vegetation structure inherent to seasonal changes.

For each National Grassland, I calculated pasture-level mean and standard error for key vegetation metrics associated with lesser prairie-chicken habitat, including: mean VOR (dm); mean percent canopy cover of grass, forbs, shrubs, and bare ground; sand sagebrush density (plants/ha); percent of observations with sand sagebrush; and percent of observations with high-value grasses (USFS 2014). High-value grasses are native species of importance to lesser prairie-chicken nesting habitat, including big bluestem, Indiangrass (*Sorghastrum nutans*), little bluestem, prairie sandreed (*Calamovilfa longifolia*), sand bluestem (*Andropogon halii*), sand lovegrass (*Eragrostis trichoides),* and switchgrass (*Panicum virgatum*; USFS 2014). Recommended management goals for vegetation metrics have been established for the Sand Sagebrush Prairie Ecoregion (USFS 2014; Ven Pelt et al. 2013, Pitman et al. 2005; Table 2.1). Nesting and brooding habitat differ structurally, with tall, dense vegetation used by incubating hens and more open, forb-dominated areas used as brooding habitat (Haukos and Zavaleta 2016, Hagen et al. 2013). Visual obstruction (VOR) incorporates both vegetation height and density and thus is an important method for assessing nesting habitat (Robel et al. 1970). Visual obstruction was measured by recording the highest point at which 100% obstruction by vegetation occurs on Robel pole marked in decimeter increments, and averaged when multiple readings (e.g., 4 cardinal directions) were recorded at a sampling point (Robel et al. 1970). Quality nesting and brooding habitat are characterized by their respective horizontal percent canopy cover of grass and forbs (Table 2.1). Percent canopy cover was measured by visually estimating percent coverage of each of the cover types within a Daubenmire frame (Daubenmire 1959). Sand sagebrush is also an important component of habitat monitoring, as successful nests in the SSPE are often located in areas with specific shrub densities (Pitman et al. 2005). Sand sagebrush density was estimated by recording the number of sagebrush occurrences along a transect, and extrapolating the number of plants per 50 m^2 to yield a number of plants per hectare. In addition to sand sagebrush shrubs, nesting habitat is also associated with a suite of selected native grasses (USFWS 2014). I calculated the percent of observations with sand sagebrush or high-value grasses from data sets that included species-level plant identification. Completeness of data sets varied among sources (Table 2.2); raw data were available for more recent surveys, while I primarily relied on summary reports or transect-level means for older data sets. Data and associated trends are presented by pasture. Where the sum of the mean and associated standard error overlapped between two time periods, I determined that no change had occurred.

Data sources

I accessed a total of 7 vegetation data sources from state and Federal biologists and from the translocation project. Raw data for Comanche National Grassland were available for 2006 and from 2017-2020; the remainder of the data were in the form of summary tables ranging in specificity from transect-level to pasture-level averages. Cimarron National Grassland raw data were available only for the translocation project data (2017-2020); the remainder were sourced from summary tables. I obtained stocking rate data from 2006 for Comanche National Grassland (D. Augustine, unpublished data) and 2018 for Cimarron National Grassland (USFS, unpublished report). I calculated AUM/ha from AUMs and area for 23 of the Cimarron pastures (USFS, unpublished report).

Colorado Parks and Wildlife (1986-1987)

I used summarized data found in annual reports for vegetation structure data collected on Comanche National Grassland during 1986-1987 (Giesen 1991). Original data were not available for Colorado Parks and Wildlife 1986-1987 reports. I omitted bird flush and nest vegetation data (1986-1990) as there were no locations provided for those data.

These data were collected during multiple “cursory vegetation measurements” per pasture for the study area (Giesen 1991). I calculated a mean of these measurements where multiple samples were taken for a pasture. Averages for each relevant year category are reported by pasture name. I identified pastures by cross-checking the township, section, and range for sample location with a map of Comanche National Grassland (Table 8 in Giesen 1991). Where a section was split between pastures, I included those “shared” values in calculating means for each of the 2 pastures.

I calculated sand sagebrush density for each pasture by averaging the density measurements for each sample point in a pasture. Of the vegetation metrics, only percent of observations with high value grass and percent of observations with sand sagebrush could not be calculated from the data set. Percent canopy cover of shrubs in this data set included only sand sagebrush, rather than all shrub species.

US Forest Service (1985, 2007)

Summaries of point-step data collected in two survey years in the Mills unit of Cimarron National Grassland, with four transects in 1985 and three transects in 2007. Only percent canopy cover and percent of observations with high value grass were available from this data set.

Cimarron and Comanche National Grasslands report (1998, 2001, 2003):

Augustine (2006) contained a summary table with average VOR (reported as VOM) for Robel pole surveys from 1998-2006 for CCNG. Raw data were unavailable. I omitted 1999 and 2006 data, as these included winter vegetation, leaving 1998, 2001, and 2003 survey years for breeding season vegetation. Pastures in this dataset were reported as "complexes", with up to three pastures in a complex. Data from one of the complexes (Rolla/Santa Fe/Wilburton) could not be used in my analyses as one of the pastures in the complex was not common to other data sets. Where possible, I combined pastures in calculating results from the other Cimarron National Grassland data sets to match the complexes in this 2006 report. To estimate percent of observations with sand sagebrush, I calculated the number of times sand sagebrush was recorded out of the total number of observations on the transect, and averaged these values within each pasture per year.

Comanche National Grassland (1998-2004, 2006)

Vegetation transect data included VOR (reported as VOM readings, in inches). Data for 2006 included raw transect data ("FY06 Standard Transects"), as well as observations on whether obstruction was due to the presence of sand sagebrush along each transect. Obstruction by yucca and sand sagebrush was reported on transects; however, no obstruction by herbaceous species was reported. Pasture-level yearly summarized mean VOR was available for 1998-2004: 1998 (April-May), 1999 (April only), 2003 (May only), and 2004 (late March-April).

Standard Transect Summaries included data from 2005 (31 transects in November) and 2006 (41 transects Feb-April); however, I excluded the 2005 data set as it was collected in the fall. I also omitted Feb 2006 data from analysis to keep consistent with breeding season measurements. I omitted pastures that were not common with other data sets (changes could not be estimated across time): Arroyo, Bethel/Sunrise, and 6C. I did not use data summaries for White Cow/Sandsage/Aubrey and Sunflower, as there were no data at the level needed to calculate standard error. Only Prairie Chicken, Mt. Carmel, Arroyo, and Sand Hills pastures were surveyed each year; other pastures were surveyed in one or two of the years. I calculated averages of transects for each pasture by year.

LPC Habitat Assessment, Comanche National Grassland (2010)

From tables in the 2010 Comanche National Grassland report assessing availability of lesser prairie-chicken habitat (Rondeau and Decker 2010), I analyzed data for all vegetation metrics except percent of observations with high value grass. Only Mt. Carmel and Sunflower pastures were reported.

Based on presented data, for each pasture I first calculated the average sand sagebrush count of the transects in each pasture. I then divided the count of observations by the area of the transect (50 m^2) and multiplied by 1,000 to estimate the number of sand sagebrush plants/ha.

Habitat Inventory, Cimarron and Comanche National Grasslands (2015)

Data were collected in 2015 for use in a lesser prairie-chicken habitat inventory assessment across CCNG (Wuenschel et al. 2016). Structural measurements were taken in April through May, while composition was measured in July and August. I sorted transects by CCNG pasture, and summed vegetation data at the pasture level. I omitted Comanche National Grassland pastures White Cow, Sandsage, and Aubrey as well as Cimarron pastures Bridge, East Artesian, and West Artesian, as these were only surveyed in recent years. I was able to calculate all vegetation metrics from this data set.

Kansas State University (2017-2020)

I accessed raw data for both point-paired and point-step transects measured throughout CCNG from 2017-2020. Both types of data were available for the majority of pastures included in my analysis, with the exception of Mills and Mt. Carmel, for which only point-paired data had been collected. Point-paired data consisted of vegetation measurements at actual used locations from lesser prairie-chickens fitted with GPS transmitters, as well as a paired survey points randomly generated within 300 m of used locations for estimation of resource selection. I also used point vegetation data randomly generated for each patch. Patches were delineated by dominant plant species, property and allotment boundaries, and soil types (Berigan 2019). I considered each point as an observation to combine the two forms of survey data (points and transects) for each pasture.

Patch data also consisted of point-step transects, which originated from randomly generated points and extended in a randomly-chosen cardinal direction within patch boundaries (L. Berigan, personal communication). I considered each step on a transect as an observation to remain consistent when combining with point-paired data. Each patch transect had 250 observations. For patch VOR, I averaged VORs recorded for each cardinal direction to yield a mean VOR per each bird location point and calculated the same for all VOR stops (at 0 m, 50 m, 100 m, 150 m, and 200 m) for each transect. I calculated VOR per pasture by averaging transect-level VOR means within a pasture. Transect data consisted of only species composition and VOR, so those metrics were calculated based on 250 * (number of transects), plus the number of points for that pasture.

I omitted data from the 7 Comanche National Grassland pastures that were not surveyed in other data sources: Boston, Boston South, Branstine, Rattlesnake, Richards, Wyche, and

Yucca. For the same reason I also omitted 11 Cimarron National Grassland pastures: Colorado Line, County Line, 81, Green, Morton, North Fork, North Lowe, Point, River, Stateline, and Richfield. Similarly, I omitted 4 Cimarron National Grassland pastures (Bridge, Lowder Knoll, East Artesian, and West Artesian) and one Comanche National Grassland grouped pasture (White Cow/Sandsage/Aubrey) for which only recent data were available. I omitted data collected in winter (Nov-Mar) from analysis. I combined data from 3 Cimarron National Grassland pastures (Steer, College, and Headquarters) to compare with data from 1998 – 2003 for those pastures, which were reported grouped together during that time period (USFS 2006). It was not possible to derive sand sagebrush density from this data set, though all other metrics could be calculated.

Results

Contemporary vegetation surveys (2015 through 2020) included 13 – 16 different pastures; however, data from the full range of pastures ultimately could not be used to estimate change over time, as most had not been sampled at or near the time of population peak or periodically previously. Only 4 Comanche National Grassland pastures (Mt. Carmel, Deweese, Prairie Chicken, and Sunflower) and 3 Cimarron National Grassland pastures (Mills, South Lowe, and Headquarters/Steer/College) were surveyed in multiple years. Therefore, I was restricted to using data from these pastures for my analysis. Additionally, variation in survey methods resulted in inconsistent coverage of vegetation metrics across the time series and among pastures. Of the Cimarron National Grassland pastures measured across years, only Headquarters/Steer/College included VOR measurements, and only Mills was sampled for percent cover (Appendix B).

When compared with lesser prairie-chicken nesting and brood-rearing habitat recommendations (USFS 2014, Van Pelt et al. 2013, Pitman et al 2005; Table 2.2), the CCNG have only partially met lesser prairie-chicken habitat requirements at the pasture scale since population peak (Tables 2.3 and 2.4). Based on the available data, none of the pastures on Comanche National Grassland met recommended visual obstruction (at least 2.54 dm [USFS 2014]) since 1985 based on vegetation monitoring records, with average VORs on Comanche National Grassland pastures decreasing over time by a range of 0.02 dm to 0.50 dm. The CCNG also appears not to have met recommended sand sagebrush density for nesting habitat over time (>6,500 plants/ha, Pitman et al. 2005). However, the CCNG appears to approach or meet management recommendations for percent canopy cover of grass for both nesting and brooding habitat. In fact, percent cover of grass and forbs increased on both National Grasslands since 1986, based on available data. However, percent cover of forbs in the Comanche National Grassland was below recommendations, while Cimarron National Grassland appears to have adequate forb cover in recent years. All vegetation cover types on Cimarron National Grassland fluctuated, but ultimately increased from 1985 – 2020 based on the available data.

Evidence suggests that >1.2 AUM/ha is the threshold after which habitat selection by female lesser prairie-chickens and potentially nest success are negatively affected (Fritts et al. 2016, Kraft et al. 2021). Comanche National Grassland stocking rates appeared to fall below this threshold as of 2006; of the available data, the greatest stocking density was 0.81 AUM/ha (D. Augustine, unpublished data). Conversely, I estimated that as of 2018, at least 17.4% of 23 Cimarron National Grassland pastures appeared to be overstocked with densities up to 1.53 AUM/ha.

Mean visual obstruction

Mean VOR did not change in the two Cimarron National Grassland pastures (South Lowe, and combined Steer/College/HQ) for which multiple years of data were available (Figure 2.1). Mean VOR decreased over time for Comanche National Grassland pastures for which multiple years of data were available (Mt. Carmel, Sunflower, Deweese, and Prairie Chicken; Figure 2.2), though Mt. Carmel did not change from 2015 - 2020. This decrease ranged from 0.02 dm to 0.50 dm among 4 pastures (Figure 2.2).

