Integrated Monitoring in Bird Conservation Regions (IMBCR) for Playa Lakes Joint Venture (PLJV): 2016 - 2020 Conservation Reserve Program Report

April 2021

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Bird Conservancy of the Rockies

Connecting people, birds and land

Mission: Conserving birds and their habitats through science, education and land stewardship

Vision: Native bird populations are sustained in healthy ecosystems

Bird Conservancy of the Rockies conserves birds and their habitats through an integrated approach of science, education, and land stewardship. Our work radiates from the Rockies to the Great Plains, Mexico and beyond. Our mission is advanced through sound science, achieved through empowering people, realized through stewardship, and sustained through partnerships. Together, we are improving native bird populations, the land, and the lives of people.

Core Values:

1. **Science** provides the foundation for effective bird conservation.
2. **Education** is critical to the success of bird conservation.
3. **Stewardship** of birds and their habitats is a shared responsibility.

Goals:

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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Grasshopper sparrow by Alan Schmierer

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Executive Summary

Bird Conservancy of the Rockies (Bird Conservancy), in conjunction with its partners, conducted landbird monitoring for the tenth year in a row for the Integrated Monitoring in Bird Conservation Regions (IMBCR) program. The IMBCR for Playa Lakes Joint Venture (PLJV) program is a collaborative partnership for evaluating and implementing wildlife conservation in the Shortgrass Prairie and Central Mixed grass Prairie Bird Conservation Regions. The partnership was designed to address management and conservation needs of a wide range of stakeholders including private landowners, conservation initiatives, federal agencies and state wildlife agencies. IMBCR uses a spatially balanced sampling design which allows inferences to avian species occurrence and population sizes at various scales, from local management units to entire Bird Conservation Regions (BCR) or states, facilitating conservation at local and national levels. The design of the IMBCR program is based on a stratified random sample obtained by separating the region into non-overlapping groups called strata, and then selecting a random sample from each stratum. The sampling design allows analysts to estimate species densities, population sizes, and occupancy rates for individual strata or biologically meaningful combinations of strata. The IMBCR design provides a spatially consistent and flexible framework for understanding the status and annual changes of bird populations. Collaboration across organizations and spatial scales increases sample sizes and improves the accuracy and precision of population estimates. Analyzing the data collectively allows us to estimate detection probabilities for species that would otherwise have insufficient numbers of detections at local scales.

The IMBCR program is well-positioned to address the conservation and management needs of a wide range of stakeholders due to the hierarchical design and IMBCR partnership. Population monitoring within BCRs can be implemented with a flexible hierarchical framework of nested units, where information on status of bird populations can be partitioned into smaller units for small-scale conservation planning, or aggregated to support large-scale conservation efforts throughout a species' geographic range. By focusing on scales relevant to management and conservation, information obtained from monitoring in BCRs can be integrated into research and management at various scales applicable to land managers. Post-stratification provides a way to stratify or group point-count plots by a factor of interest after data collection, and estimate density for different groups, such as vegetation types. Post-stratifying IMBCR point-count data by vegetation types and conservation practices provides a framework for effectiveness monitoring to learn about the success of management actions. The Conservation Reserve Program (CRP) is a voluntary program for agricultural producers administered by Farm Service Agency providing incentives to landowners to take cropland out of production and plant it back into grassland. The objectives of this report are to 1) evaluate avian population density on CRP lands relative to agricultural lands and native grasslands, and 2) estimate the contributions of CRP lands to bird populations in the PLJV region.

This report summarizes IMBCR results from 2016 through 2020 for 13 grassland priority species identified in the PLJV Landbird Team Report (2007), including a post-stratification analysis to estimate avian population density on CRP lands, agricultural lands and native grassland. To view interactive maps illustrating survey and detection locations, species counts and density, population and occupancy results, please visit Bird Conservancy's Rocky Mountain Avian Data Center (Rocky Mountain Avian Data Center, www.rmbo.org/v3/avian/ExploretheData.aspx, accessed 6 Jun 2018). Instructions for using the Avian Data Center are included in Appendix A of this report and are available on the Avian Data Center itself. Each stratum or combination of strata presented in this report's Results section contains a web

link that leads directly to the Avian Data Center with the appropriate queries already populated. Please note that not every stratum or conceivable combination of strata are summarized in this report. All individual strata and all biologically meaningful combinations of strata, or “superstrata”, can be found on the Avian Data Center.

The impact-reference comparison of population density on CRP lands relative to agricultural lands indicated large positive treatment effects for the Cassin’s sparrow (*Peucaea cassinii*) and grasshopper sparrow (*Ammodramus savannarum*). None of the priority species showed negative treatment effects for CRP plantings relative to agricultural land. The comparison of population density on CRP lands relative to native grassland suggested CRP plantings provided high habitat suitability for the northern bobwhite (*Colinus virginianus*), Cassin’s sparrow and grasshopper sparrow. Population densities were similar in CRP plantings and native grassland for the scaled quail (*Callipepla squamata*), ring-necked pheasant (*Phasianus colchicus*), common nighthawk (*Chordeiles minor*), long-billed curlew (*Numenius americanus*), Swainson’s hawk (*Buteo swainsoni*), lark sparrow (*Chondestes grammacus*), lark bunting (*Calamospiza melanocorys*), eastern meadowlark (*Sturnella magna*), dickcissel (*Spiza americana*), suggesting habitat suitability for CRP plantings and native grassland were similar. There was some indication of lower habitat suitability for CRP plantings relative to native grassland for the western meadowlark (*S. neglecta*). Population density was greater on native grassland than agricultural land for scaled quail, Cassin’s sparrow, grasshopper sparrow, lark sparrow, lark bunting and western meadowlark, suggesting these species were negatively impacted by the conversion of native grassland to agricultural land. Densities on native grassland were lower than or equal to agricultural land for the northern bobwhite, ring-necked pheasant, common nighthawk, long-billed curlew, Swainson’s hawk eastern meadowlark and dickcissel, suggesting these species were doing well in agricultural landscapes. Overall, restoring grassland by planting CRP is expected to be an effective conservation strategy to provide suitable habitat and increase the abundance of several priority grassland bird species in the PLJV.

Avian population sizes on CRP plantings suggested large contributions to regional populations of the grasshopper sparrow and Cassin’s sparrow. Land enrolled in the CRP grassland from 2016 through 2020 conserved breeding habitat for 1.2 million Cassin’s sparrows, 2.2 million grasshopper sparrows and 253,651 eastern meadowlarks per year. In addition, percentage population change from CRP plantings relative to agricultural land for the Cassin’s sparrow and grasshopper sparrow exceeded the population targets for the PLJV. Enrolling agricultural land into CRP grassland at the extent of the past 5 years is expected to increase the populations of the Cassin’s sparrow by 1.0 M and grasshopper sparrow by 1.4 M per year. The percentage of population change relative to agricultural land met PLJV population targets and increased populations of the northern bobwhite by 66,408, scaled quail by 23,690, lark bunting by 242,554 and eastern meadowlark by 88,582 per year. However, the small sample size of grid cells containing CRP plantings (*n* = 27) and high annual variation in abundance on CRP suggested more years of study may needed to adequately estimate the population change for these species. Overall, the population estimates suggested changes to land enrolled in CRP over time may have important population consequences for declining grassland bird species in the PLJV region.

Acknowledgements

We thank Playa Lakes Joint Venture for building a collaborative partnership and acquiring funding across the states within their boundary to allow for the addition of IMBCR for PLJV. Funding for surveys in the PLJV region was provided by Colorado Parks & Wildlife; Kansas Department of Wildlife, Parks & Tourism; Nebraska Game & Parks Commission; New Mexico Department of Game & Fish; Oklahoma Department of Wildlife Conservation; Texas Parks & Wildlife Department; US Department of Agriculture, Farm Service Agency (grant # AG-3151-C-16-0017); US Forest Service; Great Plains Landscape Conservation Cooperative; and US Fish and Wildlife Service, Region 2 - Southwest Region, Migratory Bird Office. Thank you to Rich Iovanna from the Farm Service for advice, collaboration and the development of research questions. Bird Conservancy of the Rockies' landowner liaison, Jenny Berven, with help from Tiffany Peeken, contacted county assessors to determine land ownership of survey locations. We thank Paul Lukacs of the University of Montana who wrote code in program R to automate data analysis for density estimation. The report benefited from thoughtful reviews by Anne Bartuszevige and Rich Iovanna. We also thank the many field technicians who collected avian and vegetation point count data and contacted private landowners to obtain access to survey locations and establish working relationships for the future. Without the efforts of these technicians and the cooperation of numerous private landowners, IMBCR partners would have been unable to conduct avian monitoring on private lands. Finally, this report benefited greatly from review by Bird Conservancy staff and IMBCR partners.

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Introduction

Monitoring is an essential component of wildlife management and conservation science (Witmer 2005, Marsh and Trenham 2008). Common goals of population monitoring are to estimate the population status of target species and to detect changes in populations over time (Thompson et al. 1998, Sauer and Knutson 2008). In addition to providing basic information on species distributions, effective monitoring programs can identify species that are at-risk due to small or declining populations (Dreitz et al. 2006); provide an understanding of how management actions affect populations (Lyons et al. 2008, Schwartz et al. 2018); and evaluate population responses to landscape alteration and climate change (Baron et al. 2008, Lindenmayer and Likens 2009).

While monitoring at local scales remains critical, there is an increasing need to monitor the consequences of environmental change over large spatial and temporal scales and address questions much larger than those that can be answered within individual management units (Jones 2011, Pavlacky et al. 2017). Reconciling disparities between the geographic scale of management actions and the scale of ecological and species-specific responses is a persistent challenge for natural resource management agencies (Conroy et al. 2012). Population monitoring of eco-regional landscapes provides an important context for evaluating population change at local and regional scales, with the potential to identify causal factors and management actions for species recovery (Manley et al. 2005, Sauer and Knutson 2008).

Before monitoring can be used by land managers to guide conservation efforts, sound program designs and analytic methods are necessary to produce unbiased population estimates (Sauer and Knutson 2008, Lindenmayer and Likens 2010). At the most fundamental level, reliable knowledge about the status of avian populations requires accounting for spatial variation and incomplete detection of the target species (Pollock et al. 2002, Rosenstock et al. 2002, Thompson 2002). Addressing spatial variation entails the use of probabilistic sampling designs, which allow population estimates to be extended over the entire area of interest (Thompson et al. 1998). Accounting for incomplete detection involves the use of appropriate sampling and analytic methods to address the fact that few, if any, species are so conspicuous that they are detected with certainty when present during a survey. Accounting for these two sources of variation ensures observed trends reflect true population changes rather than artifacts of the sampling and observation processes (Pollock et al. 2002, Thompson 2002).

The apparent large-scale declines of avian populations and the loss, fragmentation and degradation of native habitats highlight the need for extensive and rigorous landbird monitoring programs (Rich et al. 2004, US NABCI Monitoring Subcommittee 2007). The US North American Bird Conservation Initiative's (NABCI) "Opportunities for Improving Avian Monitoring" (US NABCI Monitoring Subcommittee 2007) provided goals for avian monitoring programs, including:

Goal 1: Fully integrate monitoring into bird management and conservation practices and ensure that monitoring is aligned with management and conservation priorities.
Goal 2: Coordinate monitoring programs among organizations and integrate them across spatial scales to solve conservation or management problems effectively.
Goal 3: Increase the value of monitoring information by improving statistical design.
Goal 4: Maintain bird population monitoring data in modern data management systems. Recognize

legal, institutional, proprietary, and other constraints while still providing greater availability of raw data, associated metadata, and summary data for bird monitoring programs.

With the US NABCI Monitoring Subcommittee (2007) guidelines in mind, Bird Conservancy of the Rockies and its partners initiated a broad-scale bird monitoring program in 2008, entitled “Integrated Monitoring in Bird Conservation Regions” (IMBCR, Blakesley and Hanni 2009, Pavlacky et al. 2017). See Appendix B: IMBCR Program and Stratification History for a complete history of this program. The monitoring objectives of the IMBCR partnership are to:

1. Provide robust density, population and occupancy estimates that account for incomplete detection and are comparable at different geographic extents;
2. Provide long-term status and trend data for all regularly occurring breeding landbird species throughout the study area;
3. Provide a design framework to spatially integrate existing bird monitoring efforts in the region to provide better information on distribution and abundance of breeding landbirds, especially for high priority species;
4. Provide basic habitat association data for most bird species to address habitat management issues;
5. Maintain a high-quality database that is accessible to all of our collaborators as well as to the public over the internet, in the form of raw and summarized data; and
6. Generate decision support tools that help guide conservation efforts and provide a better measure of conservation success.

The IMBCR design uses Bird Conservation Regions (BCRs) as sampling frames (Fig. 1), stratified by land ownership inside each BCR (US NABCI Monitoring Subcommittee 2007). BCRs provide a spatially consistent framework for bird conservation in North America. Each BCR represents a distinct ecological region with similar bird communities, vegetation types and resource management interests (Bird Studies Canada and NABCI 2014). Population monitoring within BCRs can be implemented with a flexible hierarchical framework of nested units, where information on status of bird populations can be partitioned into smaller units for small-scale conservation planning, or aggregated to support large-scale conservation efforts throughout a species’ geographic range. By focusing on scales relevant to management and conservation, information obtained from monitoring in BCRs can be integrated into research and management at various scales applicable to land managers (Conroy et al. 2012, Pavlacky et al. 2017). Post-stratification provides a way to stratify or group point-count plots by a factor of interest after data collection, and estimate density for different groups, such as vegetation types. The spatially balanced design of the IMBCR program samples vegetation types in proportion to their availability within strata, and post-stratification can be used to estimate population density for specific vegetation types (Thomas et al. 2010, Pavlacky et al. 2017). Post-stratification often increases the precision of the density estimates (Fewster et al. 2009), and population estimates for specific vegetation types may play a role in informing vegetation management activities. In addition, post-stratifying by specific conservation practices provides a framework for effectiveness monitoring to learn about the success of management actions (Lyons et al. 2008).

Important properties of the IMBCR design are:

- All areas are available for sampling including all vegetation types;
- Strata are based on fixed attributes, which allows us to relate changes in bird populations to changes on the landscape through time;

- Each state's portion of a BCR can be stratified differently, depending upon local needs and areas to which one wants to make inferences;
- Aggregation of strata-wide estimates to BCR- or state-wide estimates is built into the design;
- Local population trends are directly comparable to regional trends; and
- Coordination among partners reduces the costs and/or increases efficiencies of monitoring per partner.

