Effectiveness Monitoring of the Lesser Prairie-Chicken Initiative and Conservation Reserve Program for Managing the Biodiversity and Population Size of Grassland Birds

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Vision: Native bird populations are sustained in healthy ecosystems

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3. **Stewardship** of birds and their habitats is a shared responsibility.

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1. Guide conservation action where it is needed most by conducting scientifically rigorous monitoring and research on birds and their habitats within the context of their full annual cycle.
2. Inspire conservation action in people by developing relationships through community outreach and science-based, experiential education programs.
3. Contribute to bird population viability and help sustain working lands by partnering with landowners and managers to enhance wildlife habitat.
4. Promote conservation and inform land management decisions by disseminating scientific knowledge and developing tools and recommendations.

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Executive Summary

The long-term population declines of grassland birds have elevated the recovery of the grassland avifauna to among the highest conservation priorities in North America. The lesser prairie-chicken (LEPC, *Tympanuchus pallidicinctus*) is a species of greatest conservation concern, yet several other grassland bird species show long-term population declines in the Great Plains. Because a large percentage of the southern Great Plains are privately owned, the recovery of the LEPC and other grassland bird species depends on conservation initiatives with strong partnerships between private landowners and resource professionals. The Conservation Reserve Program (CRP) and Natural Resource Conservation Service, Lesser Prairie Chicken Initiative (LPCI) are two programs used to manage the abundance and distribution of the LEPC and its habitat while promoting the overall health and long-term sustainability of farming and ranching operations. The overall conservation goal of this project is to integrate the conservation needs of the LEPC with those of other grassland bird species by evaluating practices that minimize the loss, fragmentation and degradation of grasslands, promote the overall health of grazing and restored lands, and improve the long-term sustainability of farming and ranching operations. The objectives are to 1) evaluate the effectiveness of LEPC conservation practices for increasing the site occupancy and biodiversity of grassland birds, 2) to understand the mechanisms involving relationships with landscape and local vegetation structure for increasing the site occupancy and biodiversity of grassland birds and 3) the effectiveness of the practices for increasing the density and population size of grassland birds.

We found native and introduced CRP plantings to restore agricultural lands are important conservation practices for increasing the biodiversity of grassland birds in the southern Great Plains. We found both native and introduced CRP plantings increased the species richness of grassland obligates relative to agricultural lands, whereas introduced CRP plantings increased the species richness of grassland generalists. Similar to findings of other studies, species richness was similar in native and introduced CRP plantings, but a shift in species composition indicated grassland obligates showed larger positive responses to native CRP relative to introduced CRP than generalist species.

Our findings indicated LPCI prescribed grazing to improve rangeland condition is an important practice for the biodiversity of grassland obligates and species currently experiencing population declines. Lands enrolled in LPCI prescribed grazing showed greater species richness of grassland obligates than grassland generalists, and the species richness of grassland generalists was lower on LPCI rangelands than reference grasslands. However, we found little evidence of greater species richness of grassland obligates on LPCI rangelands relative to reference grasslands. Nevertheless, LPCI prescribed grazing appeared to shift species composition toward a community of grassland obligates and species that are currently declining.

The study of landscape relationships suggested declining species and grassland obligates were more sensitive to the loss of grassland than the fragmentation of native vegetation. Nevertheless, several grassland obligates favored landscapes with large patch sizes of native vegetation. There was little variation in species richness along gradients of landscape composition or configuration. However, we observed greater variation in species composition along the gradient of landscape composition than the gradient of landscape configuration, suggesting the grassland bird community may be responding to the loss rather than the fragmentation of native vegetation. This result suggested implementing CRP in a way that maximizes the area of grassland may be a more effective conservation strategy than managing the patch configuration of native vegetation.

The study of local vegetation structure suggested CRP and LPCI prescribed grazing

practices that increase the ground cover of herbaceous vegetation play an important role in increasing the biodiversity of grassland birds. The species richness of grassland obligates and generalists increased with increasing herbaceous ground cover, but not grass height. Our results suggested land enrolled in CRP and LPCI prescribed grazing practices at the low-end of shrub cover and height provided important habitat for obligate grassland species of conservation concern, and LPCI rangelands with a substantial shrub component promoted the species richness of grassland generalists. The species richness of grassland generalists increased with shrub canopy cover and height, but there was little evidence for declining species richness of grassland obligates with increasing shrub cover and height.

We investigated habitat relationships for tree canopy cover and height to predict the responses of grassland bird species to LEPC management actions to deduce the encroachment of woodland vegetation. We found little evidence for variation in species richness along gradients of tree canopy cover and height. Nevertheless, we observed a shift in species composition with a greater number of declining species and grassland obligates occurring at low levels of tree canopy cover and tree height, and a greater number of declining generalists at high levels of tree canopy cover and tree height. Our results suggest tree removal may benefit several grassland obligates currently experiencing population declines, but may be detrimental to several declining grassland generalists.

We compared avian population densities on CRP plantings and LPCI prescribed grazing relative to reference grasslands and agricultural lands to better understand how the restoration practices affected the abundance of grassland bird species in the Action Area defined by the occupied range of the LEPC plus 16 km buffer. In addition, we studied avian densities on LPCI rangelands relative to reference grasslands and CRP plantings to better understand how prescribed grazing improved rangeland condition. We then investigated avian abundance to better understand the extent that local management contributes to regional populations of grassland birds. Our results indicate the voluntary conservation practices aimed at recovering LEPC populations on private land made meaningful contributions to the regional population sizes of several declining grassland species. In 2016, the CRP plantings and LPCI prescribed grazing practices accounted for 11% of the land area in the Action Area, and the practices made proportionally larger contributions to population size relative to availability for three grassland obligates. The practices conserved 17% of the Cassin's sparrow population ($\widehat{N}$ = 518,000) that was not declining in the Great Plains, 21% of the eastern meadowlark population ($\widehat{N}$ = 244,000) that was declining by 3%, and conserved 16% of the grasshopper sparrow population ($\widehat{N}$ = 1,625,000) that was declining by 2% in the Great Plains (Sauer et al. 2017). Introduced CRP plantings and LPCI prescribed grazing made proportionally larger contributions to population size for the Cassin's sparrow and eastern meadowlark, and native CRP plantings made proportionally larger contributions to population size for the grasshopper sparrow. The three conservation practices contributed to population size in proportion to availability (11%) for 7 of the 16 grassland obligates, and 14 of the 29 grassland generalists. Of these, 4 are declining grassland obligates, including the horned lark, lark bunting, northern harrier and western meadowlark, and 10 are declining grassland generalists, including the American kestrel, Brewer's blackbird, common yellowthroat, field sparrow, lark sparrow, loggerhead shrike, mourning dove, northern bobwhite, rufous-crowned sparrow and scissor-tailed flycatcher. In contrast, the practices showed proportionally lower contributions to population size for 12 species. Of these, only grassland generalists, including the common nighthawk, eastern kingbird, killdeer and red-winged blackbird are declining in the Great Plains. Finally, monitoring the effectiveness of the conservation practices may be useful for decision making to determine the combination of management actions that best satisfy wildlife objectives in the Action Area.

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Introduction

The long-term population declines of grassland birds has elevated the recovery of the grassland avifauna to among the highest conservation priorities in North America (Vickery and Herkert 2001, Brennan and Kuvlesky 2005). The lesser prairie-chicken (LEPC, *Tympanuchus pallidicinctus*), a grassland obligate that has experienced a 90% reduction in occupied range size since European settlement is of high conservation concern in the southern Great Plains (USFWS 2012). In 2014 the LEPC was listed as threatened by the Endangered Species Act (USFWS 2014b), however, subsequent court cases resulted in the species being delisted (USFWS 2016). According to the Breeding Bird Survey, several grassland obligate bird species show long-term population declines in the Great Plains (Sauer et al. 2017), including the eastern meadowlark (*Sturnella magna*), grasshopper sparrow (*Ammodramus savannarum*), horned lark (*Eremophila alpestris*), lark bunting (*Calamospiza melanocorys*), mountain plover (*Charadrius montanus*), northern harrier (*Circus cyaneus*) and western meadowlark (*S. neglecta*). In addition, several grassland generalists are also declining in the Great Plains (Sauer et al. 2017), including the American kestrel (*Falco sparverius*), Brewer's blackbird (*Euphagus cyanocephalus*), canyon towhee (*Melozone fusca*), common nighthawk (*Chordeiles minor*), common yellowthroat (*Geothlypis trichas*), eastern kingbird (*Tyrannus tyrannus*), field sparrow (Spizella *pusilla*), killdeer (*C. vociferous*), lark sparrow (*Chondestes grammacus*), loggerhead shrike (*Lanius ludovicianus*), mourning dove (*Zenaida macroura*), northern bobwhite (*Colinus virginianus*,), rufous-crowned sparrow (*Aimophila ruficeps*), red-winged blackbird (*Agelaius phoeniceus*) and scissor-tailed flycatcher (*T. forficatus*). Habitat loss, fragmentation and degradation are widely considered to be the primary threats to the population viability of the LEPC (Van Pelt et al. 2013, Haukos and Zavaletta 2016), and other grassland bird species (Ribic et al. 2009).

Because a large percentage of the southern Great Plains are privately owned, the recovery of the LEPC and other grassland bird species depends on conservation initiatives with strong partnerships between private landowners and resource professionals (Van Pelt et al. 2013). The Natural Resource Conservation Service (NRCS), Lesser Prairie Chicken Initiative (LPCI) was established in 2010 to increase the abundance and distribution of the LEPC and its habitat while promoting the overall health of grazing lands and the long-term sustainability of ranching operations (USFWS 2011, Van Pelt et al. 2013). The core LPCI conservation practice includes Upland Wildlife Management and Prescribed Grazing is a secondary core practice when livestock are present (USFWS 2011). The Conservation Reserve Program (CRP) is a voluntary program for agricultural producers administered by Farm Service Agency, addressing a threat to the LEPC from agricultural conversion by providing incentives to landowners to take cropland out of production and plant it back into grassland (Van Pelt et al. 2013). The U. S. Fish and Wildlife Service recently ruled the implementation of CRP is consistent with the long-term recovery goals of the LEPC (USFWS 2014a). The implementation of the Prescribed Grazing or CRP practices requires the development of grazing management or conservation plans, and the NRCS provides technical and financial assistance to private landowners through the Farm Bill (USFWS 2011, Van Pelt et al. 2013).

Habitat management for the LEPC will likely improve vegetation conditions for other grassland bird species (USFWS 2011, Haukos and Boal 2016), yet monitoring data are often necessary to establish the effectiveness of umbrella species conservation for increasing biodiversity (Favreau et al. 2006, Seddon and Leech 2008). Umbrella species are those requiring large areas of habitat, and the umbrella species concept assumes protection of the species' habitat simultaneously protects other, less spatially demanding species (Roberge and

Angelstam 2004, Favreau et al. 2006). Roberge and Angelstam (2004) suggested umbrella species protects beneficiary species by conserving a range of functional attributes or scarce resources. The evaluation of species responses to available conservation measures used to manage habitat for umbrella species provides a direct evaluation of the umbrella species concept (Roberge and Angelstam 2004). Because patterns of species co-occurrence vary across different spatial scales (Favreau et al. 2006), the umbrella species hypothesis may be best addressed using a hierarchical theory for community ecology (Whittaker et al. 2001). Applying a hierarchical model of community ecology to land management activities provides a framework for linking umbrella species conservation to biodiversity at multiple scales (Bestelmeyer et al. 2003). Effectiveness monitoring (Lyons et al. 2008) to determine the ability of LEPC conservation practices for increasing the biodiversity of grassland birds may ultimately be useful for evaluating the success of Farm Bill rangeland practices toward a program of evidence-based conservation (Briske et al. 2017). The treatment effects for the effectiveness of conservation practices can be integrated into decision making (Sauer et al. 2013) and adaptive management (Williams 2011) of the grassland bird community in the southern Great Plains.

The long-term conservation goal of this project is to integrate the conservation needs of the LEPC with those of other grassland bird species of conservation concern by evaluating practices that minimize the loss, fragmentation and degradation of grasslands, promote the overall health of grazing and restored lands, and improve the long-term sustainability of farming and ranching operations. The objectives are to determine 1) the effectiveness of LEPC conservation practices for increasing the site occupancy and biodiversity of grassland birds, 2) the influence of landscape and local vegetation relationships on the site occupancy and biodiversity of grassland birds and 3) the effectiveness of the practices for increasing the density and population size of grassland birds.

Methods

Study Area

The study took place within the Action Area defined by the occupied range of the LEPC plus a 16 km buffer (SGP CHAT 2011) in Colorado, Kansas, Oklahoma, New Mexico and Texas, 2015 - 2017 (Fig. 1). The Action Area occurred within portions of the Shortgrass Prairie BCR (BCR 18) and Central Mixed-Grass Prairie BCR (BCR 19), with a small portion of the study area occurring in the Chihuahuan Desert BCR (US NABCI Committee 2000a;b). We subdivided the Action Area by four ecoregions per the LEPC range-wide conservation plan: Sand Sagebrush Prairie; Shortgrass/CRP mosaic; Mixed Grass Prairie; and Sand Shinnery Oak Prairie (Van Pelt et al. 2013).

The Sand Sagebrush Prairie (SSPR) ecoregion (Fig. 1) is characterized by a sparse to moderately dense shrub layer dominated by sand sagebrush (*Artemisia filifolia*) interspersed within a sparse to moderately dense ground cover of tall, mid-, or short grasses (USFWS 2014c). Common grass species include sand bluestem (*Andropogon hallii*), sand dropseed (*Sporobolus cryptandrus*), prairie sandreed (*Calamovilfa longifolia*), giant sandreed (*C. gigantea*), needle and thread (*Hesperostipa comata*) and gramas (*Bouteloua* spp.). Other shrub species include soapweed yucca (*Yucca glauca*), and skunkbush sumac (*Rhus trilobata*).

The Shortgrass/CRP Mosaic Prairie (SGPR) ecoregion (Fig. 1) is a mixture of native shortgrass prairie and CRP grasslands planted with a mix of native warm season grasses (USFWS 2014c). Blue grama (*B. gracilis*) and buffalograss (*B. dactyloides*) are the dominant species, and sideoats grama (*B. curtipendula*), hairy grama (*B. hirsuta*), little bluestem (*Schizachyrium scoparium*) and western wheatgrass (*Pascopyrum smithii*) are also present.

The Mixed Grass Prairie (MGPR) ecoregion (Fig. 1) is primarily comprised of blue grama and buffalograss, with blue grama as the dominant species (USFWS 2014c). Other common plant species include sideoats grama, threeawns (*Aristida* spp.), sand dropseed, vine mesquite (*P. obtusum*), little bluestem, sand bluestem, Indiangrass (*Sorghastrum nutans*), switchgrass (*Panicum virgatum*), Canada wildrye (*Elymus canadensis*) and western wheatgrass. Shrubs such as sand sagebrush, shinnery oak (*Quercus havardii*), soapweed yucca, pricklypear (*Opuntia* spp.), winterfat (*Krascheninnikovia lanata*) and skunkbush sumac are also common.

The Sand Shinnery Oak Prairie (SOPR) ecoregion (Fig. 1) is composed of flat and rolling plains interspersed with shinnery oak and sand sagebrush, incuding little bluestem, sand bluestem, soapweed yucca, purple threeawn (*A. purpurea*), hairy grama, black grama (*B. eriopoda*), fall witchgrass (*Digitaria cognata*) and New Mexico needlegrass (*Stipa neomexicana*) (USFWS 2014c). Other common grassland species include giant dropseed (*S. giganteus*), broom snakeweed (*Gutierrezia sarothrae*), honey mesquite (*Prosopis glandulosa*),, tobosa (*Hilaria mutica*), catclaw mimosa (*Mimosa aculeaticarpa*) and collegeflower (*Hymenopappus flavescens*).

Study Species

We detected 45 bird species during the course of study (Appendix, Table A1) and classified the species as obligate (16) or facultative (29) grassland species according to Vickery and Herkert (1999), and Johnsgard (2009). Facultative species are those not entirely dependent on grasslands but use grasslands as a substantial part of their habitat requirements (Vickery and Herkert 1999). Because facultative species use a variety of vegetation types in addition to grasslands, we defined facultative species as grassland generalists in the current study. We queried species detections from the Integrated Monitoring in Bird Conservation Regions (IMBCR, Box 1) database and defined the species pool as 74 grassland species, including 24 obligates and 50 generalists (Vickery and Herkert 1999, Johnsgard 2009), known to breed in the Shortgrass Prairie, Central Mixed-Grass Prairie and Chihuahuan Desert BCRs (US NABCI Committee 2000a;b).

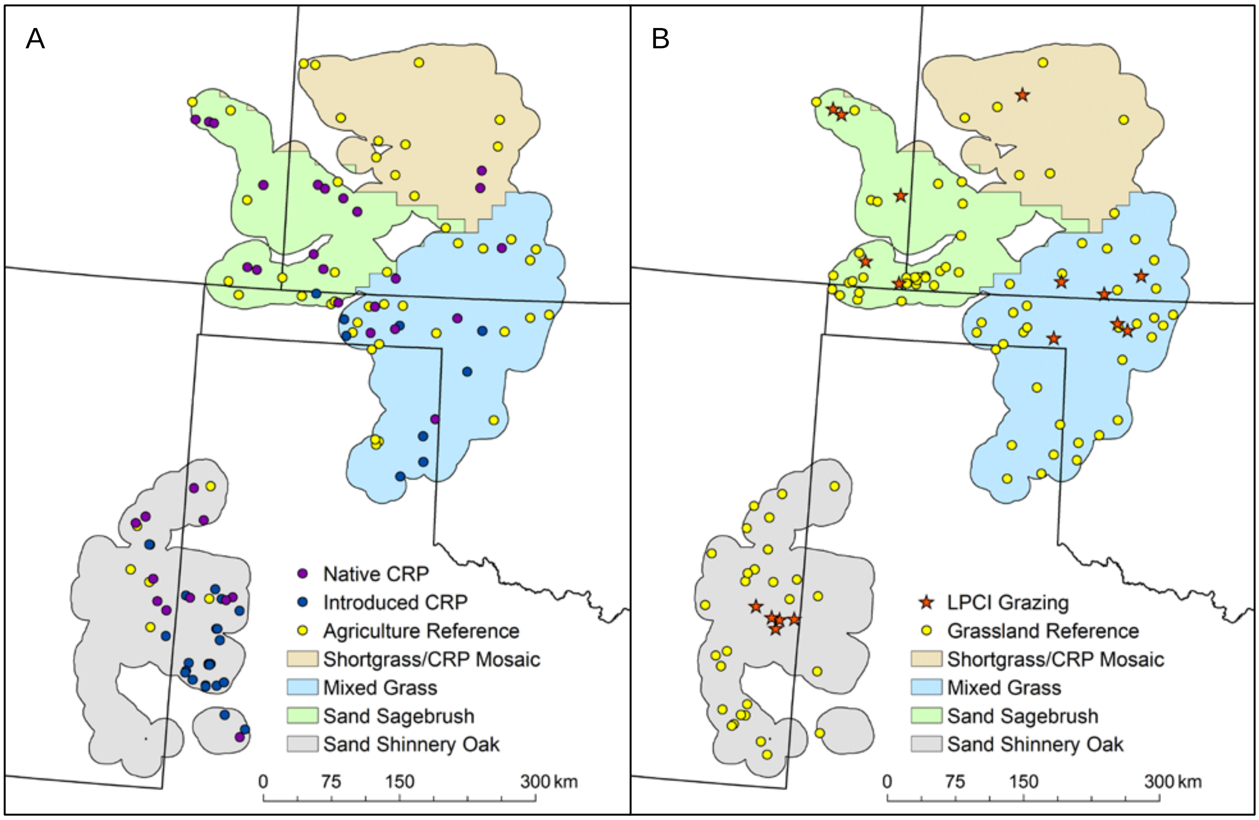

Figure 1. The location of (A) sampling grids for native Conservation Reserve Program (CRP) plantings (CP2), introduced CRP plantings (CP1) and reference agricultural lands, and (B) the general location of Lesser Prairie-Chicken Initiative (LPCI) prescribed grazing properties and sampling grids for reference grasslands within the Action Area defined by the occupied range of the lesser prairie-chicken (LEPC) plus a 16 km buffer, Colorado, Kansas, New Mexico, Oklahoma, and Texas, 2015 - 2017. Ecoregions from the LEPC Range-wide Conservation Plan are shown by the color-coded regions.

Box 1. Integrated Monitoring in Bird Conservation Regions

The Integrated Monitoring in Bird Conservation Regions (IMBCR) is a collaborative partnership between policy-makers, land managers, conservationists and scientists to leverage a common data platform over large spatial scales, promoting the efficient use of monitoring resources (Pavlacky et al. 2017). The Program was designed meet the North American Bird Conservation Initiative (NABCI) goals for improving avian monitoring and is well suited for addressing multiple management and conservation objectives (US NABCI Monitoring Subcommittee 2007). The IMBCR program uses modern sampling and analysis to provide reliable knowledge about bird populations (Pollock et al. 2002, Nichols et al. 2009). The design involves spatially balanced sampling (Stevens and Olsen 2004) to ensure representative geographic variation and data collection protocols to estimate population size and site occupancy while accounting for incomplete detection (Pavlacky et al. 2017). The spatially balanced properties of the design are maintained when sampling units are inaccessible and when sampling intensity varies between years. Accordingly, the spatially balanced design is well suited for regions with large amounts private land, when permission to access selected sampling units is denied, and is able to accommodate fluctuations in sampling intensity over time. The common data platform provides an economy of scale that allows pooling detection data across the monitoring region, allowing robust estimates of distribution and abundance in management units that have insufficient sample sizes on their own. The design of the IMBCR program provides an ecologically realistic framework for understanding hierarchical habitat use at local and landscape scales (Pavlacky et al. 2017), and the avian population metrics in local management units can be aggregated-up at multiple scales relevant to conservation and management objectives (Conroy et al. 2012).

Sampling Design

We developed an impact-reference design (Morrison et al. 2008) within the IMBCR program (Box 1, White et al. 2013, Pavlacky et al. 2017) for the Playa Lakes Join Venture (PLJV, Brennan and Kuvlesky 2005) to monitor avian community responses to treatments relative to reference lands within the Action Area (Fig. 1). The treatment strata included lands enrolled in introduced CRP plantings (CP1), native CRP plantings (CP2) and LPCI prescribed grazing, and the reference strata included random samples of grassland and agricultural lands from the IMBCR for PLJV program. The LPCI and CRP treatment strata were considered auxiliary or overlay strata because they are not integrated into the nested stratification of the base IMBCR program (White et al. 2013). Overlay projects utilize the IMBCR sampling design and field methods, and detection data from the base IMBCR program can be used in the analysis of the overlay projects.