Mean percent cover

Mean percent canopy cover of grass on Comanche National Grassland increased by 19.7 and 35.9 percentage points on Mt. Carmel and Sunflower pastures, respectively, since 1986 (Figure 2.3). Percent cover of forbs increased by 4.4 and 11.9 percentage points on Mt. Carmel and Sunflower pastures, respectively, since 1986 (Figure 2.4). In one Comanche National Grassland pasture with multiple years of shrub cover data (Mt. Carmel), mean percent shrub cover increased by 5 percentage points since 2010 (Figure 2.5). Mean percent bare ground decreased by at least 43 percentage points on two pastures since 1986 (Figure 2.6).

Data for both historic and recent measurements on Cimarron National Grassland were only available for the Mills unit, with 1985 – 2007 data consisting of 7 sample points. On the Mills unit, mean percent cover of grass increased by 64.4 percentage points between 1985 and 2017-2020 data (Figure 2.7). Mean percent forb cover did not change (Figure 2.8). Mean percent cover of shrubs and bare ground increased from 0 to 0.27, and from 0 to 2.66, respectively. (Figures 2.11, 2.12).

Occurrence of high-value grasses, sand sagebrush

Percent of observations with high-value grass species remained at 0% on the Mills unit of Cimarron National Grassland from 1985 to 2018/2020. It was not possible to calculate change in percent of observations with high-value grasses over time for Comanche National Grassland, as grass species were only identified in recent surveys (2015 and 2018/2020). For the same reason, change in percent of observations with sand sagebrush could not be estimated for Cimarron National Grassland. On Comanche National Grassland, percent of observations with sand sagebrush appeared to increase by 24.3 percentage points from 2006 to 2020 in Mt. Carmel, and 14.3 percentage points in Deweese (Figure 2.12).

Sand sagebrush density

On Comanche National Grassland, sagebrush density decreased by 4,020 shrubs/ha from 1986 – 2010 in Prairie Chicken pasture (Figure 2.13). However, shrub cover also increased by 3,463 shrubs/ha from 1986 – 2015 in Mt. Carmel pasture and increased by 1,210 shrubs/ha in Sunflower (Figure 2.13).

Discussion

Lesser prairie-chicken populations declined sharply on Cimarron and Comanche National Grasslands from 1985 to present (Haukos and Boal 2016), despite recent efforts to bolster populations via translocation (Berigan 2019, Teige 2021). It is unlikely that precipitous population declines were due to landscape-scale landcover type change throughout the ecoregion (Chapter 1) though finer-scale changes in habitat structure and composition may have occurred. Unfortunately, CCNG vegetation survey data were insufficient to adequately estimate change across the full suite of vegetation metrics. This was particularly challenging when assessing Cimarron National Grassland, for which there existed a very limited record of survey data; my results for percent cover on Cimarron National Grassland were derived solely from a single pasture (Mills). Despite the abundance of 2018-2020 CCNG vegetation survey data, most of these recent records unfortunately could not be used in my analysis, given the limited availability of pre-2015 data for comparison. Given the limited scope of data, it is unlikely that my results for Cimarron National Grassland reflect actual trends in vegetation over time.

Some variation in vegetation survey methodology is expected given the temporal extent of the collective data and resources available to agencies are often limited. Nevertheless, it is troubling that scarce habitat monitoring records exist for public lands managed for a grassland-dependent species with known vegetation structural and compositional requirements within a landscape constantly influenced by land use changes and drought cycles. It is unfortunate that after 3 decades we have only a limited retrospective picture as to whether lesser prairie-chicken habitat changed to a degree that contributed to the observed population declines. Given the precarious state of lesser prairie-chickens in the Sand Sagebrush Prairie Ecoregion, it is imperative to establish a consistent and thorough monitoring program on which to base future management decisions as well as accurately track the progress of lands managed for lesser prairie-chicken habitat.

Based on the limited data available, it appears that vegetation structure and composition may have changed on Comanche National Grassland over time. The decrease in VOR on all Comanche National Grassland pastures surveyed from 1986 through recent years, as well as the lack of meeting VOR recommendations, indicates that nesting habitat, while already lacking, may have declined further since 1985. This is consistent with recent findings that the National Grasslands currently provide limited nesting cover for lesser prairie-chickens (Berigan et al. 2022). Nesting habitat influences space use by both male and female lesser prairie-chickens;

persistence of old leks, and formation of new ones, occurs in response to female movements, which are shaped by nesting habitat availability (Aulicky 2020). Lesser prairie-chickens typically select herbaceous and shrub cover with greater height and density for nesting (Pitman et al. 2005). Nests are more successful and female survival rates during nesting are greater in vegetation characterized by moderate to high vertical structure and canopy cover (Hagen et al. 2007). Lack of nesting habitat may have driven abandonment of the landscape by lesser prairie-chickens over time, leading to a decline in populations. Recent research suggests that absence of nesting habitat on CCNG may in fact have contributed to local extinction of native birds and lack of settling by translocated birds (Berigan et al. 2022). The apparent decrease in mean percent bare ground cover on Comanche National Grassland could also have reduced the quality of brood-rearing habitat, as chicks require more open understory for movement (Van Pelt et al. 2013). An increase in mean percent grass, shrub, and forb cover on Comanche National Grassland could conceivably be a positive change for lesser prairie-chicken habitat. However, when taken in the context of inadequate visual obstruction, and lower sand sagebrush density for the Sand Sagebrush Prairie Ecoregion than recommended by Hagen et al. (2004), the CCNG has only partially met lesser prairie-chicken breeding habitat goals over time.

Though my results appear in part consistent with observed trends, my scope was limited by the available historic data. A more conclusive assessment is not possible without additional monitoring records. Given the importance of breeding season success to lesser prairie-chicken population rates, monitoring nesting and brood-rearing habitat is crucial when evaluating lesser prairie-chicken management response (Pitman et al. 2006, Hagen et al. 2009, Van Pelt et al. 2013, Ross et al. 2018).

Future monitoring on the CCNG should be purpose-driven and adaptive, based on known lesser prairie-chicken habitat requirements as well as assessments of previous monitoring outcomes - potentially following the model of the US Fish and Wildlife Service's Strategic Habitat Conservation Framework (USFWS 2008). Monitoring on the National Grasslands could be incorporated into a larger lesser prairie-chicken monitoring effort across the species' range. Surveys should include a standard set of vegetation metrics that reflect nesting and brood-rearing habitat composition and structure (Table 2.5). The U.S. Forest Service should monitor vegetation on CCNG to assess the effectiveness of their 2014 Management Plan, with the objective of detecting trends in breeding habitat over time. For this reason, the same set of pastures, or as many consistent with previous surveys as possible, should be revisited in subsequent survey years. Monitoring should also include pre- and post-treatment surveys to measure vegetation response, including woody plants, to management activities such as prescribed fire, grazing, and herbicide application. In grazed areas, dates of use and stocking rates should be recorded and drought conditions noted (USDA 2009). Monitoring vegetation response to grazing may include measuring above-ground biomass via clipped quadrats using grazing exclosures to determine grazing intensity and pasture condition (Holechek et al. 2011). From the record of vegetation survey data, managers should be able to assess how any of the metrics related to lesser prairie-chicken habitat have changed over time, and determine whether to adjust management strategies to achieve a desired outcome for vegetation characteristics.

The public lands of Cimarron and Comanche National Grasslands are the only secure potential large-scale habitat for lesser prairie-chickens in the Sand Sagebrush Prairie Ecoregion, given the temporary status of CRP fields (Chapter 1). While lesser prairie-chickens are known to select CRP, and in fact often dispersed to CRP fields following translocation to the National

Grasslands (Teige 2021), the voluntary conservation program is at best a short-term oasis for the grassland-obligate species. With the uncertain future of new CRP enrollment, and rolling expiration of current CRP contracts, it becomes increasingly necessary to pursue other strategies for promoting lesser prairie-chicken habitat. The permanence of the Cimarron and Comanche National Grasslands, its stated goal of providing lesser prairie-chicken habitat, and the capability of the Forest Service to direct land management offer a more stable avenue for restoring lesser prairie-chicken populations. Investing time and resources into monitoring, and potentially land management changes to promote quality habitat on the National Grasslands, may be the most reliable path to securing a future for lesser prairie-chickens in the Sand Sagebrush Prairie Ecoregion.

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Table 2.1. Sources of historical and contemporary vegetation survey data, including vegetation metrics measured in each set, for the U.S. Forest Service Cimarron and Comanche National Grassland in Kansas and Colorado, USA. An X indicates whether data for a given metric were collected for that vegetation metric.

				Vegetation Metrics			
Study area	**Source**	**Year(s)**	**Description**	**VOR**	**Percent Canopy Cover**	**Species Identified**	**Sand Sagebrush Density**
Comanche	Colorado Parks and Wildlife[1]	1986-1987	Summary report	X	X	na	X
	USFS[2]	1998-2004, 2006	Pasture-level data, raw data	X	na	Sand sagebrush only (2006)	na
	USFS[3], CSU[3]	2010		X	X	na	na
	USFS[4]	2015	Raw data	X	X	X	X
	Kansas State University[5]	2017-2020	Raw data	X	X	X	na
Cimarron	USFS[6]	1985	Transect summaries	na	X	X	na
		2007	Transect summaries	na	X	X	na
	USFS (2006)[7]	1998, 2001, 2003	Summary report	X	na	na	na
	USFS[4]	2015	Raw data	X	X	X	X
	Kansas State University[5]	2017-2020	Raw data	X	X	X	na

1. Ken Giesen
2. David Augustine
3. Renee Rondeau and Karin Decker
4. Amarina Wuenschel
5. David Haukos
6. Nancy Brewer
7. David Augustine (summary report)

Table 2.2. Recommended vegetation metrics for lesser prairie-chicken nesting and brooding habitat in the Sand Sagebrush Prairie Ecoregion of Kansas, Colorado, and Oklahoma, USA.

Management Plan or Recommendation	VOR	Percent Canopy Cover	Sand Sagebrush Density	Other management notes
USFS 2014	10 - 15 " (2.54 - 3.81 dm)	15 - 20% sand sagebrush	NA	Grass height 12 - 18 "
("lek-centric" 2 mi buffer zone)		40 - 50% native grasses		high-value nesting grasses (USFS 2014)
		15 - 20% native forbs		Annual monitoring until criteria are met; otherwise, every three years
Van Pelt et al. 2013			NA	
Nesting	NA	15 - 30% sand sagebrush >30% selected native grasses		Average grass height >15"
		>10% mixed native forbs		
Brood rearing	NA	10 - 25% sand sagebrush >20% selected native grasses		Understory open enough to allow movement of chicks Average grass height >15"
		>20% mixed native forbs		
Pitman et al. 2005				
Nesting	Average >2.7 dm	18 - 20% sand sagebrush	>6,500 plants/ha	Grass height > 25 cm
Hagen et al. 2013				
Nesting	NA	>60% forbs, shrubs, and grasses >40cm	NA	Less shrub and woody cover
Brood rearing	2.7 - 3.2 dm	27 - 40% herbaceous cover		Residual cover from 15 March to 15 July

Table 2.3. Summary of vegetation metrics for Cimarron National Grassland, Kansas, USA, and Comanche National Grassland, Colorado, USA, as compared to a selection of recommendations for lesser prairie-chicken nesting habitat (Table 2.2). Red indicates a value below nesting habitat standards for that metric according to all sources; blue indicates a value that meets recommendations according to some, but not all, sources; green indicates that a value meets recommendations from all of the selected sources.