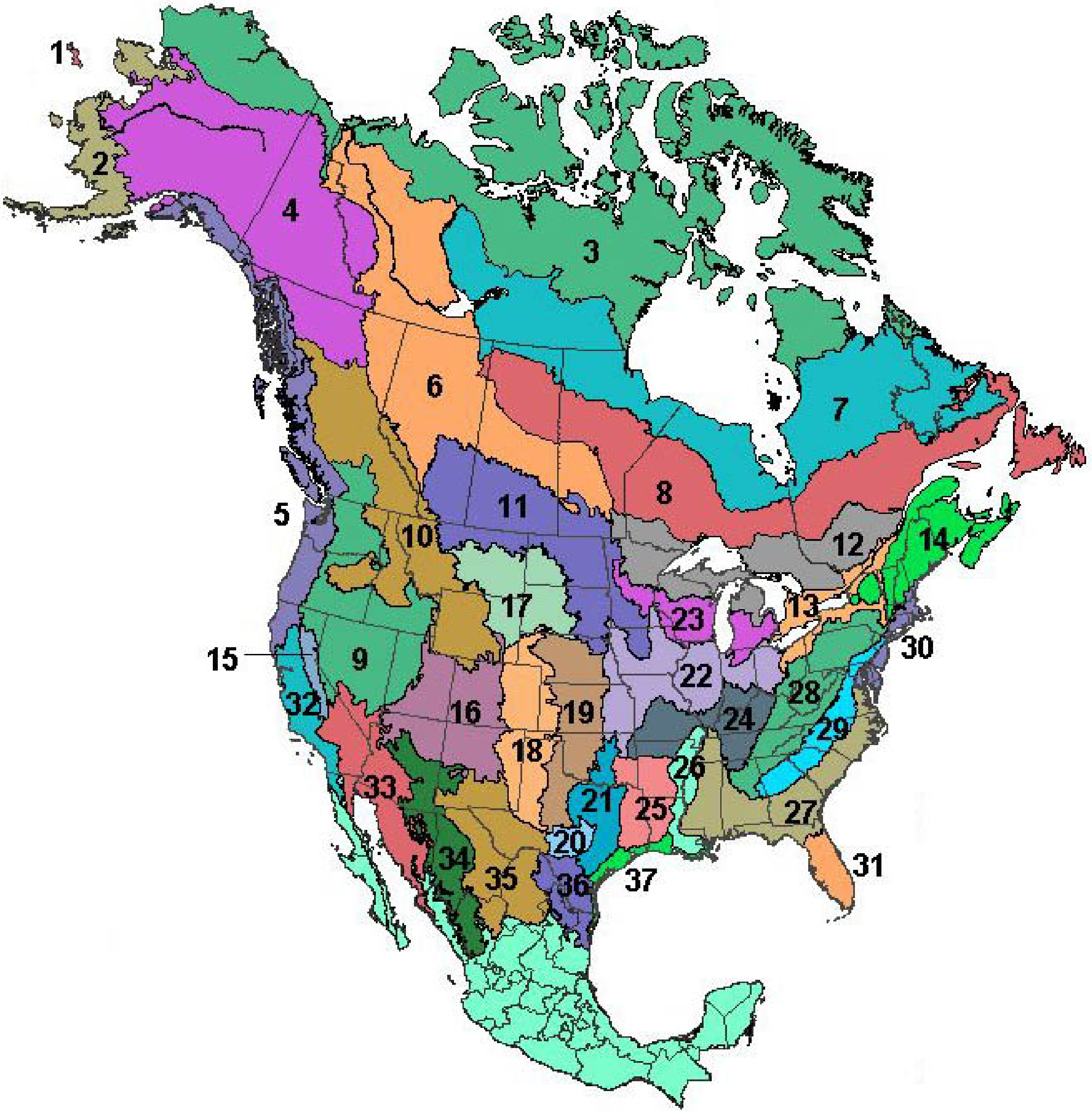

Figure 1. Bird Conservation Regions in North America, excluding Hawaii and Mexico (US North American Bird Conservation Initiative, www.nabci-us.org/resources/bird-conservation-regions-map, accessed 5 Jun 2018).

The Playa Lakes Joint Venture (PLJV) is a collaborative partnership for evaluating and implementing wildlife conservation in the Shortgrass Prairie and Central Mixed Grass Prairie BCRs (Bird Studies Canada and NABCI 2014). The partnership was designed to address management and conservation needs of a wide range of stakeholders including private landowners, initiatives such as Partner's in Flight (Carter et al. 2000), federal agencies such as the Bureau of Land Management, Farm Service Agency, Natural Resources Conservation Service, Forest Service, Department of Defense, and the state wildlife agencies of Colorado, Nebraska, New Mexico, Oklahoma and Texas. Because a large percentage of the Great Plains are privately owned, the recovery of grassland bird species depends on conservation initiatives with strong partnerships between private landowners and resource professionals (Brennan and Kuvlesky 2005). The Conservation Reserve Program (CRP) is a voluntary program for agricultural producers administered by Farm Service Agency providing incentives to landowners to take cropland out of production and plant it back into grassland (Vandever and Allen 2015). The program was designed to address a number of economic and environmental issues affiliated with agricultural land, and although the recovery of wildlife populations associated with agro-ecosystems was not a primary goal of the CRP, the program has become an important tool for managing grassland birds (Vandever and Allen 2015), including species of conservation concern such as the lesser prairie-chicken (*Tympanuchus pallidicinctus*, Van Pelt et al. 2013). Effectiveness monitoring (Lyons et al. 2008) to determine the ability of the CRP for increasing populations of grassland birds may ultimately be useful for evaluating the success of Farm Bill practices toward a program of evidence-based conservation (Briske et al. 2017). Understanding the contribution of CRP to regional bird populations provides the information to evaluate the success of the program for meeting conservation objectives in the PLJV region.

We evaluated population responses of 13 priority grassland species identified in the PLJV Landbird Team Report (2007). The objectives were to 1) evaluate avian population density on CRP grassland relative to agricultural lands and native grasslands, and 2) estimate the contributions of CRP to bird populations in the PLJV region. If habitat losses resulting from the conversion of grassland to cultivated land affected the population size of grassland birds, as reported by Stanton et al. (2018), then we predicted population densities would be greater on native grassland than agricultural land. Next, we hypothesized that CRP plantings would show increased abundance and population density of grassland birds relative to agricultural landscapes (Pavlacky et al. 2021). We also hypothesized that recent emphasis on planting native CRP seed mixes would produce high habitat suitability for grassland birds (Thompson et al. 2009) and that population densities in CRP lands would be comparable to densities on native grassland. Finally, we hypothesized that if population densities on CRP plantings were in proportion to regional population sizes over time, this would indicate high habitat suitability according to the ideal free distribution (Fretwell and Lucas 1969). According to the ideal free distribution of territorial species (Fretwell and Lucas 1969), individuals establish territories in high quality habitat first, then after high quality habitat reaches carrying capacity, individuals establish territories in low quality habitat. When population sizes of species decline over time, density of species are expected to decline in the low quality habitat first. Alternately, when population sizes increase over time, density of species increase in low quality after high quality habitat reaches carrying capacity. If CRP plantings provide high habitat suitability according to the ideal free distribution, we predicted the percentage contribution of CRP to regional population size would be relatively stable over the five years of study. Alternatively, we hypothesized changes in population densities on CRP plantings disproportionate to changes in regional population size may be indicative of low habitat suitability according to the ideal free distribution (Fretwell and Lucas 1969). If CRP plantings provide low habitat suitability under the ideal free distribution, then we predict population densities on

CRP plantings will be lower than expected when population sizes are low, and high population densities on CRP plantings are only possible when regional population sizes are high.

Methods

Study Area

In 2016, IMBCR encompassed three entire states (Colorado, Montana and Wyoming) and portions of 10 additional states (Arizona, Idaho, Kansas, North Dakota, Nebraska, New Mexico, Oklahoma, South Dakota, Texas and Utah); two entire USFS Regions (Regions 1 and 2) and portions of Regions 3 and 4; all of the Badlands and Prairies BCR and almost all of the Shortgrass Prairie BCR and portions of seven additional BCRs (Great Basin, Northern Rockies, Prairie Potholes, Southern Rockies/Colorado Plateau, Central Mixed-grass Prairie, Sonoran and Mohave Deserts, and Sierra Madre Occidental; Fig. 2).

In 2020, the IMBCR program's area of inference encompassed four entire states (Colorado, Montana, Utah, and Wyoming) and portions of 12 additional states (Arizona, California, Idaho, Kansas, Nebraska, Nevada, New Mexico, North Dakota, Oklahoma, Oregon, South Dakota, and Texas). We surveyed across US Forest Service (USFS) Regions 1, 2, and 4 and in portions of Region 3; all of the Badlands and Prairies Bird Conservation Region (BCR 17), all of the Shortgrass Prairie Bird Conservation Region (BCR 18), and portions of seven other BCRs: Great Basin (9), Northern Rockies (10), Prairie Potholes (11), Sierra Nevada (15), Southern Rockies/Colorado Plateau (16), Central Mixed Grass Prairie (19), and Sonoran and Mojave Deserts (33) (Fig. 2).

For a map and complete descriptions of the Bird Conservation Regions, see the NABCI website (US North American Bird Conservation Initiative, www.nabci-us.org/resources/bird-conservation-regions-map, accessed 5 Jun 2018).

BCR 18: Shortgrass Prairie

The Shortgrass Prairie Bird Conservation Region is characterized by unique shortgrass prairie. What was once contiguous prairie is now fragmented by agriculture and the remnant grasslands are now exposed to new grazing regimes (PLJV 2007). Numerous playa lakes dot the region and wetlands occur along major river corridors that drain the Rocky Mountains. Because of a change in the hydrology of these rivers, more shrubs and trees have encroached upon the wetlands (US NABCI Committee 2000b;a). BCR 18 stretches north-south in the rain shadow of the Rocky Mountains and covers portions of Colorado, Kansas, Nebraska, New Mexico, Oklahoma, South Dakota, Texas, and Wyoming.

This was the ninth year we implemented IMBCR within BCR 18. In BCR 18, Bird Conservancy conducted surveys throughout Colorado, Kansas, Nebraska, New Mexico, Oklahoma, Texas, and Wyoming. The only portion of BCR 18 not surveyed in 2016 was the small area within South Dakota. The effort in BCR 18 comprised 37 strata covering 381,286 km².

BCR 19: Central Mixed-grass Prairie

The Central Mixed-grass Prairie Bird Conservation Region lies between shortgrass prairie to the west and tallgrass prairie to the east (US NABCI Committee 2000b;a). This region consists of a mixture of shortgrass and tallgrass prairie habitats, with some native and hand-planted Ponderosa Pine forests in northwestern

Nebraska. BCR 19 runs north-south from the southern border of South Dakota through Nebraska, Kansas, Oklahoma, and north-central Texas.

This was the sixth year we implemented IMBCR within BCR 19. In BCR 19, Bird Conservancy conducted surveys throughout Kansas, Oklahoma, and Texas; and within USFS lands in BCR 19 in Nebraska. The effort in BCR 19 comprised 11 strata covering 274,583 km².

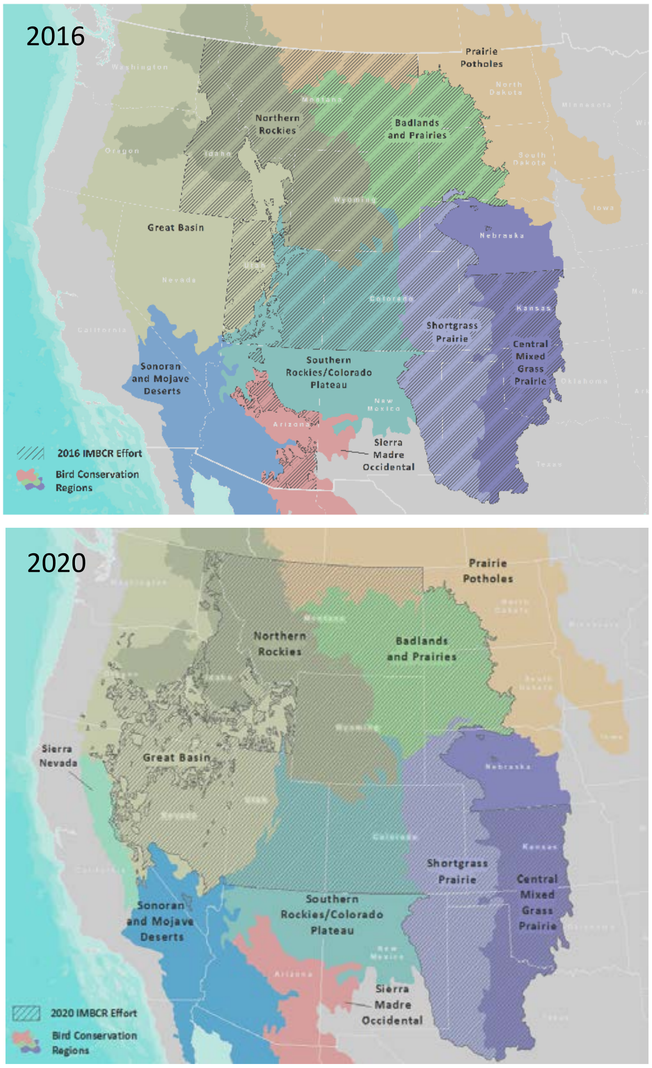

Figure 2. The spatial extent of sampled Bird Conservation Regions (BCR) using the Integrated Monitoring in Bird Conservation Regions (IMBCR) design, 2016 - 2020. The colored regions represent the BCRs and the hatched regions represent the area of inference for the IMBCR program.

Sampling Design

Sampling Frame and Stratification

A key component of the IMBCR design is the ability to infer across spatial scales, from small management units, such as individual national forests or BLM field offices, to entire states and BCRs (Pavlacky et al. 2017). This is accomplished through hierarchical (nested) stratification, which allows data from smallerorder strata to be combined to make inferences about higher-order collections of strata. For example, data from each individual national forest stratum in USFS Region 2 are combined to produce Region-wide avian population estimates; data from each individual stratum in Montana are combined to produce state-wide estimates; data from each individual stratum in BCR 17 are combined to produce BCR-wide estimates.

We defined strata based on areas to which IMBCR partners wanted to make inferences. We defined the largest sampling frame by the intersection of state and BCR boundaries (e.g., Wyoming BCR 10). We based the strata within the state-BCRs frame on fixed attributes such as land ownership boundaries, elevation zones, major river systems and wilderness/roadless designations.

Sampling Units

The IMBCR design defines sampling units as 1 km² grid cells, each containing 16 evenly spaced sample points, 250 meters apart (Fig. 3). We define potential sampling units by superimposing a uniform grid of cells over each state in the study area. We then assign each grid cell to a stratum using ArcGIS version 10.X and higher (ArcGIS Version 10, Environmental Systems Research Institute, Redlands, CA). For all stratifications developed after 2012, we used the United States National Grid, a nonproprietary alphanumeric referencing system derived from the Military Grid Reference System that was created by the Federal Geographic Data Committee.