We developed the sampling frame for reference lands in 2016 and 2017 using the base IMBCR for PLJV program (White et al. 2017, Woiderski et al. 2018). We subset the sampling frame for the IMBCR for PLJV program by superimposing the 1 km × 1 km U. S. National Grid (USNG, FGDC 2001) over the Action Area (SGP CHAT 2011) within a Geographic Information System (GIS; ArcGIS Version 10.1, Environmental Systems Research Institute, Redlands, CA). Twenty-six of the IMBCR for PLJV strata intersected the Action Area. In 2015, the sampling frame for reference grasslands comprised all grid cells within the Action Area containing ≥40% grassland or shrubland vegetation as mapped by the PLJV (2009) and Southwest Region Gap (Prior-Magee et al. 2007) spatial databases. We stratified the sampling frame for the reference lands by the SSPR, SGPR, MGPR and SOPR ecoregions (Fig. 1) from the LEPC Range-Wide

Conservation Plan (Van Pelt et al. 2013).

The sampling units for the IMBCR for PLJV design are defined by 1 km^2 grid cells, each containing 16 point count stations located 250 m apart and ≥125 m from the grid cell boundaries (White et al. 2013, Pavlacky et al. 2017). The IMBCR for PLJV program uses Generalized Random Tessellation Stratification (GRTS, Stevens and Olsen 2004) to select a spatially balanced sample. We post-stratified the point count plots by reference agriculture lands or reference grasslands using primary vegetation types collected at the point count locations (Pavlacky et al. 2018). The reference agriculture lands comprised the agricultural and rural primary vegetation type, and the reference grasslands for the site occupancy study comprised the grassland, shrub-land, and emergent wetland primary vegetation types (Fig. 1, Table 1). The plant species composition of the reference grasslands for the population density study included only the grassland primary vegetation type varied by ecoregion and is described above in the Study Area section.

Table 1. The sample sizes of 1 km^2 grid cells and 5 ha point count plots for the impact-reference design within the Action Area defined by the occupied range of the lesser prairie-chicken plus 16 km buffer, Colorado, Kansas, New Mexico, Oklahoma, and Texas, 2015 - 2017.

Impact-reference level	Grid cells			Point count plots		
	2015	2016	2017	2015	2016	2017
LPCI grazing lands	31	30	30	368	333	373
Reference grasslands	38	51	50	356	392	439
Native CRP plantings		33			293	
Introduced CRP plantings		33			322	
Reference agriculture lands	7	30	29	19	177	153

Conservation Reserve Program

The CRP program administered by the FSA plays a role in addressing habitat loss and fragmentation of the LEPC, and involves planting rangeland and critical areas in regions converted cropland to indirectly promote landscape connectivity (USFWS 2011, Van Pelt et al. 2013). We evaluated two CRP practices within the Action Area, including permanent introduced grasses and legumes (CP1) and permanent native grasses (CP2). From an extensive survey of CRP species composition in the LEPC range (Ripper et al. 2008), introduced CRP plantings (CP1) were dominated by two exotic warm-season grasses, weeping lovegrass (*Eragrostis curvula*) and old-world bluestem (*Bothriochloa ischaemum*), but also included native warm-season grasses such as big bluestem (*A. gerardii*), red three-awn (*A. purpurea*), sideoats grama, switchgrass and silver bluestem (*B. saccharoides*). Native CRP plantings (CP2) were dominated by native warm-season grasses such as sideoats grama, blue grama, switchgrass, sand dropseed, silver bluestem, big bluestem, little bluestem, tall dropseed (*S. compositus*), red three-awn, and the cool-season western wheatgrass, but also included exotic warm-season grasses such as cheatgrass (*Bromus* spp.), foxtail bristlegrass, weeping lovegrass and old-world bluestem (Ripper et al. 2008).

We used an auxiliary stratification scheme for the IMBCR for PLJV program to develop the sampling frame for CRP. We intersected the 1 km^2 USNG (FGDC 2001) and the 2015 U. S. Department of Agriculture Common Land Unit geospatial data (USDA 2014) within a GIS environment. Within the Action Area, we stratified the sampling frame for native CRP plantings (CP2) comprising all grid cells containing ≥40% land cover of native CRP and introduced CRP plantings (CP1) comprising all grid cells containing ≥40% land cover of introduced CRP. In addition, we stratified the sampling frame by BCR 18 and BCR 19 (BCR, US NABCI 2000a;b), resulting in 4 strata: native CRP-BCR 18; native CRP-BCR 19; introduced CRP-BCR 18; introduced CRP-BCR 19. We calculated the area of CRP occurring within the Action Area (80,511 km^2) as of 2016: native CRP-BCR 18 (1,711 km^2); native CRP-BCR 19 (852 km^2); introduced CRP-BCR 18 (242 km^2); introduced CRP-BCR 19 (152 km^2). We post-stratified the sampling frame by the SSPR, SGPR, MGPR and SOPR ecoregions (Fig 1.) from the LEPC Range-Wide Conservation Plan (Van Pelt et al. 2013).

We selected a spatially balanced sample of 1,200 grid cells from the native CRP (CP2) and introduced CRP (CP1) sampling frames using GRTS (Stevens and Olsen 2004) sample selection. In partnership with the Farm Service Agency (FSA), we mailed 1,430 Landowner Information Return Cards to the producers to ask permission to access the CRP lands. Of the 1,430 Return Cards, 105 producers granted permission to access the CRP lands. In 2016, we selected a spatially balanced sample of 33 introduced and 33 native granted grid cells in proportion to the areas of Shortgrass Prairie (BCR 18) and Central Mixed Grass Prairie (BCR 19) in the Action Area (Fig. 1, Table 1).

We calculated weighted means and Standard Errors (SE) of ground, shrub and tree cover variables for the vegetation types across years according to the area of the vegetation types in each of the strata in the Action Area (Table 2). We tested for differences between the vegetation variable means *i* by calculating effect sizes ($\hat{r}_i$) using the difference $\hat{r}_i = \hat{x}_{\mathrm{CRP}_i} - \hat{x}_{\mathrm{Ref}_i}$, where $\hat{x}_{\mathrm{CRP}_i}$ is the mean of vegetation variable *i* for CRP lands and $\hat{x}_{\mathrm{Ref}_i}$ is the mean of vegetation variable *i* for the reference category. We calculated the SD and 95% Confidence Intervals (CI) for the effect size using the delta method (Powell 2007) to evaluate statistical support for the effect sizes. When comparing CRP to reference grasslands, we found live grass ground cover ($\hat{r}$ = -9.5; SE = 0.8; CI = -11.0, -8.0), forb ground cover ($\hat{r}$ = -3.2; SE = 0.3; CI = -3.9, -2.4) and shrub canopy cover ($\hat{r}$ = -3.4; SE = 0.3; CI = -4.0, -2.7) were lower on native CRP plantings than reference grasslands (Table 2). In the same way, live grass ground cover ($\hat{r}$ = -11.5; SE = 0.7; CI = -12.9, -10.1), forb ground cover ($\hat{r}$ = -1.4; SE = 0.4; CI = -2.2, -0.7) and shrub canopy cover ($\hat{r}$ = -2.2; SE = 0.4; CI = -2.9, -1.4) were lower on introduced CRP plantings than reference grasslands (Table 2). Conversely, residual grass cover ($\hat{r}$ = 1.4; SE = 0.5; CI = 0.3, 2.5), residual grass height ($\hat{r}$ = 6.0; SE = 1.2; CI = 3.7, 8.3) and bare-litter ground cover ($\hat{r}$ = 14.8; SE = 0.8; CI = 13.1, 16.5) were greater on native CRP plantings than reference grasslands (Table 2). In a similar fashion, residual grass height ($\hat{r}$ = 5.3; SE = 1.1; CI = 3.1, 7.6) and bare-litter ground cover ($\hat{r}$ = 16.6; SE = 0.8; CI = 14.9, 18.3) were greater on native CRP plantings than reference grasslands, with less evidence for a difference in residual ground cover (Table 2).

When comparing CRP to agricultural lands, live grass cover and live grass height was lower on native and introduced CRP plantings than agricultural lands, and bare-litter ground cover was greater on CRP plantings than agricultural land. However, the cover and height of graminoid crops such as wheat and corn were classified as live grass in the field, and the bare-litter ground cover was primarily litter in the CRP plantings and primarily bare ground in the agricultural land (Table 2). Residual grass ground cover ($\hat{r}$ = 3.5; SE = 0.5; CI = 2.4, 4.5) and residual grass height ($\hat{r}$ = 4.2; SE = 1.2; CI = 1.8, 6.7) were greater on native CRP than agricultural land (Table 2). Likewise, residual grass ground cover ($\hat{r}$ = 1.8; SE = 0.5; CI = 0.8, 2.8) and residual grass height ($\hat{r}$ = 3.6; SE = 1.2; CI = 1.2, 5.9) were greater on introduced CRP than agricultural land (Table 2).

Table 2. The means and Standard Errors (SE) of ground, shrub and tree cover variables for point count plots classified as native Conservation Reserve Program (CRP), introduced CRP, agricultural land, Lesser Prairie-Chicken Initiative (LPCI) prescribed grazing and reference grassland, occupied range of the lesser prairie-chicken plus 16 km buffer, Colorado, Kansas, New Mexico, Oklahoma, and Texas, 2015 - 2017.

Vegetation variable	Native CRP		Intro. CRP		Ag. land		LPCI grazing		Grassland	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Grass cover (%)	10.31	0.50	8.33	0.40	13.39	0.39	13.20	0.27	19.80	0.56
Grass height (cm)	21.03	0.58	20.78	0.59	22.66	0.39	23.53	0.39	20.82	0.31
Residual cover (%)	9.50	0.47	7.85	0.44	6.04	0.20	7.81	0.21	8.10	0.26
Residual height (cm)	37.42	0.97	36.78	0.92	33.19	0.73	42.79	0.79	31.39	0.64
Forb cover (%)	2.96	0.23	4.69	0.25	4.62	0.24	5.24	0.18	6.12	0.25
Bare-litter cover (%)	76.74	0.65	78.56	0.63	74.80	0.44	67.98	0.45	61.94	0.53
Shrub cover (%)	0.80	0.09	2.00	0.19	1.35	0.08	10.72	0.42	4.15	0.30
Shrub height (m)	0.28	0.02	0.57	0.03	0.35	0.02	0.56	0.02	0.48	0.03
Tree cover (%)	0.02	0.01	0.54	0.16	0.25	0.04	0.62	0.11	0.36	0.06
Tree height (m)	0.11	0.05	0.79	0.12	0.63	0.08	1.13	0.11	0.56	0.06

When comparing native CRP to introduced CRP, forb ground cover ($\hat{r}$ = -1.7; SE = 0.3; CI = -2.4, -1.0), bare-litter ground cover ($\hat{r}$ = -1.8; SE = 0.9; CI = -3.7, 0.0), shrub canopy cover ($\hat{r}$ = -1.2; SE = 0.2; CI = -1.7, -0.7) and shrub canopy height ($\hat{r}$ = -0.3; SE < 0.1; CI = -0.4, -0.2) was lower on native CRP than introduced CRP (Table 2). In contrast, live grass ground cover ($\hat{r}$ = 2.0; SE = 0.6; CI = 0.7, 3.3) and residual grass ground cover ($\hat{r}$ = 1.7; SE = 0.6; CI = 0.3, 3.0) was greater on native CRP than introduced CRP.

When comparing LPCI prescribed grazing to reference grasslands, live grass ground cover ($\hat{r}$ = -6.6; SE = 0.6; CI = -7.9, -5.3) was lower on LPCI grazing lands than reference grasslands (Table 2). Conversely, live grass height ($\hat{r}$ = 2.7; SE = 0.5; CI = 1.7, 3.7), residual grass height ($\hat{r}$ = 11.4; SE = 1.0; CI = 9.4, 13.4), bare-litter ground cover ($\hat{r}$ = 6.0; SE = 0.7; CI = 4.6, 7.5), shrub canopy cover ($\hat{r}$ = 6.6; SE = 0.5; CI = 5.5, 7.6) and shrub canopy height ($\hat{r}$ = 0.1; SE < 0.1; CI = 0.0, 0.2) was greater on LPCI grazing lands than reference grasslands (Table 2).

When comparing LPCI rangelands to CRP plantings, residual grass ground cover ($\hat{r}$ = -1.7; SE = 0.5; CI = -2.7, -0.6) and bare-litter ground cover ($\hat{r}$ = -8.8; SE = 0.8; CI = -10.3, -7.2) was lower on LPCI rangelands than native CRP plantings (Table 2). In the same way, bare-litter ground cover was lower on LPCI rangelands than introduced CRP plantings ($\hat{r}$ = -10.6; SE = 0.9; CI = -12.4, -8.7), except there was little evidence for differences in residual grass ground cover (Table 2). Live grass ground cover ($\hat{r}$ = 2.9; SE = 0.6; CI = 1.7, 4.1), live grass height ($\hat{r}$ = 2.5; SE = 0.7; CI = 1.1, 3.9), residual grass height ($\hat{r}$ = 5.4; SE = 1.2; CI = 2.9, 7.9), forb ground cover ($\hat{r}$ = 2.3; SE = 0.3; CI = 1.7, 2.9), shrub canopy cover ($\hat{r}$ = 9.9; SE = 0.4; CI = 9.0, 10.8) and shrub canopy height ($\hat{r}$ = 0.3; SE < 0.1; CI = 0.2, 0.4) was greater on LPCI rangelands than native CRP plantings (Table 2). In a similar fashion, live grass ground cover ($\hat{r}$ = 4.9; SE = 0.7; CI = 3.5, 6.3), live grass height ($\hat{r}$ = 2.8; SE = 0.9; CI = 1.0, 4.5), residual grass height ($\hat{r}$ = 6.0; SE = 1.2; CI = 3.5, 8.5) and shrub canopy cover ($\hat{r}$ = 9.9; SE = 0.4; CI = 9.0, 10.8) was greater on LPCI rangelands than native CRP plantings, but there was little evidence for differences in forb ground cover and shrub canopy height (Table 2).

Prescribed Grazing

The LPCI prescribed grazing practice plays a role in addressing habitat degradation of the LEPC and is defined as managing the harvest of vegetation with grazing and/or browsing animals (USFWS 2011). The practice involves the management of stocking rates, rotation patterns, grazing intensity and duration, and includes an objective to meet nesting and brood rearing habitat requirements of the LEPC (Van Pelt et al. 2013). Recommendations for grazing management of LEPC nesting habitat in sand sagebrush vegetation include maintaining suitable vegetation structure for plant height >25 cm (western range) or >40 cm (eastern range), plant foliar cover >60 %, and shrub canopy cover ~15 % (Hagen et al. 2013). Recommendations for grazing management in shinnery oak vegetation include maintaining plant height >36 cm (western range) or >50 cm (eastern range), plant foliar cover >35 %, and shrub canopy cover ~20 % (Hagen et al. 2004).

We developed the sampling frame for LPCI prescribed grazing using an auxiliary stratification scheme for the IMBCR for PLJV program. We recruited landowners participating the LPCI prescribed grazing program within a partnership between the National Fish and Wildlife Foundation, and the NRCS state offices of Colorado, Kansas, Oklahoma and New Mexico. We intersected the 1 km^2 USNG (FGDC 2001) and project boundaries for 17 ranches enrolled in the LPCI prescribed grazing practice within a GIS environment (Fig. 1). The sampling frame for LPCI rangelands included all grid cells completely contained within the project boundaries of the 17 LPCI ranches. We stratified the sampling frame by the SSPR, SGPR, MGPR and SOPR ecoregions (Fig. 1) from the LEPC Range-Wide Conservation Plan (Van Pelt et al. 2013). Because the length of a typical LPCI grazing contract is five years, we calculated the area of LPCI prescribed grazing occurring within the Action Area (80,511 km^2) for years 2015, 2016 and 2017 as a five-year running total (U. S. Department of Agriculture, NRCS, unpublished report; Table 3).

Table 3. The five-year running total (km^2) of Lesser Prairie-Chicken Initiative prescribed grazing within the Action Area defined by occupied range of the lesser prairie-chicken, Colorado, Kansas, New Mexico, Oklahoma, and Texas, 2015 - 2017.

Ecoregion	2011-2015	2012-2016	2013-2017
Mixed Grass Prairie	634	800	940
Sand Sagebrush Prairie	146	281	319
Shinnery Oak Prairie	540	687	937
Shortgrass/CRP Mosaic	102	118	101
Total	1,422	1,887	2,296

In 2015, we selected a spatially balanced sample of 31 grid cells within the Action Area using GRTS (Stevens and Olsen 2004), and ensured ≥1 grid cell was selected in each of the 17 ranches (Fig.1, Table 1). In 2016 and 2017, we selected 30 grid cells from 16 ranches enrolled in LPCI prescribed grazing becasue one of the ranches unsubsribed from the program.

Data Collection

We sampled avian occurrence using 6 min point counts (Buckland 2006) between one-half hour before sunrise and 1100 h at each accessible point count location, and measured the distance to each bird detection using a laser rangefinder (White et al. 2013, Pavlacky et al. 2017). We binned the 6 min point count duration into three, two min time occasions in order to maintain a constant detection rate in each interval and ensure a monotonic decline in the detection frequency histogram through time (Pavlacky et al. 2012). Before beginning each 6 min point count, surveyors recorded vegetation data within a 50 m radius of the point rapid using ocular estimation. The vegetation data included primary vegetation type, and percent cover and mean height of trees and shrubs; as well as ground cover and grass height.

We measured 2 continuous covariates at the level of 1 km^2 grid cells using remotely sensed data to represent landscape configuration and composition. We quantified the mean patch size of native vegetation and land cover of shrub-land (Table 4) within a GIS environment using the LANDFIRE spatial data (USGS 2014). We measured 6 continuous covariates at the level of point count plots using data collected in the IMBCR for PLJV monitoring program to represent local vegetation structure. We quantified the canopy cover and height of ground, shrub and tree vegetation with 50 m radius of the point count location (Table 4). We measured a covariate representing the start time of each point count survey using data recorded in the IMBCR for PLJV program (Table 4). In addition to the continuous covariates, we considered a treatment factor effect with levels for native CRP (CP2), introduced CRP (CP1), LPCI prescribed grazing, agriculture reference and grassland reference, and an ecoregion factor covariate with levels for the SSPR, SGPR, MGPR and SOPR ecoregions. We standardized the continuous covariates using the *z*-transformation (Sokal and Rohlf 1981, Schielzeth 2010).

Table 4. The name, description, and mean and range of covariates for grid cells and point count plots within the Action Area defined by the occupied range of the lesser prairie-chicken, Colorado, Kansas, New Mexico, Oklahoma, and Texas, 2015 - 2017.

Covariate	Description	Mean (range)
Patch size	Mean patch size of native vegetation (km^2) within 1 km^2 grid cells, including grassland and shrub-land.	0.42 km^2 (0.00 km^2 - 1.00 km^2)
Shrub-land	Proportion (*P*) of shrub-land cover within 1 km^2 grid cells.	0.25 *P* (0.00 *P* - 1.00 *P*)
Herb cover	Percentage (%) of herbaceous live grass and forb ground cover within a 50 m radius of the point count locations.	18.6 % (0.0 % - 90.0 %)
Grass height	Mean height (cm) of live or residual grass cover within a 50 m radius of the point count locations.	36.8 cm (0.0 cm - 142.0 cm)
Shrub cover	Percentage (%) of shrub canopy cover within a 50 m radius of the point count locations.	6.8 % (0.0 % - 60.0 %)
Shrub height	Mean height (m) of shrubs within a 50 m radius of the point count locations.	0.46 m (0.00 m - 2.75 m)
Tree cover	Percentage (%) of tree canopy cover within a 50 m radius of the point count locations.	0.2 % (0.0 % - 40.0 %)
Tree height	Mean height (m) of trees within a 50 m radius of the point count locations.	0.45 m (0.00 m - 25.00 m)
Time	Start time (h) of the point count survey measured as fractional time.	6.98 h (4.73 h - 10.80 h)

Hypotheses and Model Justification

Site Occupancy and Species Richness

We used predictive models and the method of multiple working hypotheses (Chamberlin 1965) to evaluate *a priori* hypotheses for the effects of CRP plantings, LPCI prescribed grazing and vegetation structure on avian community structure at multiple scales. We defined the spatial scales using the grain and extent of ecological processes (Turner et al. 2001) operating in the southern Great Plains. The landscape scale corresponded to a grain of 1 km^2 grid cells, the local scale corresponded to a grain of 5 ha point count plots and the extent for both scales corresponded to the Action Area defined by the occupied range of the LEPC. We estimated the site occupancy of all grassland bird species in the community at landscape and local scales (Pavlacky et al. 2012), and estimated gamma species richness at the landscape scale and alpha species richness at the local scale (Whittaker et al. 2001, Bestelmeyer et al. 2003).

At the landscape scale, we used patterns of vegetation composition to evaluate hypotheses for the effects of native vegetation loss, and patterns of patch configuration to evaluate hypotheses about the effects of native vegetation fragmentation on the biodiversity of grassland birds (Fischer and Lindenmayer 2007). The landscape covariate for the proportion of shrub-land vegetation quantified the relative composition of shrub-land or grassland vegetation, and this covariate predicted biodiversity responses to the loss of shrub-land or grassland vegetation (Table 4). We hypothesized gamma species richness of grassland obligates would decline with increasing proportion of shrub-land vegetation in landscapes with low land cover of grassland vegetation. In contrast, we hypothesized species richness of grassland generalists would increase with increasing proportion of shrub-land vegetation in landscapes with high land cover of shrub-land vegetation. In general, we hypothesized community composition of grassland obligates would show negative occupancy responses, and grassland generalists would show positive occupancy responses to the proportion of shrub-land vegetation. The landscape covariate for the mean patch size of native vegetation quantified the extent of patch discontinuity, and this covariate predicted biodiversity responses to the fragmentation of native vegetation (Table 4). We hypothesized gamma species richness of grassland obligates would decline with increasing fragmentation of native vegetation, and hypothesized that species richness of generalists would be insensitive or increase with the fragmentation of native vegetation. In general, we hypothesized the community composition of grassland obligates would show negative occupancy responses to the fragmentation of native vegetation, and grassland generalists would show positive or stable occupancy responses to the fragmentation of native vegetation. In addition, we evaluated an ecoregion factor covariate with levels for the SSPR, SGPR, MGPR and SOPR regions (Fig. 1) from the LEPC Range-Wide Conservation Plan (Van Pelt et al. 2013). We hypothesized species richness of grassland obligates would be greatest in the SGPR ecoregion, which is most important to LEPC occupancy, followed by the MGPR, SOPN and SSPR ecoregions (Hagen et al. 2016).

At the local scale, we used the LEPC core conservation practices and patterns of vegetation structure to evaluate hypotheses for the effects of vegetation degradation and condition on the biodiversity of grassland birds (Fischer and Lindenmayer 2007). The comparison of CRP plantings and agricultural reference lands represent treatment effects for enrollment of cropland into the CRP program within the Action Area. We hypothesized the alpha species richness of grassland obligates would be greater on native than introduced CRP plantings (Bakker and Higgins 2009), and greater on native CRP plantings than agricultural reference lands. In contrast, we predicted the species richness of grassland generalists would be lower on native than introduced CRP plantings, and greater on introduced CRP plantings than agricultural reference lands. In general, we hypothesized community composition of grassland obligates would show positive occupancy responses to native CRP plantings, and grassland generalists would show positive occupancy responses to introduced CRP plantings. In addition, we predicted species richness and composition of native CRP plantings would be more similar to reference grasslands than species richness and composition of introduced CRP plantings because the dominant plants found in native CRP plantings were similar to those in the reference grasslands (Bakker and Higgins 2009).