Location	Data Source	Time Period	Pasture	Mean VOR (dm)	SE	Avg % Grass	SE	Avg % Forb	SE	Avg % Shrub	SE	Avg % Bare Ground	SE	Sand Sagebrush Density ([plants/ha)	SE	% Obs with High-Value Grass	% Obs with Sand Sagebrush
Comanche	Colorado Parks and Wildlife	1986-1987	Prairie Chicken	1.31	0.15	17.10	5.60	11.55	0.05	7.70*	2.30	71.35	5.75	5,820.00	1,660.00	NA	NA
			Mt Carmel	0.95	0.12	23.90	9.30	5.12	1.40	3.81*	1.02	70.88	3.00	2346.78	892.80	NA	NA
			Sunflower	1.23	0.45	18.60	1.90	1.25	0.15	3.30*	1.80	80.15	1.75	2940.0	37.42	NA	NA
	USFS	1998-2004, 2006	Prairie Chicken	0.91	0.15	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
			Mt Carmel	0.88	0.11	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	11.00
			Deweese	0.97	0.06	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	24.60
	Rondeau and Decker	2010	Prairie Chicken	0.81	0.19	63.20	6.28	0	NA	8.40	4.40	23.60	2.14	1,800.00	1,062.07	NA	NA
			Mt Carmel	1.32	0.15	51.94	3.04	3.29	1.30	11.03	1.48	24.45	1.55	2,509.68	375.12	NA	NA
	USFS	2015	Mt Carmel	0.24	0.03	39.22	2.96	10.83	1.50	30.77	3.70	14.80	1.97	5810.0	1308.55	1.79	0
			Sunflower	0.5	0.06	50.08	2.59	11.60	1.18	34.53	4.09	18.53	2.05	4150.0	930.97	1.72	0.06
			Deweese	0.75	0.12	41.48	2.56	13.38	1.20	35.16	2.27	14.21	1.67	12470.0	2045.81	4.68	10.27
	Kansas State University	2017-2020	Mt Carmel	0.93	0.21	43.64	4.25	9.54	1.85	15.53	2.99	27.11	2.99	NA	NA	0	35.29
			Sunflower	0.44	0.11	54.47	6.79	13.18	3.59	6.02	2.22	18.04	2.97	NA	NA	0	5.36
Cimarron	USFS	1985	Mills	NA	NA	42.25	4.75	3.25	1.80	0.00	NA	0	NA	NA	NA	0	

	2007	Mills	NA	NA	69.00	4.73	14.33	2.91	7.00	2.00	6.33	1.86	NA	NA	0	
USFS	1998, 2001, 2003	Steer, College, and HQ	1.19	0.16	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
		South Lowe	0.71	0.14	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
USFS	2015	College	0.17	0.03	28.35	3.10	24.65	2.16	30.79	3.45	26.15	1.87	2580			
		Wilburton	0.44	0.06	31.49	2.64	22.48	1.90	32.52	2.22	21.98	1.78	14770			
		South Lowe	0.62	0.08	38.08	2.45	16.42	1.22	35.19	3.34	21.79	1.60	4530			
		Steer	0.18	0.04					30.77	2.83			6000			
Translocation	2018-2020	Steer, College, and HQ	1.30	0.10									NA	NA		10.46
		Mills	0.61	0.15	74.37	3.51	5.26	1.45	0.27	0.23	2.66	0.77	NA	NA	0	0
		Sandhills	1.42		39.65		26.59		8.65		9.73		NA	NA	0	10.88
		South Lowe	0.78	0.06	39.91		24.69		4.68		15.34		NA	NA	0.09	8.20

Table 2.4. Summary of vegetation metrics for Cimarron National Grassland, Kansas, USA, and Comanche National Grassland, Colorado, USA, as compared to a selection of recommendations for lesser prairie-chicken brood habitat (Table 2.2). Red indicates a value below brood habitat standards for that metric according to all sources; blue indicates a value that meets recommendations according to some, but not all, sources; green indicates that a value meets recommendations from all of the selected sources.

Location	**Data Source**	**Time Period**	**Pasture**	**Mean VOR**	**SE**	**Avg % Grass**	**SE**	**Avg % Forbs**	**SE**	**Avg % Shrubs**	**SE**	**Avg % Bare Ground**	**SE**	**Sand sagebrush density (plants/ha)**	**SE**	**% obs with high-value grass**	**% obs with sand sagebrush**
Comanche	Giesen	1986-1987 only	Prairie Chicken	1.31	0.15	17.10	5.60	11.55	0.05	7.70*	2.30	71.35	5.75	5,820.00	1,660.00	NA	NA
			Mt Carmel	0.95	0.12	23.90	9.30	5.12	1.40	3.81*	1.02	70.88	3.00	2346.78	892.80	NA	NA
			Sunflower	1.23	0.45	18.60	1.90	1.25	0.15	3.30*	1.80	80.15	1.75	2940.0	37.42	NA	NA
	USFS	1998-2004, 2006	Prairie Chicken	0.91	0.15	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
			Mt Carmel	0.88	0.11	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	11.00
			Sand Hills			NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	24.00
			Deweese	0.97	0.06	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	24.60
	Rondeau and Decker	2010	Prairie Chicken	0.81	0.19	63.20	6.28	0	NA	8.40	4.40	23.60	2.14	1,800.00	1,062.07	NA	NA
			Mt Carmel	1.32	0.15	51.94	3.04	3.29	1.30	11.03	1.48	24.45	1.55	2,509.68	375.12	NA	NA
	USFS	2015	Mt Carmel	0.24	0.03	39.22	2.96	10.83	1.50	30.77	3.70	14.80	1.97	5810.0	1308.55	1.79	0
			Sunflower	0.50	0.06	50.08	2.59	11.60	1.18	34.53	4.09	18.53	2.05	4150.0	930.97	1.72	0.06
			Deweese	0.75	0.12	41.48	2.56	13.38	1.20	35.16	2.27	14.21	1.67	12470.0	2045.81	4.68	10.27
	Kansas State University	2017-2020	Mt Carmel	0.93	0.21	43.64	4.25	9.54	1.85	15.53	2.99	27.11	2.99	NA	NA	0	35.29
			Sunflower	0.44	0.11	54.47	6.79	13.18	3.59	6.02	2.22	18.04	2.97	NA	NA	0	5.36

Cimarron	Brewer point step data	1985	Mills	NA	NA	42.25	4.75	3.25	1.80	0.00	na	0	na	NA	NA	0	
		2007	Mills	NA	NA	69.00	4.73	14.33	2.91	7.00	2.00	6.33	1.86	NA	NA	0	
	USFS	1998, 2001, 2003	Steer, College, and HQ	1.19	0.16	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
			South Lowe	0.71	0.14	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	USFS	2015	College	0.17	0.03	28.35	3.10	24.65	2.16	30.79	3.45	26.15	1.87	2580			
			Wilburton	0.44	0.06	31.49	2.64	22.48	1.90	32.52	2.22	21.98	1.78	14770			
			South Lowe	0.62	0.08	38.08	2.45	16.42	1.22	35.19	3.34	21.79	1.60	4530			
			Steer	0.18	0.04					30.77	2.83			6000			
	Kansas State University	2018-2020	Steer, College, and HQ	1.30	0.10									NA	NA		
			Mills*	0.61	0.15	74.37	3.51	5.26	1.45	0.27	0.23	2.66	0.77	NA	NA	0	0
			Sandhills	1.42		39.65		26.59		8.65		9.73		NA	NA	0	10.88
			South Lowe	0.78	0.06	39.91		24.69		4.68		15.34		NA	NA	0.09	8.20

Table 2.5. Guidelines for monitoring lesser prairie-chicken breeding habitat in the Sand Sagebrush Prairie Ecoregion of Kansas, Colorado, and Oklahoma, USA.

Habitat component or response	**Metric**	**Method**	**Timing**
Structure	Visual obstruction	VOR with Robel pole[1] (dm); eight meters in four cardinal directions; points and point-step transects at patch or pasture level	Annually to every 3 years; May-June (breeding habitat conditions)
	Vegetation height	Tallest vegetation in Daubenmire frame[2] (cm) at survey points	
Cover composition	Canopy cover	Daubenmire[2] frame (% grass, forbs, shrubs, bare ground) at survey points	
	High-value nesting grasses (see USFS 2014)	Record species ID along point-step transects at patch or pasture level	
	Sand sagebrush density	Estimate plants/ha from count per belt transect (along point-step transect path) at patch or pasture level	
	Invasive species presence	Record species ID along point-step transects at patch or pasture level	
Grazing utilization	Above-ground biomass	Paired quadrats (clipped prior and clipped post-grazing); photos[3]	Dormant season (fall/winter)
Woody encroachment	Density (plants/ha)	Detection from aerial imagery via spatial covariance[4]	Every 5 years

1. Robel et al. 1970
2. Daubenmire et al. 1959
3. Holechek et al. 2011
4. Roberts et al. 2021

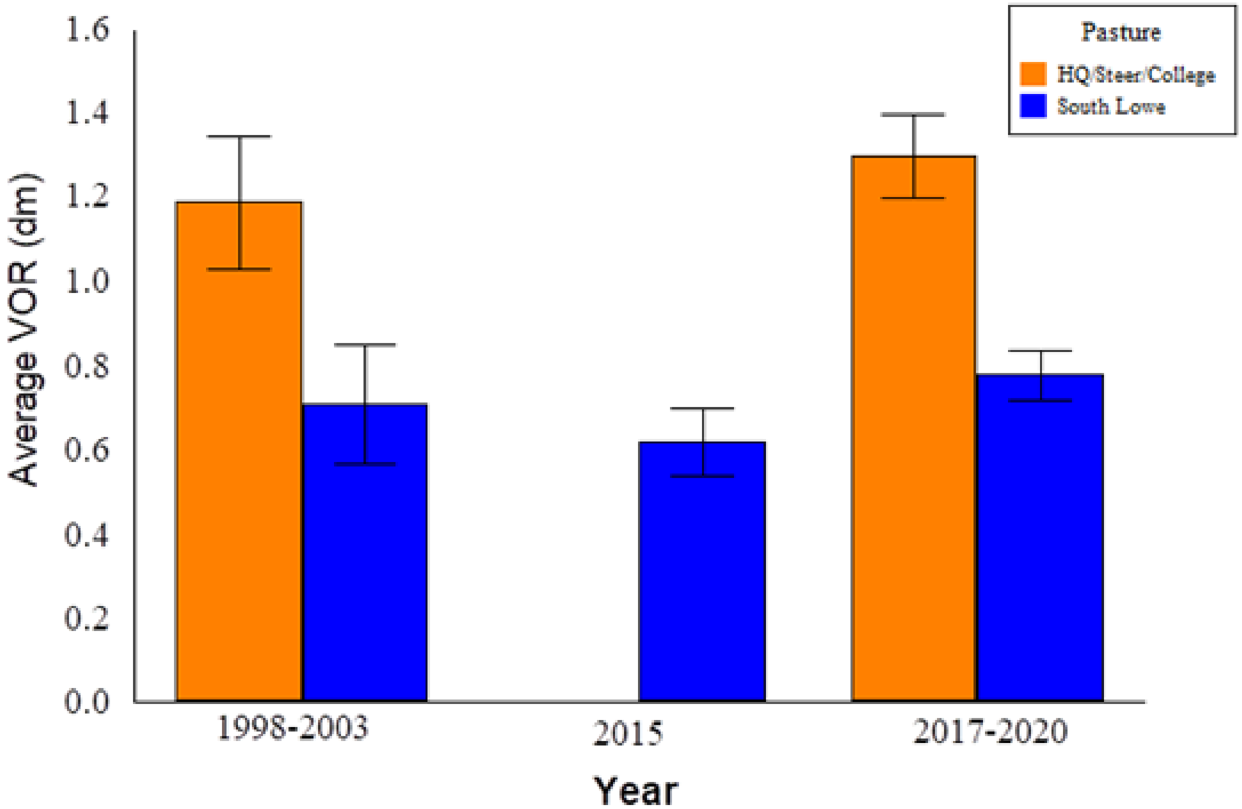

Figure 2.1. Average visual obstruction readings (100% VOR, in dm) and associated standard errors for two pastures on Cimarron National Grassland, Kansas, USA, between 1998 and 2020.

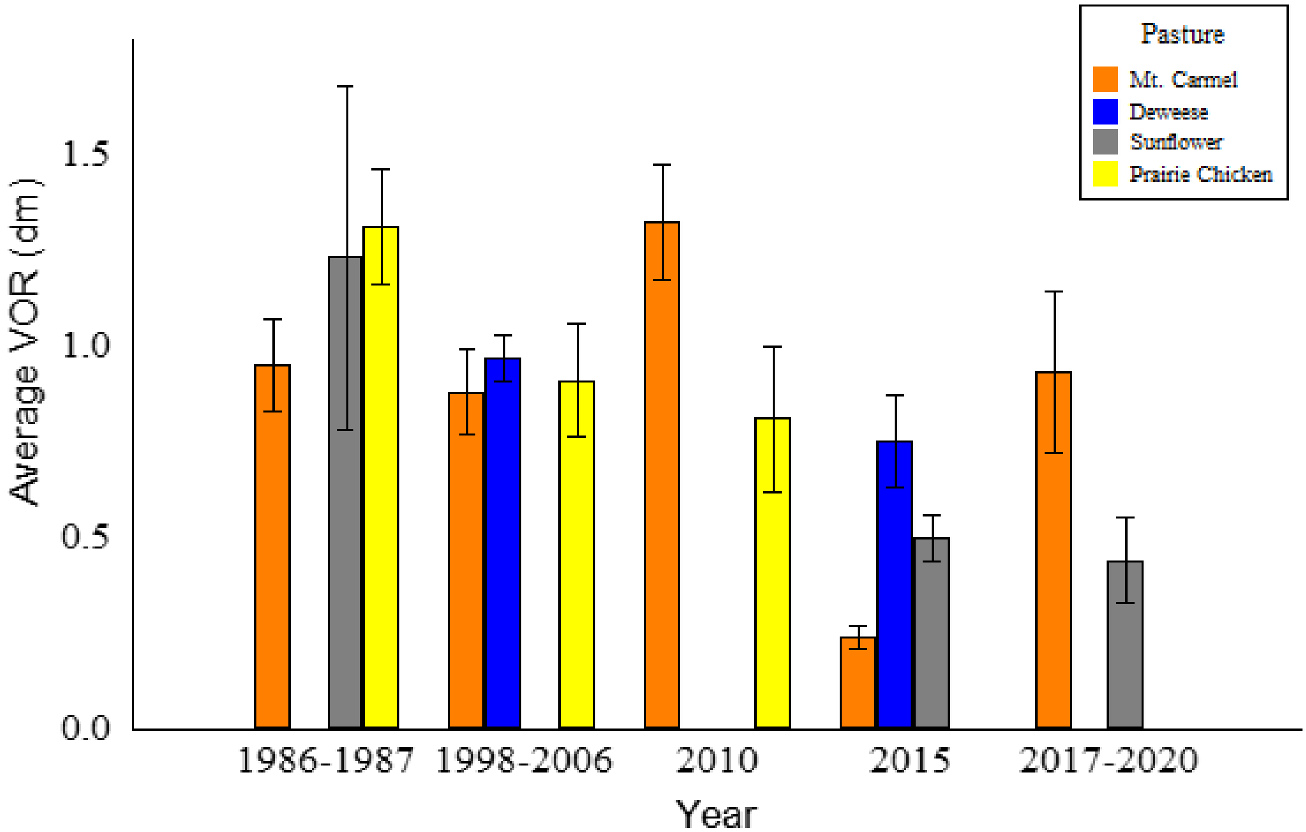

Figure 2.2. Average visual obstruction readings (VOR, in dm) and associated standard errors for four pastures in Comanche National Grassland, Colorado, USA, between 1986 and 2020.