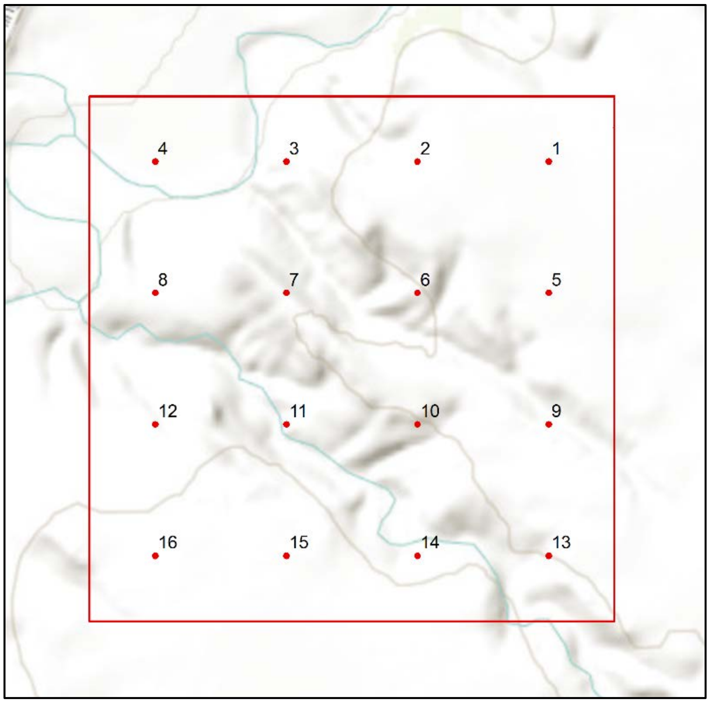

Figure 3. Example 1 km² sampling unit using the Integrated Monitoring in Bird Conservation Regions design.

Sample Selection

Within each stratum, the IMBCR design used generalized random-tessellation stratification (GRTS), a spatially balanced sampling algorithm, to select sample units (Stevens and Olsen 2004). The GRTS design has some appealing properties with respect to long-term monitoring of birds at large spatial scales:

- Spatially balanced sampling is generally more efficient than simple random sampling of natural resources (Stevens and Olsen 2004). Incorporating information about spatial autocorrelation in the data can increase precision in density estimates; and
- All sample units in the sampling frame are ordered, such that any set of consecutively numbered units is a spatially well-balanced sample (Stevens and Olsen 2004). In the case of fluctuating budgets, IMBCR partners can adjust the sampling effort among years within each stratum while still preserving a random, spatially balanced sampling design. In addition, the spatially-balanced property of the sample is maintained when access to sampling units are not possible, such as when private landowners deny access permission or dangerous terrain exists.

A minimum of two sampling units within each stratum are required to estimate the variances of population parameters. However, reliable stratum-level occupancy estimates require larger samples sizes, with a minimum of approximately 10 samples per stratum. Furthermore, additional samples may be required for strata comprising large geographic areas. Because we estimate regional density and occupancy using an area-weighted mean, adding more samples to a particular stratum does not bias the overall estimate, it simply increases the precision. After the initial two sampling units were selected, the remaining allocation of sampling effort among strata was based on the priorities of the funding partners.

Sampling Methods

IMBCR surveyors (also referred to as field technician, technician or observer in this report), with excellent aural and visual bird identification skills, conducted field work. Prior to conducting surveys, technicians completed an intensive training program to ensure full understanding of the field protocol; review bird and plant identification; and practice distance estimation in a variety of habitats.

Field technicians conducted point counts (Buckland 2006) following protocols established by IMBCR partners (Hanni et al. 2018). Observers conducted surveys in the morning, beginning one-half hour before sunrise and concluding no later than five hours after sunrise. Technicians recorded the start time for every point count conducted. For every bird detected during the six-minute period, observers recorded species; sex; horizontal distance from the observer; minute; type of detection (e.g., call, song, visual); whether the bird was thought to be a migrant; and whether the observer was able to visually identify each record.

Observers measured distances to each bird using laser rangefinders, when possible. When it was not possible, observers estimated the distance by measuring to some object near the bird using a laser rangefinder. In addition to recording all bird species detected in the area during point counts, observers recorded birds flying over but not using the immediate surrounding landscape. Technicians considered all non-independent detections of birds (i.e., flocks or pairs of conspecific birds together in close proximity) as part of a “cluster” rather than as independent observations. Observers recorded the number of birds detected within each cluster along with a letter code to distinguish between multiple clusters.

At the start and end of each survey, observers recorded time, ambient temperature, cloud cover, precipitation, and wind speed. Technicians navigated to each point using hand-held Global Positioning System units. Before beginning each six-minute count, surveyors recorded vegetation data within a 50 m radius of the point via ocular estimation. Vegetation data included the dominant vegetation type and relative abundance, percent cover and mean height of trees and shrubs by species, as well as grass height and ground cover types. Technicians recorded vegetation data quietly to allow birds time to return to their normal habits prior to beginning each count.

The comparison of avian population density on CRP lands relative to agricultural lands represents an impact design-reference (Morrison et al. 2008) for estimating the effect of restoring agricultural lands to CRP lands. The impact-reference design for the comparison of avian population density on CRP lands and native grassland provides a way to evaluate habitat suitability of the CRP for various bird species. To evaluate the influence of CRP lands on bird populations in the PLJV region, we post-stratified (Thomas et al. 2010) the point count plots by three vegetation types: agricultural lands, native grassland and CRP grassland (Table 1). The impact-reference question may be better addressed with a covariate analysis of the IMBCR data, but comes at higher cost above and beyond the annual IMBCR analysis. Post-stratification provides a low-cost analysis method that extends the base IMBCR program to address partner management questions related to vegetation type. We used the primary vegetation type collected in the field through the IMBCR program to classify the majority land cover of each point-count plot according to agricultural land and native grassland. We defined agricultural lands as agricultural or rural land planted for food production or ornamental purposes in sparsely developed areas (Hanni et al. 2016). We defined native grassland as grassland vegetation types with <10% shrub cover (Hanni et al. 2016). We defined CRP lands according to the Common Land Unit (CLU) geospatial dataset (USDA 2014) depicting the spatial distribution of lands enrolled in the CRP. We classified 4.9 ha point-count plots as CRP points when >50% of the plots were covered by grassland CRP types within a Geographic Information System environment (ArcGIS Version 10.1, Environmental Systems Research Institute, Redlands, CA). We classified the IMBCR point-count plots according to the previous year of CRP data to account for situations when CRP plantings were established after the bird surveys were conducted in a given year.

For all years, we calculated the area of grassland CRP types (km^2) within the 43 strata in the PLJV region (Fig. 5). Overall, the area of the PLJV region was 642,782 km^2. Using the CLU data from 2016 through 2020, active contacts for grassland CRP types comprised 4.8% of the PLJV Region in 2016 (31,078 km^2), 4.6% in 2017 (29,407 km^2), 4.6% in 2018 (29,868 km^2), 4.0% in 2019 (25,736 km^2) and 4.1% of the Region in 2020 (26,191 km^2).

Table 1. The sample sizes for the numbers of grid cells and point count plots for the post-stratification of the Playa Lakes Joint Venture region, 2016 - 2020.

Year	CRP		Grassland		Agricultural land	
	Grid	Point	Grid	Point	Grid	Point
2016	24	142	202	1,332	105	570
2017	26	132	222	1,469	105	538
2018	29	157	217	1,374	129	668
2019	39	123	136	906	95	508
2020	18	111	122	942	74	455

We calculated weighted means and Standard Errors (SE) of ground and shrub cover variables for the vegetation types across years according to the area of the vegetation types in each of the PLJV strata (Table 2). We tested for differences between the vegetation variable means *i* by calculating effect sizes ($\hat{\theta}_i$) using the difference $\hat{\theta}_i = \hat{x}_{\text{CRP}_i} - \hat{x}_{\text{Ref}_i}$, where $\hat{x}_{\text{CRP}_i}$ is the mean of vegetation variable *i* for CRP lands and $\hat{x}_{\text{Ref}_i}$ is the mean of vegetation variable *i* for the reference category. We calculated the SE and 95% Confidence Intervals (CI) for the effect size using the delta method (Powell 2007) to evaluate statistical support for the effect sizes. We found that live grass ground cover [$\hat{\theta}$ = -5.93; SE = 0.82; CI = -7.54, -4.32], shrub canopy cover [$\hat{\theta}$ = -0.96; SE = 0.12; CI = -1.19, -0.73] and shrub height [$\hat{\theta}$ = -0.21; SE = 0.01; CI = -0.23, -0.18] were lower on CRP lands than native grasslands. In contrast, live grass height [$\hat{\theta}$ = 7.23; SE = 0.65; CI = 5.95, 8.51], residual grass height [$\hat{\theta}$ = 7.61; SE = 0.92; CI = 5.79, 9.42], herbaceous ground cover [$\hat{\theta}$ = 3.42; SE = 0.39; CI = 2.65, 4.20] and bare-litter ground cover [$\hat{\theta}$ = 3.19; SE = 0.86; CI = 1.50, 4.89] were greater on CRP lands than native grasslands (Table 2). There was no difference between residual grass ground cover for CRP and native grassland (Table 2). Live grass height [$\hat{\theta}$ = -5.45; SE = 1.19; CI = -7.80, -3.11], bare-litter ground cover [$\hat{\theta}$ = -10.27; SE = 1.04; CI = -12.31, -8.24] and shrub height [$\hat{\theta}$ = -0.50; SE = 0.02; CI = -0.54, -0.46] were lower on CRP lands than agricultural lands. Conversely, residual grass ground cover [$\hat{\theta}$ = 5.64; SE = 0.53; CI = 4.60, 6.68], residual grass height [$\hat{\theta}$ = 7.40; SE = 1.08; CI = 5.28, 9.53], herbaceous ground cover [$\hat{\theta}$ = 3.02; SE = 0.53; CI = 1.98, 4.06] and shrub canopy cover [$\hat{\theta}$ = 0.82; SE = 0.10; CI = 0.62, 1.02] were greater on CRP lands than agricultural reference lands (Table 2). Live grass ground cover on CRP and agricultural lands were not considerably different (Table 2).

For more detailed information about survey methods and vegetation data collection protocols, refer to Bird Conservancy's Field Protocol for Spatially Balanced Sampling of Landbird Populations on our Avian Data Center (Rocky Mountain Avian Data Center, www.rmbo.org/v3/avian/ExploretheData.aspx, accessed 5 June 2018). There you will find links to past and current protocols and data sheets.

Table 2. The means and Standard Errors (SE) of ground and shrub cover variables for point-count plots classified as Conservation Reserve Program (CRP) lands, native grasslands and agricultural lands, Playa Lakes Joint Venture (PLJV) region, 2016 - 2020.

Vegetation variables	CRP lands		Native grasslands		Agricultural lands	
	Mean	SE	Mean	SE	Mean	SE
Live grass ground cover (%)	14.79	0.73	20.72	0.37	13.58	0.66
Live grass height (cm)	27.35	0.53	20.12	0.37	32.80	1.07
Residual grass ground cover (%)	8.98	0.42	8.44	0.18	3.34	0.32
Residual grass height (cm)	40.30	0.66	32.70	0.64	32.90	0.86
Herbaceous ground cover (%)	8.18	0.36	4.76	0.16	5.16	0.39
Bare-litter ground cover (%)	67.33	0.75	64.14	0.42	77.61	0.71
Shrub canopy cover (%)	1.09	0.09	2.05	0.07	0.27	0.04
Shrub height (m)	0.54	0.01	0.75	0.01	1.04	0.02

Data Analysis

Distance Sampling Analysis Assumptions

Distance sampling theory was developed to account for the decreasing probability of detecting an object of interest (e.g., a bird) with increasing distance from the observer to the object (Buckland et al. 2001). The detection probability is used to adjust the count of birds to account for birds that were present but undetected. 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 (Buckland et al. 2001, Thomas et al. 2010); 4) cluster sizes are recorded without error; and 5) the sampling units are representative of the entire survey region (Buckland et al. 2008).

Distance Sampling Analysis

We developed a Bayesian, zero-inflated *N*-mixture model (Royle 2004, Sillett et al. 2011) to estimate density and abundance for all strata and biologically meaningful combinations of strata (superstrata) across all species with sufficient data. We used distance sampling to estimate detection probabilities and adjust counts accordingly. For a detailed description of statistical analyses performed, see (Appendix A).

Bayesian approaches to density estimation provide several benefits over traditional distance sampling analyses, while providing similar and unbiased estimates of density and abundance. First, with the nested design of IMBCR, point count locations within a 1-km^2 grid cell are not independent (Fig. 4). Therefore, with traditional methods, it is necessary to treat each point as a spatial replicate within the grid cell (i.e., average counts across points). However, it is unlikely that bird densities are uniform within a grid cell, and a better solution would be to estimate density at the point count location. Bayesian models provide the flexibility to do this, while correctly accounting for the lack of independence among points. The second benefit, also provided by this flexibility, is the ability to include covariates to explain changes in density. This allows us to explicitly estimate the response of bird density to variables, such as habitat variables, management actions, or time (i.e., trend). Finally, Bayesian approaches allow for sharing of information across parameters. This can assist in obtaining estimates at sites with little data or provide measures of uncertainty when no birds were detected, such as at low densities and/or small sample sizes.

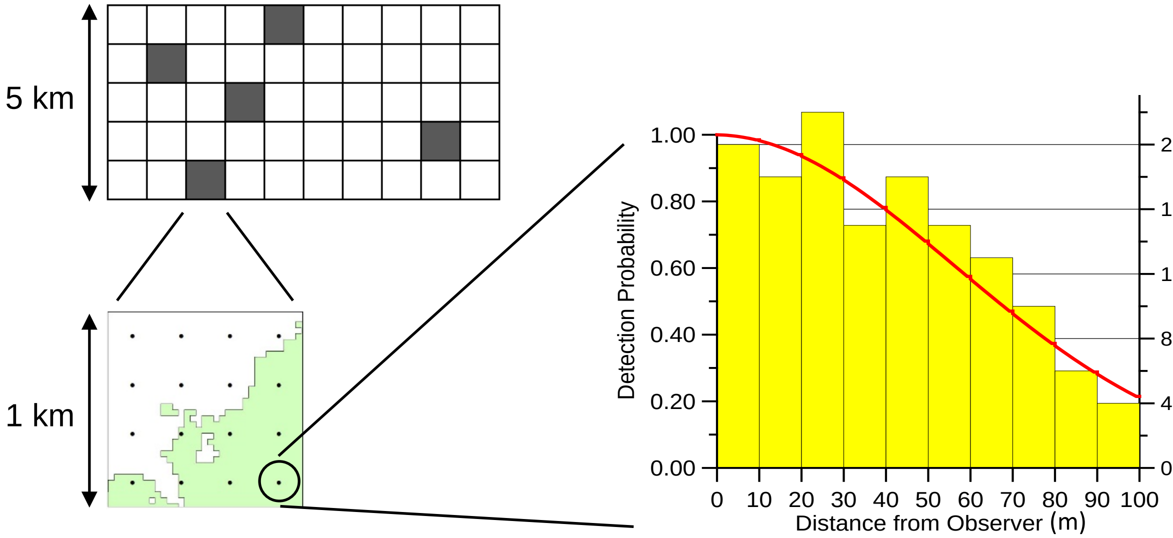

Figure 4. Distance sampling from the Integrated Monitoring in Bird Conservation Regions program, with grid cells nested within strata, point count plots nested within grid cells and distances nested within point count plots. The detection probability on the *y*-axis of the graph corresponds to the red-colored 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 fit a series of models to the data from each species that had the same model structure describing density estimation but varied in detection structure (see *Observation process* section below). We used zero-inflation to account for excess zeros in the data, where abundance at a point count location (*N*) is conditional on the point's true occupancy state (*z*) of a species at the point count location, and the mean abundance within a 1-km^2 grid cell was modeled as a function of year to estimate stratum-specific trends. All points within a grid cell shared a mean abundance to account for the lack of independence of those points, but abundance was allowed to vary spatially within a grid cell (i.e., by point) through Poisson variation. To avoid predicting species occurrence outside of observed ranges, we fixed occupancy probabilities to 0 for all strata in which the species was never observed and used a prior informed by the observed proportion of grid-year combinations in a stratum in which the species was detected.