The comparison of LPCI prescribed grazing lands to reference grasslands represent the departure of LPCI grazing lands from average grassland conditions in the Action Area. Because the LPCI prescribed grazing practice employed a rotational grazing system, we hypothesized grassland obligates would show greater alpha species richness on lands enrolled in prescribed grazing than reference grasslands (Derner et al. 2009). In contrast, we hypothesized the species richness of grassland generalists would be lower on lands enrolled in prescribed grazing than reference grasslands. In general, we hypothesized the community composition of grassland obligates would show positive occupancy responses, and grassland generalists would show negative occupancy responses to LPCI prescribed grazing.

We used local scale covariates for the canopy cover and height of herbaceous, shrub and tree vegetation to represent hypotheses for heterogeneity in grassland condition (Derner et al. 2009). We hypothesized that alpha species richness of obligates and generalists would increase with increasing cover and height of herbaceous ground cover. In general, we hypothesized the community composition of grassland species would vary according to the known habitat associations of the species (Knopf 1996). We hypothesized alpha species richness of grassland obligates would decline, and generalist grassland species would increase with increasing cover and height of shrubs and trees (Coppedge et al. 2001). Overall, we hypothesized the community composition of grassland obligates would show negative occupancy responses, and grassland generalists would show positive occupancy responses to the cover and height of trees and shrubs.

We accounted for the incomplete observation of avian species using covariates to explain temporal and spatial variation in the detection probabilities of species (Table 4). We hypothesized the year factor would explain differences in detection due to annual turn-over in the field crew and variable bird abundance in different years. We hypothesized the ecoregion factor covariate would explain spatial differences in detection due to variation in the geographic ranges and abundances of the species. The shrub cover and height covariates represented hypotheses that increasing shrub cover and height may interfere with the ability of the observers to detect the bird species. In addition, we hypothesized that increasing grass at the point count plot would inhibit the ability of the observers to detect birds. The time of day covariate represented the hypothesis that the singing frequency of bird species would decline later in the morning.

Density and Population Size

We hypothesized avian population densities on introduced and native CRP plantings relative to reference agricultural land would correspond to the habitat relationships of the species (Knopf 1996). For species requiring moderate to tall residual grass cover, such as the Cassin's sparrow (*Peucaea cassinii*), dickcissel (*Spiza americana*), eastern meadowlark, lark bunting and grasshopper sparrow, we predicted abundance would be greater on CRP plantings than agricultural land. For species preferring bare ground or agricultural ecotones, such as the brown-headed cowbird (*Molothrus ater*), horned lark, killdeer, mourning dove, red-winged blackbird, ring-necked pheasant (*Phasianus colchicus*), western kingbird (*T. verticalis*) we predicted abundance would be lower on CRP planting than agricultural lands. Because grass species richness is greater in native than introduced CRP plantings (Ripper et al. 2008), we predicted the abundance of grassland bird species requiring habitat heterogeneity, such as the eastern meadowlark, grasshopper sparrow (Hovick et al. 2015) and lark bunting, would be greater on native than on introduced CRP plantings.

We hypothesized avian population densities on LPCI rangelands relative to reference grasslands would correspond to the habitat relationships and sensitivity to livestock grazing of the species (Knopf 1996, Derner et al. 2009). For species requiring tall grass structure afforded by conservative grazing systems, such as the Cassin's sparrow, dickcissel, eastern meadowlark, lark bunting and grasshopper sparrow, we predicted abundance would be greater on LPCI rangelands than reference grasslands. In contrast, we predicted species requiring short grass conditions and greater habitat heterogeneity afforded by less conservative grazing, such as the horned lark and western meadowlark, would have lower abundance on LPCI rangelands than reference grasslands. Because shrub cover and height was greatest on LPCI rangelands, intermediate on reference grasslands, and lowest on CRP plantings, we predicted species requiring a shrub component, such as Cassin's sparrow, lark bunting, lark sparrow, northern bobwhite and scaled quail (*Callipepla squamata*) (Rodewald 2019), would be greater on LPCI rangelands than reference grasslands, and much greater on LPCI rangelands than CRP plantings.

Statistical Analysis

Site Occupancy and Species Richness

We extended the hierarchical Bayes multi-scale occupancy model of Mordecai et al. (2011) to accommodate multiple species (Dorazio and Royle 2005, Royle and Dorazio 2008) and two spatial scales (Pavlacky et al. 2012). For each species, we estimated the probability of large-scale occupancy (ψ) for grid cells, probability of small-scale occupancy (θ) for point count plots given presence at the grid cells, and probability of detection (p) in minute intervals given presence at point count plots (Pavlacky et al. 2012). We used a state-space formulation (Royle and Dorazio 2008) composed of two sub-models for partially observed processes of large-scale and small-scale occupancy and an observation model for repeated detections (Mordecai et al. 2011). The latent state z_{itk} is the estimated presence (z = 1) or absence (z = 0) of species i, year t and grid cell k, and the latent state u_{itkj} is the estimated presence (u = 1) or absence (u = 0) of species i, year t, grid cell k and point j. The observations y_{itkj} are the frequency of detections for species i, year t, grid cell k and point j using a removal design for 3, 2-minute time occasions (Pavlacky et al. 2012, MacKenzie et al. 2018). The state process model is comprised of two equations, one for the occupancy state of grid cells $z_{itk}|w_i \sim \text{Bernoulli}(\psi_{itk} w_i)$, where the latent variable w_i is explained below, and the other for the occupancy state of point count plots conditional on the occupancy of grid cells $u_{itkj}|z_{itk} \sim \text{Bernoulli}(\theta_{itk} z_{itk})$. The observation model for the frequency of detections $y_{itkj}|u_{itkj} \sim \text{Binomial}(p_{itkj} u_{itkj}, J_{itkj})$ is conditional on the occupancy state of point count plots, where J_{itkj} is the time occasion in which species i was first detected for year t, grid cell k and point j using a removal design (Pavlacky et al. 2012, MacKenzie et al. 2018). When a species was not detected, or when a species was detected on the last time occasion, J = 3.

We used a series of logistic regression equations to model the effects of treatment, vegetation structure, ecoregion and year on large-scale (ψ) and small-scale (θ) occupancy, and the effects vegetation structure, ecoregion and year on the probability of detecting the species (p):

$$\text{logit}(\psi_{itk}) = d_{0i} + d_{1i}x_{1k} + \cdots + d_{hi}x_{hk},$$
$$\text{logit}(\theta_{itkj}) = b_{0i} + b_{1i}x_{1kj} + \cdots + b_{hi}x_{hkj},$$
$$\text{logit}\left(p_{itkj}\right) = a_{0i} + a_{1i}x_{1kj} + \cdots + a_{hi}x_{hkj},$$

where d_{0i} is the random intercept, d_{li} is the beta coefficient of covariate x_l (l = 1, . . . , h) for the large-scale occupancy of species i, year t and grid cell k. The parameters b_{0i} and a_{0i} are the random intercepts, b_{li} and a_{li} are the beta coefficients of covariate x_l (l = 1, . . . , h) for the small-scale occupancy and detection, respectively of species i, year t, grid cell k and point j.

We used data augmentation to estimate the number of unobserved species in the community (Dorazio et al. 2006, Iknayan et al. 2014). The observed data comprised encounter histories for 45 species and we augmented the observed data with "all zero" encounter histories for 29 unobserved species known to breed in the region, resulting in a species pool of 74 grassland species. The latent state w_i is the estimated presence (w = 1) or absence (w = 0) of species i in the community of species. The state process for the membership of unobserved species in the community is $w_i \sim \text{Bernoulli}(\Omega_g)$, where Ω_g is the probability a species in the augmented data set is a member of the community of species that are present and vulnerable to detection (Dorazio et al. 2011) for guild g. We estimated the membership of species in the avian community by sharing information and accounting for correlation between the large-scale occupancy, small-scale occupancy and detection of the species (Dorazio et al. 2006, Iknayan et al. 2014).

We used the multivariate normal distribution to specify the variation and correlation of occupancy and detection probabilities among bird species (Dorazio et al. 2011),

$$\begin{bmatrix} a_{0i} \\ b_{0i} \\ d_{0i} \end{bmatrix} \sim \text{Normal}\left(\begin{bmatrix} \alpha_0 \\ \beta_0 \\ \delta_0 \end{bmatrix}, \begin{bmatrix} \sigma^2_{a_0} & \rho_{ab}\sigma_{a_0}\sigma_{b_0} & 0 \\ \rho_{ab}\sigma_{a_0}\sigma_{b_0} & \sigma^2_{b_0} & \rho_{bd}\sigma_{b_0}\sigma_{d_0} \\ 0 & \rho_{bd}\sigma_{b_0}\sigma_{d_0} & \sigma^2_{d_0} \end{bmatrix} \right),$$

where α_0 is mean detection, β_0 is mean small-scale occupancy, and δ_0 is mean large-scale occupancy among the i species. The parameters $\sigma^2_{a_0}$, $\sigma^2_{b_0}$ and $\sigma^2_{d_0}$ represent the variance of detection, small-scale occupancy and large-scale occupancy, respectively among the i species. The parameter ρ_{ab} estimates the correlation between detection and small-scale occupancy, and the ρ_{bd} estimates the correlation between small-scale occupancy and large-scale occupancy. The parameters a_{0i}, b_{0i}, d_{0i} represent the random intercepts for detection, small-scale occupancy and large-scale occupancy, respectively for species i.

In addition, we assumed the species-level beta coefficients were drawn from normal distributions for the 74 species in the community (Dorazio et al. 2006, Royle and Dorazio 2008). We defined the community-level random effects according to:

$$d_{li} \sim \text{Normal}\left(\mu_{d_l}, \sigma^2_{d_l}\right),$$

$$b_{li} \sim \text{Normal}\left(\mu_{b_l}, \sigma^2_{b_l}\right),$$

and

$$a_{li} \sim \text{Normal}\left(\mu_{a_l}, \sigma^2_{a_l}\right),$$

where μ is the mean and σ^2 is the variance for the l beta coefficients of large-scale occupancy (d), small-scale occupancy (b) and detection (a) for species i.

We estimated model parameters using Markov Chain Monte Carlo (MCMC) simulation implemented in program JAGS (Plummer 2003, JAGS Version 4.3.0, www.sourceforge.net, accessed 5 April 2018) using package jagsUI in the R statistical computing environment (R Version 3.4.3, www.r-project.org, accessed 5 April 2018). We used vague and weakly informative prior distributions for all estimated parameters (Dorazio et al. 2011):

$$\Omega_g \sim \text{Uniform}(0,1),$$
$$\rho_{ab},\ \rho_{bd} \sim \text{Uniform}(-1,1),$$
$$\alpha_0,\ \beta_0,\ \delta_0,\ \mu_{a_l},\ \mu_{b_l},\ \mu_{d_l} \sim t(\sigma,v),$$

and

$$\sigma_{a_0}, \sigma_{b_0}, \sigma_{d_0},\ \sigma_{a_l},\ \sigma_{b_l},\ \sigma_{d_l} \sim \text{half-Cauchy}(\alpha),$$

where the scale parameter $\sigma = 1.566$ and degrees of freedom $v = 7.763$ for the t-distribution. The scale parameter α for the half-Cauchy distribution has the probability density function $f(\alpha) = 2/[\pi(1+\alpha^2)]$, and we used the Student t-distribution prior approximation to the half-Cauchy distribution with mean $\mu = 0$, degrees of freedom = 1 and non-centrality parameter $\delta = 1$ (Dorazio et al. 2011). We generated 10,000 MCMC samples, specified a burn-in period of 5,000 iterations, and used $\widehat{R} < 1.1$ as an indication of model convergence (Gelman and Rubin 1992).

We estimated the parameters using the mean and standard deviation of the MCMC samples of the posterior distributions, calculated 95% Credible Intervals (CI) using the quantiles of the posterior distributions, and calculated Bayesian *P*-values for beta regression coefficients greater than or less than zero using posterior predictive distributions (Hobbs and Hooten 2015). The Bayesian CI indicated there was a two-tailed probability of 0.95, given the data, that true value of the beta coefficient fell within the credible region. The Bayesian *P*-values provided support for one-tailed hypotheses and represented the probabilities, given the data, that the beta regression coefficients were greater than [e.g., $P(b_{li} > 0)$] or less than [e.g., $P(b_{li} < 0)$] zero. We considered beta coefficients with *P*-values >0.90 as considerable support for the one-tailed hypotheses. The Bayesian *P*-value may be easier to interpret than the frequentist *P*-value, which represents the probability of obtaining a test statistic at least as extreme as the observed one, given the null hypothesis is true (Anderson et al. 2000)

Box 2. Species richness and composition at multiple-scales

Species richness is defined as the number of species in the community. And while species richness is an important aspect of biodiversity, the usefulness of species richness to conservation is limited by lack of attention to geographic scale and species composition (Whittaker et al. 2001). Alpha species richness represents the number of species at local scales, gamma richness corresponds to the number of species at landscape scales and epsilon richness represents the total number of species at regional scales. Beta species richness represents the turnover among species between the sampling locations at local or landscape scales (Whittaker et al. 2001). We did not report measures of epsilon and beta richness in the present study, but the approach easily accommodates these additional measures (Dorazio et al. 2011). Adopting a hierarchical framework for species richness allows the study of biodiversity along environmental gradients at multiple scales that are relevant to land management and species conservation (Bestelmeyer et al. 2003). Because bird species composition may shift in response management with no apparent changes in species richness, we developed a multi-species approach that estimates both species richness and composition from the bottom-up responses of individual species (Iknayan et al. 2014). In the present study, alpha species richness at the local scale corresponds to the average number of species among 5 ha point count plots. At the landscape scale, gamma richness represents the average number of species among 1 km^2 grid cells. This approach allowed us to simultaneously investigate biodiversity responses of the grassland bird community to environmental gradients operating at two spatial scales. In the present study, we investigated gamma richness along gradients of grassland loss and fragmentation at the landscape scale, and alpha richness in response to conservation practices and vegetation condition at the local scale.

We estimated gamma and alpha species richness for the 24 grassland obligates and 50 grassland generalists in the avian community. We derived estimates of species richness from the posterior MCMC samples (Hobbs and Hooten 2015) by summing the species occupancy estimates for each treatment and ecoregion, and along gradients of the continuous covariates while holding the other effects in the model constant at mean covariate values. We estimated gamma species richness at the landscape scale (Whittaker et al. 2001) by $\hat{\Gamma}=\sum_{m=1}^{M}\hat{\psi}_m$, where M is the number of m species in each guild and $\hat{\psi}$ is the estimate of large-scale occupancy for each species (Zipkin et al. 2009, MacKenzie et al. 2018), and this corresponds to the mean number of species among 1 km^2 grid cells (Box 2). We estimated alpha species richness at the local scale (Whittaker et al. 2001) by $\hat{\Upsilon}=\sum_{m=1}^{M}\hat{\psi}_m\hat{\theta}_m$, where M is the number of m species in each guild, $\hat{\psi}$ is the estimate of large-scale occupancy and $\hat{\theta}$ is the estimate of small-scale occupancy for each species, and this corresponds to the mean number of species among 5 ha point count plots (Box 2).

Density and Population Size

We used distance sampling theory to estimate population density while accounting for the decreasing probability of detecting individuals of a bird species with increasing distance from the observer (Buckland et al. 2001, Thomas et al. 2010). The detection probability is used to adjust the count of birds to account for birds that were present but undetected (Fig. 2). The detection function model [$g(y)$] for the y distance data is of the general form $g(y)=\frac{k(y)[1+s(y)]}{k(0)[1+s(0)]}$, where $k(y)$ is a parametric key function and $s(y)$ is a series expansion that may be used to improve the fit of the function if necessary (Buckland et al. 2001, Marques et al. 2007). The denominator ensures detection probability is one at distance zero [$g(0) = 1$]. We considered two key functions to model the detection data including the half-normal [$\exp\left(-y^2/2\sigma^2\right)$] and hazard-rate $\left\{1-\exp\left[-(y/\sigma)^{-b}\right]\right\}$ key functions. Both key functions have a scale parameter σ, which determines the rate at which the function decreases with increasing *y*, and the hazard-rate function has an additional shape parameter *b* (Buckland et al. 2001, Marques et al. 2007). The simple functional forms of the key functions may not adequately describe $g(y)$. Therefore, the shape of $g(y)$ can be adjusted by one or more series expansion terms and we considered the cosine term adjustment $\left[s(y)=\sum_{j=2}^{m} a_j \cos\left(j\pi y_s\right)\right]$, where *m* is the number of the *j* expansion terms and y_s are scaled values of the distance data *y* (Buckland et al. 2001, Marques et al. 2007). Application of distance sampling theory requires that five critical assumptions be met: 1) all birds at and near the sampling location (distance = 0) are detected; 2) distances to birds are measured accurately; 3) birds do not move in response to the observer's presence; 4) cluster sizes are recorded without error; and 5) the sampling units are representative of the entire survey region (Buckland et al. 2008).

Analysis of distance data includes fitting a detection function to the distribution of recorded distances (Buckland et al. 2001, Thomas et al. 2010). The distribution of distances can be a function of characteristics of the object (e.g., for birds, size and color, movement, volume of song or call and frequency of call), the surrounding environment (e.g., density of vegetation) and observer ability. Because detectability varies among species, we analyzed these data separately for each species. The development of robust density estimates typically requires 80 or more independent detections within the entire sampling area. We excluded birds flying over but not using the immediate surrounding landscape, birds detected while migrating (not breeding), juvenile birds and birds detected between points from analyses.

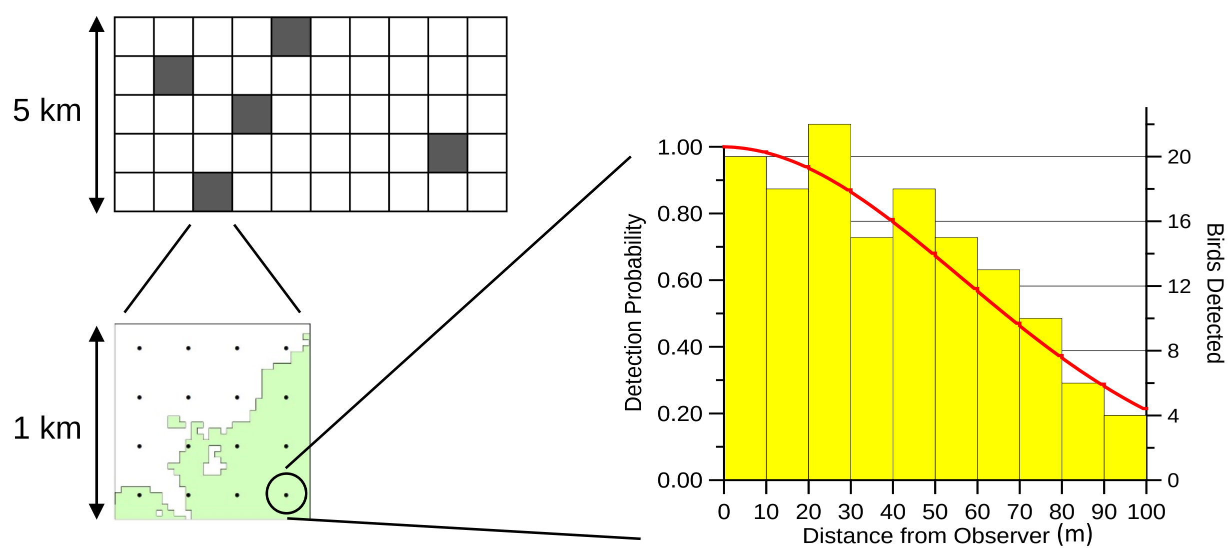

Figure 2. Distance sampling from the Integrated Monitoring in Bird Conservation Regions program, with grid cells nested within strata, point count plots nested within grid cells and distances nested within point count plots. The detection probability on the *y*-axis of the graph corresponds to the red-colored line for the detection function and birds detected on the *z*-axis corresponds to the histogram of the frequency of detections represented by the filled bars.

We estimated density for each species using a sequential framework where 1) year-specific detection functions were applied to species with greater than or equal to 80 detections per year ($n \geq 80$), 2) global detection functions were applied to species with less than 80 detections per year ($n < 80$) and greater than or equal to 80 detections over the life of the project ($n \geq 80$), and 3) remedial measures were used for species with moderate departures from the assumptions of distance sampling (Buckland et al. 2001).

We fit continuous models with no series expansions to all species and using the recommended 10% truncation for point transects. Truncating the largest 10% of the distance data shortened the tail and simplified the shape of the distributions, and this reduced the need to fit series expansions to accommodate distributions with long tails and complex shapes. For the year-specific detection functions, we fit Conventional Distance Sampling models using the half-normal and hazard-rate key functions with no series expansions (Thomas et al. 2010). For the global detection functions, in addition to the above models, we fit Multiple Covariate Distance Sampling models using half-normal and hazard-rate key function models with a categorical year covariate and no series expansions (Thomas et al. 2010). We selected the best detection function for each species using Akaike's Information Criterion adjusted for sample size (AIC_c, Burnham and Anderson 2002, Thomas et al. 2010) and considered the most parsimonious model as the estimation model.

We reviewed the highest ranking detection function for each species to check the shape criteria, evaluate the fit of the model and identify species with moderate departure from the assumptions of distance sampling (Buckland et al. 2001). First, we checked the shape criteria of the histogram to make sure the detection data exhibited a "shoulder" that fell away at increasing distances from the point. Second, we evaluated the fit of the model using the Kolmogorov-Smirnov goodness-of-fit test. Finally, we visually inspected the detection histograms to identify species that demonstrated evasive movement and/ or measurement errors. We looked for a type of measurement error involving the heaping of detections at certain distances that occurs when observers round detection distances. We also looked for histograms with detections that were highly skewed to the right, which may indicate a pattern of evasive movement (Buckland et al. 2001).

For species with moderate departures from the assumptions and shape criteria, we used two sequential remedial measures. First, we truncated the data to the point where detection probability was approximately 0.1 [$g(y) \sim 0.1$] and second order cosine series-expansion terms [$s(y)$] were applied to the half-normal and hazard-rate key function [$k(y)$] models {$k(y)\times[1+s(y)]$} to accommodate additional wiggle in the distance distributions (Buckland et al. 2001). We did not include detection function models with a single cosine expansion term because the half-normal and hazard-rate models require the order of the terms are > 1 (Buckland et al. 2001). Second, when the goodness-of-fit test and/ or inspection of the detection histogram continued to suggest evasive movement and/or measurement errors, we grouped the distance data into four to eight bins and applied custom truncation and second order expansion terms to the half-normal and hazard rate models. These remedial measures can ameliorate problems associated with moderate levels of evasive movement and/or distance measurement errors (Buckland et al. 2001).