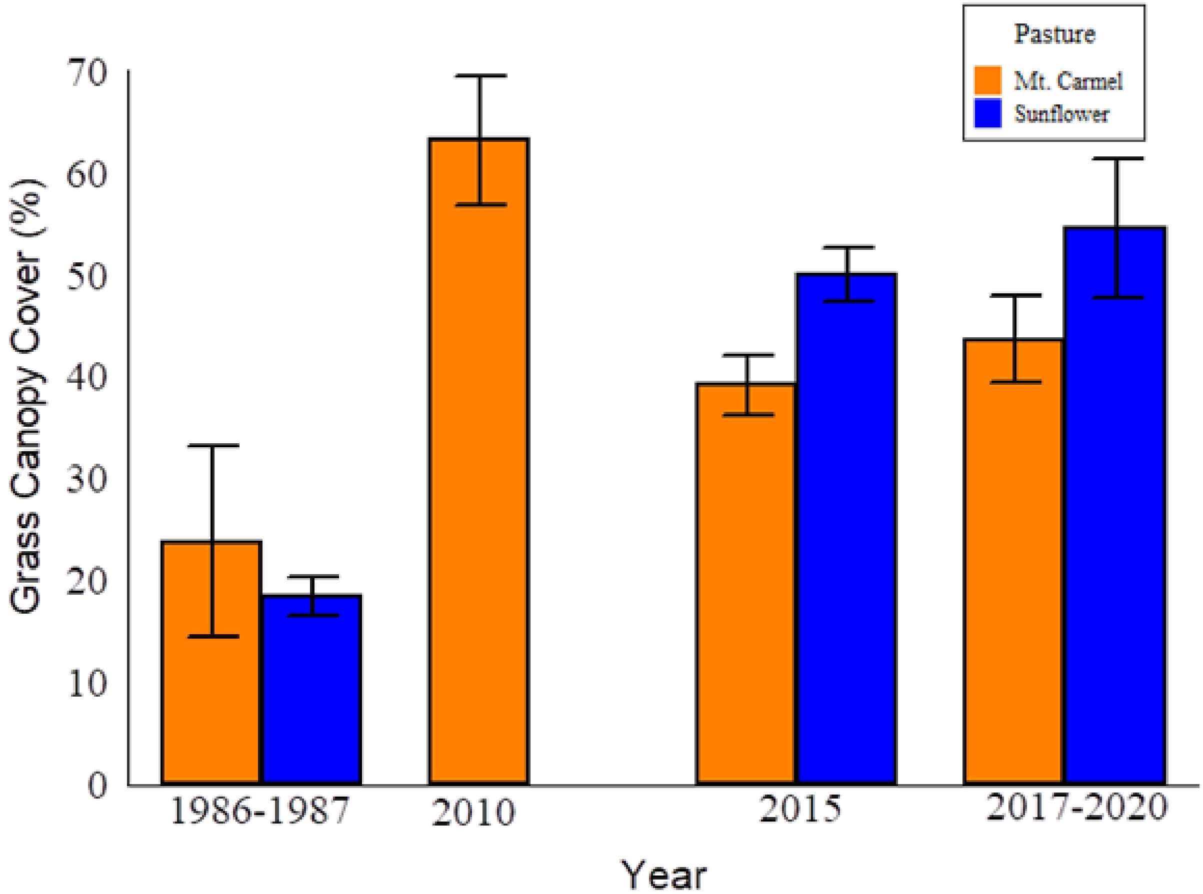

Figure 2.3. Average percent canopy cover of grass and associated standard errors in two pastures on Comanche National Grassland, Colorado, USA, between 1986 and 2020.

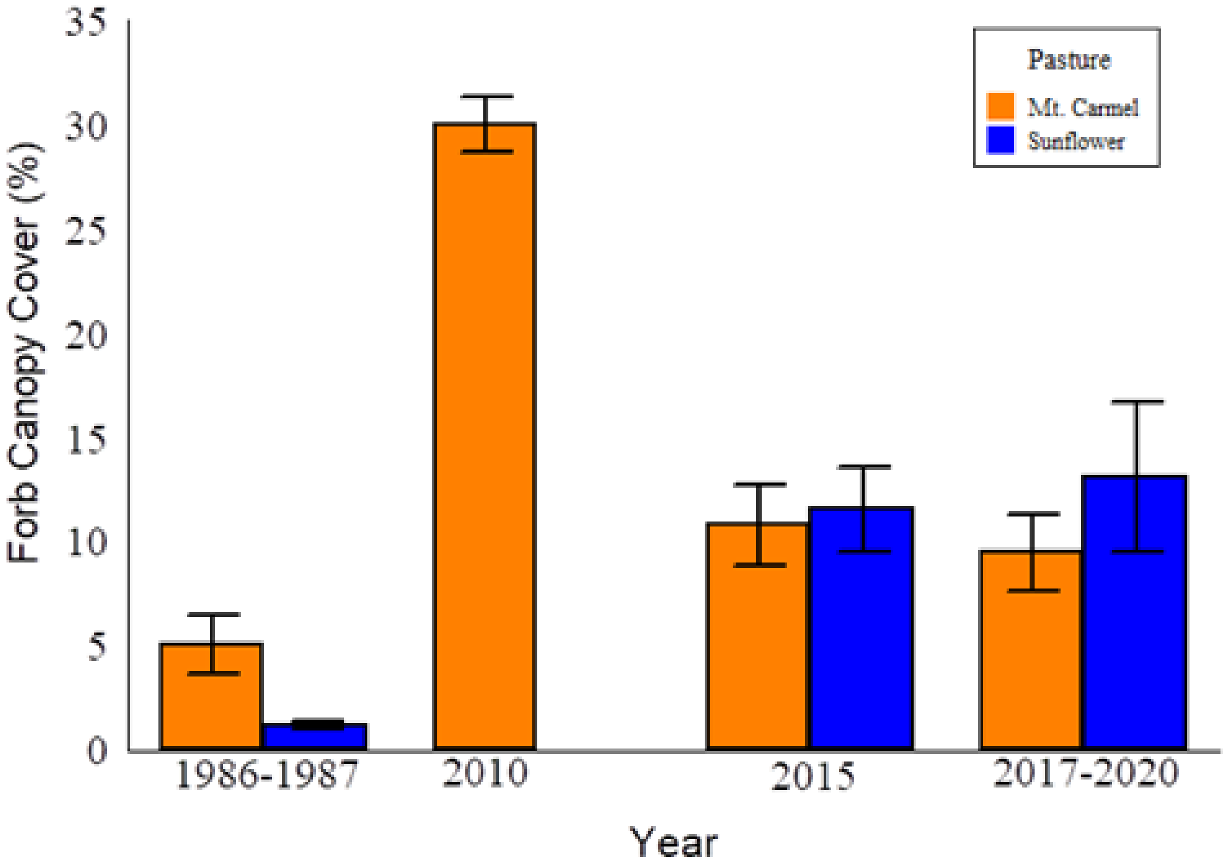

Figure 2.4. Average percent canopy cover of forbs and associated standard errors in two pastures on Comanche National Grassland, Colorado, USA, between 1986 and 2020.

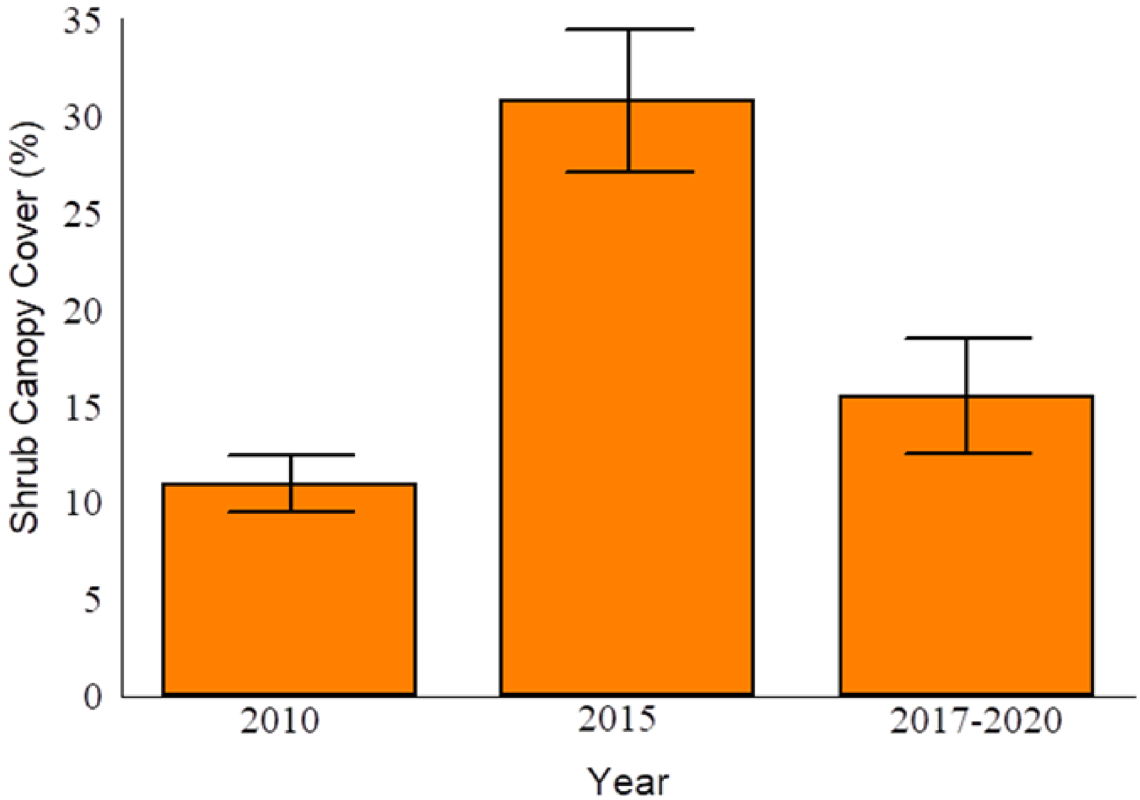

Figure 2.5. Average percent canopy cover and associated standard errors of shrubs in two pastures on Comanche National Grassland, Colorado, USA, between 1986 and 2020.

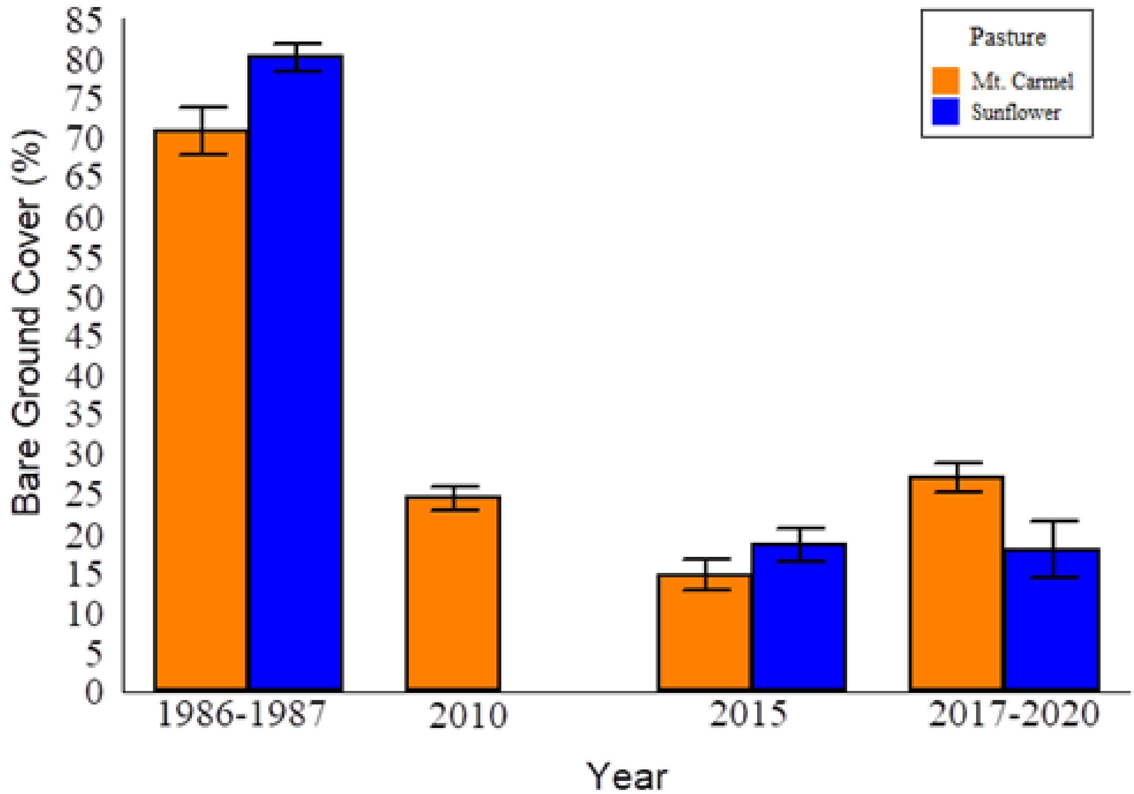

Figure 2.6. Average percent canopy cover of bare ground and associated standard errors in two pastures on Comanche National Grassland, Colorado, USA, between 1986 and 2020.