We derived density at the point count location by dividing the estimated abundance by the area of the point count circle (see *Observation process* section below) and multiplying by cluster size. We derived stratum-level density estimates by averaging all point-level density estimates within each stratum, and we took the area-weighted average of strata estimates to obtain superstratum estimates.

Observation process

We estimated the probability of detecting an independent cluster of individuals by fitting distance functions to the distance data collected during surveys (Buckland et al. 2001). We fit four detection models including: 1) half-normal constant [HN(.)], 2) hazard rate constant [Haz(.)], 3) half-normal year [HN(*t*)], and 4) hazard rate year [Haz(*t*)].

We removed the furthest 10% of observed detection distances from the data set and binned the remaining detections into 10 evenly spaced distance classes. The furthest remaining detection distance became the radius of the point count circle with which we estimated density.

Detection model selection

To minimize computing time but find the most parsimonious detection function, we fit detection-only models to the distance data, using the four model structures described above. We used the Watanabe-Akaike Information Criterion (WAIC; Watanabe 2010, Hooten and Hobbs 2015) to select the most parsimonious detection structure and then used that structure for detection probabilities in the full model to estimate density and abundance.

Post-stratification

In addition to the general analysis above, we estimated the densities and population sizes for priority grassland species in the PLJV Region (2007) with >25 detections from 2016 through 2020 by post-stratifying the point-count data into three mutually exclusive vegetation types (Thomas et al. 2010): agricultural lands; CRP lands; and native grassland. We saved 500 Markov chain Monte Carlo (MCMC) iterations of the posterior distribution for population density at the point count level from models for each species. We estimated density and population size for each of the 43 strata within the PLJV region, and aggregated the post-stratified density estimates within the PLJV region according to a stratified random estimator (Pavlacky et al. 2017) weighted by area for each stratum and year $\widehat{D}_{jk} = \sum_{i=1}^{n} \sum_{j=1}^{t} w_{ij} \widehat{d}_{ijk} \big/ \sum_{j=1}^{t} w_{ij}$, where $\widehat{D}_{jk}$ is aggregated density for year j and iteration k, n is the number of strata, t is the number of years, w_{ij} is proportional areas of stratum i and year j, and $\widehat{d}_{ijk}$ is estimated density for stratum i, year j and MCMC iteration k. We estimated overall density from 2016 through 2020 ($\widehat{D}_{\mathrm{tot}}$) for each species accounting for annual variation by calculating the median, SD and 90% Credible Intervals (CI) across the matrix of annual density estimates of the posterior iterations $\widehat{D}_{jk}$. We estimated treatment effects for each species ($\widehat{\Delta}_j$) according to $\widehat{\Delta}_j = \widehat{D}_{\mathrm{trt}_j} - \widehat{D}_{\mathrm{ref}_j}$, where $\widehat{D}_{\mathrm{trt}_j}$ and $\widehat{D}_{\mathrm{ref}_j}$ are estimated population densities for treatment and reference strata for year j, respectively. We estimated relative population change for CRP by year ($\widehat{N}_{\mathrm{rel}_j}$) according to $\widehat{N}_{\mathrm{rel}_j} = \widehat{\Delta}_{\mathrm{CRP}_j} A_{\mathrm{CRP}_j}$, where $\widehat{\Delta}_{\mathrm{CRP}_j}$ is the CRP treatment effect on density and A_{CRP_j} is the regional area of CRP for year j. In addition, we estimated absolute population size each year ($\widehat{N}_{\mathrm{abs}_j}$) according to $\widehat{N}_{\mathrm{abs}_j} = \widehat{D}_{\mathrm{CRP}_j} A_{\mathrm{CRP}_j}$, where $\widehat{D}_{\mathrm{CRP}_j}$ is median density on CRP lands and A_{CRP_j} is the regional area of

CRP for year *j*. We estimated regional population size according to $\widehat{N}_{\text{tot}_j} = \widehat{D}_{\text{tot}_j} A_{\text{tot}}$, where $\widehat{D}_{\text{tot}_j}$ is median density for year *j* and A_{tot} is the area of the PLJV region (Buckland et al. 2001). We estimated percentage relative change from treatment effects ($\hat{\delta}_{\text{rel}_j}$) and absolute percentage of population conserved by year ($\hat{\delta}_{\text{abs}_j}$) according to $\hat{\delta}_j = \left(\widehat{N}_j / \widehat{N}_{\text{tot}_j}\right) \times 100$, where $\widehat{N}_j$ is relative or absolute population size from CRP and $\widehat{N}_{\text{tot}_j}$ is population size in the PLJV region in year *j*. As above, we estimated overall treatment and population effects from 2016 through 2020 for each species accounting for annual variation by calculating the median, SD and 90% CIs across the matrix of annual estimates and posterior iterations.

We evaluated statistical support for the effect sizes by evaluating 90% CIs for the difference in the median estimates relative to zero. We presented 90% CIs to evaluate two-tailed hypotheses for effect sizes different from zero, and one-tailed probabilities (*f*) that percentages were greater than [$P(\hat{\delta} > 4.4\%)$] or less than [$P(\hat{\delta} > 4.4\%)$] the overall availability of CRP in the PLJV region, or that effect sizes were greater than [$P(\widehat{\Delta} > 0)$] or less than [$P(\widehat{\Delta} > 0)$] zero. We considered coefficients with *f*-values >0.9 as considerable support for the one-tailed hypotheses. We considered the contribution to population size to be in proportion to the availability of CRP when the CI included the percentage of CRP in the PLJV region. We determined evidence of meeting regional population targets by evaluating CIs for relative percent change over space with respect to trends for percent change over time used to set PLJV (2007) population objectives, as well as trends for percent change in the region.

Trend Estimates

We estimated trends for individual strata by calculating the least-squares regression mean and standard errors for the intercept and slope of the log densities across the monitoring period. We calculated these parameters for every Bayesian iteration to account for uncertainty around density estimates.

We developed a post-hoc approach to estimate trends for superstrata. Using the rolled-up estimates of density for a superstratum, we fit a general linear model (GLM) to the samples from each Bayesian iteration. Fitting a GLM across iterations allowed us to incorporate uncertainty in superstratum trends due to uncertainty around density estimates, but it did not account for temporal variation. To incorporate this second form of variation, we sampled a random intercept and slope for each iteration using the mean and standard error estimated using the GLM and made inference on the distribution of the resampled values.

Automated Analysis

We recently updated our analytical methods and are using Bayesian hierarchical models specifically designed for analysis of IMBCR data. We performed all data and output manipulation in R (R Core Team, 2019) and model fitting in JAGS (Plummer 2003, 2017) using the R package jagsUI (Kellner 2018). The R code called the raw data from the IMBCR Structured Query Language (SQL) server database and reformatted the data into a form usable with the JAGS code. We allowed the input of all data collected in a manner consistent with the IMBCR design to increase the number of detections available for estimating global detection rates for population density and site occupancy. The R code provided an automated framework for combining stratum-level estimates of population density and site occupancy at multiple spatial scales, as well as estimating the standard deviations and credible intervals for the combined estimates.

We fit initial models to all species with at least 30 detections for density estimation and 10 detections for occupancy estimation. For density estimation, we fit the full model after determining whether there were enough detections based on results from the detection-only model fits. In some cases for both density and occupancy estimation, it was necessary to use a less parsimonious detection structure or simplified model structure to facilitate model convergence. We currently maintain version control of the automated analysis code in the Bird Conservancy repository (Atlassian Stash, version 3.6.1).

Results

Playa Lakes Joint Venture

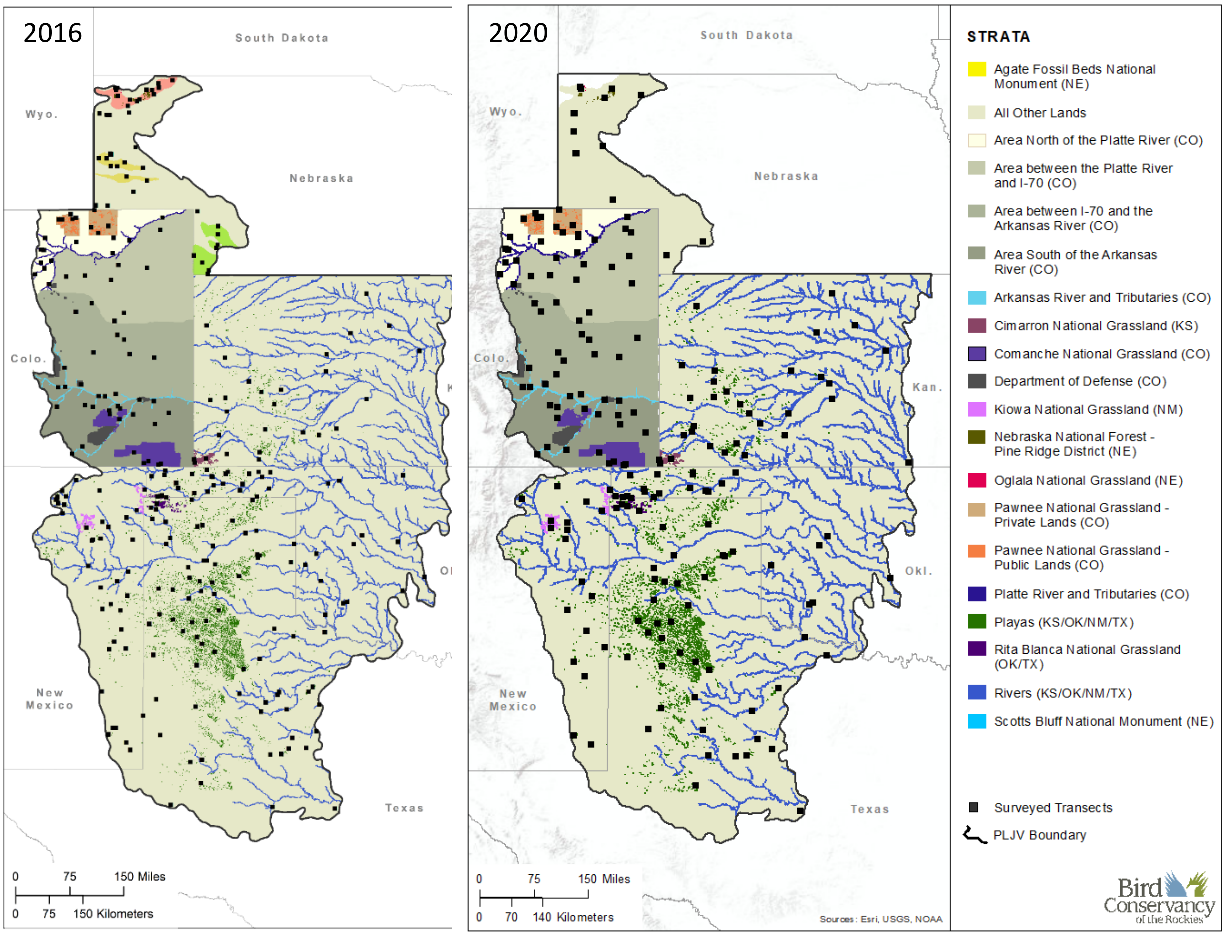

Figure 5. Survey locations and strata in the Playa Lakes Joint Venture (PLJV) region during 2016 and 2020. The black square symbols represent the survey locations and the color coded regions represent the strata.

Playa Lakes Joint Venture Survey Effort 2016 - 2020

In 2016, the Playa Lakes Joint Venture (PLJV) coordinated a partnership between several state wildlife agencies and Bird Conservancy to expand sampling in five of the joint venture's six states: Nebraska, Kansas, New Mexico, Oklahoma, and Texas. PLJV's sixth state, Colorado, was already included in the IMBCR program starting in 2008. This expansion now provides the program with nearly complete coverage of two BCRs that were only sparsely covered in past years: Shortgrass Prairie (BCR 18) and Central Mixed Grass Prairie (BCR 19). The BCR 18 and 19 portions of these 5 states were divided into several strata, including, playas, rivers, biologically unique landscapes in Nebraska, and all other lands.

With the expansion of IMBCR throughout the PLJV region, several existing strata needed to be fit to the US National Grid to make them consistent with the rest of the IMBCR program in the region: Cimarron, Kiowa, and Rita Blanca National Grasslands in Kansas, Oklahoma, New Mexico, and Texas. In addition, we determined that the portion of Rita Blanca National Grassland that fell in New Mexico was actually managed by Kiowa National Grassland, so that so that portion was moved to the Kiowa National Grasslands stratum. All DoD lands in Colorado BCR18 were combined into one stratum. This was the same stratification used prior to 2015. We obtained results for the Playa Lakes Joint Venture area by compiling and jointly analyzing data from 43 Strata in six states (Fig. 5). A summary of the planned surveys in the PLJV from 2016 through 2020 is presented in Table 3.

Table 3. Summary of planned surveys in the Playa Lakes Joint Venture, 2016 – 2020.

Survey summary	2016	2017	2018	2019	2020
Planned surveys	330	330	359	233	197
Competed surveys	300	330	359	230	199
Percentage of surveys completed	91	100	100	99	99
Point-count surveys	2,847	3,169	3,296	2,155	2,029
Bird species detected	226	220	236	210	217

Weather patterns in the Great Plains indicated annual precipitation declined over the study period, and was above average in 2015 and 2016, average in 2017, above average in 2018 and 2019, and below average in 2020 (NOAA National Centers for Environmental information 2021) (Fig. 6).