We estimated the population density for LPCI prescribed grazing within 4 auxiliary strata defined by the ecoregions in the LEPC Conservation Plan (Fig. 1) and 4 CRP strata defined by introduced and native CRP, and BCR 18 and 19. We estimated population density in the Action Area using the 26 stratum-level estimates from the IMBCR for PLJV program that intersected the region (White et al. 2017, Woiderski et al. 2018). In addition, we post-stratified the point-count data (Thomas et al. 2010) to estimate population density for native grassland and agricultural land in 2016 and 2017. We used the post-stratified density estimates for the 26 strata intersecting the Action Area (Pavlacky et al. 2018) and used the proportion of post-stratified points to estimate the area of grassland or agricultural land within the region.

We estimated overall density and population size using a one-stage stratified random estimator weighted by the area of the resource in each stratum $\widehat{D} = \sum_{i=1}^{n} w_i \widehat{d}_i$, where $\widehat{D}$ is the aggregated density estimate, n is the number of strata, w_i is the proportion of area in stratum i and $\widehat{d}_i$ is the density estimate for stratum i (Thompson et al. 1998, Pavlacky et al. 2017). We estimated population size for the Action Area using the stratum-level density estimates from the IMBCR for PLJV program weighted by the area of the strata intersecting the region (White et al. 2017, Woiderski et al. 2018). We estimated population size ($\widehat{N}$) as $\widehat{N} = \widehat{D} \times A$, where $\widehat{D}$ was the estimated population density and A was the area of the resource of interest. We estimated the variance for the aggregated density and population size estimates using the delta method (Powell 2007), and calculated Satterthwaite 90% CIs for the estimates (Buckland et al., 2001).

We estimated effect sizes ($\widehat{\Delta}$) for differences in mean population density between the conservation treatments and reference lands during 2016 using $\widehat{\Delta} = \left(\widehat{D}_{\text{Trt}} - \widehat{D}_{\text{Range}}\right)$, where $\widehat{D}_{\text{Trt}}$ is the estimated population density (km^{-2}) for a conservation treatment and $\widehat{D}_{\text{Range}}$ is the estimated population density (km^{-2}) for agricultural lands or native grassland in the Action Area. We evaluated statistical support for the effect sizes by evaluating 90% CIs for the difference in the means relative to zero. We estimated the percent contribution of the conservation treatments to the regional bird population in the Action Area ($\widehat{\delta}$) using $\widehat{\delta} = \left(\widehat{N}_{\text{Trt}} / \widehat{N}_{\text{Range}}\right) \times 100$, where $\widehat{N}_{\text{Trt}}$ is the estimated population size of a conservation treatment and $\widehat{N}_{\text{Range}}$ is the estimated population size in the Action Area. We estimated the standard errors for the effect sizes and percent contribution using the delta method (Powell 2007, Pavlacky et al. 2017). We presented symmetric 90% CIs for the effect sizes and asymmetric logit CIs for the percent contribution to the population sizes. We considered the contribution to population size to be in proportion to the availability of CRP when the CI included the percentage of CRP implemented in the PLJV region.

Results

Site Occupancy and Species Richness

As outlined in the Hypotheses and Model Justification Section, the beta regression coefficients represented hypotheses for species and community responses to conservation practices and vegetation structure at two spatial scales. We presented 95% CIs to indicate the range of beta regression coefficients falling within the two-tailed credible region, and Bayesian *P*-values to indicate one-tailed hypotheses that represent the probabilities the beta regression coefficients were greater than or less than zero. We considered beta coefficients with *P*-values >0.9 as considerable support for the one-tailed hypotheses. When the CI covered zero and the $P > 0.9$, we inferred the beta coefficient was not different from 0 at 95% confidence, but the beta coefficient was greater or less than 0 with 90% confidence.

We presented species richness responses to the ecoregions in 2016 because presenting results by year would require additional space. Because the effects are additive, the relationships are similar in the other years. The alpha species richness of obligates on reference grasslands was greater in the Sand Sagebrush Prairie than in the Sand Shinnery Oak Prairie ($\beta = 0.96$, SD = 0.24, CI = [0.52, 1.50], $P > 0.99$) and Mixed Grass Prairie ($\beta = 0.62$, SD = 0.35, CI = [-0.17, 1.31], $P = 0.95$, Fig. 2A). The alpha species richness of generalists on reference grasslands was greater in the Mixed Grass Prairie than the Sand Shinnery Oak Prairie ($\beta = 1.40$, SD = 0.54, CI = [0.49, 2.55], $P > 0.99$) and Sand Sagebrush Prairie ($\beta = 1.40$, SD = 0.64, CI = [0.19, 2.72], $P = 0.99$, Fig. 2B). In addition, the alpha species richness of obligates was greater than generalists on reference grasslands in the Sand Sagebrush Prairie ($\beta = 1.01$, SD = 0.70, CI = [-0.47, 2.27], $P = 0.92$, Fig. 2).

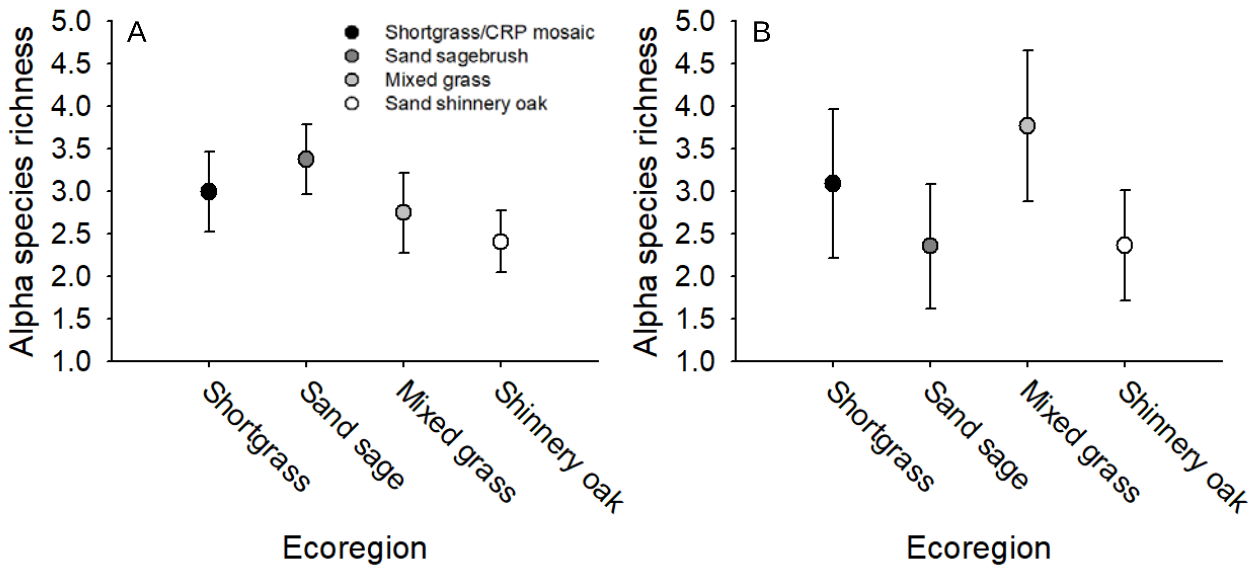

Figure 2. The alpha species richness of 5 ha point count plots for reference grasslands by ecoregion for grassland (A) obligates and (B) generalists within the Action Area defined by the occupied range of the lesser prairie-chicken, Colorado, Kansas, New Mexico, Oklahoma, and Texas, 2016. The round symbols represents the mean number of species among point count plots and the error bars are 1 standard deviation. The effects are additive and show parallel responses other years (not shown).

Species composition varied by ecoregion in the Action Area (Fig. 3, Table S1, available in Supporting Information). Several obligates showed high small-scale occupancy on reference grasslands in the Sand Sagebrush Prairie relative to the other ecoregions, including the Cassin's sparrow ($P \leq 0.99$), horned lark ($P \leq 0.99$), lark bunting ($P \leq 0.99$) and western meadowlark ($P \leq 0.99$, Fig. 3). Generalists showing low small-scale occupancy on reference grasslands in the Sand Sagebrush Prairie relative to the other ecoregions included the common nighthawk ($P \leq 0.98$), eastern kingbird ($P \leq 0.93$), killdeer ($P \leq 0.99$), lark sparrow ($P \leq 0.99$), red-winged blackbird ($P \leq 0.92$), scissor-tailed flycatcher ($P \leq 0.97$) and western kingbird ($P \leq 0.98$, Fig. 3). Generalists with greater small-scale occupancy on reference grasslands in the Mixed Grass Prairie relative to the Sand Sagebrush Prairie included the eastern kingbird ($P = 0.92$), killdeer ($P > 0.99$), lark sparrow ($P > 0.99$), red-winged blackbird ($P = 0.92$) and scissor-tailed flycatcher ($P = 0.97$, Fig. 3C).

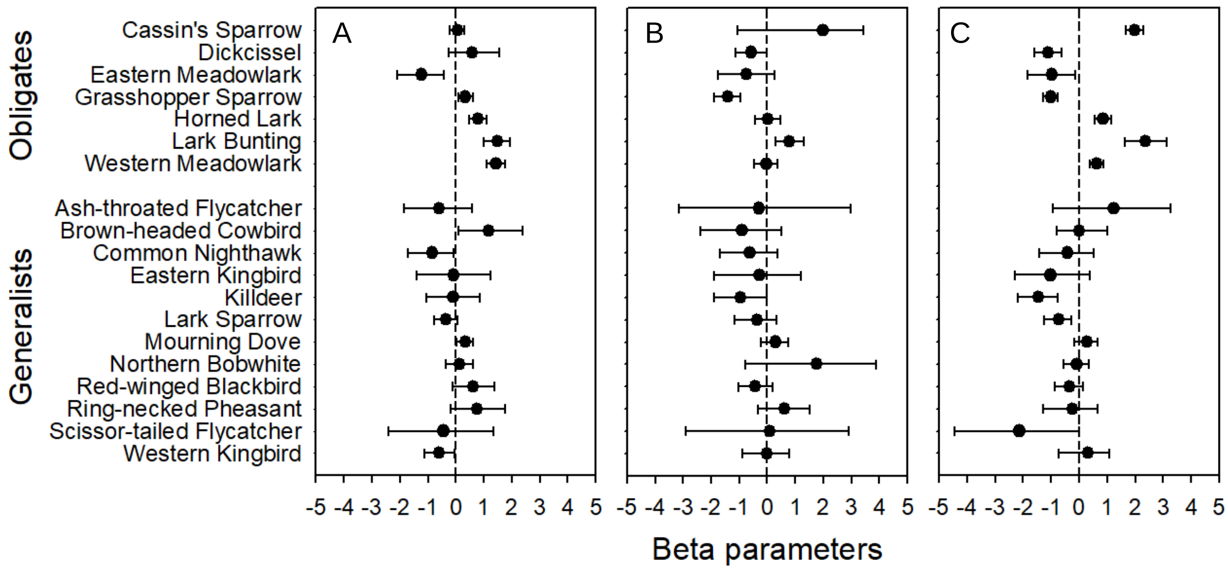

Figure 3. The beta coefficients for the small-scale occupancy of 5 ha point count plots for reference grasslands within the Sand Sagebrush Prairie relative to (A) Shinnery Oak Prairie ecoregion, (B) Shortgrass/CRP Mosaic and (C) Mixed Grass Prairie ecoregions for grassland obligates and generalists within the Action Area defined by the occupied range of the lesser prairie-chicken, Colorado, Kansas, New Mexico, Oklahoma, and Texas. The round symbols represent beta coefficients graphed relative to zero and the error bars are 95% credible intervals.

Conservation Reserve Program

We presented species richness responses to CRP for the Sand Shinnery Oak Prairie ecoregion in 2016 because presenting results by ecoregion and year would require a large amount of space. Because the effects are additive, the relationships are similar in the other ecoregions and years. We confirmed the hypothesis that alpha species richness of grassland obligates was greater on native CRP plantings ($\beta = 0.65$, SD = 0.17, CI = [0.28, 1.00], $P > 0.99$) and introduced CRP plantings ($\beta = 0.59$, SD = 0.16, CI = [0.28, 0.93], $P > 0.99$) than agricultural reference lands (Fig. 4A). In contrast, there was little support for the hypothesis that alpha species richness of obligates was greater on native CRP than on introduced CRP ($P = 0.73$, Fig. 4A), or the hypothesis that the alpha species richness of generalists was lower on native CRP than on introduced CRP ($P = 0.84$, Fig. 4A). We found evidence for greater alpha species richness of generalists in introduced CRP plantings relative to agricultural reference lands (β =

0.35, SD = 0.26, CI = [-0.14, 0.93], P = 0.92), but little evidence for a difference between native CRP plantings and agricultural lands (P = 0.56, Fig. 4A).

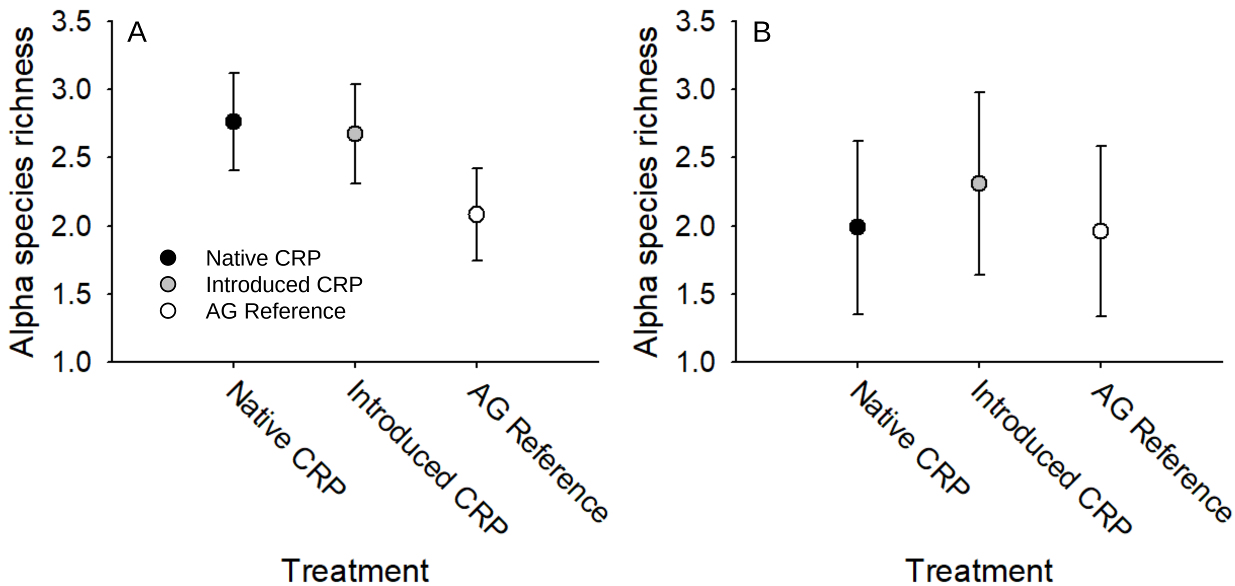

Figure 4. The alpha species richness of 5 ha point count plots for native Conservation Reserve Program (CRP) plantings, introduced CRP plantings and agricultural reference lands for grassland (A) obligates and (B) generalists within the Sand Shinnery Oak Prairie ecoregion, occupied range of the lesser prairie-chicken, New Mexico and Texas, 2016. The round symbols represents the mean number of species among point count plots and the error bars are 1 standard deviation. The effects are additive and show similar responses in the other ecoregions and years (not shown).

We found large shifts in species composition in the CRP treatments relative to agricultural lands within the Action Area (Fig. 5, Table S1, available in Supporting Information). The dickcissel (P > 0.99), eastern meadowlark (P > 0.99), grasshopper sparrow (P > 0.99), brown-headed cowbird (P = 0.98) and mourning dove (P > 0.99) showed greater small-scale occupancy on native CRP plantings than agricultural lands (Fig. 5A). The small-scale occupancy of the Cassin's sparrow (P > 0.99), eastern meadowlark (P > 0.99), grasshopper sparrow (P > 0.99), lark bunting (P > 0.99), mourning dove (P > 0.99) and northern bobwhite (P > 0.99) was greater on introduced CRP plantings than agricultural lands (Fig. 5A). In contrast, the common nighthawk (P ≤ 0.92), eastern kingbird (P ≤ 0.91), killdeer (P > 0.99), western kingbird (P ≤ 0.99), horned lark (P > 0.99) and red-winged blackbird (P > 0.99) showed greater small-scale occupancy on agricultural lands than the CRP plantings (Fig 5A, Fig.5B).

Although there was little evidence for differences in alpha species richness of obligates and generalists between the native and CRP plantings, there was strong evidence of shifts in species composition in the Action Area (Fig. 5C, Table S1, available in Supporting Information). The small-scale occupancy of the dickcissel (P = 0.97), grasshopper sparrow (P > 0.99), western meadowlark (P = 0.96) and brown-headed cowbird (P = 0.96) was greater on native CRP plantings than introduced CRP plantings (Fig. 5C). The Cassin's sparrow (P > 0.99), lark bunting (P = 0.98), killdeer (P = 0.91), mourning dove (P = 0.98), northern bobwhite (P = 0.99) and western kingbird (P = 0.94) showed greater small-scale occupancy on introduced CRP plantings than native CRP plantings (Fig. 5C).

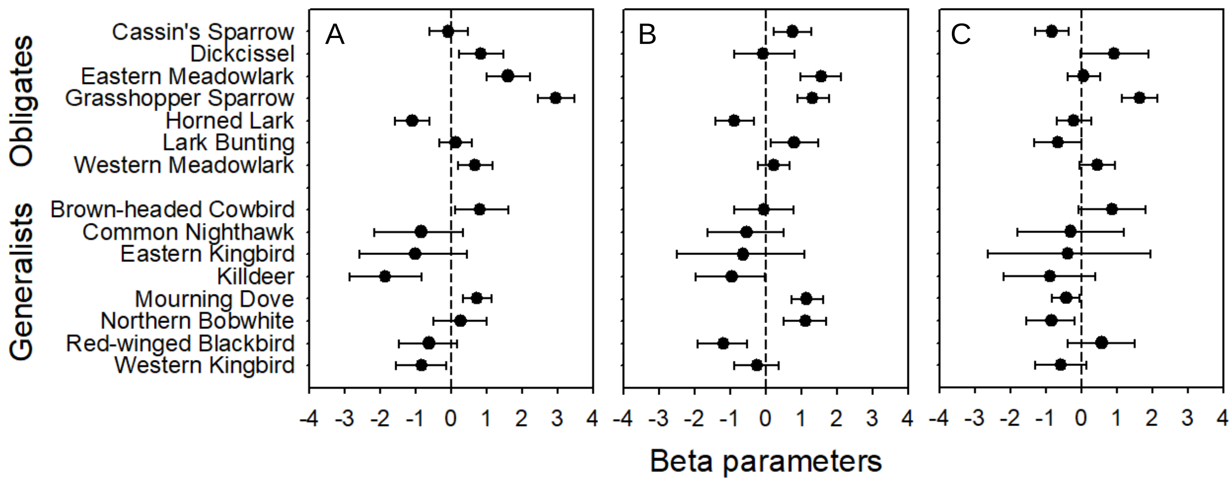

Figure 5. The beta coefficients for the small-scale occupancy of 5 ha point count plots for (A) native Conservation Reserve Program (CRP) plantings relative to agricultural lands, (B) introduced CRP plantings relative to agricultural lands, and (C) native CRP plantings relative to introduced CRP plantings for grassland obligates and generalists within the Action Area defined by the occupied range of the lesser prairie-chicken, Colorado, Kansas, New Mexico, Oklahoma, and Texas. The round symbols represent beta coefficients graphed relative to zero and the error bars are 95% credible intervals.

Prescribed Grazing

We presented species richness responses to CRP for the Sand Shinnery Oak Prairie ecoregion in 2016 because presenting results by ecoregion and year would require an inordinate amount of space. Because the effects are additive, the relationships are similar in the other ecoregions and years. There was little support for the hypothesis that alpha species richness of grassland obligates was greater on LPCI prescribed grazing lands than on reference grasslands (P = 0.86, Fig. 6A, Table S1, available in Supporting Information). However, we confirmed the hypothesis for lower alpha species richness of generalists in LPCI prescribed grazing lands relative to reference grasslands (β = 0.55, SD = 0.34, CI = [-0.06, 1.28], P = 0.96, Fig. 6B). We confirmed the hypothesis for greater alpha species richness of obligates than generalists in LPCI grazing lands (β = 0.82, SD = 0.63, CI = [-0.65, 1.90], P = 0.90, Fig. 6).

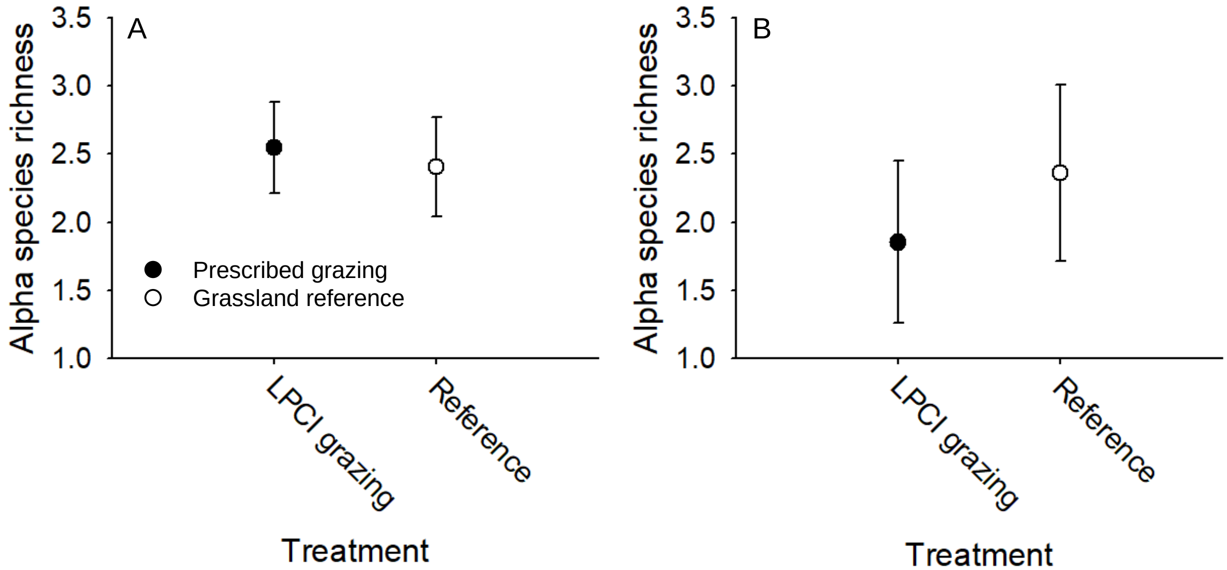

Figure 6. The alpha species richness of 5 ha point count plots for Lesser Prairie-Chicken Initiative prescribed grazing and reference grasslands for grassland (A) obligates and (B) generalists within the Sand Shinnery Oak Prairie ecoregion, occupied range of the lesser prairie-chicken, New Mexico and Texas, 2016. The round symbols represents the mean number of species among point count plots and the error bars are 1 standard deviation. The effects are additive and show similar responses in the other ecoregions and years (not shown).