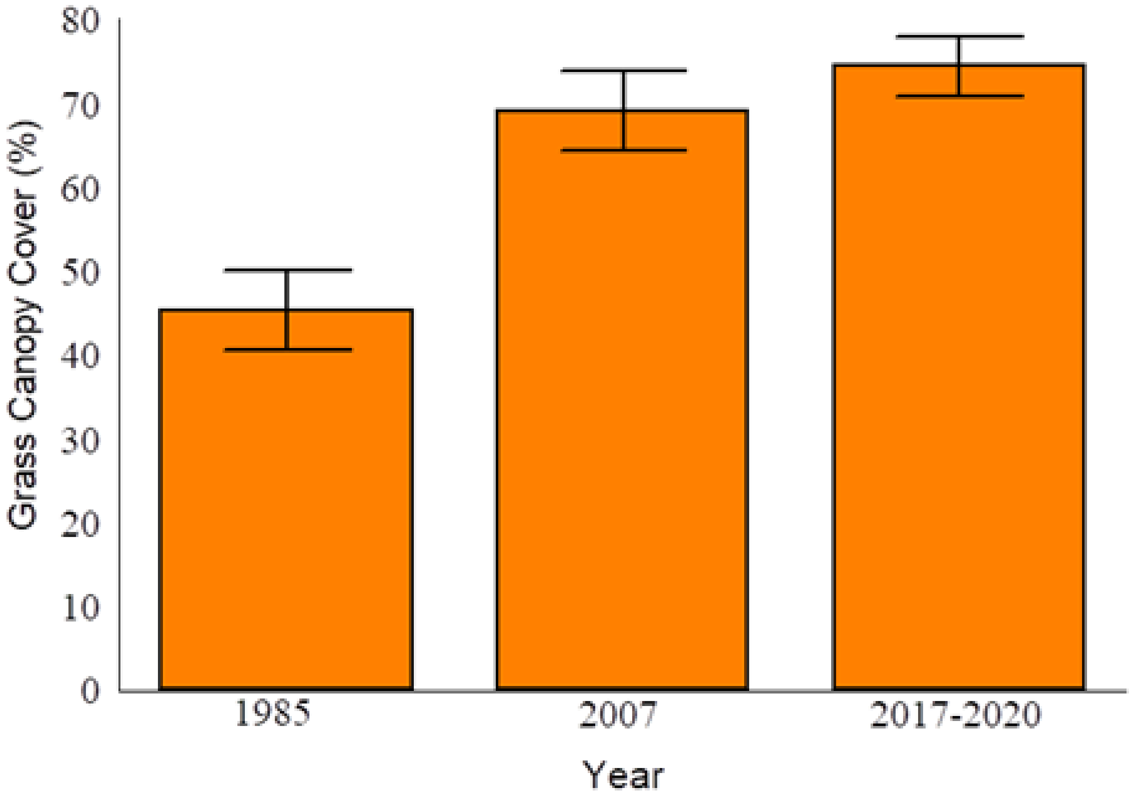

Figure 2.7. Average percent canopy cover and associated standard errors of grass in the Mills pasture on Cimarron National Grassland, Kansas, USA, between 1985 and 2020.

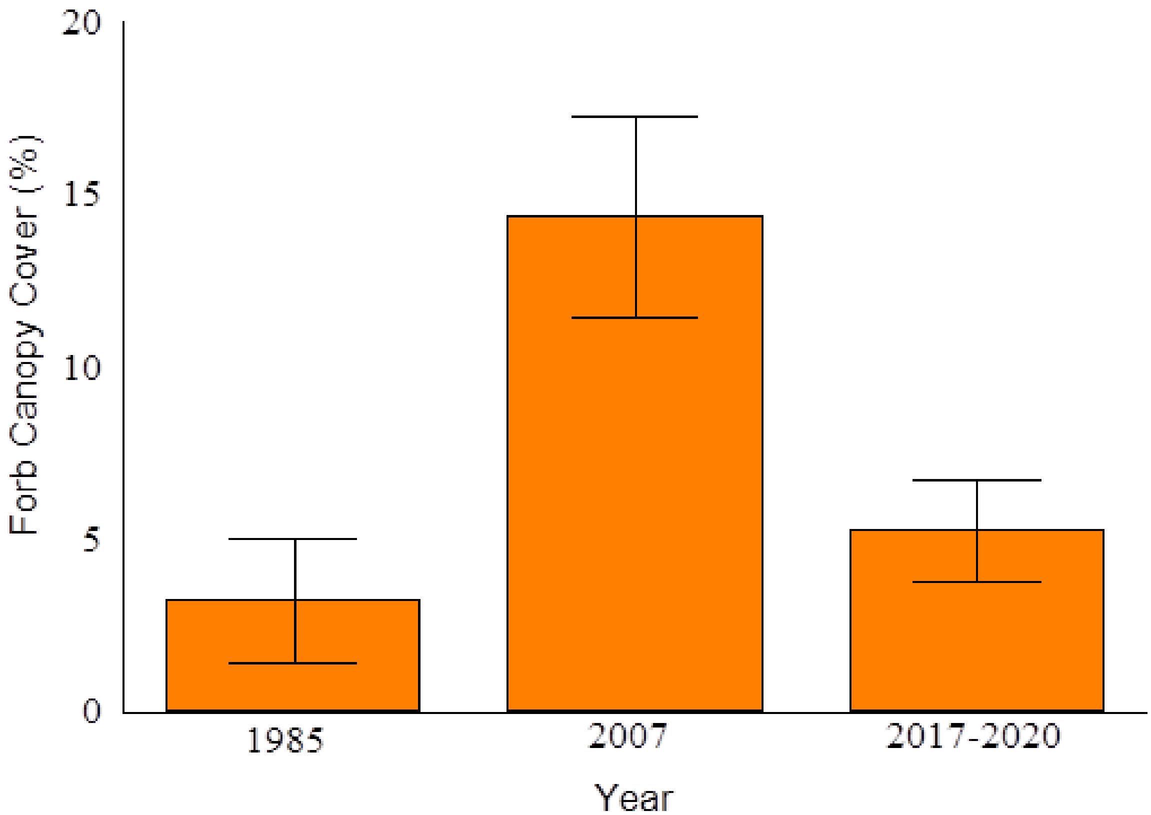

Figure 2.8. Average percent canopy cover and associated standard errors of forbs in the Mills pasture on Cimarron National Grassland, Kansas, USA, between 1985 and 2020.

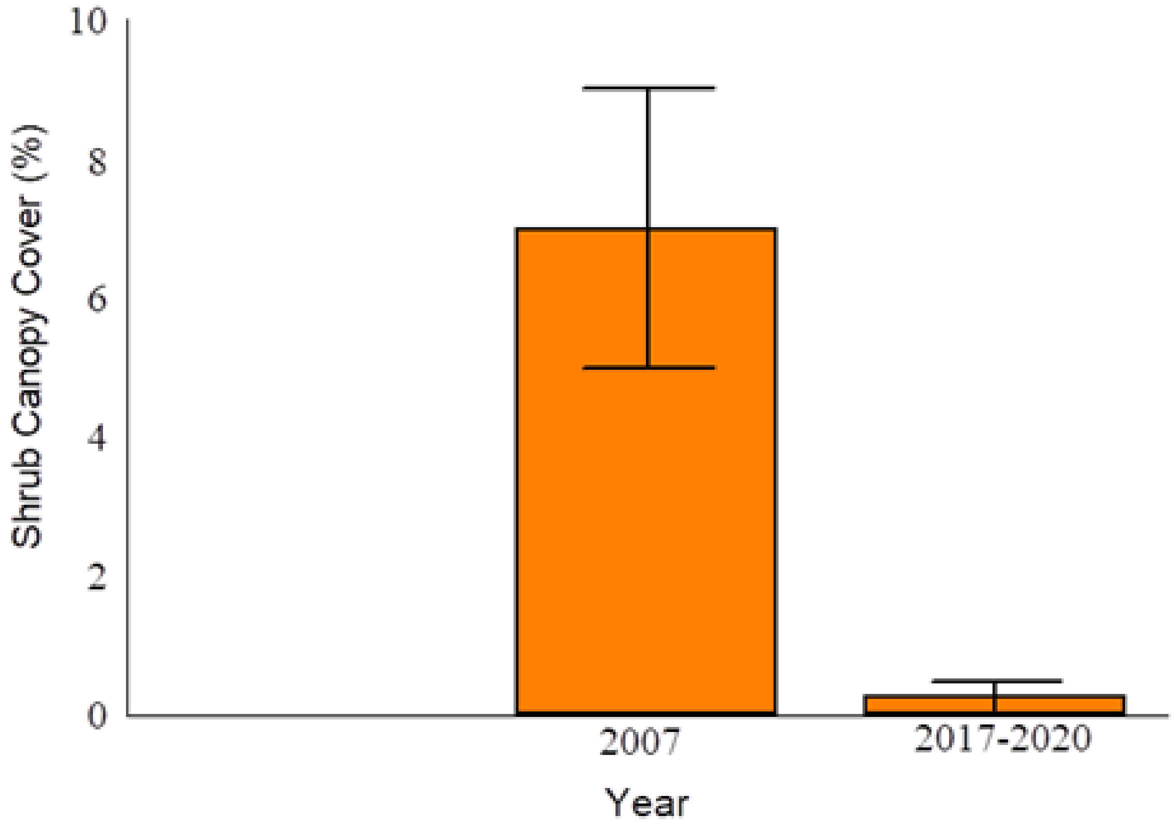

Figure 2.9. Average percent canopy cover and associated standard errors of shrubs in the Mills pasture on Cimarron National Grassland, Kansas, USA, between 1985 and 2020.

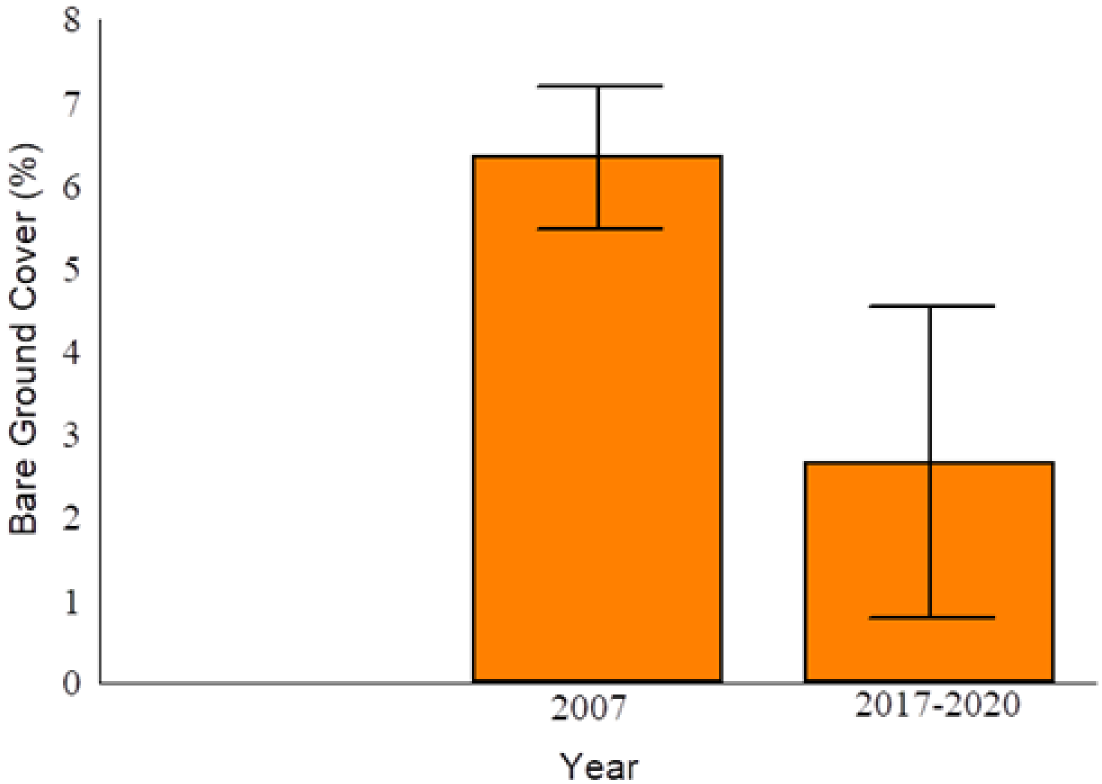

Figure 2.10. Average percent bare ground cover and associated standard errors in the Mills pasture on Cimarron National Grassland, Kansas, USA, between 1985 and 2020.