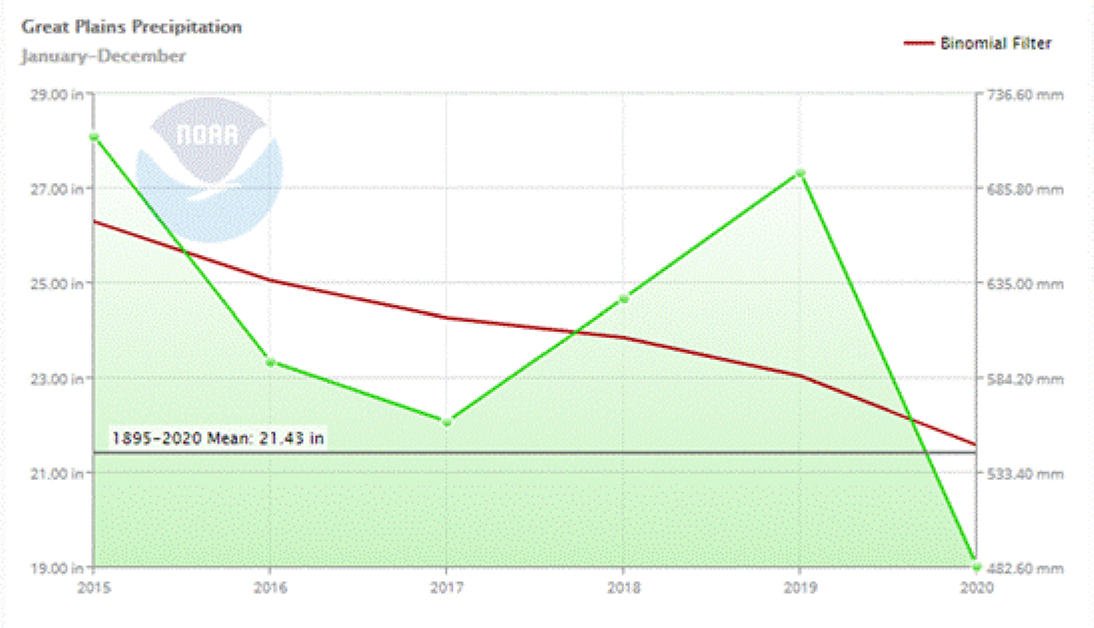

Figure 6. Annual precipitation time series in the Great Plains, 2015 - 2020. Annual precipitation is shown in green, binomial smoothed time series is depicted by the red trend line, and the historical mean is shown by the gray horizontal line.

Avian Density in CRP relative to Agricultural Lands and Native Grassland

We estimated avian population densities for CRP lands, agricultural lands and native grassland within the PLJV region from 2016 through 2020 (Table 3). We present the effect sizes for the comparison of avian population density in CRP plantings relative to agricultural lands, CRP lands relative to native grassland, and native grassland relative to agricultural land in Table 3.

Avian population densities on CRP plantings were greater than densities in agricultural land for Cassin's sparrow (*Peucaea cassinii*) and grasshopper sparrow (*Ammodramus savannarum*) (Table 3). These effects can be considered expected treatment effects for implementing CRP plantings in agricultural landscapes. Although the 2-tailed CI for the treatment effect covered zero (Table 3), there was a >0.9 probability that the densities of ring-necked pheasant (*Phasianus colchicus*) were lower on CRP lands than on agricultural lands (f = 0.95).

We found northern bobwhite (*Colinus virginianus*), Cassin's sparrow and grasshopper sparrow densities were greater on CRP plantings than they were on native grassland (Table 3), suggesting CRP provided high habitat suitability for these species. Population densities on CRP plantings were not statistically different from those on native grasslands (Table 3) for the scaled quail (*Callipepla squamata*), ring-necked pheasant, common nighthawk (*Chordeiles minor*), long-billed curlew (*Numenius americanus*), Swainson's hawk (*Buteo swainsoni*), lark sparrow (*Chondestes grammacus*) and lark bunting (*Calamospiza melanocorys*), suggesting habitat suitability on CRP plantings and native grassland was similar for these species. Western meadowlark (*Sturnella neglecta*) density was lower on CRP plantings than on native grassland (Table 3), suggesting CRP provided low habitat suitability for this species.

Table 4. The effect sizes for differences in avian population density between CRP plantings, native grasslands and agricultural lands within the Playa Lakes Join Venture Region, from 2016 through 2020. The CRP – agriculture effect represents density on CRP lands minus density on agricultural lands (km^{-2}), CRP – grassland represents density on CRP lands minus density on native grassland (km^{-2}), and grassland – agriculture represents density on native grassland minus agricultural land. The Standard Error (SE), and Lower (LCL) and Upper (UCL) 90% Credible Limits, respectively represent the precision of the effect sizes. The bold values represent measureable effect sizes with 90% Confidence Intervals excluding zero.

Species	CRP – agriculture			CRP - grassland			Grassland - agriculture		
	Effect	LCL	UCL	Effect	LCL	UCL	Effect	LCL	UCL
Northern bobwhite	2.26	-1.74	5.36	**3.56**	**0.65**	**6.44**	**1.33**	**0.23**	**2.76**
Scaled quail	0.83	-0.20	2.53	-0.37	-2.44	1.34	**-1.45**	**-2.35**	**-0.72**
Ring-necked pheasant	-0.72	-1.66	0.01	0.94	-0.23	1.91	**1.59**	**1.28**	**2.05**
Common nighthawk	-0.57	-1.87	0.21	-0.13	-0.87	0.39	0.45	-0.72	1.76
Long-billed curlew	0.00	-0.04	0.53	-0.01	-0.06	0.49	-0.03	-0.07	0.03
Swainson's hawk	0.03	-0.15	0.40	0.06	-0.11	0.38	0.01	-0.07	0.12
Cassin's sparrow	**37.43**	**28.39**	**47.61**	**16.25**	**4.74**	**24.12**	**-21.74**	**-26.87**	**-17.89**
Grasshopper sparrow	**50.48**	**32.60**	**66.23**	**27.17**	**9.17**	**47.52**	**-21.40**	**-30.33**	**-15.21**
Lark sparrow	2.89	-3.51	20.66	-1.07	-7.79	15.58	**-4.01**	**-7.28**	**-2.07**
Lark bunting	8.44	-3.36	12.28	-8.23	-12.16	2.35	**-14.40**	**-21.14**	**-5.77**
Eastern meadowlark	3.14	-1.70	11.31	2.69	-1.85	6.95	-2.01	-4.96	2.36
Western meadowlark	1.43	-1.45	6.09	**-3.52**	**-7.64**	**-0.39**	**-5.99**	**-8.49**	**-3.00**
Dickcissel	-11.34	-29.06	6.72	-0.18	-6.80	16.36	**13.25**	**1.34**	**23.49**

Avian population densities were lower on agricultural land than on native grassland for the scaled quail, Cassin's sparrow, grasshopper sparrow, lark sparrow, lark bunting and western meadowlark (Table 3), suggesting the conversion of native grassland to agricultural landscapes negatively affected densities of these species. Densities were greater on agricultural land than native grassland for northern bobwhite, ring-necked pheasant and dickcissel (Table 3), suggesting these species have relatively high densities in agricultural landscapes.

Contributions to Regional Population Sizes

The land enrolled in grassland CRP types declined by 16% from 4.8% of the PLJV region in 2016 to 4.1% of the region in 2020, with a mean of 4.4%. We presented results for grassland bird species (Vickery and Herkert 1999) that were included in the PLJV (2007) priority species list with greater than 25 detections on CRP land from 2016 through 2020. We estimated population size on CRP plantings in the PLJV region and the percentage of population in the PLJV region as a measure of the breeding population conserved by CRP. A percentage contribution to population size with CIs covering the 4.4% availability of CRP in the Region was interpreted as conserving breeding habitat in proportion to the aavialiability of CRP in the Region. Percentage contributions with CIs above the 4.4% availability of CRP in the region were interpreted as contributions greater than the availability of CRP in the PLJV region. In addition, we measured the percentage change for the CRP treatment effect relative to agricultural land as a measure of population response to implementing CRP in agricultural landscapes. Estimates of the percentage change from CRP treatments relative to agricultural land with CIs including the overall 4.4% availability of CRP were considered in proportion to the availability of CRP in the region.

The northern bobwhite population declined in the PLJV region between 2016 and 2020 ($\hat{\lambda}$ = 0.78; CI = 0.70, 0.86; $f > 0.99$; Fig. 7A). Population densities of northern bobwhite on CRP lands (Fig. 7B) were disproportional to population sizes in the PLJV over time (Fig. 7A), as reflected by variability in the percentage of the population conserved by CRP over time (Fig. 7C). Overall, the CRP program conserved 8.2% (CI = 3.7, 20.4) of the northern bobwhite population and the overall percentage of the population on CRP land was in proportion to the availability of CRP in the region ($f = 0.86$). The percentages were relatively stable over the first 4 years of study (Fig. 7C), and was greatest when population size was low, suggesting moderate habitat suitability of CRP. Overall, CRP conserved breeding habitat for 0.16 M (CI = 0.09, 0.32) northern bobwhite per year. The population change from the CRP treatment relative to agricultural land was in proportion to the availability of CRP in the region ($\hat{\delta}$ = 2.7; SD = 5.6; $f = 0.75$), and the CRP treatments increased population size by 66,408 (SD = 64,087) northern bobwhite per year.

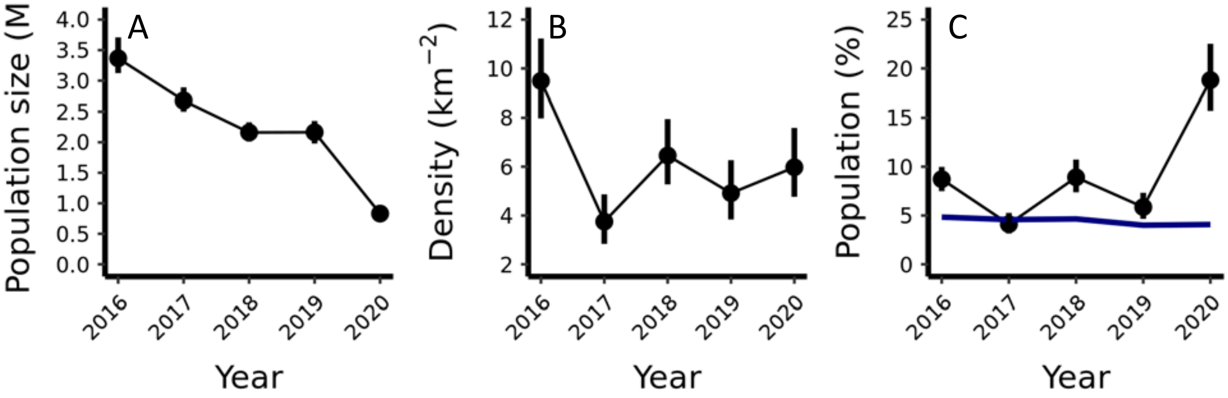

Figure 7. The northern bobwhite population size (million, M) in the Playa Lakes Joint Venture (PLJV) region (A), density (km^{-2}) on Conservation Reserve Program (CRP) plantings (B), and percentage (%) of breeding population conserved by CRP (C), 2016 – 2020. The error bars are 90% Credible Intervals and the blue line represents the percentage of CRP land in the PLJV region (C).

The scaled quail population declined in the PLJV region between 2016 and 2020 ($\hat{\lambda}$ = 0.77; CI = 0.68, 0.87; *f* > 0.99; Fig. 8A). Population densities of scaled quail on CRP lands (Fig. 8B) were disproportional to population sizes in the PLJV over time (Fig. 8A), as depicted by variability in the percentage of the population conserved by CRP over time (Fig. 8C). Overall, the CRP program conserved 2.1% (CI = 0.0, 5.9) of the scaled quail population (Fig. 8C) and the overall percentage of the population on CRP land was in proportion to the availability of CRP in the region (*f* = 0.78). The percentages varied over time (Fig. 8C), with contributions in 3 years in proportion to availability and lower than expected contributions in other years. The variability in the contributions of CRP did not suggest low quality habitat from the ideal free distribution, but the small contributions suggested marginal habitat suitability. Overall, CRP conserved breeding habitat for 36,409 (CI = 23, 96,915) scaled quail per year. The percentage population change from CRP treatments relative to agricultural land was in proportion to availability of CRP in the region ($\hat{\delta}$ = 1.6; SD = 2.0; *f* = 0.88) and the CRP treatment effects increased population size by 23,690 (SD = 35,823) scaled quail per year.

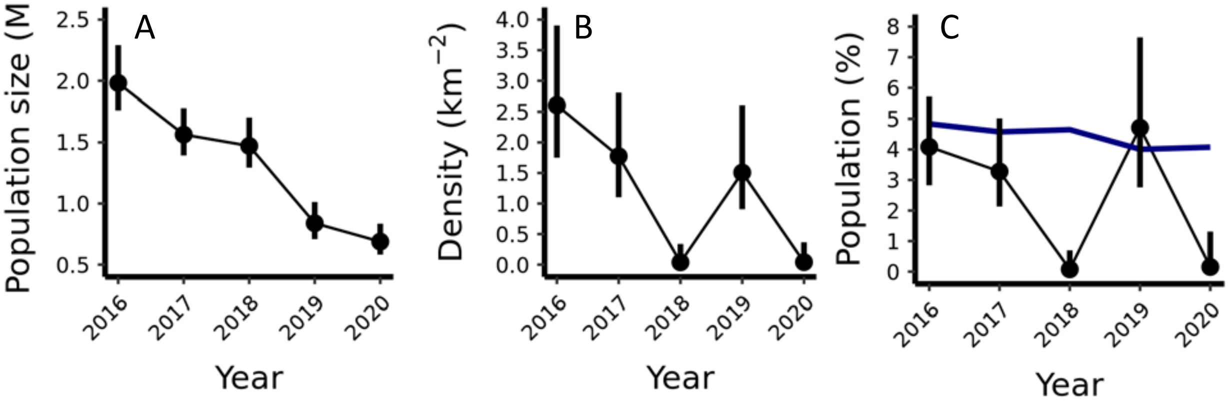

Figure 8. The scaled quail population size (million, M) in the Playa Lakes Joint Venture (PLJV) region (A), density (km^{-2}) on Conservation Reserve Program (CRP) plantings (B), and percentage (%) of breeding population conserved by CRP (C), 2016 – 2020. The error bars are 90% Credible Intervals and the blue line represents the percentage of CRP land in the PLJV region (C).

The ring-necked pheasant population declined in the PLJV region between 2016 and 2020 ($\hat{\lambda}$ = 0.87; CI = 0.81, 0.92; *f* > 0.99; Fig. 9A). Population densities of ring-necked pheasant on CRP lands (Fig. 9B) were proportional to population sizes in the PLJV (Fig. 9A), where the percentage of the population conserved by CRP was stable over the first 3 years and lower than expected over the last 2 years (Fig. 9C). Overall, the CRP program conserved 7.6% (CI = 0.8, 9.8) of the ring-necked pheasant population (Fig. 9C) and the overall percentage of the population on CRP land was in proportion to the availability of CRP in the region (*f* = 0.60). The percentages varied over the five years of study (Fig. 9C), but the temporal pattern was not consistent with low habitat suitability, with contributions suggesting moderate habitat suitability of CRP. Overall, CRP conserved breeding habitat for 44,538 (CI = 4,128, 91,004) ring-necked pheasants per year. However, the percentage population change from CRP treatments relative to agricultural land was less than the availability of CRP in the region ($\hat{\delta}$ = -3.4; SD = 2.8; *f* > 0.99).