We found strong evidence for shifts in species composition between LPCI prescribed grazing lands and reference grasslands within the Action Area (Fig. 7, Table S1, available in Supporting Information). The small-scale occupancy of the Cassin's sparrow ($P > 0.99$), common nighthawk ($P = 0.99$), dickcissel ($P = 0.91$), eastern meadowlark ($P = 0.99$), killdeer ($P = 0.94$), lark bunting ($P > 0.99$), lesser prairie-chicken ($P = 0.99$) and eastern kingbird ($P = 0.98$) was greater on LPCI prescribed grazing lands than reference grasslands (Fig. 7). In contrast, the ash-throated flycatcher (*Myiarchus cinerascens*, $P > 0.99$), canyon towhee ($P = 0.98$), field sparrow ($P > 0.99$), grasshopper sparrow ($P = 0.98$), lark sparrow ($P = 0.98$), loggerhead shrike ($P = 0.99$), mallard (*Anas platyrhynchos*, $P = 0.92$), northern bobwhite ($P = 0.93$), red-winged blackbird ($P = 0.99$), ring-necked pheasant ($P = 0.94$), scaled quail ($P > 0.99$), scissor-tailed flycatcher ($P > 0.99$), vesper sparrow (*Pooecetes gramineus*, $P = 0.90$) and western kingbird ($P > 0.99$) showed lower small-scale occupancy in LPCI prescribed grazing lands than reference grasslands (Fig. 7).

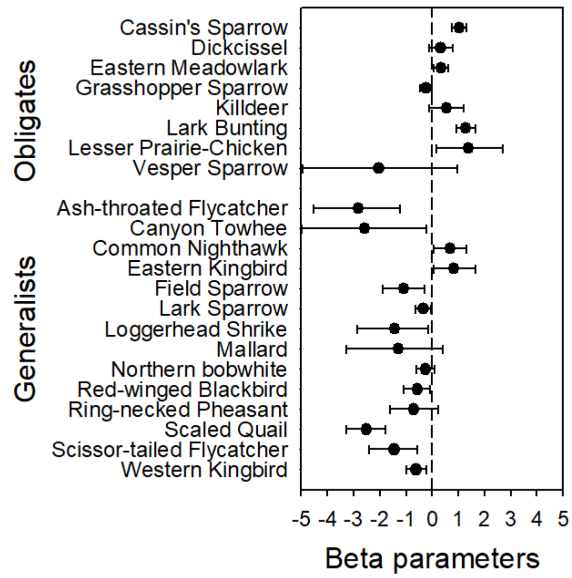

Figure 7. The beta coefficients for the small-scale occupancy of 5 ha point count plots for Lesser Prairie-Chicken Initiative prescribed grazing relative to reference grasslands for grassland obligates and generalists within the Action Area defined by the occupied range of the lesser prairie-chicken, Colorado, Kansas, New Mexico, Oklahoma, and Texas. The round symbols represent beta coefficients graphed relative to zero and the error bars are 95% credible intervals.

Landscape and Local Vegetation Structure

We presented species richness responses to CRP for the Sand Shinnery Oak Prairie ecoregion in 2016. Because the effects are additive, the relationships are similar in the other ecoregions and years. At the landscape scale, we found little evidence for the hypothesis that gamma species richness of grassland obligates declined with increasing fragmentation of native vegetation ($P = 0.51$), or the hypothesis that gamma species richness of grassland generalists increased with increasing fragmentation of native vegetation ($P = 0.29$). Likewise, we found little evidence for the hypothesis that gamma species richness of grassland obligates declined with increasing land-cover of shrub-land vegetation ($P = 0.72$), or the hypothesis that gamma species richness of grassland generalists increased with increasing land-cover of shrub-land vegetation ($P = 0.28$).

Nevertheless, species composition varied by the fragmentation of native vegetation in the Action Area (Fig. 8A, Table S1, available in Supporting Information) and the relative land-cover of shrubland vegetation (Fig. 8B). The large-scale occupancy of the Cassin's sparrow ($P = 0.98$), eastern meadowlark ($P = 0.90$), canyon towhee ($P = 0.92$), Chihuahuan raven (*Corvus cryptoleucus*, $P = 0.92$), field sparrow ($P = 0.93$) and lark sparrow ($P = 0.96$) increased with increasing mean patch size of native vegetation (Fig. 8A). In contrast, the dickcissel ($P = 0.93$), killdeer ($P = 0.94$), northern bobwhite ($P = 0.99$), red-winged blackbird ($P > 0.99$) and ring-necked pheasant ($P = 0.97$) showed declining large-scale occupancy with increasing mean patch size of native vegetation (Fig. 8A). Along the shrubland landcover gradient, the large-scale occupancy of the burrowing owl (*Athene cunicularia*, $P = 0.97$), vesper sparrow ($P = 0.90$), ash-throated flycatcher ($P = 0.99$), field sparrow ($P = 0.91$), scaled quail ($P > 0.99$), scissor-tailed flycatcher ($P > 0.99$) and western kingbird ($P = 0.97$) increased with increasing land-cover of shrub-land vegetation (Fig. 8B). In contrast, the dickcissel ($P > 0.99$), grasshopper sparrow

(*P* > 0.99), horned lark (*P* > 0.99), long-billed curlew (*Numenius americanus*, *P* = 0.94), western meadowlark (*P* > 0.99), killdeer (*P* > 0.99), lark sparrow (*P* = 0.95), red-winged blackbird (*P* = 0.93) and ring-necked pheasant (*P* > 0.99) showed declining large-scale occupancy with increasing land-cover of shrubland vegetation (Fig. 8B).

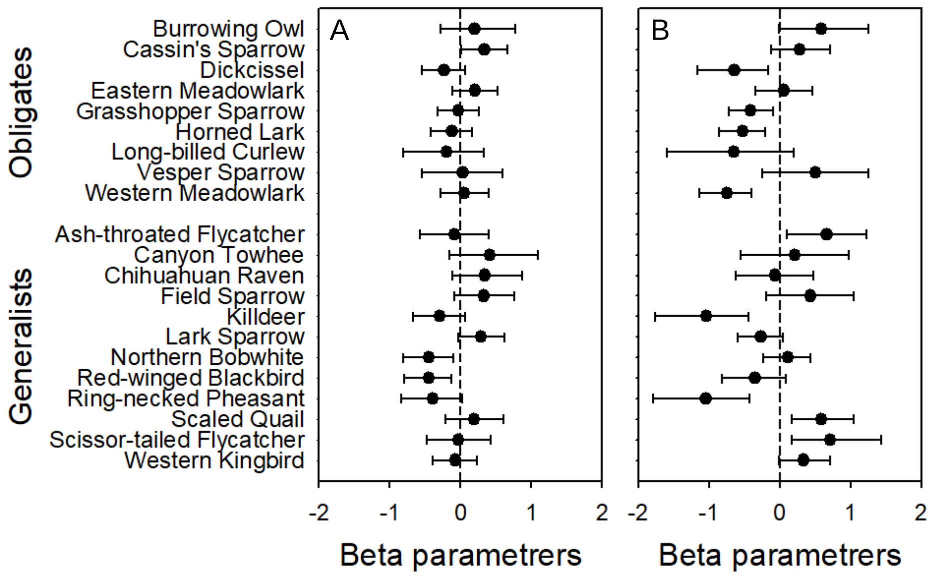

Figure 8. The beta coefficients for the large-scale occupancy of 1 km^2 grid cells for the (A) mean patch size of native vegetation and (B) relative land-cover of shrubland vegetation for grassland obligates and generalists within the Action Area defined by the occupied range of the lesser prairie-chicken, Colorado, Kansas, New Mexico, Oklahoma, and Texas. The round symbols represent beta coefficients graphed relative to zero and the error bars are 95% credible intervals.

At the local scale, we confirmed the hypothesis that the alpha species richness of grassland obligates in the Action Area (β = 0.10, SD = 0.04, CI = [0.02, 0.19], *P* = 0.99) and generalists (β = 0.08, SD = 0.04, CI = [0.00, 0.17], *P* = 0.98) increased with increasing ground cover of herbaceous vegetation (Fig. 9). In contrast, there was little evidence that the alpha species richness of grassland obligates (*P* = 0.56) and generalists (*P* = 0.73) increased with increasing grass height.

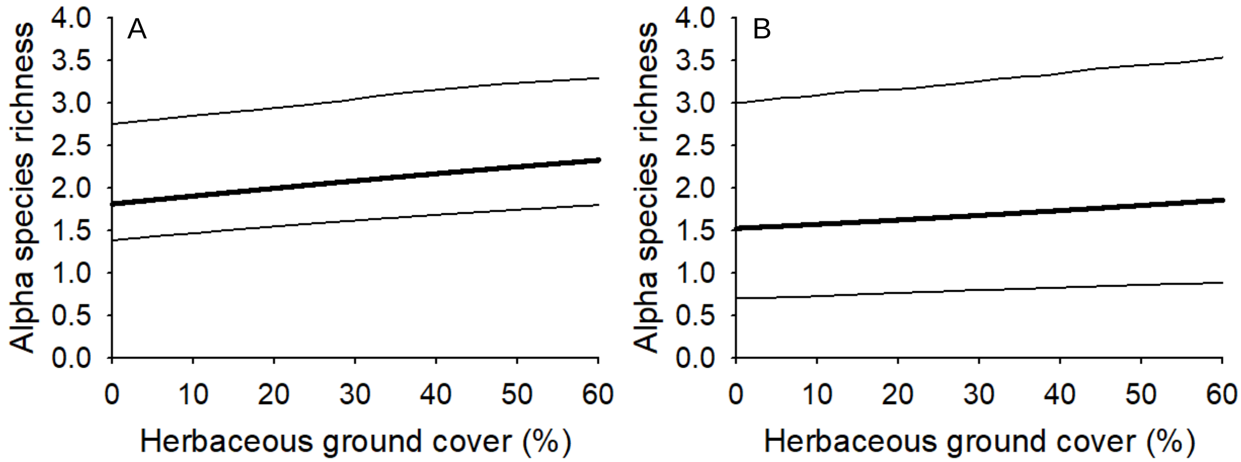

Figure 9. The alpha species richness of 5 ha point count plots by herbaceous ground cover for grassland (A) obligates and (B) generalists in reference grasslands within the Sand Shinnery Oak Prairie ecoregion, occupied range of the lesser prairie-chicken, New Mexico and Texas, 2016. The bold trend line represents the mean number of species among point count plots and the bounding lines are 95% credible intervals. The effects are additive and show similar responses in the other ecoregions and years (not shown).

We found evidence for shifts in species composition according to variation in herbaceous ground cover within the Action Area (Fig. 10A, Table S1, available in Supporting Information) and grass height (Fig. 10B, Table S1, available in Supporting Information). The small-scale occupancy of the Cassin's sparrow ($P > 0.99$), grasshopper sparrow ($P = 0.98$), horned lark ($P = 0.99$), western meadowlark ($P > 0.99$), brown-headed cowbird ($P = 0.97$), northern bobwhite ($P > 0.99$), red-winged blackbird ($P = 0.97$) and scaled quail ($P = 0.91$) increased with increasing herbaceous ground cover (Fig. 10A). In terms of grass height, the dickcissel ($P > 0.99$), lark bunting ($P = 0.97$), mallard ($P = 0.93$), mourning dove ($P > 0.99$), red-winged blackbird ($P > 0.99$) and ring-necked pheasant ($P = 0.98$) showed increasing small-scale occupancy with increasing grass height, whereas horned lark ($P = 0.96$) and western meadowlark ($P = 0.96$) occupancy declined with increasing grass height (Fig. 10B).

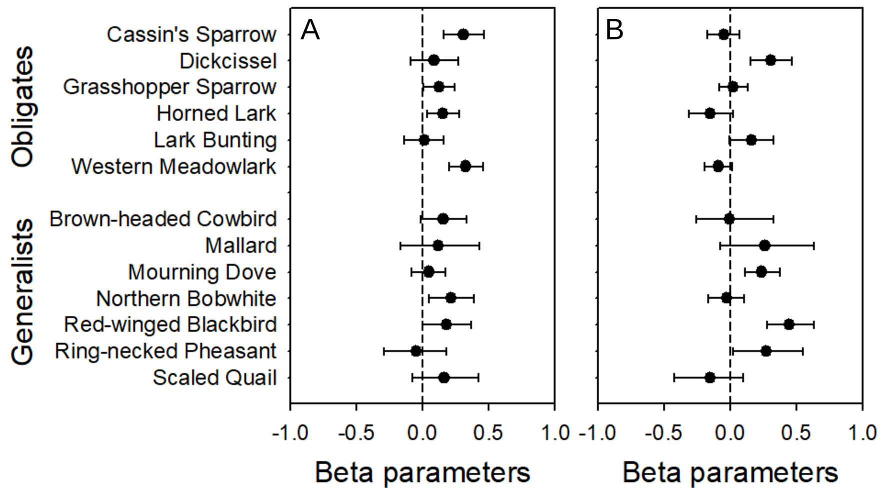

Figure 10. The beta coefficients for the small-scale occupancy of 5 ha point count plots for (A) herbaceous ground cover and (B) mean grass height for grassland obligates and generalists within the Action Area defined by the occupied range of the lesser prairie-chicken, Colorado, Kansas, New Mexico, Oklahoma, and Texas. The round symbols represent beta coefficients graphed relative to zero and the error bars are 95% credible intervals.

We presented species richness responses to CRP for the Sand Shinnery Oak Prairie ecoregion in 2016. Because the effects are additive, the relationships are similar in the other ecoregions and years. With respect to the shrub component, we found little support for declining alpha species richness of grassland obligates with increasing shrub canopy cover (P = 0.42) and shrub height (P = 0.19). In contrast, we confirmed the hypothesis for increasing species richness of generalists with increasing shrub cover (β = 0.09, SD = 0.06, CI = [-0.03, 0.20], P = 0.93, Fig. 11A) and shrub height (β = 0.15, SD = 0.06, CI = [0.03, 0.27], P = 0.99, Fig. 11B).

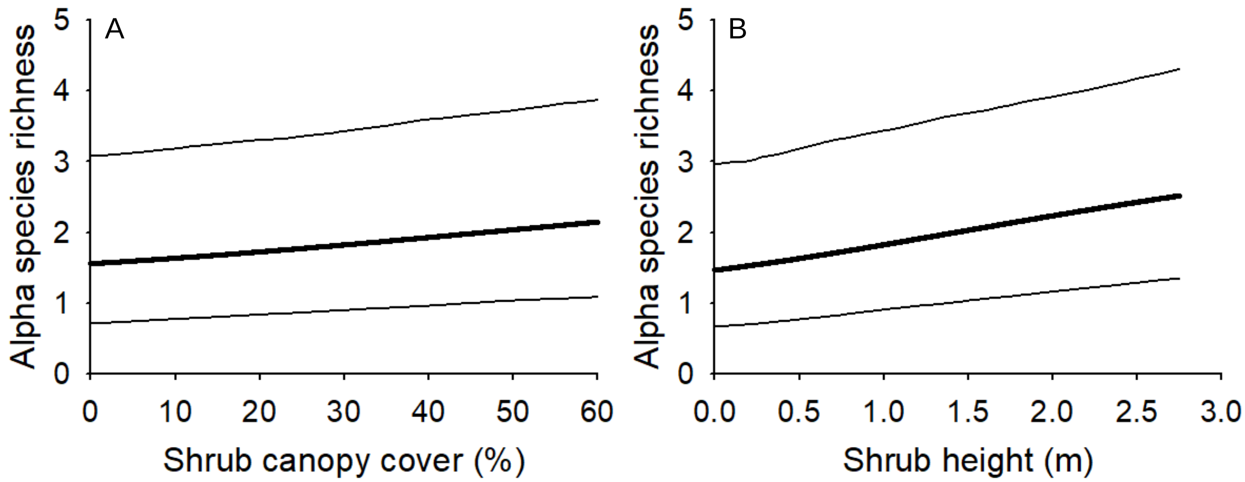

Figure 11. The alpha species richness of 5 ha point count plots for grassland generalists by shrub canopy cover and height in reference grasslands within the Sand Shinnery Oak Prairie ecoregion, occupied range of the lesser prairie-chicken, New Mexico and Texas, 2016. The bold trend line represents the mean number of species among point count plots and the bounding lines are 95% credible intervals. The effects are additive and show similar responses in the other ecoregions and years (not shown).

We found strong evidence for shifts in species composition along gradients of shrub cover within the Action Area (Fig. 12A, Table S1, available in Supporting Information) and shrub height (Fig. 12B, Table S1, available in Supporting Information). The Cassin's sparrow ($P >$ 0.99), lesser prairie-chicken ($P = 0.97$), field sparrow ($P > 0.99$), mourning dove ($P = 0.99$), northern bobwhite ($P > 0.99$), scaled quail ($P > 0.99$), scissor-tailed flycatcher ($P = 0.99$) and western kingbird ($P = 0.90$) showed increasing small-scale occupancy with increasing shrub cover, whereas grasshopper sparrow ($P > 0.99$), lark bunting ($P = 0.99$) and western meadowlark ($P = 0.98$) occupancy declined with increasing shrub cover (Fig. 12A). Along the gradient of shrub height, the small-scale occupancy of the Cassin's sparrow ($P > 0.99$), eastern meadowlark ($P = 0.99$), vesper sparrow ($P = 0.90$), American kestrel ($P > 0.99$), ash-throated flycatcher ($P = 0.99$), canyon towhee ($P = 0.91$), eastern kingbird ($P = 0.99$), field sparrow ($P =$ 0.97), lark sparrow ($P > 0.99$), loggerhead shrike ($P = 0.96$), mourning dove ($P = 0.99$), northern bobwhite ($P > 0.99$), Say's phoebe (*Sayornis saya*, $P = 0.92$), scaled quail ($P > 0.99$) and western kingbird ($P > 0.99$) increased with increasing shrub height (Fig. 12B). In contrast, the grasshopper sparrow ($P > 0.99$), horned lark ($P > 0.99$), western meadowlark ($P = 0.98$) and red-winged blackbird ($P = 0.99$) showed declining small-scale occupancy with increasing shrub height (Fig. 12B).

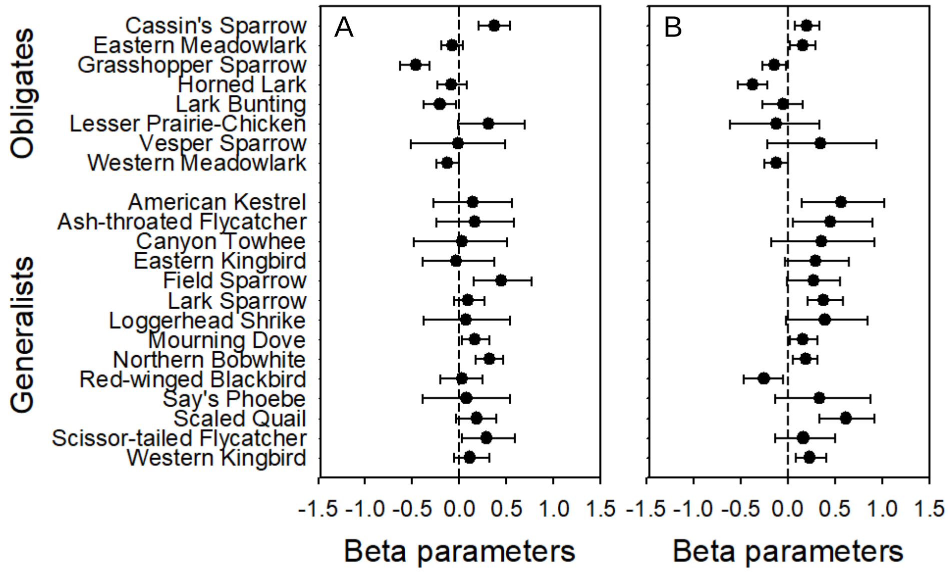

Figure 12. The beta coefficients for the small-scale occupancy of 5 ha point count plots for (A) shrub canopy cover and (B) mean shrub height for grassland obligates and generalists within the Action Area defined by the occupied range of the lesser prairie-chicken, Colorado, Kansas, New Mexico, Oklahoma, and Texas. The round symbols represent beta coefficients graphed relative to zero and the error bars are 95% credible intervals.

We presented species richness responses to CRP for the Sand Shinnery Oak Prairie ecoregion in 2016. Because the effects are additive, the relationships are similar in the other ecoregions and years. With respect to the tree component, we found little support for declining alpha species richness of grassland obligates with increasing tree canopy cover (P = 0.85) and tree height (P = 0.82). In a similar fashion, there was little support for increasing alpha species richness of grassland generalists with increasing tree canopy cover (P = 0.33) and tree height (P = 0.86).

We observed evidence for shifts in species composition along the gradients of tree canopy cover in the Action Area (Fig. 13A, Table S1, available in Supporting Information) and tree height (Fig. 13B, Table S1, available in Supporting Information). Along the tree canopy cover gradient, the small-scale occupancy of the grasshopper sparrow (P = 0.96), horned lark (P = 0.93) and western meadowlark (P = 0.92) declined, whereas the occupancy of the northern bobwhite (P = 0.98) increased with increasing tree canopy cover (Fig. 13A). Along the gradient of tree height, the Cassin's sparrow (P > 0.99), grasshopper sparrow (P > 0.99), horned lark (P > 0.99), lark bunting (P = 0.99), western meadowlark (P > 0.99) and killdeer (P = 0.97) showed declining small-scale occupancy with increasing tree height, whereas American kestrel (P = 0.91), ash-throated flycatcher (P = 0.98), eastern kingbird (P > 0.99), lark sparrow (P > 0.99), loggerhead shrike (P = 0.90), mourning dove (P > 0.99), red-winged blackbird (P = 0.97) and rufous-crowned sparrow (P = 0.92) occupancy increased with increasing tree height (Fig. 13B).

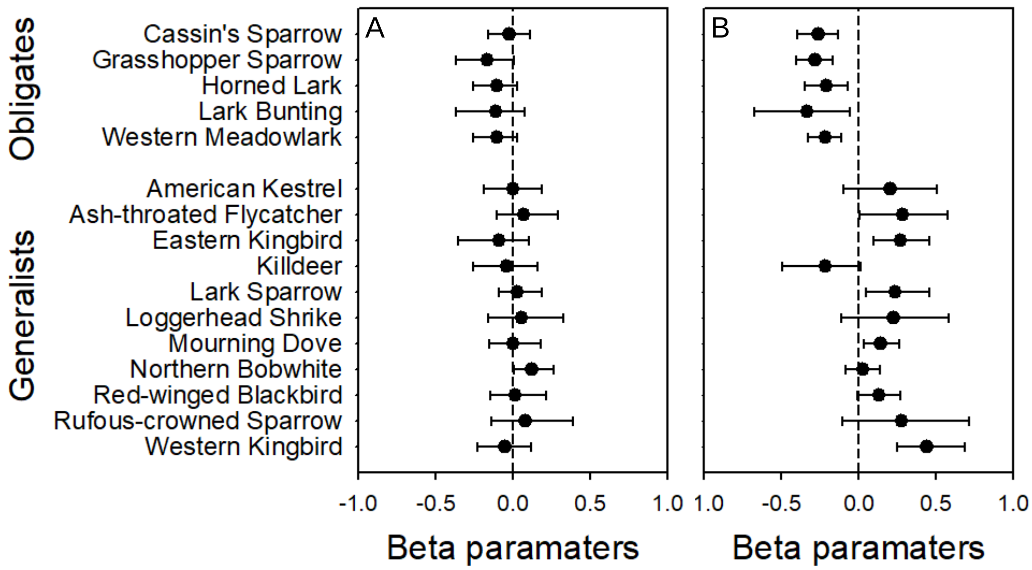

Figure 13. The beta coefficients for the small-scale occupancy of 5 ha point count plots for (A) tree canopy cover and (B) mean tree height for grassland obligates and generalists within the Action Area defined by the occupied range of the lesser prairie-chicken, Colorado, Kansas, New Mexico, Oklahoma, and Texas. The round symbols represent beta coefficients graphed relative to zero and the error bars are 95% credible intervals.