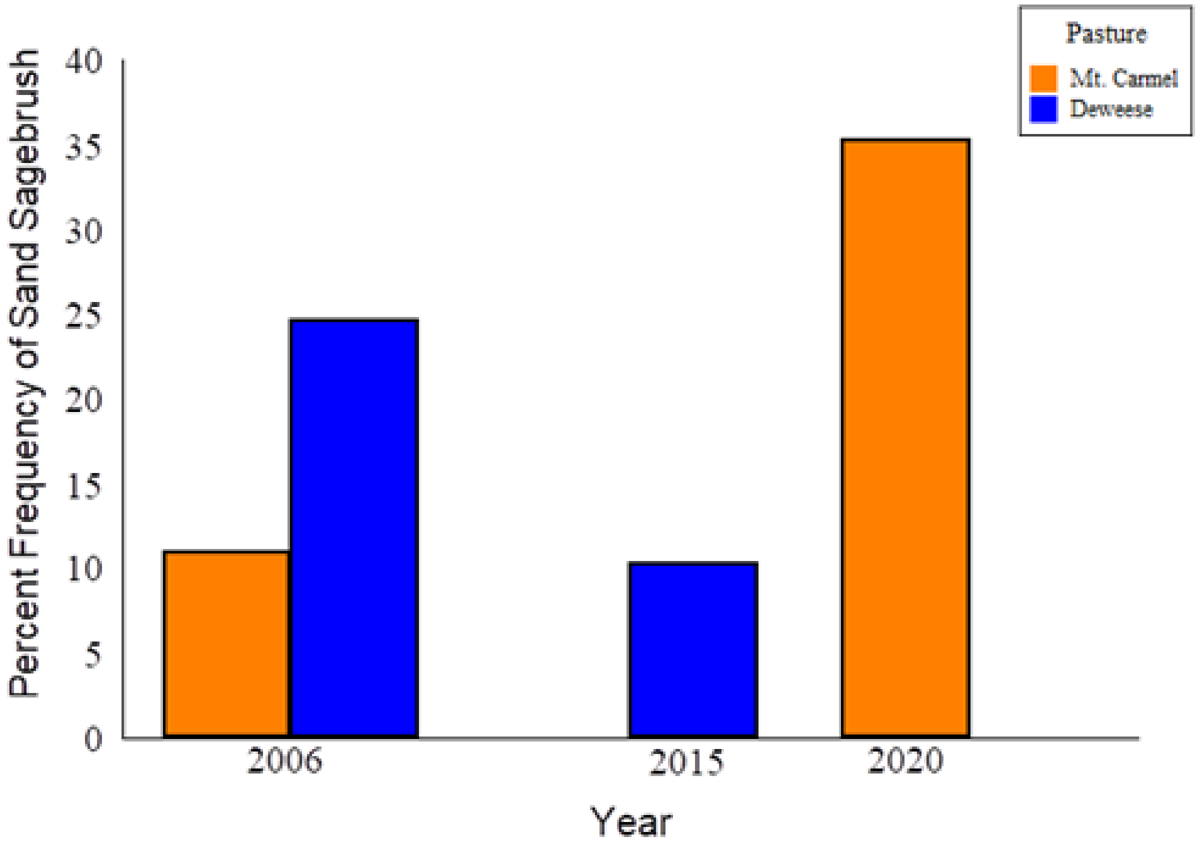

Figure 2.11. Proportion of observations with sand sagebrush on two pastures on Comanche National Grassland in Colorado, USA.

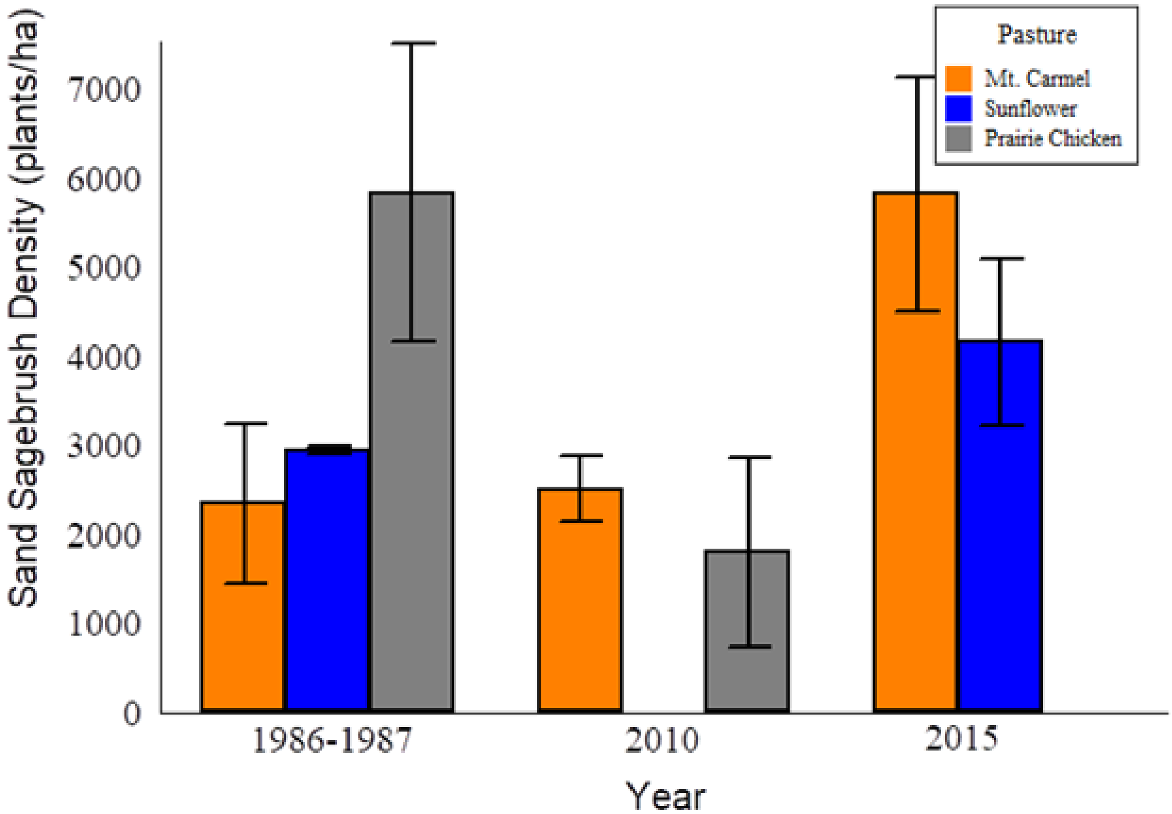

Figure 2.12. Average sand sagebrush density (plants/ha) and associated standard errors on three pastures on Comanche National Grassland, Colorado, USA.

Chapter 3 - Characteristics of Conservation Reserve Program (CRP) Fields Used by Lesser Prairie-Chickens in the Sand Sagebrush Prairie Ecoregion

Introduction

The lesser prairie-chicken (*Tympanuchus pallidicinctus*) is the iconic prairie grouse of the southwestern Great Plains (Boal and Haukos 2016). The species inhabits distinctive ecological sites within this semi-arid region, usually represented by sandy soils and mid- to tall grasses or native shrubs (e.g., sand sagebrush [*Artemisia filifolia*] and sand shinnery oak [*Quercus havardii*]) within the larger landscapes of short-grass or mid-grass prairie (Haukos and Zavaleta 2016, Spencer et al. 2017). Currently, lesser prairie-chickens occur in a patchy distribution among four distinct ecoregions (Short-Grass Prairie/CRP Mosaic, Mixed-Grass Prairie, Sand Sagebrush Prairie, and Sand Shinnery Oak Prairie) across five states (Kansas, Colorado, Oklahoma, Texas, and New Mexico, USA; McDonald et al. 2014; Figure 1.1).

For much of the past 50 years, the Sand Sagebrush Prairie Ecoregion of southwestern Kansas and southeastern Colorado supported the greatest density of lesser prairie-chickens among the range-wide ecoregions (Haukos et al. 2016). Garton et al. (2016) estimated >86,000 birds in the ecoregion during the 1970s; population estimates were relatively consistent from the mid-1970s through mid-late 1980s. However, populations have dramatically decreased since then, as the Sand Sagebrush Prairie Ecoregion currently has among the lowest density and total abundance of lesser prairie-chickens of any ecoregion. The estimated population of 3,083 in 2018 represented a >96% decline since the 1970s and 69% below the population objective of

10,000 for the ecoregion. Furthermore, populations declined again between 2018 and 2022, with only 1,713 estimated in 2022 (90% CI = 209, 3,681; Nasman et al. 2022).

From 2016-2019, following a period of above-average precipitation across the Sand Sagebrush Prairie Ecoregion, 411 lesser prairie-chickens were translocated from the Short-Grass Prairie/CRP Mosaic Ecoregion in northwest Kansas to a >125,000 ha release area in southwestern Kansas and southeastern Colorado (Aulicky 2020, Teige 2021, Berigan et al. 2022). Release sites included historical leks in the sand sagebrush prairie on U.S. Forest Service Comanche and Cimarron National Grasslands (~45,000 ha), with birds dispersing into surrounding private lands with native prairie, cropland, and interspersed U.S. Department of Agriculture's Conservation Reserve Program (CRP) land. Though the objective was to bolster local populations on the National Grasslands, >95% of 115 lesser prairie-chickens fitted with GPS transmitters translocated to the National Grasslands in 2018 and 2019 dispersed >5 km from their initial release sites within days of release (Berigan et al. 2022). While translocated birds initially created new leks and bolstered existing ones, male high counts at all leks decreased by approximately 43% by 2021 (Teige 2021).

Drivers of declines in lesser prairie-chicken population abundance and occupied range in the Sand Sagebrush Prairie Ecoregion are unclear. It is possible that increasing anthropogenic structures across the ecoregion may have contributed to downward population trends (Chapter 1). Additionally, there is evidence that vegetation on the Cimarron and Comanche National Grasslands, once a core area for lesser prairie-chickens, no longer provides wide-spread quality breeding habitat for lesser prairie-chickens including required vegetation visual obstruction and high-quality grass species (Chapter 2; Berigan et al. 2022, USFS 2014). Availability of quality nesting habitat is likely limited throughout the ecoregion. However, grassland composition and structure that satisfies lesser prairie-chicken nesting habitat requirements in the Sand Sagebrush Prairie Ecoregion may be found in CRP fields similar to other ecoregions (Sullins et al. 2018).

Many private grasslands in the Sand Sagebrush Prairie Ecoregion were historically converted to cropland (Haukos et al. 2016). However, the CRP was established in 1986, through which landowners could convert crop fields in highly erodible soils to perennial grasses in exchange for annual rental payments (Stubbs 2014). The CRP is recognized as providing habitat for grassland species including the lesser prairie-chicken (Stubbs 2014, Elmore and Dahlgren 2016). In fact, recent research suggests native and translocated lesser prairie-chickens in the Sand Sagebrush Prairie Ecoregion select CRP over public and private native grassland (Teige 2021, Berigan et al. 2022) and that CRP can provide critical cover for adults and nesting females particularly during drought (Sullins et al. 2018). The full extent of the influence of CRP on movements, home range establishment, and resource selection by lesser prairie-chickens is unclear. Determination of characteristics of CRP fields used by lesser prairie-chickens will provide initial insights into the role of CRP in population occupancy and abundance. There is evidence that CRP provides the vegetation composition and structure needed by nesting lesser prairie-chickens in a mosaic landscape (Sullins et al. 2018).

My objective was to compare characteristics of CRP fields used and apparently unused by lesser prairie-chickens, at both vegetation and landscape scales. I hypothesized that used CRP fields would have a greater proportion of CRP and grassland within 1 km (that is, in fields immediately surrounding the CRP), and apparently unused fields would have greater proportion of cropland within 1 km. I also hypothesized that used CRP fields would be larger than apparently unused fields, include a greater proportion of associated with at least 60% grassland within 5 km [Crawford and Bolen 1976]), and closer to release sites. I also hypothesized that used CRP fields would have greater mean visual obstruction reading (VOR) and grass and forb cover than apparently unused CRP fields.

Study Area

The Sand Sagebrush Prairie Ecoregion spanned 32,516 km^2 in 21 counties of southwest Kansas, southeast Colorado, and northern Oklahoma panhandle, including both public (U.S. Forest Service Cimarron and Comanche National Grasslands) and private lands. The latter were composed primarily of grazed working grasslands, CRP grasslands, and row-crop agriculture (Berigan 2019). Grazing intensity in the region was typically moderate to high, with grazing pressure on the Cimarron National Grassland having nearly doubled since the mid-1950s (Berigan et al. 2022.) Bands of sandy soils such as those running parallel to the Cimarron and Arkansas River drainages, characteristic of the sand sagebrush prairie, supported the eponymous sand sagebrush shrubs as well as mid- to tall grasses including big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), sideoats grama (*Bouteloua curtipendula*), and sand dropseed (*Sporobolus cryptandrus*; Haukos et al. 2016). Short-grass prairie species, supported in clayey and loamy soils, included blue grama (*Bouteloua gracilis*) and buffalograss (*B. dactyloides*; Berigan et al. 2022).