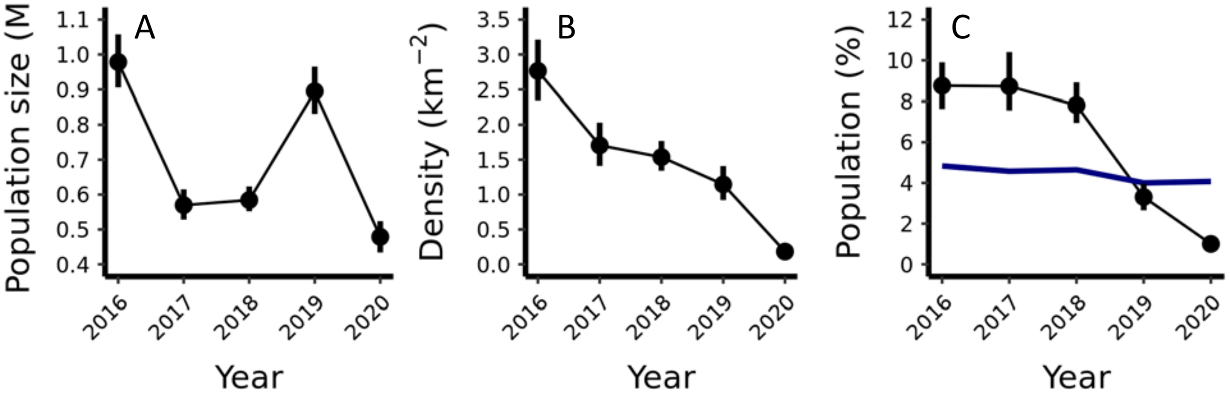

Figure 9. The ring-necked pheasant population size (million, M) in the Playa Lakes Joint Venture (PLJV) region (A), density (km^{-2}) on Conservation Reserve Program (CRP) plantings (B), and percentage (%) of breeding population conserved by CRP (C), 2016 – 2020. The error bars are 90% Credible Intervals and the blue line represents the percentage of CRP land in the PLJV region (C).

The common nighthawk population was stable in the PLJV region between 2016 and 2020 ($\hat{\lambda}$ = 1.11; CI = 0.85, 1.45; f = 0.76; Fig. 10A). Population densities of common nighthawk on CRP lands (Fig. 10B) were proportional to population sizes in the PLJV over time (Fig. 10A), as reflected by stability in the percentage of the population conserved by CRP over time (Fig. 10C). Overall, the CRP program accounted for 2.8% (CI = 1.2, 5.8) of the common nighthawk population (Fig. 10C) and the overall percentage of the population on CRP land was in proportion to the availability of CRP in the region (f = 0.86). The percentages were relatively stable over the five years of study (Fig. 10C), suggesting CRP provided consistent habitat over time, with contributions suggesting marginal suitability. Overall, CRP conserved breeding habitat for 18,145 (CI = 6,105, 37,218) common nighthawks per year. However, the percentage population change from CRP treatments relative to agricultural land was less than the availability of CRP in the region ($\hat{\delta}$ = -2.3; 2.5; f = 0.99).

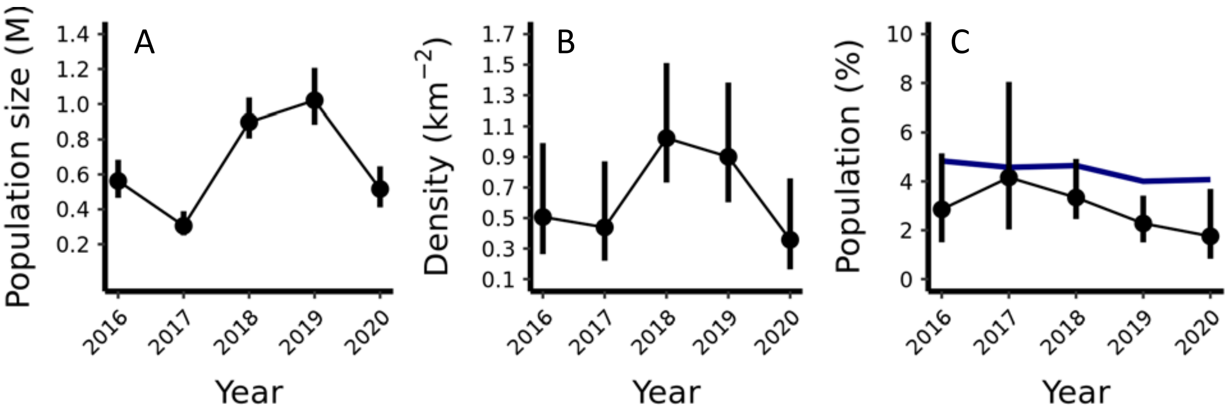

Figure 10. The common nighthawk population size (million, M) in the Playa Lakes Joint Venture (PLJV) region (A), density (km^{-2}) on Conservation Reserve Program (CRP) plantings (B), and percentage (%) of breeding population conserved by CRP (C), 2016 – 2020. The error bars are 90% Credible Intervals and the blue line represents the percentage of CRP land in the PLJV region (C).

The long-billed curlew population was stable in the PLJV region between 2016 and 2020 ($\hat{\lambda}$ = 0.79; CI = 0.56, 1.13; f = 0.87; Fig. 11A). Population densities of long-billed curlew on CRP lands (Fig. 11B) were disproportional to population sizes in the PLJV over time (Fig. 11A), as shown by variability in the percentage of the population conserved by CRP over time (Fig. 11C). Overall, the CRP program accounted for 1.4% (CI = 0.0, 22.8) of the long-billed curlew population (Fig. 11C) and the overall percentage of the population on CRP land was in proportion to the availability of CRP in the region (f = 0.58). The percentages varied over the five years of study (Fig. 11C), the contribution was high in the first 2 years when population size was large, and lower than expected from 2018 to 2020 when population sizes were low, suggesting low habitat suitability from the ideal free distribution. Overall, low population sizes and high annual variation prevented strong inference about the population of long-billed curlews conserved by CRP per year ($\widehat{N}$ = 187; CI = 12, 16,252). The percentage population change from CRP treatments relative to agricultural land exhibited very high annual variation and prevented strong inference about the proportion relative to the availability of CRP in the region ($\hat{\delta}$ = 0.3; SD = 10.5; f = 0.59).

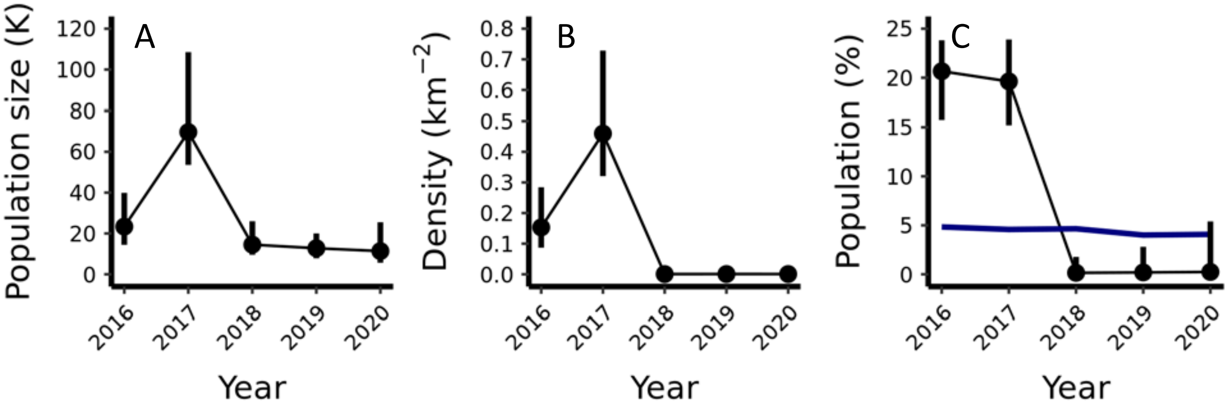

Figure 11. The long-billed curlew population size (thousand, K) in the Playa Lakes Joint Venture (PLJV) region (A), density (km^{-2}) on Conservation Reserve Program (CRP) plantings (B), and percentage (%) of breeding population conserved by CRP (C), 2016 – 2020. The error bars are 90% Credible Intervals and the blue line represents the percentage of CRP land in the PLJV region (C).

The Swainson's hawk population was stable in the PLJV region between 2016 and 2020 ($\hat{\lambda} = 0.90$; CI = 0.75, 1.07; $f = 0.86$; Fig. 12A). Population densities of Swainson's hawk on CRP lands (Fig. 12B) were disproportional to population sizes in the PLJV over time (Fig. 12A), as reflected by variability in the percentage of the population conserved by CRP over time (Fig. 12C). Overall, the CRP program conserved 6.1% (CI = 0.9, 16.2) of the Swainson's hawk population (Fig. 12C) and the overall percentage of the population on CRP land was in proportion to the availability of CRP in the region ($f = 0.62$). The percentages varied over time (Fig. 12C), with high contributions of CRP plantings in some years and marginal contributions in other years, the temporal pattern did not correspond to low habitat suitability from the ideal free distribution, with contributions suggesting moderate habitat suitability. Overall, CRP conserved breeding habitat for 5,041 (CI = 793, 13,431) Swainson's hawks per year. The percentage population change from CRP treatments relative to agricultural land exhibited very high annual variation and prevented strong inference about the proportion relative to the availability of CRP in the region ($\hat{\delta} = 1.1$; SD = 5.8; $f = 0.65$).

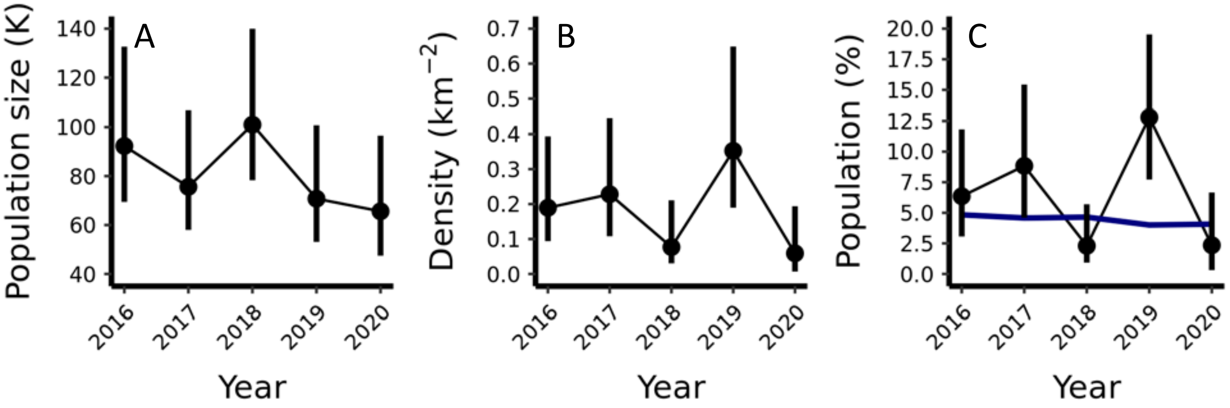

Figure 12. The Swainson's hawk population size (thousand, K) in the Playa Lakes Joint Venture (PLJV) region (A), density (km^{-2}) on Conservation Reserve Program (CRP) plantings (B), and percentage (%) of breeding population conserved by CRP (C), 2016 – 2020. The error bars are 90% Credible Intervals and the blue line represents the percentage of CRP land in the PLJV region (C).

The Cassin's sparrow population declined in the PLJV region between 2016 and 2020 ($\hat{\lambda}$ = 0.95; CI = 0.91, 0.99; $f > 0.99$; Fig. 13A). Except for 2017, population densities of Cassin's sparrow on CRP lands (Fig. 13B) were proportional to population sizes in the PLJV over time (Fig. 13A), as shown by stability in the percentage of the population conserved by CRP over time (Fig. 13C). Overall, the CRP program accounted for 9.6% (CI = 6.6, 11.0) of the Cassin's sparrow population (Fig. 13C) and the overall percentage of the population conserved by CRP land exceeded the availability of CRP in the region ($f > 0.99$). Except for 2017, the percentages were stable over the five years of study (Fig. 13C), suggesting CRP provided high habitat suitability. Overall, CRP conserved breeding habitat for 1.19 M (CI = 0.99, 1.51) Cassin's sparrows per year. The percentage population change from CRP treatments relative to agricultural land was greater than the availability of CRP in the region ($\hat{\delta}$ = 8.5; SD = 1.4; $f > 0.99$) and the mean population increase attributed to CRP was 1.03 M (SD = 0.20) per year.

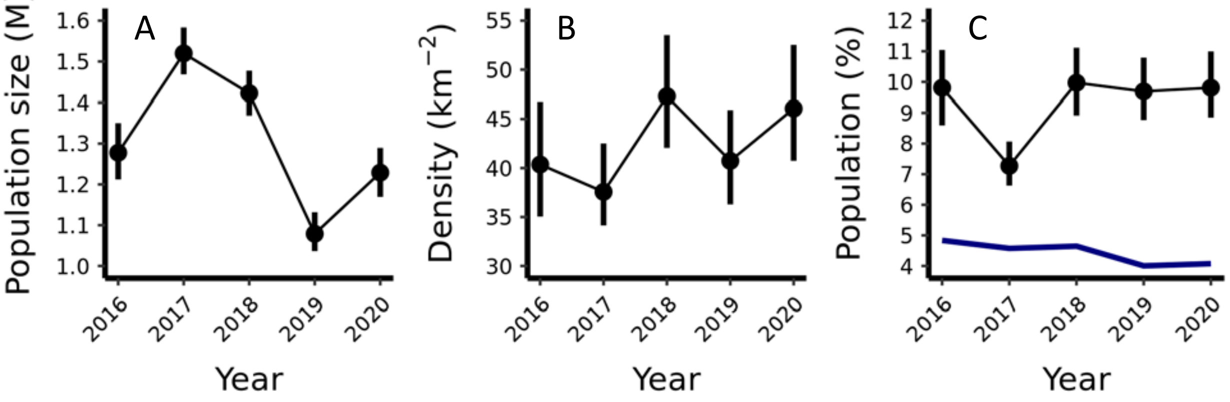

Figure 13. The Cassin's sparrow population size (million, M) in the Playa Lakes Joint Venture (PLJV) region (A), density (km^{-2}) on Conservation Reserve Program (CRP) plantings (B), and percentage (%) of breeding population conserved by CRP (C), 2016 – 2020. The error bars are 90% Credible Intervals and the blue line represents the percentage of CRP land in the PLJV region (C).