Population Density

Conservation Reserve Program

We evaluated hypotheses for differences in avian densities on CRP plantings relative to agricultural lands and reference grasslands in the Action Area. Population densities were greater on native than introduced CRP plantings for the grasshopper sparrow, horned lark, lark bunting and ring-necked pheasant (Table 5, Table 6, Table S2, available in Supporting Information). In contrast, densities on native CRP were lower than introduced CRP for the American kestrel, Cassin's sparrow, eastern meadowlark, mourning dove, northern bobwhite and western kingbird (Table 5, Table 6).

Table 5. The mean population densities (D, km^{-2}) and Standard Errors (SE) for Lesser Prairie-Chicken Initiative (LPCI) prescribed grazing, native Conservation Reserve Program (CRP) plantings, introduced CRP plantings, reference grasslands and agricultural lands within the Action Area defined by the occupied range of the lesser prairie-chicken plus 16 km buffer, Colorado, Kansas, New Mexico, Oklahoma and Texas, 2015 - 2017. Population densities for agricultural land was estimated for 2016 and 2017, and densities for native and introduced CRP was estimated for 2016.

Species	LPCI		Native CRP		Intro. CRP		Grassland		Ag. land	
	D	SE	*D*	SE	*D*	SE	*D*	SE	*D*	SE
American Kestrel	0.15	0.07	0.04	0.04	0.44	0.22	0.13	0.05	0.06	0.03
Ash-throated Flycatcher	0.04	0.04	-	-	0.80	0.62	0.18	0.13	-	-
Brown-headed Cowbird	8.85	1.09	1.51	0.84	3.93	1.25	8.15	1.58	25.71	4.12
Brewer's Blackbird	0.06	0.06	-	-	0.17	0.17	-	-	-	-
Burrowing Owl	0.08	0.04	0.02	0.03	-	-	0.19	0.20	-	-
Blue-winged Teal	-	-	-	-	-	-	0.06	0.06	0.01	0.01
Cassin's Kingbird	0.04	0.05	-	-	-	-	0.05	0.05	-	-
Canada Goose	0.01	0.01	-	-	-	-	-	-	-	-
Cassin's Sparrow	47.05	1.83	26.35	4.45	47.74	5.12	33.67	4.92	2.55	0.46
Chihuahuan Raven	0.19	0.07	0.20	0.11	0.32	0.18	0.31	0.14	0.09	0.04
Common Yellowthroat	0.12	0.07	-	-	-	-	-	-	-	-
Dickcissel	11.30	2.75	3.08	1.52	1.54	0.91	8.78	1.92	26.45	5.40
Eastern Bluebird	0.33	0.14	-	-	0.27	0.16	-	-	-	-
Eastern Kingbird	3.36	1.04	0.25	0.27	-	-	0.24	0.15	-	-
Eastern Meadowlark	18.63	1.70	13.06	4.04	25.09	3.40	6.85	1.30	4.64	1.08
Field Sparrow	1.32	0.39	0.07	0.05	1.07	0.65	0.14	0.08	-	-
Grasshopper Sparrow	44.83	5.22	104.62	13.82	71.97	13.64	64.38	5.80	33.63	6.65
Horned Lark	19.11	2.29	47.65	10.77	23.10	8.79	47.75	5.78	78.02	7.44
Killdeer	1.40	0.37	0.53	0.30	0.80	0.26	1.37	0.28	6.09	0.88
Lark Bunting	9.32	1.14	25.23	9.93	3.85	1.82	14.76	3.43	12.36	5.66
Lark Sparrow	16.48	1.88	2.72	1.38	6.01	2.07	18.25	3.56	2.70	0.76
Long-billed Curlew	0.06	0.06	0.04	0.04	-	-	0.17	0.15	0.06	0.05
Lesser Prairie-Chicken	0.81	0.32	0.05	0.03	0.20	0.14	0.13	0.07	0.00	0.00
Loggerhead Shrike	0.12	0.06	0.07	0.07	0.20	0.14	0.07	0.04	-	-
Mallard	0.02	0.02	-	-	-	-	0.08	0.05	0.32	0.14
Mourning Dove	10.25	0.73	13.02	1.56	20.22	1.97	7.36	0.61	10.12	0.74
Mountain Plover	-	-	-	-	-	-	0.14	0.15	-	-
Northern Bobwhite	5.05	0.60	3.60	1.13	9.06	1.43	3.00	0.41	3.70	0.60
Northern Harrier	0.03	0.03	0.22	0.14	0.07	0.07	-	-	-	-
Rufous-crowned Sparrow	0.34	0.23	-	-	-	-	-	-	-	-
Ring-necked Pheasant	0.36	0.11	1.09	0.27	0.35	0.16	0.33	0.07	3.80	0.37
Red-winged Blackbird	5.36	0.68	1.41	0.60	4.97	2.19	5.19	1.41	32.61	5.43
Say's Phoebe	0.03	0.03	0.11	0.06	0.03	0.03	0.03	0.03	-	-
Scaled Quail	0.79	0.23	1.30	0.65	2.17	0.75	1.97	0.68	0.29	0.15
Scissor-tailed Flycatcher	1.16	0.44	0.99	0.72	1.94	0.62	1.91	0.47	1.24	0.31
Swainson's Hawk	0.09	0.03	0.12	0.07	0.15	0.08	0.04	0.01	0.20	0.08
Turkey Vulture	1.67	0.72	-	-	0.05	0.05	0.26	0.08	0.19	0.15
Western Kingbird	3.52	0.53	3.07	1.14	12.08	2.59	4.94	1.07	5.58	1.50
Western Meadowlark	12.31	1.03	19.92	2.69	13.32	3.20	22.29	1.73	21.43	2.04

Table 6. The effect sizes (Δ) for density (km^{-2}), and Lower (LCL) and Upper (UCL) 90% Confidence Limits, respectively for differences between native Conservation Reserve Program (CRP) and introduced CRP plantings, native CRP plantings and reference grasslands, and native CRP plantings and agricultural lands within the Action Area defined by the occupied range of the lesser prairie-chicken plus 16 km buffer, Colorado, Kansas, New Mexico, Oklahoma and Texas, 2015 – 2017. The bold values represent measurable effects sizes with Confidence Intervals excluding zero.

Species[a]	Native CRP - intro. CRP			Native CRP - grassland			Native CRP - ag. land		
	Δ	LCL	UCL	Δ	LCL	UCL	Δ	LCL	UCL
American Kestrel	**-0.40**	**-0.77**	**-0.03**	-0.09	-0.20	0.01	-0.02	-0.10	0.05
Brown-headed Cowbird	-2.42	-4.91	0.07	**-6.64**	**-9.60**	**-3.69**	**-24.20**	**-31.12**	**-17.28**
Cassin's Sparrow	**-21.39**	**-32.55**	**-10.23**	-7.33	-18.24	3.58	**23.80**	**16.44**	**31.16**
Dickcissel	1.54	-1.37	4.45	**-5.70**	**-9.73**	**-1.67**	**-23.37**	**-32.60**	**-14.14**
Eastern Meadowlark	**-12.03**	**-20.72**	**-3.33**	6.21	-0.78	13.20	**8.42**	**1.54**	**15.31**
Grasshopper Sparrow	**32.65**	**0.70**	**64.59**	**40.24**	**15.58**	**64.90**	**70.99**	**45.76**	**96.21**
Horned Lark	**24.55**	**1.67**	**47.43**	-0.10	-20.21	20.02	**-30.36**	**-51.90**	**-8.82**
Killdeer	-0.27	-0.93	0.39	**-0.84**	**-1.52**	**-0.16**	**-5.57**	**-7.09**	**-4.04**
Lark Bunting	**21.38**	**4.76**	**38.00**	10.46	-6.83	27.76	12.87	-5.94	31.69
Lark Sparrow	-3.29	-7.39	0.80	**-15.53**	**-21.82**	**-9.24**	0.01	-2.58	2.61
Mourning Dove	**-7.21**	**-11.34**	**-3.07**	**5.66**	**2.89**	**8.42**	**2.90**	**0.05**	**5.74**
Northern Bobwhite	**-5.46**	**-8.48**	**-2.45**	0.60	-1.38	2.59	-0.10	-2.22	2.01
Ring-necked Pheasant	**0.74**	**0.22**	**1.26**	**0.76**	**0.30**	**1.23**	**-2.71**	**-3.47**	**-1.95**
Red-winged Blackbird	-3.56	-7.30	0.18	**-3.77**	**-6.31**	**-1.24**	**-31.20**	**-40.19**	**-22.20**
Western Kingbird	**-9.01**	**-13.66**	**-4.35**	-1.87	-4.46	0.71	-2.51	-5.61	0.60

[a] Only species with measureable effect sizes are shown.

The population densities were greater on native CRP plantings than grasslands for the grasshopper sparrow, mourning dove and ring-necked pheasant (Table 5, Table 6, Table S2, available in Supporting Information). Conversely, densities were lower on native CRP plantings than grasslands for the brown-headed cowbirds, dickcissel, killdeer, lark sparrow and red-winged blackbird (Table 5, Table 6). In addition, the population densities were greater on native CRP plantings than agricultural lands for the Cassin's sparrow, eastern meadowlark, grasshopper sparrow and mourning dove (Table 5, Table 6). In contrast, the densities were lower on native CRP plantings than agricultural lands for the brown-headed cowbird, dickcissel, horned lark, killdeer, ring-necked pheasant and red-winged blackbird (Table 5, Table 6).

Finally, we evaluated hypotheses for differences in avian densities on introduced CRP plantings relative to grasslands and agricultural lands in the Action Area. Population densities were greater on introduced CRP plantings than grasslands for the Cassin's sparrow, eastern meadowlark, mourning dove, northern bobwhite and western kingbird (Table 4, Table 6, Table S2, available in Supporting Information). In contrast, densities were lower on introduced CRP plantings than grasslands for the brown-headed cowbird, dickcissel, horned lark, lark bunting, lark sparrow, turkey vulture (*Cathartes aura*) and western meadowlark (Table 5, Table 7).

Table 7. The effect sizes (Δ) for density (km^{-2}), and Lower (LCL) and Upper (UCL) 90% Confidence Limits, respectively for differences between introduced Conservation Reserve Program (CRP) plantings and reference grasslands, and introduced CRP plantings and agricultural lands within the Action Area defined by the occupied range of the lesser prairie-chicken plus 16 km buffer, Colorado, Kansas, New Mexico, Oklahoma and Texas, 2015 – 2017. The bold values represent measurable effects sizes with Confidence Intervals excluding zero.

Species[a]	Introduced CRP - grassland			Introduced CRP - ag. land		
	Δ	LCL	UCL	Δ	LCL	UCL
American Kestrel	0.31	-0.07	0.69	**0.38**	**0.00**	**0.75**
Brown-headed Cowbird	**-4.23**	**-7.55**	**-0.90**	**-21.78**	**-28.87**	**-14.70**
Cassin's Sparrow	**14.07**	**2.39**	**25.75**	**45.19**	**36.74**	**53.65**
Dickcissel	**-7.24**	**-10.74**	**-3.74**	**-24.91**	**-33.92**	**-15.90**
Eastern Meadowlark	**18.23**	**12.24**	**24.23**	**20.45**	**14.57**	**26.32**
Grasshopper Sparrow	7.59	-16.80	31.99	**38.34**	**13.37**	**63.31**
Horned Lark	**-24.65**	**-41.96**	**-7.33**	**-54.91**	**-73.86**	**-35.97**
Killdeer	-0.57	-1.21	0.08	**-5.30**	**-6.80**	**-3.79**
Lark Bunting	**-10.91**	**-17.31**	**-4.52**	**-8.51**	**-18.29**	**1.28**
Lark Sparrow	**-12.24**	**-19.02**	**-5.46**	3.30	-0.32	6.93
Mourning Dove	**12.86**	**9.46**	**16.26**	**10.10**	**6.64**	**13.57**
Northern Bobwhite	**6.07**	**3.61**	**8.52**	**5.36**	**2.80**	**7.92**
Ring-necked Pheasant	0.02	-0.27	0.32	**-3.45**	**-4.12**	**-2.78**
Red-winged Blackbird	-0.22	-4.51	4.08	**-27.64**	**-37.28**	**-18.00**
Scaled Quail	0.20	-1.47	1.87	**1.87**	**0.61**	**3.14**
Turkey Vulture	**-0.21**	**-0.38**	**-0.05**	-0.14	-0.40	0.12
Western Kingbird	**7.14**	**2.52**	**11.75**	**6.50**	**1.58**	**11.42**
Western Meadowlark	**-8.98**	**-14.97**	**-2.98**	**-8.11**	**-14.36**	**-1.86**

[a] Only species with measureable effect sizes are shown.

Population densities were greater on introduced CRP plantings than agricultural lands for the American kestrel, Cassin's sparrow, eastern meadowlark, grasshopper sparrow, mourning dove, northern bobwhite, scaled quail and western kingbird (Table 5, Table 7, Table S2, available in Supporting Information). Conversely, densities were lower on introduced CRP plantings than agricultural lands for the brown-headed cowbird, dickcissel, horned lark, killdeer, ring-necked pheasant, red-winged blackbird and western meadowlark (Table 5, Table 7).

Prescribed Grazing

Next, we evaluated hypotheses for differences in avian densities on LPCI prescribed grazing lands relative to CRP lands and grasslands in the Action Area. Avian population densities were greater on LPCI grazing lands than grasslands in the region for the Cassin's sparrow, eastern kingbird, eastern meadowlark, field sparrow, lesser prairie-chicken, mourning dove, northern bobwhite and turkey vulture (Table 5, Table 8, Table S2, available in Supporting Information). Conversely, population densities on LPCI grazing lands were lower than grasslands for the grasshopper sparrow, horned lark, scaled quail and western meadowlark (Table 5, Table 8).

Table 8. The effect sizes (Δ) for density (km^{-2}), and Lower (LCL) and Upper (UCL) 90% Confidence Limits, respectively for differences between Lesser Prairie-Chicken Initiative (LPCI) prescribed grazing and reference grasslands, LPCI prescribed grazing and native Conservation Reserve Program (CRP) plantings, and LPCI prescribed grazing and introduced CRP plantings within the Action Area defined by the occupied range of the lesser prairie-chicken plus 16 km buffer, Colorado, Kansas, New Mexico, Oklahoma and Texas, 2015 – 2017. The bold values represent measurable effects sizes with Confidence Intervals excluding zero.

Species[a]	LPCI - grassland			LPCI - native CRP			LPCI - introduced CRP		
	Δ	LCL	UCL	Δ	LCL	UCL	Δ	LCL	UCL
Brown-headed Cowbird	0.70	-2.47	3.87	**7.34**	**5.07**	**9.62**	**4.93**	**2.19**	**7.66**
Burrowing Owl	-0.11	-0.45	0.23	0.06	-0.03	0.15	**0.08**	**0.01**	**0.16**
Cassin's Sparrow	**13.37**	**4.73**	**22.01**	**20.70**	**12.78**	**28.62**	-0.69	-9.64	8.26
Dickcissel	2.51	-3.01	8.03	**8.22**	**3.05**	**13.38**	**9.76**	**4.99**	**14.52**
Eastern Kingbird	**3.11**	**1.38**	**4.84**	**3.10**	**1.34**	**4.87**	**3.36**	**1.65**	**5.07**
Eastern Meadowlark	**11.78**	**8.25**	**15.31**	5.57	-1.64	12.79	**-6.45**	**-12.71**	**-0.19**
Field Sparrow	**1.18**	**0.53**	**1.84**	**1.25**	**0.61**	**1.90**	0.26	-0.99	1.50
Grasshopper Sparrow	**-19.54**	**-32.39**	**-6.69**	**-59.78**	**-84.09**	**-35.48**	**-27.14**	**-51.17**	**-3.10**
Horned Lark	**-28.64**	**-38.87**	**-18.41**	**-28.55**	**-46.67**	**-10.42**	-4.00	-18.94	10.95
Killdeer	0.04	-0.74	0.81	**0.88**	**0.09**	**1.66**	0.60	-0.15	1.36
Lark Bunting	-5.45	-11.40	0.51	-15.91	-32.36	0.55	**5.47**	**1.93**	**9.00**
Lark Sparrow	-1.77	-8.39	4.86	**13.76**	**9.93**	**17.60**	**10.47**	**5.87**	**15.07**
Lesser Prairie-Chicken	**0.69**	**0.15**	**1.23**	**0.77**	**0.24**	**1.29**	**0.62**	**0.05**	**1.19**
Mourning Dove	**2.89**	**1.32**	**4.45**	-2.77	-5.61	0.07	**-9.98**	**-13.44**	**-6.52**
Northern Bobwhite	**2.05**	**0.85**	**3.25**	1.45	-0.67	3.56	**-4.02**	**-6.58**	**-1.45**
Ring-necked Pheasant	0.03	-0.18	0.25	**-0.73**	**-1.21**	**-0.25**	0.01	-0.31	0.33
Red-winged Blackbird	0.17	-2.41	2.76	**3.95**	**2.45**	**5.45**	0.39	-3.39	4.17
Scaled Quail	**-1.18**	**-2.36**	**0.00**	-0.51	-1.65	0.63	**-1.38**	**-2.68**	**-0.08**
Turkey Vulture	**1.40**	**0.21**	**2.60**	-	-	-	**1.62**	**0.43**	**2.81**
Western Kingbird	-1.42	-3.40	0.56	0.45	-1.63	2.53	**-8.56**	**-12.90**	**-4.21**
Western Meadowlark	**-9.98**	**-13.30**	**-6.66**	**-7.61**	**-12.34**	**-2.87**	-1.00	-6.54	4.54

[a] Only species with measureable effect sizes are shown.

Population densities for LPCI prescribed grazing lands were greater than native CRP lands for the brown-headed cowbird, Cassin's sparrow, dickcissel, eastern kingbird, field sparrow, killdeer, lark sparrow, lesser prairie-chicken and red-winged blackbird (Table 5, Table 8, Table S2, available in Supporting Information). In contrast, densities for LPCI prescribed grazing lands were lower than native CRP lands for the grasshopper sparrow, horned lark, ring-necked pheasant and western meadowlark (Table 5, Table 8).

Population densities for LPCI prescribed grazing lands were greater than introduced CRP lands for the brown-headed cowbird, burrowing owl, dickcissel, eastern kingbird, lark bunting, lark sparrow, lesser prairie-chicken and turkey vulture (Table 5, Table 8, Table S2, available in Supporting Information). Conversely, densities for LPCI prescribed grazing lands were lower than introduced CRP lands for the eastern meadowlark, grasshopper sparrow, mourning dove, northern bobwhite, scaled quail and western kingbird (Table 5, Table 8).

Population Size

We estimated avian population sizes in 2016 to determine the contributions of the conservation practices to bird populations in the Action Area. The Action Area encompassed a region covering 161,761 km^2 (62,456 mi^2). In the five-year period between 2012 and 2016, 1,887 km^2 (728 mi^2) of LPCI prescribed grazing was implemented, corresponding to 1.2% of the Action

Area. As of 2016, 13,718 km^2 (5,296 mi^2) of native CRP and 1,425 km^2 (550 mi^2) of introduced CRP was implemented, corresponding to 8.5% and 0.9% of the Action Area, respectively. Overall, 17,030 km^2 (6,575 mi^2) of the land area was enrolled in the three conservation practices, corresponding to 10.5% of the Action Area. We calculated the percentage of population size in the Action Area for each conservation practice and considered 90% CIs excluding the regional availability of the practice as evidence for disproportionate population responses to the practices.

Conservation Reserve Program

In 2016, introduced CRP plantings showed proportionally larger contributions to the population size relative to availability for the Cassin's sparrow, eastern meadowlark, mourning dove, northern bobwhite and western kingbird (Fig. 14, Table S2, available in Supporting Information). The introduced CRP plantings conserved 2.3% (SE = 0.5; CI = 1.5, 3.2) of the Cassin's sparrow population in the Action Area, representing a population size of $\widehat{N}$ = 68,053 (SE = 7,296; CI = 57,078, 81,138). Introduced CRP conserved 3.1% (SE = 0.8; CI = 1.8, 4.6) of the eastern meadowlark population, corresponding to a population size of $\widehat{N}$ = 35,760 (SE = 4,849; CI = 28,640, 44,651). The introduced CRP plantings conserved 1.5% (SE = 0.2; CI = 1.1, 1.9) of the mourning dove population in the Action Area, representing a population size of $\widehat{N}$ = 28,825 (SE = 2,808; CI = 24,566, 33,823). Introduced CRP conserved 1.7% (SE = 0.4; CI = 1.1, 2.4) of the northern bobwhite population, corresponding to a population size of $\widehat{N}$ = 12,921 (SE = 2,044; CI = 9,976, 16,734). The introduced CRP plantings conserved 1.8% (SE = 0.5; CI = 1.0, 2.7) of the western kingbird population in the Action Area, representing a population size of $\widehat{N}$ = 17,215 (SE = 3,687; CI = 12,150, 24,391).