Common forbs in the ecoregion included annual buckwheat (*Eriogonum annuum*), blazing star (*Liatris* spp.), western ragweed (*Ambrosia psilostachya*), prairie sunflower (*Helianthus periolaris*), annual sunflower (*H. annuus*), Russian thistle (*Salsola tragus*), camphorweed *(Heterotheca subaxillaris),* Indian blanket flower (*Gaillardia pulchella*), tansy aster (*Machaeranthera tanacetifolia*), buffalo bur (*Solanum rostratum*), buffalo gourd (*Cucurbita foetidissima*), wax goldenweed (*Grindelia papposa*), prickly lettuce (*Lactuca serriola),* and marestail (*Conzya canadensis*; Berigan et al. 2022).

Newly enrolled CRP fields in Kansas and Colorado were initially seeded with a native grass mixture; seed mixtures did not include forbs until later years (Rodgers 2016). Historically, CRP plantings in Kansas were limited to native warm-season mid- and tall grass species such as little bluestem, sideoats grama, and switchgrass (*Panicum virgatum*). Other grass species in CRP throughout the ecoregion included big bluestem, western wheatgrass (*Pascopyrum* spp.), blue grama, and Indiangrass (*Sorghastrum nutans*). Forbs in CRP fields included alfalfa (*Medicago sativa*), sweet clover (*Melilotus* spp.), Maximillian sunflower (*Helianthus maximiliani*), prairie bundleflower (*Desmanthus illinonensis*), purple prairie-clover (*Dalea purpurea*), and upright prairie coneflower (*Ratibida columnifera*; Berigan at al. 2022).

The Cimarron and Comanche National Grasslands (CCNG) consisted of former pasture and croplands largely restored to sand sagebrush prairie after the prolonged drought and desolation of the 1930s Dust Bowl (Guest 1968, Olson 1997, Berigan et al. 2022). The CCNG encompassed 45,300 ha of the Sand Sagebrush Prairie Ecoregion in Kansas and Colorado, comprising the largest parcel of public lands in the lesser prairie-chicken range (Elmore and Dahlgren 2016, Berigan 2019). The CCNG was managed under a multi-use strategy including livestock grazing, recreation, and energy development. Grazing intensity in the region was typically moderate to high, with annual grazing pressure on the Cimarron having nearly doubled since the mid-1950s (Guest 1968, Berigan et al. 2022).

Average annual precipitation for Elkhart, Kansas, in Morton County from 1985-2020 was 43.8 cm, varying from 24.2 cm to 67.6 cm (data from High Plains Regional Climate Center 2021). Drought in this region occurred on a roughly 5-year cycle (Haukos et al. 2016).

Methods

I compared landscape-scale and field-scale characteristics between “used” and “apparently unused” CRP fields in which vegetation surveys had been conducted via random sampling during the study of habitat use by translocated lesser prairie-chickens with GPS transmitters (Teige 2021). I defined “used” fields as those in which GPS points representing translocated bird locations overlapped with identified CRP fields. I defined fields without GPS bird points as “apparently unused”. Of the 79 randomly sampled fields, I identified 64 used and 15 apparently unused fields. I focused on configuration and juxtaposition of three cover categories at the landscape scale: cropland, grass/pasture (non-CRP, non-public grass), and CRP. At the within-CRP field (finer) scale, I assessed differences in percent canopy cover of grass, forbs, shrubs, and bare ground, as well as VOR.

Data sources

GPS locations

Lesser prairie-chickens were captured with permission on private lands in Gove, Lane, Ness, and Finney counties in the Short-Grass Prairie/CRP Mosaic Ecoregion of Kansas during fall 2016 and spring 2017 – 2919 (Berigan et al. 2022). A subset of 279 birds were fitted with very-high-frequency (VHF) transmitters, and another 115 fitted with GPS transmitters, before being released at 5 sites across Cimarron and Comanche National Grasslands (Berigan et al. 2022).

I obtained locations of all GPS-marked translocated lesser prairie-chickens (Teige 2021, Berigan et al. 2022). I cross-referenced GPS bird locations with the record of bird mortalities, and eliminated locations that occurred after the mortality detection date. Where records were unclear or I could not find the specific bird entry, I eliminated points that were clearly repetitive in a long series that indicated the transmitter had been stationary for an extended period (i.e., the bird was dead or the transmitter slipped), leaving a total of 72,617 used locations in the Sand Sagebrush Prairie Ecoregion.

CRP

I used a 2014 CRP shapefile (US Farm Service Agency) to initially define CRP fields. In order to create an updated CRP layer for 2018, I cross-checked the 2014 polygons with a 2018 landcover raster compiled by Berigan (2019) from USDA-NRCS Ecological Site descriptions (USDA-NRCS) and USDA cropland data layers (USDA NASS 2019), which represented dominant landcover types including CRP, grassland, cropland, and Cimarron and Comanche National Grasslands. I also cross-checked the 2014 CRP fields with Landsat aerial imagery via Google Earth from 2018. Using the 2018 sources, I was able to manually add 2 additional CRP fields enrolled since 2018. This resulted in a CRP layer for the 79 CRP fields in which vegetation surveys occurred, which I used for my field-scale analysis. I would later merge boundaries between adjacent fields for landscape-scale analysis.

Field scale

Vegetation data

I obtained vegetation survey data collected between 2017-2020 via points (both point-paired and random) and point-step transects. Point-paired surveys occurred at randomly selected GPS locations within 300 m of a location used by GPS-marked individuals (Teige 2021). Random vegetation points, including 1 origin point per each of the 2, 250-m transects, were generated for each patch type throughout the study area (Berigan et al. 2022). Patch types were delineated based on USDA cropland data layers (USDA NASS 2019) or Ecological Site and plant community (Caudle et al. 2013, Berigan et al. 2022). For points, VOR was measured with a

Robel pole (Robel 1970) and canopy cover of grasses, forbs, shrubs, and bare ground was estimated with a Daubenmire frame (60 x 60 cm) in 4 cardinal directions (Daubenmire 1959). Field personnel measured VOR every 50 m along transects beginning at a randomly generated point.

Landscape scale

CRP field adjustment

I used Dissolve Boundaries in ArcGIS Pro to merge neighboring CRP fields (not separated by a road) from the field-scale analyses as those would likely be perceived as a single field by lesser prairie-chickens, resulting in creation of 56 fields for landscape-scale analyses. I used the Calculate Geometry Attributes tool in ArcGIS Pro to estimate total area for each field, for both unmerged CRP (field-scale metrics) and merged CRP (landscape-scale).

Landcover within 1 km

I created separate rasters for each cover type (i.e., cropland, grass/pasture, and CRP) from the compiled 2018 landcover raster (Berigan 2019). I attempted to create field boundaries that would translate best into a polygon form. I used Segment Mean Shift to identify misclassified pixels in the cropland raster, then used Reclassify to change the scattered isolated pixel group to reflect the dominant local cover type and essentially "clean up" the raster. I created polygon shapefiles from the rasters for each cover type.

I created 1-km buffers from the borders of each CRP field polygon, and Intersected landcover polygons with each CRP field-specific buffer. I summed the area (ha) of each landcover type from the Intersect output per cover type within a buffer, and cross-checked that the area for each landcover for that buffer matched the total acreage when added together.

Distance of CRP fields to release sites

I used the Near tool to calculate distance from nearest CRP field edge to each translocation release site location. I averaged distances from all release sites for each CRP field.

Percent grassland within 5 km

To identify proportion of grassland at the 5-km scale, I first created a new raster from the 2018 landcover raster compiled by Berigan (2019). The new raster was comprised of only Grass, CRP, and National Grassland categories, which were all designated as 1, while other cover type set to value of 0. I followed the same steps from Chapter 1, Appendix A for Focal Statistics, including keeping the default "Mean" setting, but checked "Ignore NoData" as my raster did not include a "cutout" around public lands that would need to be avoided during the analysis. The resulting raster showed the study area in terms of 5-km windows, each shaded to represent the average pixel value to depict landscape composition.

Using Extract by Mask, I obtained a raster of only the portions of the moving window raster that fell within CRP boundaries (that is, the overlap between the CRP polygon and the Focal Statistics output). I used Raster Calculator, specifying ">=.6", to identify how much of each field and its surroundings could be categorized as "≥60% grassland within 5 km". This resulted in a binary raster of fields identifiable by cell values 1 (≥60% grassland within 5 km) or 0 (areas < 60% grassland). In comparing used and apparently unused CRP, I used only fields that contained exclusively "1" or exclusively "0" cell values.

<u>Statistical analyses</u>

I calculated mean VOR and mean percent cover for each vegetation survey point from the four measurements at each cardinal direction for that point. I then used point- and transect-level averages associated with a particular CRP identifier to calculate field-level averages, doing so separately for points and transects. I defined experimental unit for comparisons as the individual CRP field.

I used a two-sample *t*-test assuming unequal variances to compare field area, landcover within 1 km, distance to release sites, VOR, and percent cover between the used and apparently unused CRP. I derived the proportion of each CRP category that was represented by ≥60% grassland within 5 km.

Results

A total of 1,698 survey points and 250 transects occurred across the set of 79 CRP fields I used for my field-level analysis. There were 79 fields in the original CRP grouping used for field-scale analysis, with 64 used and 15 apparently unused CRP fields (Table 3.1). There were 56 fields in the landscape-level subset after I dissolved borders of adjacent CRP polygons. The dissolved subset included 12 apparently unused and 44 used CRP fields (Table 3.2). I found few differences between used and apparently unused CRP at either the field or landscape scale.

Field scale

Mean area of used and apparently unused non-dissolved CRP fields was similar (Table 3.3). Mean VOR from point data was statistically similar between used and apparently unused CRP (Table 3.3). Mean percent cover of grass, forbs, shrubs, and bare ground was similar between used and apparently unused CRP (Table 3.3). However, mean transect VOR of used CRP was greater than apparently unused fields by 0.49 dm (Table 3.3).

Landscape scale

Mean area of fields and mean proportion of landcover types within 1 km were similar between used and apparently unused CRP fields (Table 3.4). Mean distance to translocation release sites was approximately 17 km greater for apparently unused CRP fields than used CRP fields (Table 3.4). Proportion of fields associated with ≥ 60% grassland was greater for used CRP fields (50%) than for apparently unused CRP fields (33%).

Discussion

Private native grasslands and croplands in the Sand Sagebrush Prairie Ecoregion largely fail to provide nesting habitat requirements for lesser prairie-chickens (Berigan et al. 2022). Although monitoring data of available lesser prairie-chicken habitat on the US Forest Service Cimarron and Comanche National Grasslands are limited (Chapter 2), recent findings suggest that the National Grasslands fall short of recommended vegetation metrics for nesting lesser prairie-chickens (Wuenschel 2016, Berigan et al. 2022). With its focus on native mid- and tall grasses that meet nesting thresholds and long-term contracts, CRP is potentially the strongest conservation option for the private-lands-dominated landscape in the western range of the lesser prairie-chicken (Sullins et al. 2018).

I found that VOR was greater in transect vegetation of used CRP fields relative to unused CRP. This, together with the lack of difference in canopy cover types between used and apparently unused CRP at the pasture scale, could indicate that structure had greater influence than composition in determining lesser prairie-chicken use. However, the finding of greater VOR within transect vegetation conflicts with the VOR results from random point vegetation sampling, which did not indicate any differences in VOR between used and apparently unused CRP. This inconsistency may be related to the lower sample size of transect vegetation, which also may have been a factor in the pronounced difference between transect used and apparently unused CRP. Interestingly, even the comparatively greater average VOR from used CRP transects at the pasture scale was still below the 2.7 dm recommended for nesting habitat (Pitman et al. 2005).

The significance of distance to release sites, with average distance from release sites being shorter for used CRP, could be interpreted in 2 ways. Release sites were located on or near CCNG existing or historical leks, with the assumption that these sites could be viewed as a proxy for nearness to grassland or nearness to a lek resulting in nearby settling by translocated lesser prairie-chickens. In that case, it could be inferred that CRP fields closer to CCNG form a larger landscape-scale "grassland" cover type area perceived by lesser prairie-chickens. However, use of CRP fields nearest the release site could be an artifact of dispersal behavior post-release; nearer CRP fields may be the most convenient or provide the closest habitat for dispersing translocated lesser prairie-chickens. Further context on birds' movements would indicate whether used CRP fields were in fact the first ones reached, or if movements extended beyond those fields but birds ultimately settled there.