The grasshopper sparrow population declined in the PLJV region between 2016 and 2020 ($\hat{\lambda}$ = 0.86; CI = 0.82, 0.90; *f* > 0.99; Fig. 14A). Population densities of grasshopper sparrow on CRP lands (Fig. 14B) were proportional to population sizes in the PLJV over time (Fig. 14A), as illustrated by stability in the percentage of the population conserved by CRP over time (Fig. 14C). Overall, the CRP program accounted for 9.4% (CI = 7.6, 11.1) of the grasshopper sparrow population (Fig. 14C) and the overall percentage of the population on CRP land was greater than the availability of CRP in the region (*f* > 0.99). The percentages were relatively stable over the five years of study (Fig. 14C), suggesting CRP provided high habitat suitability. Overall, CRP conserved breeding habitat for 2.22 M (CI = 1.58, 3.27) grasshopper sparrows per year. The percentage population change from CRP treatments relative to agricultural land was greater than the availability of CRP in the region ($\hat{\delta}$ = 5.6; SD = 0.9; *f* = 0.96) and the mean population increase attributed to CRP was 1.44 M (SD = 0.33) per year.

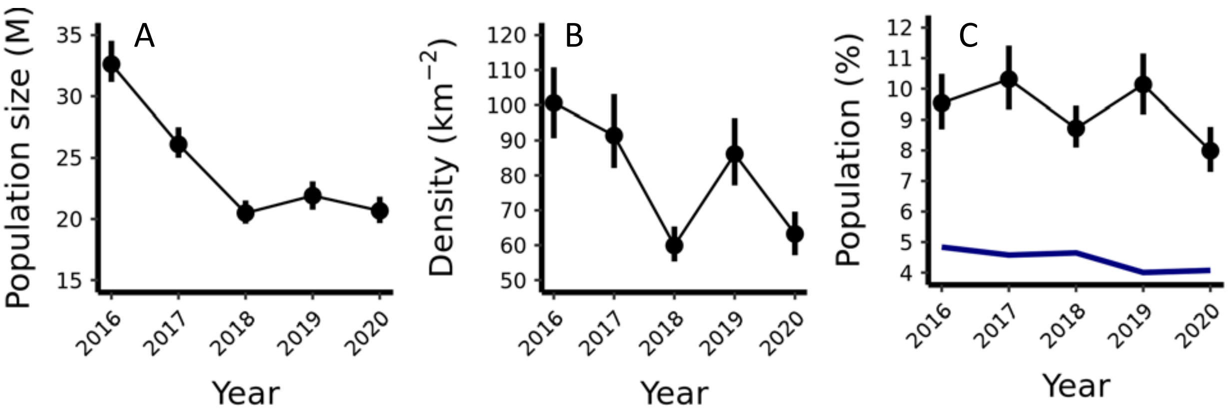

Figure 14. The grasshopper sparrow population size (million, M) in the Playa Lakes Joint Venture (PLJV) region (A), density (km^{-2}) on Conservation Reserve Program (CRP) plantings (B), and percentage (%) of breeding population conserved by (C), 2016 – 2020. The error bars are 90% Credible Intervals and the blue line represents the percentage of CRP land in the PLJV region (C).

The lark sparrow population increased in the PLJV region between 2016 and 2020 ($\hat{\lambda}$ = 1.09; CI = 0.99, 1.21; *f* > 0.94; Fig. 15A). Population densities of lark sparrow on CRP lands (Fig. 15B) were disproportional to population sizes in the PLJV over time (Fig. 15A), as depicted by variability in the percentage of the population conserved by CRP over time (Fig. 15C). Overall, the CRP program accounted for 4.8% (CI = 1.3, 11.0) of the lark sparrow population (Fig. 15C) and the overall percentage of the population on CRP land was in proportion to the availability of CRP in the region (*f* = 0.54). The percentages varied over time (Fig. 15C), with low or marginal contributions of CRP in some years and high contributions in years with high population sizes, which corresponded to low habitat suitability from the ideal free distribution. Overall, CRP conserved breeding habitat for 0.23 M (CI = 0.05, 0.70) lark sparrows per year. The percentage population change from CRP treatments relative to agricultural land exhibited high annual variation and prevented reliable inference about the proportion relative to availability of CRP in the region ($\hat{\delta}$ = 1.7; SD = 3.6; *f* = 0.76).

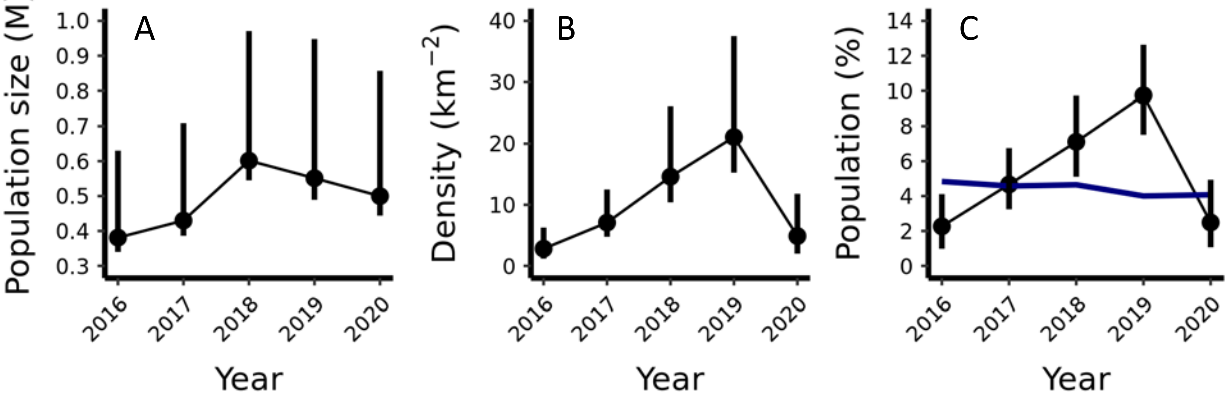

Figure 15. The lark sparrow population size (million, M) in the Playa Lakes Joint Venture (PLJV) region (A), density (km^{-2}) on Conservation Reserve Program (CRP) plantings (B), and percentage (%) of breeding population conserved by CRP (C), 2016 – 2020. The error bars are 90% Credible Intervals and the blue line represents the percentage of CRP land in the PLJV region (C).

The lark bunting population declined in the PLJV region between 2016 and 2020 ($\hat{\lambda}$ = 0.73; CI = 0.59, 0.88; *f* > 0.99; Fig. 16A). Population densities of lark bunting on CRP lands were disproportional to population sizes in the PLJV over time (Fig. 16A, Fig. 16B), as shown by variability in the percentage of the population conserved by CRP over time (Fig. 16C). Overall, the CRP program accounted for 5.2% (CI = 1.3, 9.6) of the lark bunting population (Fig. 16C) and the overall percentage of the population on CRP land was in proportion to the availability of CRP in the region (*f* = 0.61). The percentages varied over the five years of study (Fig. 16C), with greater than expected contributions when population size was low, and lower than expected contributions when population sizes were high, which suggested moderate habitat suitability. Overall, CRP conserved breeding habitat for 0.37 M (CI = 0.07, 0.73) lark buntings per year. The percentage population change from CRP treatments relative to agricultural land was in proportion to the availability of CRP in the region ($\hat{\delta}$ = 2.4; SD = 2.7; *f* = 0.80), and the mean population increase attributed to CRP was 0.24 M (SD = 0.41) per year.

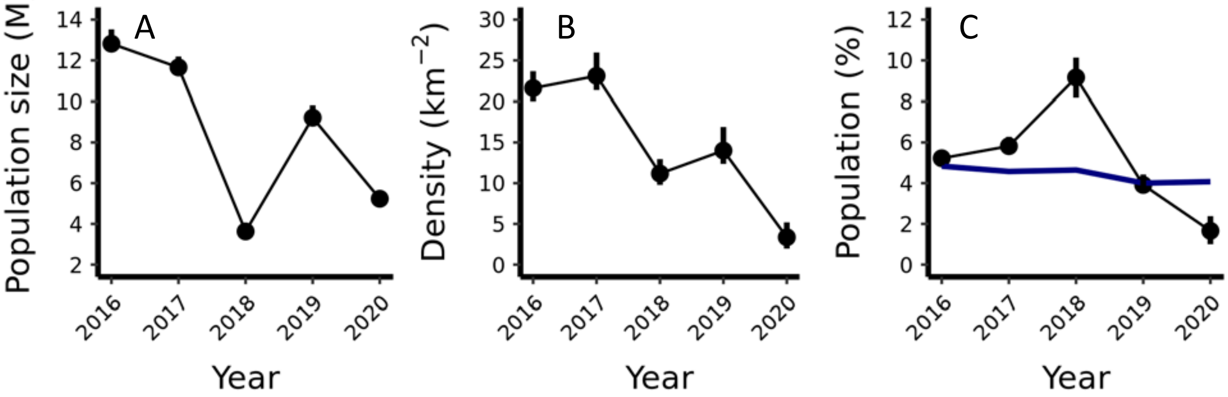

Figure 16. The lark bunting population size (million, M) in the Playa Lakes Joint Venture (PLJV) region (A), density (km^{-2}) on Conservation Reserve Program (CRP) plantings (B), and percentage (%) of breeding population conserved by CRP (C), 2016 – 2020. The error bars are 90% Credible Intervals and the blue line represents the percentage of CRP land in the PLJV region (C).

The eastern meadowlark population declined in the PLJV region between 2016 and 2020 ($\hat{\lambda}$ = 0.88; CI = 0.83, 0.93; *f* > 0.99; Fig. 17A). Population densities of eastern meadowlark on CRP lands (Fig. 17B) were disproportionate to population sizes in the PLJV over time (Fig. 17A), as depicted by variability in the percentage of the population conserved by CRP over time (Fig. 17C). Overall, the CRP program accounted for 7.8% (CI = 4.2, 11.3) of the eastern meadowlark population (Fig. 17C) and the overall percentage of the population on CRP land was greater than the availability of CRP in the region (*f* = 0.92). The percentages showed cyclic dynamics over time (Fig. 17C), with high contributions in 4 of the 5 years and marginal contributions in 2017. The temporal pattern did not correspond to low habitat suitability from the ideal free distribution, and the contributions suggested high habitat suitability. Overall, CRP conserved breeding habitat for 0.25 M (CI = 0.13, 0.46) eastern meadowlarks per year. The percentage population change from CRP treatments relative to agricultural land was in proportion to the availability of CRP in the region ($\hat{\delta}$ = 2.6; SD = 3.4; *f* = 0.60), and the mean population increase attributed to CRP was 88,582 (SD = 123,756) per year.

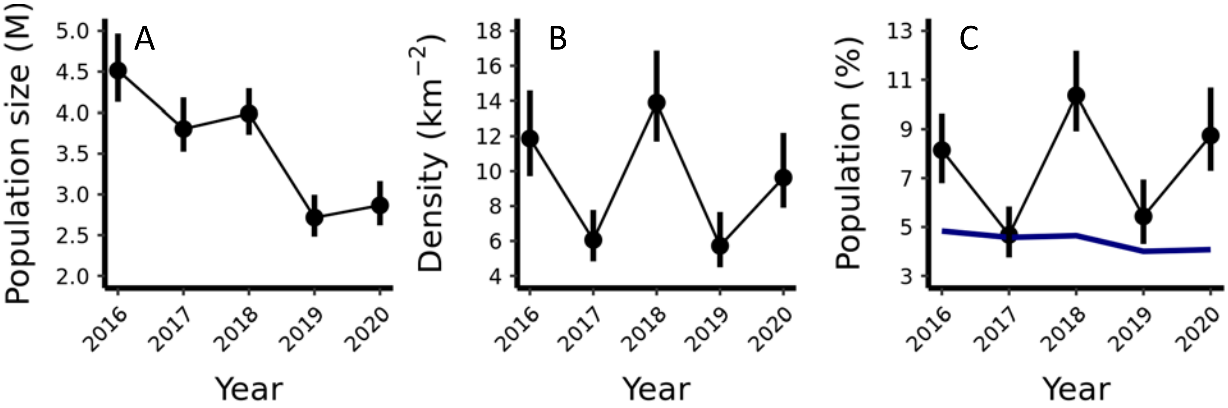

Figure 17. The eastern meadowlark population size (million, M) in the Playa Lakes Joint Venture (PLJV) region (A), density (km^{-2}) on Conservation Reserve Program (CRP) plantings (B), and percentage (%) of breeding population conserved by (C), 2016 – 2020. The error bars are 90% Credible Intervals and the blue line represents the percentage of CRP land in the PLJV region (C).

The western meadowlark population declined in the PLJV region between 2016 and 2020 ($\hat{\lambda}$ = 0.90; CI = 0.86, 0.95; *f* > 0.99; Fig. 18A). Population densities of western meadowlark on CRP lands (Fig. 18B) were proportional to population sizes in the PLJV over time (Fig. 18A), as illustrated by stability in the percentage of the population conserved by CRP over time (Fig. 18C). Overall, the CRP program conserved 5.1% (CI = 4.2, 5.7) of the western meadowlark population (Fig. 18C) and the overall percentage of the population on CRP land was in proportion to the availability of CRP in the region (*f* = 0.86). Except for 2020, the percentages were stable over time (Fig. 18C), suggesting high habitat suitability of CRP. Overall, CRP conserved breeding habitat for 0.62 M (CI = 0.49, 71) western meadowlarks per year. However, the percentage population change from CRP treatments relative to agricultural land was less than the availability of CRP in the region ($\hat{\delta}$ = 0.3; CI = -0.4, 1.3; *f* > 0.99), and the mean population increase attributed to CRP was 41,585 (SD = 62,862) per year.

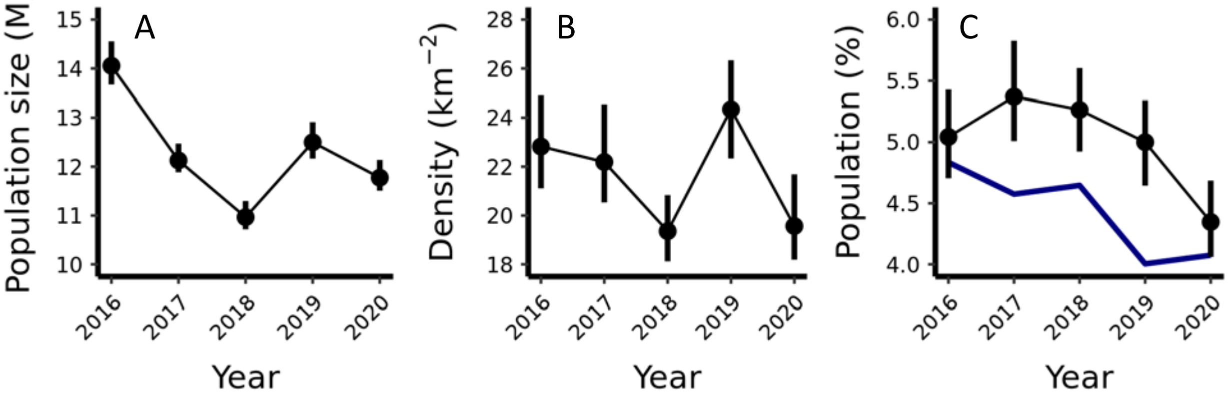

Figure 18. The western meadowlark population size (M) in the Playa Lakes Joint Venture (PLJV) region (A), density (km^{-2}) on Conservation Reserve Program (CRP) plantings (B), and percentage (%) of breeding population conserved by CRP (C), 2016 – 2020. The error bars are 90% Credible Intervals and the blue line represents the percentage of CRP land in the PLJV region (C).