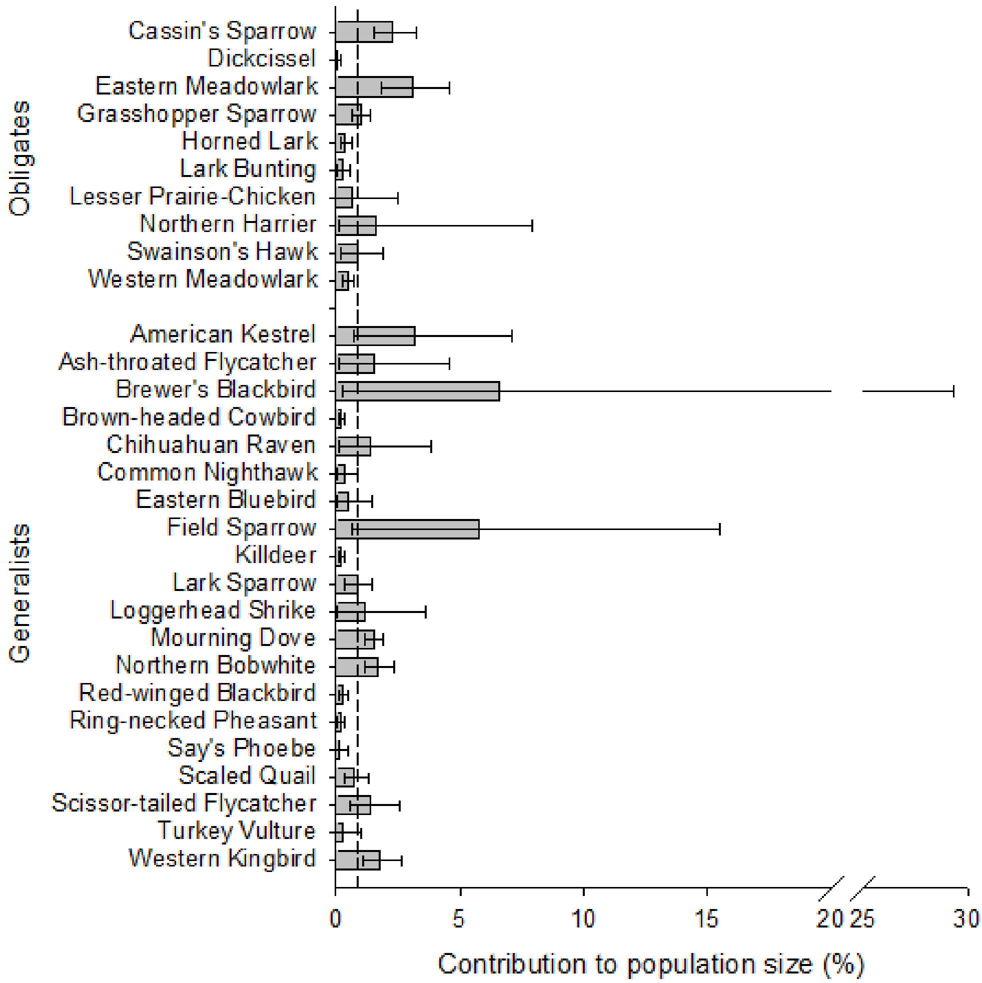

Figure 14. The percentage contribution of introduced Conservation Reserve Program (CRP) plantings to bird populations in the Action Area defined by the occupied range of the lesser prairie-chicken plus 16 km buffer, Colorado, Kansas, New Mexico, Oklahoma and Texas, 2016. The error bars represent 90% Confidence Intervals for the percentage contribution and the vertical dashed line represents the availability of introduced CRP plantings in the Action Area (0.9%).

Other grassland species showed proportionally lower contributions to population size relative to the availability of introduced CRP plantings in 2016, including the brown-headed cowbird, common nighthawk, dickcissel, horned lark, killdeer, lark bunting, red-winged blackbird, ring-necked pheasant, Say's pheobe and western meadowlark (Fig. 14, Table S2, available in Supporting Information). The remainder of the grassland species showed contributions to population size in proportion to the availability of introduced CRP in the Action Area (Fig. 14, Table S2, available in Supporting Information).

In 2016, native CRP plantings showed proportionally larger contributions to the population size relative to availability for the grasshopper sparrow (Fig. 15, Table S2, available in Supporting Information). The native CRP plantings conserved 14.0% (SE = 2.6; CI = 9.9, 18.6) of the grasshopper sparrow population in the Action Area, representing a population size of $\widehat{N}$ = 1,435,084 (SE = 189,528; CI = 1,155,936, 1,781,643).

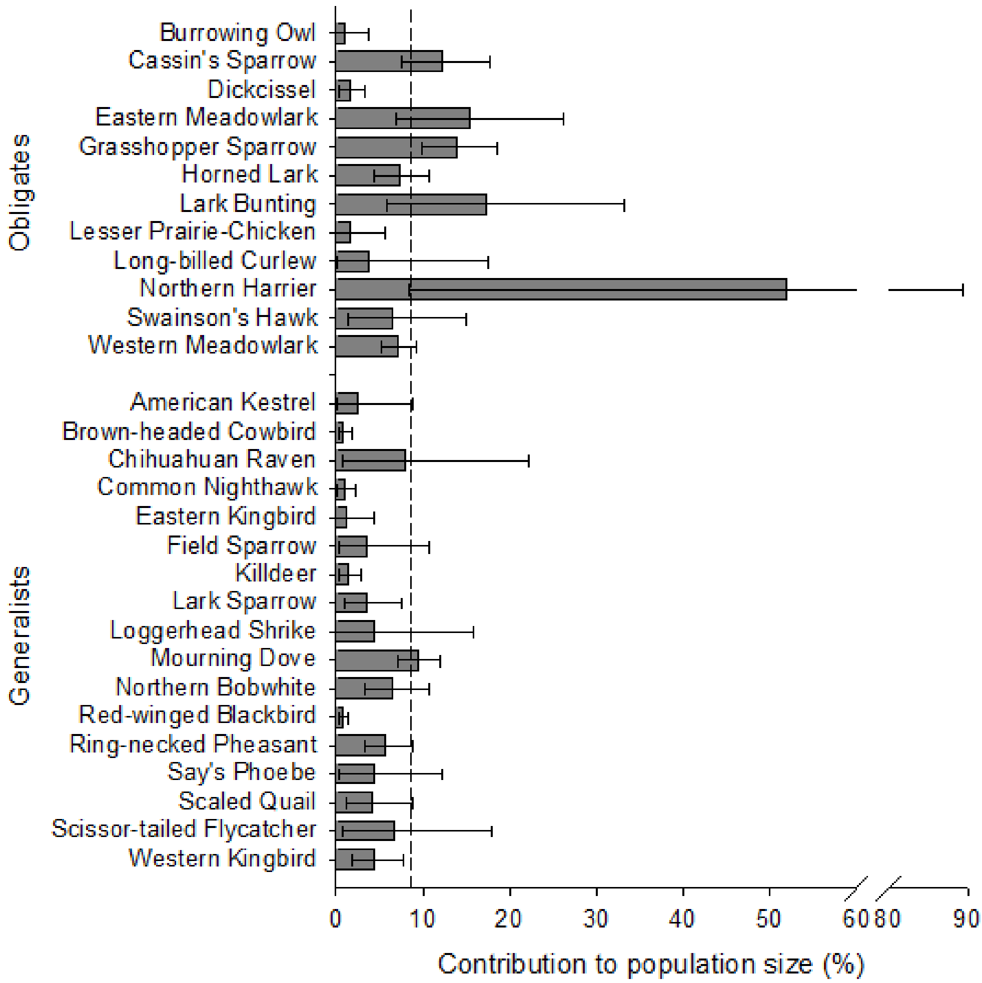

Figure 15. The percentage contribution of native Conservation Reserve Program (CRP) plantings to bird populations in the Action Area defined by the occupied range of the lesser prairie-chicken plus 16 km buffer, Colorado, Kansas, New Mexico, Oklahoma and Texas, 2016. The error bars represent 90% Confidence Intervals for the percentage contribution and the vertical dashed line represents the availability of native CRP plantings in the Action Area (8.5%).

Several grassland species showed proportionally lower contributions to population size relative to the availability of native CRP plantings in 2016, including the brown-headed cowbird, burrowing owl, common nighthawk, dickcissel, eastern kingbird, killdeer, lark sparrow, lesser prairie-chicken, red-winged blackbird and western kingbird (Fig. 15, Table S2, available in Supporting Information). The remainder of the grassland species showed contributions to population size in proportion to the availability of native CRP in the Action Area (Fig. 15, Table S2, available in Supporting Information).

Prescribed Grazing

In 2016, LPCI prescribed grazing showed proportionally larger contributions to the population size relative to availability for the Cassin's sparrow, eastern meadowlark, field sparrow, lark sparrow and turkey vulture (Fig. 16, Table S2, available in Supporting Information). The LPCI prescribed grazing practice conserved 3.0% (SE = 0.6; CI = 2.0, 4.1) of the Cassin's sparrow population in the Action Area, representing a population size of $\widehat{N}$ = 88,666 (SE = 5,292; CI =

80,381, 97,804). Prescribed grazing conserved 2.5% (SE = 0.7; CI = 1.4, 3.9) of the eastern meadowlark population, corresponding to a population size of $\widehat{N}$ = 29,238 (SE = 4,880; CI = 22,260, 38,404). The LPCI grazing practice conserved 7.0% (SE = 4.4; CI = 1.5, 16.0) of the field sparrow population, corresponding to a population size of $\widehat{N}$ = 26,284 (SE = 13,355; CI = 11,948, 57,819). Prescribed grazing conserved 3.2% (SE = 1.1; CI = 1.6, 5.2) of the lark sparrow population, representing a population size of $\widehat{N}$ = 32,978 (SE = 8,135; CI = 22,110, 49,188). The LPCI grazing practice conserved 22.9% (SE = 16.2; CI = 3.2, 53.1) of the turkey vulture population, corresponding to a population size of $\widehat{N}$ = 5,531 (SE = 3,320; CI = 2,221, 13,775).

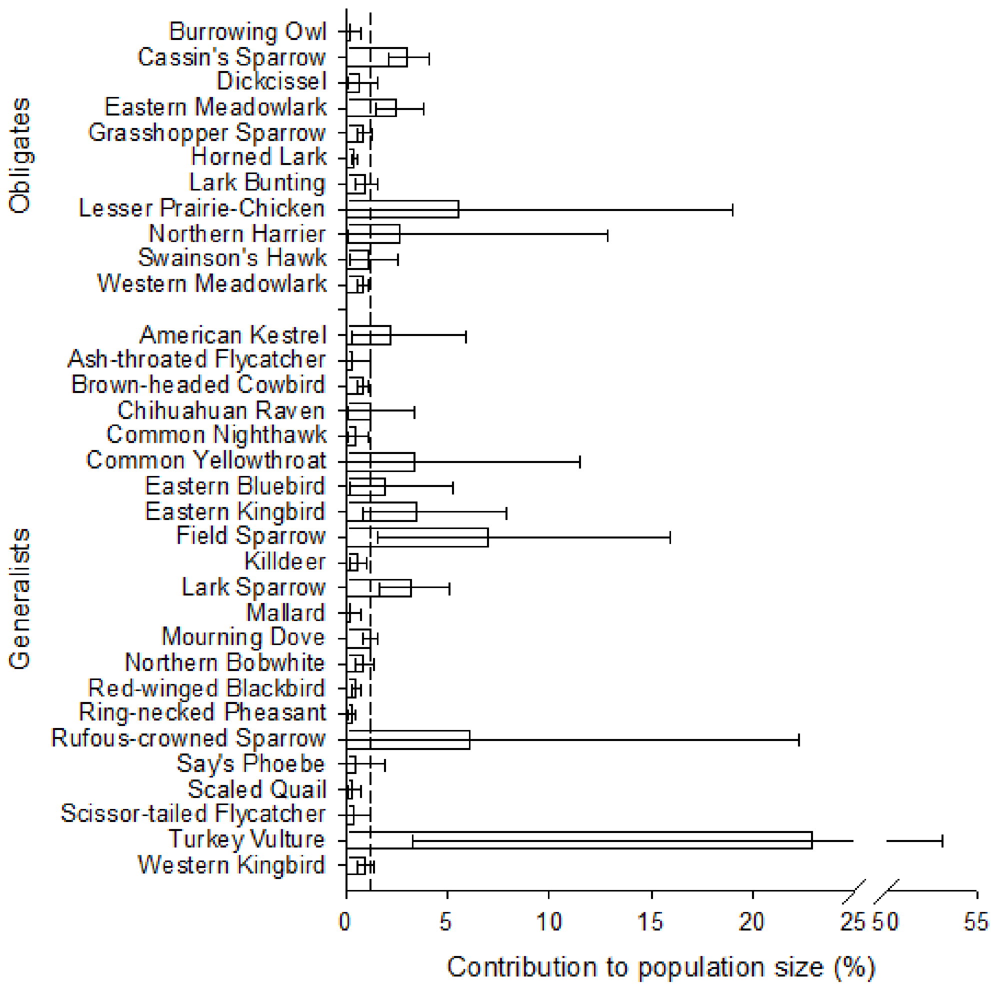

Figure 16. The percentage contribution of Lesser Prairie-Chicken Initiative (LPCI) prescribed grazing to bird populations in the Action Area defined by the occupied range of the lesser prairie-chicken plus 16 km buffer, Colorado, Kansas, New Mexico, Oklahoma and Texas, 2016. The error bars represent 90% Confidence Intervals for the percentage contribution and the vertical dashed line represents the availability of LPCI prescribed grazing in the Action Area (1.2%).

Several grassland species showed proportionally lower contributions to population size relative to the availability of LPCI grazing in 2016, including the brown-headed cowbird, burrowing owl, common nighthawk, horned lark, killdeer, mallard, red-winged blackbird, ring-necked pheasant, scaled quail and western meadowlark (Fig. 16, Table S2, available in Supporting Information). The remainder of the grassland species showed contributions to population size in proportion to the availability of LPCI prescribed grazing in the Action Area (Fig. 16, Table S2, available in Supporting Information).

Overall Conservation

In 2016, the three conservation practices showed proportionally larger contributions to population size relative to availability for the Cassin's sparrow, eastern meadowlark and grasshopper sparrow (Fig. 17, Table S2, available in Supporting Information). The three conservation practices accounted for 17.4% (SE = 4.0; CI = 11.3, 24.5) of the Cassin's sparrow population in the Action Area, representing a population size of $\widehat{N}$ = 518,117 (SE = 61,681; CI = 426,263, 629,766). Overall conservation accounted for 20.9% (SE = 6.9; CI = 10.8, 33.2) of the eastern meadowlark population in the Action Area, representing a population size of $\widehat{N}$ = 244,146 (SE = 55,848; CI = 168,386, 353,991). The three conservation practices accounted for 15.8% (SE = 2.8; CI = 11.4, 20.7) of the grasshopper sparrow population in the Action Area, representing a population size of $\widehat{N}$ = 1,625,418 (SE = 191,598; CI = 1,339,809, 1,971,911).

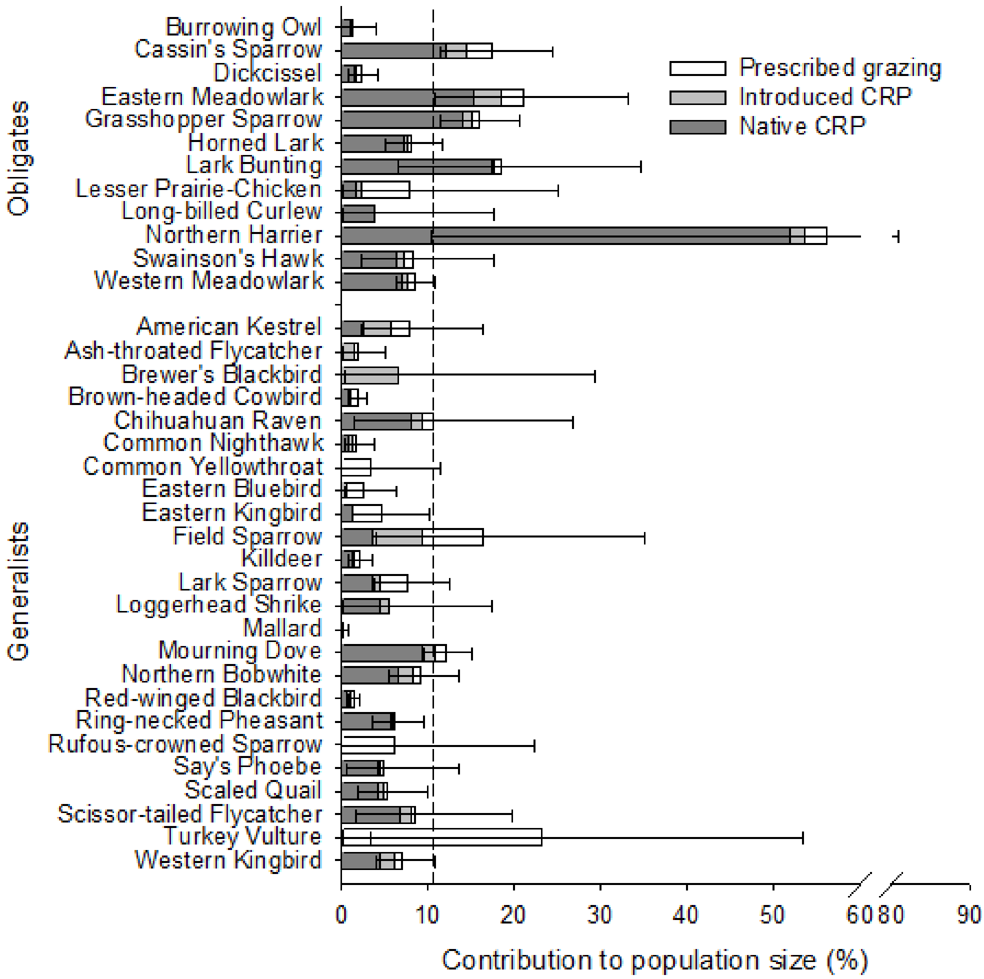

Figure 17. The percentage contribution of Lesser Prairie-Chicken prescribed grazing, introduced Conservation Reserve Program (CRP) and native CRP to bird populations in the Action Area defined by the occupied range of the lesser prairie-chicken plus 16 km buffer, Colorado, Kansas, New Mexico, Oklahoma and Texas, 2016. The error bars represent 90% Confidence Intervals for the percentage contribution and the vertical dashed line represents the availability of the three conservation practices in the Action Area (10.5%).

In 2016, several grassland species showed proportionally lower contributions to population size relative to the availability of the three conservation practices, including the ash-throated flycatcher, brown-headed cowbird, burrowing owl, common nighthawk, dickcissel, eastern bluebird (*Sialia sialis*), eastern kingbird, killdeer, mallard, red-winged blackbird, ring-necked pheasant and scaled quail (Fig. 17, Table S2, available in Supporting Information). The remainder of the grassland species showed contributions to population size in proportion to the availability of the three conservation practices in the Action Area (Fig. 16, Table S2, available in Supporting Information).

Discussion

The objectives of the study were to determine 1) the effectiveness of LEPC conservation practices for increasing the site occupancy and biodiversity of grassland birds, 2) the influence of landscape and local vegetation relationships on site occupancy and biodiversity and 3) the effectiveness of the conservation practices for increasing the density and population size of grassland birds. We used the life histories of grassland bird species (Knopf 1996, Vickery and Herkert 1999, Rodewald 2019) to develop predictions for the responses of grassland birds to conservation practices. We studied the main Farm Bill practices (Briske et al. 2017) available to manage LEPC habitat on private land, including CRP and LPCI prescribed grazing (USFWS 2011, Van Pelt et al. 2013). The results of the study provided an evaluation of the effectiveness of conservation practices for increasing the distribution and abundance of grassland birds. Monitoring the effectiveness of the conservation practices was ultimately useful for evaluating the extent that the LEPC serves as an umbrella species for the conservation for grassland birds (Roberge and Angelstam 2004, Favreau et al. 2006). We conclude, by discussing the usefulness of the results for informing the conservation of biodiversity at multiple scales, and the extent that local management actions contribute to regional bird populations.

Site Occupancy and Species Richness

Conservation Reserve Program

Our results indicated native and introduced CRP plantings to restore agricultural lands are important conservation practices for increasing the biodiversity of grassland bird in the southern Great Plains. Land enrolled in the CRP program is an important conservation practice for increasing the large-scale occupancy of the LEPC in the SGPR and MGPR ecoregions (Carlisle et al. 2018). We evaluated community and species responses to the conservation practices in terms of grassland obligate or facultative (generalist) species (Vickery and Herkert 1999). In addition, we evaluated shifts in species composition in terms of species showing apparent population declines in the central region of the Breeding Bird Survey (Sauer et al. 2017). We found both native and introduced CRP plantings increased the alpha species richness of grassland obligates relative to agricultural lands. In contrast to the findings of Bakker and Higgins (2009), introduced CRP plantings increased the species richness of grassland generalists, and species richness of generalists was similar in native CRP and agricultural lands. Similar to the findings of Thompson et al. (2009), we were unable to confirm the hypothesis for greater species richness in native CRP plantings relative to introduced CRP plantings, but a shift in species composition indicated grassland obligates showed larger positive responses to native CRP plantings relative to introduced CRP plantings than generalist species. Of the 4 species favoring native over introduced CRP plantings, 3 are obligates and 2 are declining in the Great Plains. Of the 6 species favoring introduced over native CRP plantings, 2 are obligates and 4 are declining. Overall, the treatment effects for taking cropland out of production and planting CRP was important for the species composition of grassland obligates and declining species. Of the 5 species favoring native CRP over agricultural reference lands, 3 are obligates and 3 are declining. Of the 6 species favoring introduced CRP over agricultural reference lands, 4 are obligates and 5 are declining.

Prescribed Grazing

Our findings indicated LPCI prescribed grazing to improve rangeland condition is an important practice for increasing the biodiversity of grassland obligates and declining species. Although unmanaged grazing may pose a threat to LEPC nesting habitat (Hagen et al. 2004), conservative grazing practices are known to be compatible with LEPC nesting habitat and nest survival (Fritts et al. 2016). The rest-rotation grazing systems employed by the LPCI prescribed grazing practice were expected to produce heterogeneity in grassland structure and increase the biodiversity of grassland birds (Derner et al. 2009). Lands enrolled in LPCI prescribed grazing showed greater species richness of grassland obligates relative to grassland generalists. The species richness of grassland generalists was lower on LPCI rangelands than reference grasslands. However, we were unable to confirm the hypothesis for greater alpha species of grassland obligates on LPCI rangelands relative to reference grasslands. Nevertheless, LPCI prescribed grazing appeared to shift species composition toward a community of grassland obligates and species that are currently declining. Of the 8 species favoring LPCI rangelands over reference grasslands, 5 are obligates and 6 are declining. In contrast, of the 14 species favoring reference grasslands over LPCI rangelands, 2 are obligates and 8 are declining in the Great Plains.

Landscape and Local Vegetation Structure

We investigated landscape relationships to better understand biodiversity responses to the loss and fragmentation of native vegetation in the Action Area. Our results suggested declining grassland obligates were more sensitive to the loss of grassland land-cover than the fragmentation of native vegetation, but more grassland obligates favored landscapes with large patches of native vegetation. The mean patch size of native grassland and land-cover of shrub-land are important drivers of the large-scale occupancy distribution of the LEPC (Carlisle et al. 2018). However, we were unable to confirm hypotheses for changes in gamma species richness of grassland obligates or generalists along gradients of mean patch size of native vegetation or land-cover of shrub-land vegetation. Nevertheless, we observed greater variation in species composition along the gradient of landscape composition than the gradient of landscape configuration, suggesting the grassland bird community may be responding to the loss of grassland rather than the fragmentation of native vegetation (Fahrig 2003). Future research to investigate the interactions between landscape composition and configuration may reveal additional insight into the fragmentation of grassland and shrub-land vegetation. The relative importance of habitat loss and fragmentation to biodiversity has important conservation implications (Fischer and Lindenmayer 2007). For example, when habitat loss is more important than habitat fragmentation, implementing CRP in a way that maximizes the percentage of suitable habitat in any configuration may be a more effective conservation strategy than managing the patch configuration of native vegetation. Of the 6 species favoring large patches of native vegetation, 2 are obligates and 4 are declining. Of the 5 species favoring small patches of native vegetation, 1 is an obligate and 3 are declining. In terms of landscape composition, of the 9 species favoring grassland land cover, 5 are obligates and 6 are declining. Of the 7 species increasing with shrubland land cover, 2 are obligates and 1 is declining.