The apparent lack of importance of CRP field area is not consistent with what is understood about lesser prairie-chicken populations requiring large, contiguous grasslands (Haukos and Zavaleta 2016). It has been suggested that CRP fields are more likely to be used when in close proximity with existing grasslands (Sullins et al. 2018). Although I dissolved boundaries between CRP fields sharing a single border for the landscape-scale analyses, I did not connect fields separated by roads. Previous research suggests that smaller county roads, such as those common in the study area, are not avoided by lesser prairie-chickens (Sullins et al. 2019). In that case, smaller roads might not be an obstacle for a lesser prairie-chicken, and there might be some "misidentified" fields in terms of total area.

The relatively greater proportion of used CRP fields being associated with ≥60% grass at a 5-km scale was consistent with previous research on lesser prairie-chicken use of CRP (Sullins et al. 2018). However, the similarity in proportion of cover types within 1 km between used and apparently unused CRP, as well as the low proportion (~10%) of grasslands at the 1-km scale was unexpected. Given the hypothesis that CRP fields may be more likely to be used when adjacent to other grasslands (Sullins et al. 2018), I had predicted that used CRP fields would have more grass and CRP cover within 1 km than unused fields. On average, used CRP buffers consisted of about 33% CRP. The lack of native grass immediately surrounding the subset of used CRP fields, and greater presence of surrounding CRP by comparison, potentially indicates lesser prairie-chickens are seeking the vegetation structure provided by CRP for nesting cover as a primary determinant for use during dispersal following translocation. It is possible that there is some effect at the 1-km scale for broods traveling from CRP fields, but that lesser prairie-chickens take cues for finding nesting cover from the 5 km landscape. Native pasture was not a factor determining use of cover types by translocated lesser prairie-chickens. Whether this apparent selection for CRP over native prairie (Sullins et al. 2018, Berigan 2019, Teige 2021, Berigan et al. 2022), as well as greater documented nest densities in CRP (Sullins et al. 2018), reflects broader benefits to lesser prairie-chickens at all life-history stages is not certain. While my preliminary results provide some additional support for the need to preserve or establish CRP with greater proportions of grassland at the 5-km scale, as well as those in close proximity to the CCNG, it will be necessary for subsequent analyses to expand to a full analyses of the scope of CRP effects on lesser prairie-chicken populations in the Sand Sagebrush Prairie Ecoregion.

Availability of CRP grasslands remains imperative for conserving contemporary lesser prairie-chicken populations in the western part of their range (Sullins et al. 2018). Based on the finding that nearly all translocated lesser prairie-chickens dispersed from the CCNG after release and exhibited selection for CRP (Teige 2021, Berigan et al. 2022), it is apparent that CRP currently provides quality nesting habitat in an otherwise cropland-dominated landscape of the

Sand Sagebrush Prairie Ecoregion. It is possible that implementing changes to land management on public and private grasslands, including adjusting grazing intensity and frequency (Guest 1968, Kraft et al. 2021), could improve nesting habitat for lesser prairie-chickens in the long-term. The Cimarron and Comanche National Grasslands should pursue adjustments to their grazing practices if they still consider lesser prairie-chickens a conservation priority, as once outlined in their management plan (USFS 2014). In the near-term, however, it is unlikely that the Cimarron and Comanche National Grasslands and associated private grasslands will provide sufficient quality nesting habitat for significantly increasing lesser prairie-chicken populations in the Sand Sagebrush Prairie Ecoregion. Despite the apparent success of CRP for nesting, the reliability of CRP as a conservation strategy is precarious. The loss of habitat as expiring CRP fields are converted to row crops, coupled with inevitable stress of extreme drought events, could be the final nails in the coffin for lesser prairie-chickens of the Sand Sagebrush Prairie Ecoregion. Pursuing financial incentives for CRP enrollment and re-enrollment could potentially address the problem of CRP declines. However, until more is known about the benefits of CRP beyond nesting, even abundant CRP likely cannot be considered a replacement for native prairie in conserving lesser prairie-chickens in the Sand Sagebrush Prairie Ecoregion.

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Table 3.1. Comparison of field-scale metrics ($\bar{x}$ ± SE), including area (ha), 100% visual obstruction reading (dm, VOR) for data collected on random transects and random points, and percent cover of grass, forb, shrub, and bare ground, associated with Conservation Reserve Program (*n* = 79) fields used and apparently unused by translocated lesser prairie-chickens with GPS transmitters during 2018-2021 in the Sand Sagebrush Prairie Ecoregion of southwestern Kansas and southeastern Colorado.

Field Metric	Used CRP	Apparently Unused CRP	t_{77}	*P*
Area	82.79 ± 5.13	93.22 ± 19.61	0.74	0.46
VOR Transect*	0.91 ± 0.10	0.42 ± 0.10	-2.23	0.03
VOR Point	0.75 ± 0.07	0.99 ± 0.20	1.39	0.17
Percent Grass	63.45 ± 2.45	72.15 ± 3.83	1.61	0.11
Percent Forb	9.36 ± 6.83	8.68 ± 1.89	-0.34	0.73
Percent Shrub	3.03 ± 0.35	2.64 ± 0.37	-0.48	0.62
Percent Bare Ground	10.90 ± 1.08	7.40 ± 2.11	-1.43	0.15

*DF = 31

Table 3.2. Comparison of landscape-scale metrics ($\bar{x}$ ± SE) associated with Conservation Reserve Program (n = 56) fields used and apparently unused by translocated lesser prairie-chickens with GPS transmitters during 2018-2021 in the Sand Sagebrush Prairie Ecoregion of southwestern Kansas and southeastern Colorado.

Landscape Metric	Used CRP	Apparently Unused CRP	t_{54}	P
Distance to Release Site (km)	33.83 ± 1.18	51.69 ± 4.75	5.38	<0.001
Area (ha)	132.83 ± 15.90	95.27 ± 25.36	-1.13	0.26
Percent Crop within 1 km	33.23 ± 3.30	40.49 ± 8.77	0.93	0.35
Percent Grass within 1 km	10.99 ± 1.82	11.02 ± 3.65	0.008	0.99
Percent CRP within 1 km	33.25 ± 2.45	38.17 ± 5.64	0.89	0.38

Appendix A - Landscape-Scale Analyses

<u>Calculating CRP from LCMAP rasters:</u>

In Raster Calculator for every pair of time steps, I used the Map Algebra expression: "([layer for year a]*1000) + [layer for year b])" to yield a history of change for land cover types. I created a landcover change raster for each time step ([bounded by 2 LCMAP rasters, 5 years apart]) in Raster Calculator in ArcGIS Pro, which yielded rasters with categories for "2002" (started as crops, or category "2", and remained as cropland through end of time step), "2003" (crop that changed to grass/shrub, or "3", cover type), etc. I did this to estimate change between consecutive time steps. Classifying the land cover history in this manner would identify which cover types remained the same, and which cover types changed during the 5 -year interval – with the objective of identifying specifically "2003" coded cover categories; that is, crop (2) that changed to apparent grass/shrub (3). For this reason, for each of the 2 landcover change rasters I created another Raster Calculator [binary raster] showing only "2003" (crop to new grass/forb) as 1, and everything that was not "2003" designated as 0. This would identify crop that was changed into grass at a given time step; that is, new CRP.

Next in Raster Calculator, I built the expression: " Con((("estimatedCRP_8590_2.tif" ==1) & ("Clip_LCMAP_CU_1995_V01_LCPRI_MinusPublic" ==3)) | ("binrasCRP9095" ==1),1,0)" to yield all grid cells identified as "2003" (or crop to grass; that is, CRP) from the previous time step that would also be identified in LCMAP as grass/shrub ("3") through 1995 – this finds all CRP but does not include expired fields turned back to cropland; in other words, a summary of non-expired CRP as of 1995. This expression also asks to include anything with code "2003" (crop that changed to grass; theoretically, CRP) in the next consecutive time step (here, 1990-95). This results in a product that includes all the previously identified, non-expired CRP, plus new enrollment.

To incorporate the newly estimated CRP pixels into the LCMAP raster categories, I built a statement to yield a raster made up of "2003" (CRP) pixels, with the LCMAP pixels in place of wherever there is "non CRP": Con("estimatedCRP_8590_2.tif" == 1 ,9,"Clip_LCMAP_CU_1990_V01_LCPRI_MinusPublic").

I summed the total number of pixels of each cover type of interest (i.e., cropland, prairie, CRP) for each time step and converted to hectares.

Estimating percent prairie:

I created a set of binary rasters (1=grass/shrub, 0=everything else, noData=NoData) based on using my “LCMAP [year] plus CRP, filtered” named series of layers. I ran each through Focal Statistics in ArcGIS Pro with the following parameters: circle, 5000, Map; and unchecked the box to “Ignore NoData in calculations”. I found it was necessary to uncheck “Ignore NoData”, otherwise the analysis would extrapolate over the “cut out” public lands. I plugged the Focal Statistics rasters into Raster Calculator specifying “[raster name for given year]>.3”, and separately, “[raster for that given year]>.6”, and recorded the count value associated with the “1” category for each raster. I identified only ≥ 30% and also ≥ 60%, but used Mosaic to New Raster to create a single raster of “0,1,2” categories for each time step.

Appendix B - National Grasslands Data

Table B 1. Vegetation metrics, including source and time period of collection, for pastures surveyed in multiple years on the U.S. Forest Service Cimarron National Grassland, Kansas, USA, and Comanche National Grassland, Colorado, USA

Location	Data Source	Time Period	Pasture	Mean VOR (dm)	SE	Mean % Grass Cover	SE	Mean % Forb Cover	SE	Mean % Shrub Cover	SE	Mean % Bare Ground Cover	SE	Sand Sagebrush density (plants/ha)	SE	% obs with high-value grasses	% obs with sand sagebrush
Comanche	Colorado Parks & Wildlife	1986-1987 only	Prairie Chicken	1.31	0.15	17.10	5.60	11.55	0.05	7.70*	2.30	71.35	5.75	5,820.00	1,660.00	NA	NA
			Mt Carmel	0.95	0.12	23.90	9.30	5.12	1.40	3.81*	1.02	70.88	3.00	2346.78	892.80	NA	NA
			Sunflower	1.23	0.45	18.60	1.90	1.25	0.15	3.30*	1.80	80.15	1.75	2940.0	37.42	NA	NA
	USFS	1998-2004, 2006	Prairie Chicken	0.91	0.15	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
			Mt Carmel	0.88	0.11	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	11.00**
			Deweese	0.97	0.06	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	24.60**
	Rondeau and Decker	2010	Prairie Chicken	0.81	0.19	63.20	6.28	0	NA	8.40	4.40	23.60	2.14	1,800.00	1,062.07	NA	NA
			Mt Carmel	1.32	0.15	51.94	3.04	3.29	1.30	11.03	1.48	24.45	1.55	2,509.68	375.12	NA	NA
	USFS	2015	Mt Carmel	0.24	0.03	39.22	2.96	10.83	1.50	30.77	3.70	14.80	1.97	5810.0	1308.55	1.79	0
			Sunflower	0.50	0.06	50.08	2.59	11.60	1.18	34.53	4.09	18.53	2.05	4150.0	930.97	1.72	0.06
			Deweese	0.75	0.12	41.48	2.56	13.38	1.20	35.16	2.27	14.21	1.67	12470.0	2045.81	4.68	10.27
	Kansas State University	2017-2020	Mt Carmel	0.93	0.21	43.64	4.25	9.54	1.85	15.53	2.99	27.11	2.99	NA	NA	0	35.29***

			Sunflower	0.44	0.11	54.47	6.79	13.18	3.59	6.02	2.22	18.04	2.97	NA	NA	0	5.36
Cimarron	USFS	1985	Mills	NA	NA	42.25	4.75	3.25	1.80	0.00	NA	0	NA	NA	NA	0	NA
		2007	Mills	NA	NA	69.00	4.73	14.33	2.91	7.00	2.00	6.33	1.86	NA	NA	0	NA
	USFS	1998, 2001, 2003	Steer, College, and HQ (C2)	1.19	0.16	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
			South Lowe	0.71	0.14	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
	USFS	2015	College	0.17	0.03	28.35	3.10	24.65	2.16	30.79	3.45	26.15	1.87	2580	NA****	NA****	NA****
			Wilburton	0.44	0.06	31.49	2.64	22.48	1.90	32.52	2.22	21.98	1.78	14770	NA****	NA****	NA****
			South Lowe	0.62	0.08	38.08	2.45	16.42	1.22	35.19	3.34	21.79	1.60	4530	NA****	NA****	NA****
			Steer	0.18	0.04									6000	NA****	NA****	NA****
	Kansas State University	2017-2020	Steer, College, and HQ	1.30	0.10	NA****	NA****	NA****	NA****	NA****	NA****	NA****	NA****	NA	NA	NA****	NA****
			Mills*	0.61	0.15	74.37	3.51	5.26	1.45	0.27	0.23	2.66	0.77	NA	NA	0	0
			Sandhills	1.42		39.65	NA****	26.59	NA****	8.65	NA****	9.73	NA****	NA	NA	0	10.88
			South Lowe	0.78	0.06	39.91	NA****	24.69	NA****	4.68	NA****	15.34	NA****	NA	NA	0.09	8.20

* sand sagebrush only

**2006 data only

***Point data only; based on nearest shrub

****Not calculated; not needed for analysis