We investigated local vegetation relationships to better understand the mechanisms for biodiversity responses to the LEPC conservation practices. Herbaceous ground cover and grass height, as well as shrub cover and height, have important implications for LEPC nesting habitat and nest survival (Hagen et al. 2004). In addition, variation in species responses to heterogeneity in vegetation structure is expected to result in high species richness at larger spatial scales (Derner et al. 2009, Hovick et al. 2015). Our results suggested CRP and LPCI prescribed grazing practices that increase the ground cover of herbaceous vegetation play an important role in increasing the biodiversity of grassland birds. The alpha species richness of grassland obligates and generalists increased with increasing herbaceous ground cover of grass and forbs, but the species richness of obligates and generalists did not vary with grass height. Of the 8 species favoring high herbaceous ground cover, 4 are obligates and 5 are declining. None of the grassland species declined with increasing herbaceous ground cover. Of the 6 species favoring tall grass height, 2 are obligates and 3 are declining. Of the 2 species favoring short grass height, all 2 are obligates and both are declining.

In terms of the shrub component, the alpha species richness of grassland generalists increased with shrub cover and height, but we were unable to confirm hypotheses for declining species richness of grassland obligates with increasing shrub cover and height. However, the majority of species responding negatively to shrub cover and height were grassland obligates currently experiencing population declines. Our results suggested land enrolled in CRP and LPCI prescribed grazing practices at the low-end of shrub cover and height provide important habitat for obligate grassland species of conservation concern, and LPCI rangelands with a substantial shrub component promote the species richness of grassland generalists. Of the 8 species increasing with shrub canopy cover, 2 are obligates and 4 are declining. Of the 3 species declining with increasing shrub cover, all 3 are obligates and are declining. Of the 15 species increasing with shrub height, 3 are obligates and 9 were declining. Of the 4 species declining with increasing shrub height, 3 are obligates and all 4 are declining.

We investigated habitat relationships for tree canopy cover and height to predict the responses of grassland bird species to LEPC management actions for the encroachment of woodland vegetation. The large-scale occupancy distribution of the LEPC is negatively influenced by woodland cover (Carlisle et al. 2018), and tree removal may facilitate grassland habitat restoration and range expansion of the LEPC (Lautenbach et al. 2017). We were unable to confirm hypotheses for declining species richness of grassland obligates or increasing species richness of generalists along gradients of tree canopy cover and height. The range of covariate values for tree canopy cover and height (Table 4) may not have contained enough information to evaluate hypotheses for variation in species richness along gradients of woody vegetation. Alternately, because we did not investigate tree cover at the landscape scale, species responses to tree cover at the landscape scale may constrain the responses to tree canopy cover at the local scale. Nevertheless, similar to the findings of Coppedge et al. (2001), we observed a shift in species composition with a greater number of declining grassland obligates occurring at low levels of tree canopy cover and tree height, and a greater number of declining generalists at high levels of tree canopy cover and tree height. Our results suggest tree removal may benefit several grassland obligates currently experiencing population declines, but may be detrimental to several species of declining grassland generalists. Of the 3 species declining with increasing tree canopy cover, all 3 are obligates and all 3 are declining. Only one species, increased with increasing tree canopy cover, the northern bobwhite, and this species is a declining grassland generalist. In terms of tree height, of the 6 species declining with increasing tree height, 5 are obligates and 5 are declining. Of the 8 species increasing with tree height, none are obligates and 6 are declining.

Density and Population Size

Conservation Reserve Program

We compared avian population densities on CRP plantings relative to reference grasslands and agricultural lands to better understand how the practices affected the abundance and population size of grassland bird species in the Action Area. The population densities of the grasshopper sparrow and mourning dove were greater on native CRP plantings than reference grasslands and agricultural lands, suggesting native CRP is important for managing the abundance of this species. In 2016, the estimates of population size showed native CRP contributed 1,435,000 grasshopper sparrows, which corresponded to 14% of the population in the Action Area. Because, native CRP plantings accounted for 9% of the Action Area, native CRP plantings made a proportionally larger contribution to the grasshopper sparrow population relative to availability in the region. Considering the annual trend for the grasshopper sparrow was declining by 2% (CI = -2.7, -1.4) in the BBS central region (Sauer et al. 2017), the contribution to the breeding population in the southern Great Plains represents considerable conservation value for this grassland obligate. In addition, native CRP made proportional contributions to population size for 8 of the 16 grassland obligates, and 10 of the 29 grassland generalists (Fig. 15). A proportional 9% contribution of native CRP to population size represents meaningful conservation gains for grassland species declining in the Great Plains. Of these, 5 are declining obligates, including the eastern meadowlark, horned lark, lark bunting, northern harrier and western meadowlark, and 6 are declining grassland generalists, such as the American kestrel, field sparrow, loggerhead shrike, mourning dove, northern bobwhite and scissor-tailed flycatcher (Sauer et al. 2017). In contrast, the densities of the dickcissel, brown-headed cowbird, killdeer, lark sparrow and red-winged blackbird were lower on native CRP than reference grasslands or agricultural lands. Overall, native CRP made proportionally lower contributions to population size relative to availability for 9 grassland species (Fig. 15). Of these, only grassland generalists are declining in the Great Plains (Sauer et al. 2017), including the common nighthawk, eastern kingbird, killdeer, lark sparrow and red-winged blackbird. Because these species require other vegetation types in addition to grassland, native CRP is not expected to provide conservation benefits for these species (Vickery and Herkert 1999).

Introduced CRP made up only 1% of the Action Area, yet several species showed greater population densities on the introduced plantings relative to reference grasslands and agricultural lands, including the Cassin's sparrow, eastern meadowlark, mourning dove, northern bobwhite and western kingbird. The introduced CRP plantings conserved 2% of the Cassin's sparrow population ($\widehat{N}$ = 68,000) that was not declining in the Great Plains, 3% of eastern meadowlark population ($\widehat{N}$ = 36,000) that was declining by 3% (CI = -3.4, -2.5), 2% of the mourning dove population ($\widehat{N}$ = 29,000) that was declining by <1% (CI = -0.6, -0.2), and conserved 2% of the northern bobwhite population ($\widehat{N}$ = 13,000) that was declining by 3% (CI = -3.1, -2.2) in the Great Plains (Sauer et al. 2017). Despite making-up a small percentage of land area in the Action Area, the introduced plantings contributed to the conservation of a declining grassland obligate, and 2 important game species in the region. In addition, introduced CRP plantings contributed to the regional populations in proportion to availability (1%) for 4 of 16 grassland obligates and 11 of 29 grassland generalists.

Of these, 3 were declining grassland obligates, including the grasshopper sparrow, lesser prairie-chicken and northern harrier, and 6 were declining grassland generalists, including the American kestrel, Brewer's blackbird, field sparrow, lark sparrow, loggerhead shrike and scissor-tailed flycatcher. Conversely, avian population densities were lower on introduced CRP relative to reference grasslands or agricultural lands for the brown-headed cowbird, common nighthawk, dickcissel, horned lark, killdeer, lark bunting, red-winged blackbird, ring-necked pheasant and western meadowlark. Overall, introduced CRP made lower contributions to population size relative to availability for 10 grassland species. Of these, 4 species are declining grassland obligates, including the horned lark, lark bunting and western meadowlark, and 3 were declining grassland generalists, including the common nighthawk, killdeer and red-winged blackbird. Introduced CRP plantings are not expected to provide a conservation benefit for these species.

Because native CRP plantings have more diverse seed mixes than introduced CRP plantings (Ripper et al. 2008), we predicted the abundance of grassland bird species requiring heterogeneous vegetation structure (Hovick et al. 2015), such as the eastern meadowlark, grasshopper sparrow and lark bunting, would be greater on native than on introduced CRP plantings. In general, native CRP was more important to declining grassland obligates, whereas introduced CRP was more important for declining grassland generalists. We discovered the population densities of the grasshopper sparrow, horned lark, lark bunting and ring-necked pheasant were greater on native than introduced CRP plantings. All three of the grassland obligates favoring native CRP, the grasshopper sparrow, horned lark and lark bunting, are declining in the Great Plains (Sauer et al. 2017), and the grassland generalist, ring-necked pheasant, is an important socio-economic game species in the region. In contrast, the densities of the American kestrel, Cassin's sparrow, eastern meadowlark, mourning dove, northern bobwhite and western kingbird were lower on native than introduced CRP. Of the species favoring introduced CRP, one species, the eastern meadowlark, is a declining grassland obligate, whereas 3 species, the American kestrel, mourning dove and northern bobwhite, are declining grassland generalists. Because native CRP is 9 times more prevalent than introduced CRP in the region, native CRP has a much greater cumulative impact on declining grassland obligates such as the grasshopper sparrow and lark bunting, whereas introduced CRP provides limited contributions to populations of declining grassland generalists.

Prescribed Grazing

We compared avian population densities on LPCI rangelands relative to reference grasslands and CRP plantings to better understand how prescribed grazing to improve rangeland condition affected the abundance and population size of grassland bird species in the Action Area. The LPCI grazing practice made up only 1% of the Action Area, yet several species showed greater population densities on LPCI rangelands relative to reference grasslands, including the Cassin's sparrow, eastern meadowlark, field sparrow, lesser prairie-chicken, mourning dove, northern bobwhite and turkey vulture. The LPCI rangelands conserved 3% of eastern meadowlark population ($\widehat{N}$ = 26,000) that was declining by 3% in the Great Plains (CI = -3.4, -2.5), 7% of the field sparrow population ($\widehat{N}$ = 29,000) that was declining by 2% (CI = -2.0, -1.1), and conserved 3% of the lark sparrow population ($\widehat{N}$ = 33,000) that was declining by 1% (CI = -1.5, -0.6) in the Great Plains (Sauer et al. 2017). In 2016, LPCI grazing practice contributed 89,000 Cassin's sparrows, which corresponded to 3% of the population for this grassland obligate in the Action Area. The positive response of the Cassin's sparrow to LPCI prescribed grazing is notable because this grassland obligate is sensitive to intensive grazing pressure (Rodewald 2019). In addition, LPCI prescribed grazing contributed to the regional populations in proportion to availability (1%) for 7 of the 16 grassland obligates, and 14 of the 29 grassland generalists. Of these, 4 are declining grassland obligates, including the grasshopper sparrow, lark bunting, lesser prairie-chicken and northern harrier, and are declining grassland generalists, including the American kestrel, common nighthawk, common yellowthroat, eastern kingbird, mourning dove, northern bobwhite, rufous-crowned sparrow and scissor-tailed flycatcher. In contrast, avian population densities were lower on LPCI rangelands relative to reference grasslands for the grasshopper sparrow, horned lark, scaled quail and western meadowlark. Overall, introduced CRP made lower contributions to population size relative to availability for 10 grassland species. Of these, the horned lark and western meadowlark are declining grassland obligates, and the killdeer and red-winged blackbird are declining grassland generalists.

The apparent habitat suitability of LPCI rangelands was greater than native CRP plantings for the brown-headed cowbird, Cassin's sparrow, dickcissel, eastern kingbird, field sparrow, killdeer, lesser prairie-chicken and red-winged blackbird. In contrast, avian densities were lower on LPCI rangelands than native CRP plantings for the grasshopper sparrow, horned lark and ring-necked pheasant. The habitat suitability of LPCI rangelands was greater than introduced CRP plantings for the brown-headed cowbird, burrowing owl, dickcissel, eastern kingbird, lark bunting, lark sparrow, lesser-prairie-chicken and turkey vulture. Conversely, population densities were lower on LPCI rangelands than introduced CRP plantings for the eastern meadowlark, grasshopper sparrow, mourning dove, northern bobwhite, scaled quail and western kingbird. The positive responses of the Cassin's sparrow, dickcissel, eastern kingbird, field sparrow, lark bunting, lark sparrow and lesser prairie-chicken to LPCI prescribed grazing relative to the CRP plantings correspond to the known habitat affinities of the species (Rodewald 2019), and may be related to the greater heterogeneity in vegetation structure (Derner et al. 2009) provided by the combination rotational grazing and increased shrub cover. The negative responses to LPCI prescribed grazing relative to native CRP plantings suggested the vertical structure of managed rangelands may be too tall for some grassland obligates, such as the grasshopper sparrow, horned lark and western meadowlark (Rodewald 2019).

Conclusions

Monitoring to understanding the effectiveness of conservation practices plays an important role in the management of natural resources (Lyons et al. 2008). Long-term trends are useful in conservation planning to prioritize and assess the vulnerability of species (Brennan and Kuvlesky 2005, Rosenberg et al. 2017), but because restoration may be necessary to stabilize grassland bird populations (Vickery and Herkert 1999), understanding the relative performance of available management actions to achieve wildlife objectives is crucial for conservation (Nichols and Williams 2006). Monitoring the effectiveness of Farm Bill conservation practices increases the confidence of resource professionals and promotes accountability in the public trust toward meeting intended objectives (Briske et al. 2017). Because a large percentage of the southern Great Plains are privately owned, the recovery of the LEPC and other grassland bird species depends on conservation initiatives with strong partnerships between private landowners and resource professionals (Van Pelt et al. 2013). Conservation success for grassland birds in the southern Great Plains may well depend on rewarding private landowners for conserving public interests (Briske et al. 2017).

Habitat management for the LEPC is expected to benefit other grassland bird species (USFWS 2011, Haukos and Boal 2016), but because each species has different habitat requirements (Lindenmayer et al. 2002), monitoring is necessary to determine which species are benefitted by umbrella species conservation (Favreau et al. 2006, Seddon and Leech 2008). Although the effectiveness of umbrella species conservation often investigates overlap in habitat use,(Favreau et al. 2006), umbrella species are defined as a species whose conservation confers protection to a number of naturally co-occurring species (Roberge and Angelstam 2004). By studying the effectiveness of conservation practices for increasing biodiversity and abundance of grassland birds, our study provides more direct test of the umbrella species hypothesis than merely investigating overlap in habitat use.

We developed a hierarchical framework for evaluating the responses of avian biodiversity (Whittaker et al. 2001, Bestelmeyer et al. 2003) to Farm Bill conservation practices (Briske et al. 2017) aimed at the recovery of the LEPC (USFWS 2011, Van Pelt et al. 2013). At the landscape scale, our results suggested declining grassland obligates were more sensitive to the loss of grassland than the fragmentation of native vegetation. This suggested a conservation strategy to minimize the loss of grassland may be more effective for grassland obligates than landscape conservation to maintain large patches of grassland. At the local scale, conservation practices to increase herbaceous ground cover is expected to increase the species richness of grassland birds. The multi-species habitat relationships suggested that shrub management to reduce shrub cover and height may increase the site-occupancy of declining grassland obligates, but may negatively affect the species richness of grassland generalists that also require shrub components in addition to grassland.

We found native and introduced CRP plantings to restore agricultural lands and LPCI prescribed grazing to improve rangeland condition were important conservation practices for increasing the biodiversity of grassland birds in the Action Area. Species richness was greater on CRP plantings than agricultural lands, but was similar in native and introduced CRP plantings. The species richness of grassland generalists was lower on native than introduced plantings, whereas grassland obligates showed stronger responses to native than introduced plantings. The species richness of grassland obligates was similar on LPCI rangelands and reference grasslands, but LPCI prescribed grazing appeared to shift species composition toward a community of declining grassland obligates. The species richness of grassland generalists was lower on LPCI rangelands than reference grasslands, and LPCI rangelands supported more grassland obligate species than generalist species.

The results suggested declining enrollment of native CRP and concomitant increases in agricultural lands have the potential to reduce the biodiversity of grassland obligate bird species. Increases in introduced CRP relative to native CRP may benefit game species such the mourning dove and northern bobwhite, but may shift species composition from grassland obligates to a community dominated by grassland generalists. The five-year enrollment of LPCI prescribed grazing increased from 2015 (1,422 km^2) to 2017 (2,296 km^2), and this is expected to shift species composition on working rangelands from a community of grassland generalists to a community of grassland obligates that are declining in the region.

We investigated avian abundance to better understand the extent that local management contributes to regional populations of grassland birds (Pavlacky et al. 2017). Our results indicate the voluntary conservation practices aimed at recovering LEPC populations on private land have made meaningful contributions to the regional population sizes of several declining grassland species. In 2016, the CRP plantings and LPCI prescribed grazing practices accounted for 11% of the land area in the Action Area, and the practices made proportionally larger contributions to population size relative to availability for three grassland obligates. The practices conserved 17% of the Cassin's sparrow population ($\widehat{N}$ = 518,000) that was not declining in the Great Plains, 21% of the eastern meadowlark population ($\widehat{N}$ = 244,000) that was declining by 3% (CI = -3.4, -2.5), and conserved 16% of the grasshopper sparrow population ($\widehat{N}$ = 1,625,000) that was declining by 2% (CI = -2.7, -1.4) in the Great Plains (Sauer et al. 2017). Introduced CRP plantings and LPCI prescribed grazing provided proportionally larger contributions to population size for the Cassin's sparrow and eastern meadowlark, and native CRP plantings made proportionally larger contributions to population size for the grasshopper sparrow. The three conservation practices contributed to population size in proportion to availability (11%) for 7 of the 16 grassland obligates, and 14 of the 29 grassland generalists. Of these, 4 are declining grassland obligates, including the horned lark, lark bunting, northern harrier and western meadowlark, and 10 are declining grassland generalists, including the American kestrel, Brewer's blackbird, common yellowthroat, field sparrow, lark sparrow, loggerhead shrike, mourning dove, northern bobwhite, rufous-crowned sparrow and scissor-tailed flycatcher. In contrast, the practices showed proportionally lower contributions to population size for 12 species. Of these, only grassland generalists, including the common nighthawk, eastern kingbird, killdeer and red-winged blackbird are declining in the Great Plains (Sauer et al. 2017).

Finally, monitoring the effectiveness of the conservation practices may be useful for decision making to determine the combination of management actions that best satisfy wildlife objectives in the Action Area (Nichols and Williams 2006, Lyons et al. 2008). For example, a conservation planning objective could be developed to set thresholds for the population performance of an umbrella species (Nicholson and Possingham 2006), and then above the threshold, maximize the cumulative occupancy of target species (Sauer et al. 2013) and maximize the abundance of declining species (Bunnell 2004). Because a large percentage of the southern Great Plains are privately owned, conservation success in the region will depend on defining objectives representing the values and interests of private landowners. Changes in the enrollment of the voluntary conservation practices are expected to have large effects on avian biodiversity, and declining eastern meadowlark and grasshopper sparrow populations. Because the Cassin's sparrow is endemic to the southern Great Plains (Rodewald 2019), reductions in the enrollment of the voluntary conservation practices may place the Cassin's sparrow population in jeopardy. We recommend a proactive conservation strategy that addresses the "*what to do*" and "*where to do it*" questions in conservation planning (Wilson et al. 2009), considers the effectiveness of available conservation practices (Lyons et al. 2008), and provides the best outcomes for private landowner interests and wildlife conservation in the region.

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Appendix

Table A1. The common name, scientific name, guild (Vickery and Herkert 1999, Johnsgard 2009) and declining trends in the Great Plains from the Breeding Bird Survey (Sauer et al. 2017) for the 45 grassland bird species observed in the study, occupied range of the lesser prairie-chicken, Colorado, Kansas, New Mexico, Oklahoma, and Texas, 2015 - 2017.

Common name	Scientific name	Guild	Trend
Burrowing Owl	*Athene cunicularia*	Obligate	No
Cassin's Sparrow	*Peucaea cassinii*	Obligate	No
Dickcissel	*Spiza americana*	Obligate	No
Eastern Meadowlark	*Sturnella magna*	Obligate	Yes
Ferruginous Hawk	*Buteo regalis*	Obligate	No
Grasshopper Sparrow	*Ammodramus savannarum*	Obligate	Yes
Horned Lark	*Eremophila alpestris*	Obligate	Yes
Lark Bunting	*Calamospiza melanocorys*	Obligate	Yes
Lesser Prairie-Chicken	*Tympanuchus pallidicinctus*	Obligate	-
Long-billed Curlew	*Numenius americanus*	Obligate	No
Mountain Plover	*Charadrius montanus*	Obligate	Yes
Northern Harrier	*Circus cyaneus*	Obligate	Yes
Short-eared Owl	*Asio flammeus*	Obligate	No
Swainson's Hawk	*Buteo swainsoni*	Obligate	No
Vesper Sparrow	*Pooecetes gramineus*	Obligate	No
Western Meadowlark	*Sturnella neglecta*	Obligate	Yes
American Kestrel	*Falco sparverius*	Facultative	Yes
Ash-throated Flycatcher	*Myiarchus cinerascens*	Facultative	No
Blue-winged Teal	*Anas discors*	Facultative	No
Brewer's Blackbird	*Euphagus cyanocephalus*	Facultative	Yes
Brown-headed Cowbird	*Molothrus ater*	Facultative	No
Canyon Towhee	*Melozone fusca*	Facultative	Yes
Cassin's Kingbird	*Tyrannus vociferans*	Facultative	No
Cattle Egret	*Bubulcus ibis*	Facultative	No
Chihuahuan Raven	*Corvus cryptoleucus*	Facultative	No
Common Nighthawk	*Chordeiles minor*	Facultative	Yes
Common Yellowthroat	*Geothlypis trichas*	Facultative	Yes
Eastern Bluebird	*Sialia sialis*	Facultative	No
Eastern Kingbird	*Tyrannus tyrannus*	Facultative	Yes
Field Sparrow	*Spizella pusilla*	Facultative	Yes
Killdeer	*Charadrius vociferus*	Facultative	Yes
Lark Sparrow	*Chondestes grammacus*	Facultative	Yes
Loggerhead Shrike	*Lanius ludovicianus*	Facultative	Yes
Mallard	*Anas platyrhynchos*	Facultative	No
Mourning Dove	*Zenaida macroura*	Facultative	Yes
Northern Bobwhite	*Colinus virginianus*	Facultative	Yes
Northern Shoveler	*Anas clypeata*	Facultative	No
Red-winged Blackbird	*Agelaius phoeniceus*	Facultative	Yes
Ring-necked Pheasant	*Phasianus colchicus*	Facultative	No
Rufous-crowned Sparrow	*Aimophila ruficeps*	Facultative	Yes
Say's Phoebe	*Sayornis saya*	Facultative	No

Common name	Scientific name	Guild	Trend
Scaled Quail	*Callipepla squamata*	Facultative	No
Scissor-tailed Flycatcher	*Tyrannus forficatus*	Facultative	Yes
Turkey Vulture	*Cathartes aura*	Facultative	No
Western Kingbird	*Tyrannus verticalis*	Facultative	No