The dickcissel population was stable in the PLJV region between 2016 and 2020 ($\hat{\lambda} = 0.93$; CI = 0.83, 1.1; $f = 0.89$; Fig. 19A). Population densities of dickcissel on CRP lands were disproportional to population sizes in the PLJV over time (Fig. 19A, Fig. 19B), as shown by variability in the percentage of the population conserved by CRP over time (Fig. 18C). Overall, the CRP program accounted for 3.6% (CI = 0.9, 12.0) of the dickcissel population (Fig. 19C) and the overall percentage of the population on CRP land was in proportion to the availability of CRP in the region ($f = 0.42$). The percentages varied over the five years of study (Fig. 19C), with low contributions in 3 years and large contributions in 2016 and 2018. The temporal pattern did not correspond to low habitat suitability from the ideal free distribution, with contributions suggesting marginal habitat suitability. Overall, CRP conserved breeding habitat for 0.28 M (CI = 0.05, 0.74) dickcissels per year. However, the percentage population change from CRP treatments relative to agricultural land was less than the availability of CRP in the region ($\hat{\delta} = -3.8$; SD = 3.0; $f = 0.99$).

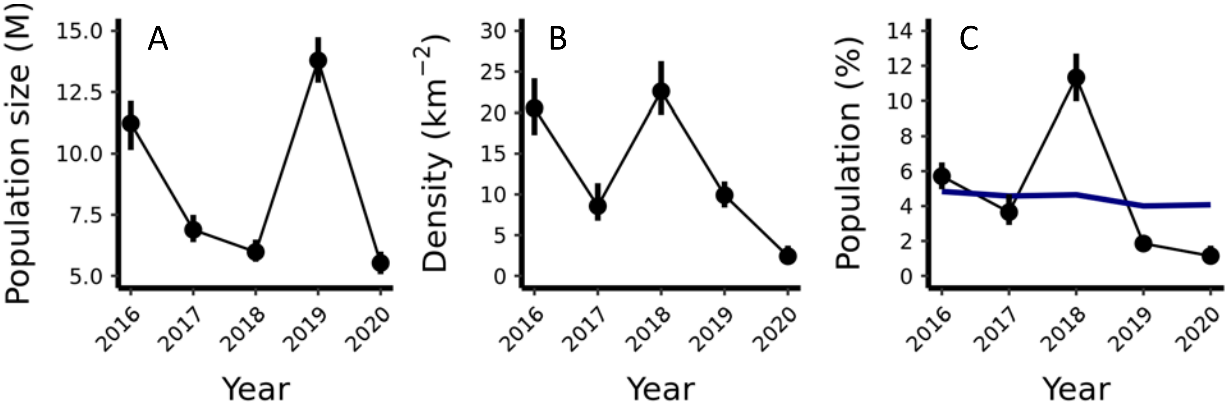

Figure 19. The dickcissel population size (million, M) in the Playa Lakes Joint Venture (PLJV) region (A), density (km^{-2}) on Conservation Reserve Program (CRP) plantings (B), and percentage (%) of breeding population conserved by (C), 2016 – 2020. The error bars are 90% Credible Intervals and the blue line represents the percentage of CRP land in the PLJV region (C).

Discussion

We developed a post-stratification framework for the IMBCR program to monitor the effectiveness of CRP for increasing the abundance of grassland birds relative to agricultural lands and native grassland. We estimated density and population size to evaluate contributions of CRP lands to bird populations in the PLJV region. We hypothesized that habitat loss from the conversion of native grassland to agricultural landscapes produced population declines of grassland birds (Stanton et al. 2018). We found evidence for population declines in agricultural landscapes for 6 species: scaled quail, Cassin's sparrow, grasshopper sparrow, lark sparrow, lark bunting and western meadowlark. Except for the lark sparrow, populations of these species are declining in the PLJV region over the last 5 years. Three species, the northern bobwhite, ring-necked pheasant and dickcissel, were more abundant in agricultural landscapes than native grassland. Of these species, the northern bobwhite and ring-necked pheasant are declining in the region, which suggested processes such as habitat degradation in native grassland or ecological traps in agricultural landscapes may be threatening process for these species (Battin 2004).

Next, we hypothesized that CRP plantings in agricultural landscapes would increase the abundance of grassland birds. We found evidence that CRP treatments increased Cassin's sparrow and grasshopper sparrow abundance, confirming findings for these species from an intensive, but local-scale, single-year study in the range of the lesser prairie chicken (Pavlacky et al. 2021). However, we were unable to detect a positive CRP treatment effect for the eastern meadowlark at a large spatial and temporal scale that was detected by Pavlacky et al. (2021) at a smaller spatial and temporal scale. In contrast to Pavlacky et al. (2021), we did not detect negative CRP treatment effects for the ring-necked pheasant, common nighthawk and dickcissel over bioregional-scales and five-years of study. Because abundance over time often shows high annual variation due to stochastic processes unrelated to conservation effects of interest (Joseph et al. 2006, Pollock 2006), monitoring over large temporal and spatial scales may be necessary to estimate robust treatment effects for some species (Pavlacky et al. 2017). For example, several species of grassland birds are known to be nomadic in response to annual variability in weather patterns (George et al. 1992, Niemuth et al. 2008), and the resulting annual variation may interfere with estimating CRP treatment effects. While the spatial extent of the PLJV region is likely large enough to subsume nomadic movements of grassland birds, high levels of observed annual variation may reduce statistical power and require additional years of data to adequately resolve treatment effects for some species. We were unable to detect treatment effects for CRP plantings relative to agricultural lands for 11 of the 13 species, suggesting it may be incorrect to assume agricultural landscapes provide little value as habitat for the grassland species in this study. However, the agricultural land designation from the IMBCR program includes both cropland and rural vegetation along ditches, fence rows and corners of center-pivot irrigation, and future post-stratification may be improved by a GIS exercise to separate cropland and rural vegetation.

Recent emphasis on planting native CRP seed mixes led us to hypothesize that CRP planting would produce high habitat suitability for grassland birds (Thompson et al. 2009). We found evidence of high habitat suitability on CRP plantings relative to native grassland for the northern bobwhite, Cassin's sparrow and grasshopper sparrow, confirming findings for these species from a local-scale, single-year study in the range of the lesser prairie chicken (Pavlacky et al. 2021). We found abundance on CRP plantings were similar to abundance on native grasslands for 9 of the remaining 10 species, suggesting CRP provided suitable habitat for scaled quail, ring-necked pheasant, common nighthawk, long-billed curlew, Swainson's hawk, lark sparrow and lark bunting. However, we caution the treatment effects or lack of effects on abundance may not translate to habitat quality resulting in higher survival and reproduction of the species (Van Horne 1983). Only 1 of the 13 species, the western meadowlark, demonstrated lower abundance on CRP plantings relative to native grasslands.

We estimated population sizes for grassland priority species on CRP plantings relative to population sizes for the PLJV region to determine the contribution of CRP to regional bird populations. We found consistent and large contributions of CRP plantings to Cassin's sparrow and grasshopper sparrow populations and consistent and smaller contributions to western meadowlark populations in the PLJV region. The percentage contribution to population size for these species was relatively stable across years, suggesting CRP plantings provided high habitat suitability according to the ideal free distribution (Fretwell and Lucas 1969). According to the ideal free distribution of territorial species (Fretwell and Lucas 1969), individuals establish territories in high quality habitat first, then after the high quality habitat is fully occupied, individuals establish territories in low quality habitat. When population sizes of species decline over time, density of species are expected to decline in the low quality habitat first. Alternately, when population sizes increase over time, density of species increase in low quality after high quality habitat reaches carrying capacity. Because the idea free distribution predicts relatively stable density in high quality habitat and volatile density in low quality habitat when populations vary over time, we predicted a stable proportion of population conserved by CRP over time would be indicative of high quality of restored habitat.

We found large, but variable contributions of CRP to eastern meadowlark populations over time, suggesting high habitat suitability on CRP. The cyclic population dynamics on CRP plantings did not reflect the population decline occurring in the region suggesting CRP is making considerable contributions to population size in some years when population size is low. In addition, we found variable, but moderate contributions of CRP to population size for the northern bobwhite, ring-necked pheasant, Swainson's hawk and lark bunting, suggesting CRP provides important contributions to population size in years when population size is low. The scaled quail and common nighthawk demonstrated high variability and low contributions of CRP to regional populations, suggesting CRP provide marginal habitat for these species.

Two species, the long-billed curlew and lark sparrow, showed temporal patterns in contribution of CRP that suggested low habitat suitability of CRP. For these species, CRP provided large contributions to population size in years with large populations sizes, and lower than expected contributions when population size in the region was small.

Land enrolled in CRP plantings conserved breeding habitat for 1.2 million Cassin's sparrows per year, and conserved 10% of the population, which was greater than the 4.4% availability of CRP in the PLJV region. In addition, CRP program conserved 2.2 million grasshopper sparrows per year and conserved 9% of the population at a rate proportionally greater than the availability of CRP in the region. Land enrolled in CRP plantings conserved breeding habitat for 253,651 eastern meadowlarks per year, and conserved 9% of the population at a rate greater than the 4.4% availability of CRP in the PLJV region.

Land enrolled in CRP plantings made considerable contributions to grassland bird populations by conserving 158,459 northern bobwhites per year (8% of the population), 44,538 ring-necked pheasants per year (8% of the population), 5,041 Swainson's hawks per year (6% of the population), 229,699 lark sparrows per year (5% of the population), 368,788 lark buntings per year (5% of the population), and 620,278 western meadowlarks per year (5% of the population). We found smaller contributions to breeding populations per year for the scaled quail, common nighthawk, long-billed curlew and dickcissel.

Management Implications

Monitoring is integral to the management and conservation of wildlife populations (Marsh and Trenham 2008, Jones 2011), and is a key part of decision making and adaptive management, providing the means to assess impacts of management actions and improve understanding of system dynamics (Nichols and Williams 2006, Lyons et al. 2008). The hierarchical design of the IMBCR program provides a framework for determining species responses to conservation practices and understanding how local conservation efforts scale-up to influence regional bird populations (Pavlacky et al. 2017). We used post-stratification (Thomas et al. 2010) within an impact-reference design (Morrison et al. 2008) to evaluate the effectiveness of CRP relative to agricultural lands and native grassland. By collecting seamless monitoring data over large regions composed of public, tribal and private land ownership, the PLJV partnership provided the framework for estimating avian population sizes for CRP relative to population sizes in the region.

Effectiveness monitoring is useful for learning about the success of management actions, and also plays important roles in decision making and adaptive management (Lyons et al. 2008). The impact-reference treatment effects (Morrison et al. 2008) for evaluating avian population densities on CRP relative to agricultural land (Table 3) provides predictions for increases in abundance expected from taking cropland out of production and planting CRP grassland. For example, enrolling agricultural land into CRP grassland at the extent of the past 5 years is expected to increase the populations of the Cassin's sparrow by 1.0 M and grasshopper sparrow by 1.4 M per year. The percentage population change for CRP relative to agricultural land exceeded the PLJV (2007) annual population targets for these species. The percentage population change relative to agricultural land for the grasshopper sparrow (5.6%, CI = 4.4, 7.5) and Cassin's sparrow (8.5%, CI = 5.6, 10.1) exceeded the 2.3% annual population target for the PLJV (2007). This suggested a similar investment in CRP conservation over the 30 year PLJV planning cycle would increase populations of the grasshopper sparrow by 43.2 M (CI = 33.3, 53.1) and Cassin's sparrow by 31.0 M (CI = 25.1, 36.9), exceeding the population target over the PLJV (2007) planning cycle. Although the grasshopper sparrow and Cassin's sparrow were the only species with CRP treatment effects greater than the availability of CRP in the region, CRP treatment effects for the northern bobwhite, scaled quail, lark bunting and eastern meadowlark were in proportion to the availability of CRP in the region, suggesting linear population responses to implementing CRP in the PLJV region.

The CLU data for active CRP contracts indicate CRP enrollment has declined by 16% in the PLJV region. Declining CRP may explain the population declines of species with high or moderate habitat suitability of CRP, including the northern bobwhite, ring-necked pheasant, grasshopper sparrow, lark bunting and eastern meadowlark. The CIs for the population decline of these species include the 16% decline of CRP, indicating the population declines are of the same magnitude as the CRP decline. In contrast, the CIs for the population declines of the Cassin's sparrow and western meadowlark do not include the 16% decline of CRP in the region, suggesting declining CRP may not be having a large effect on populations of these species. However, it may be important to recognize that a large percentage of expired CRP in the Southern Plains retain restored grassland after the contracts expire (C. A. Hagen, Oregon State University, personal communication), suggesting other potential process may be contributing to the population trajectories of the species. We found the annual variation of the ring-necked pheasant (Fig. 9A) and western meadowlark (Fig. 18A) showed higher densities following wet years and lower densities following dry years (Fig. 6, NOAA National Centers for Environmental information 2021). Annual variation of the Cassin's sparrow population

(Fig. 13A) showed higher densities in dry years, whereas declines of the northern bobwhite (Fig. 7A), grasshopper sparrow (Fig. 14A) and eastern meadowlark populations (Fig. 17A) may be tracking the overall time series for declining precipitation in the Great Plains (Fig. 6, NOAA National Centers for Environmental information 2021).

Nevertheless, we found evidence that CRP at the mean extent over the past five years increased populations of the northern bobwhite by 66,408, scaled quail by 23,690, lark bunting by 242,554 and eastern meadowlark by 88,582 per year. The percentage population change relative to agricultural land for these species met the PLJV (2007) annual population objectives for the northern bobwhite, scaled quail, lark bunting and eastern meadowlark. However, high annual variation in the population responses to CRP prevented strong inference to population change for the other species. The small sample size of grid cells containing CRP plantings (*n* = 27) and high annual variation in abundance on CRP plantings suggested more years of study may needed to adequately estimate the population change for these species. The impact-reference design used in this study often show bias relative to true experiments with random assignment of treatments to experimental units (Adams et al. 2019). Although the before-after-control-impact design is better able to tease apart temporal and spatial variation than the impact-reference design (Morrison et al. 2008), the results of this study suggested CRP may provide suitable habitat for several grassland bird species and be an effective conservation strategy for increasing the abundance of priority species in the PLJV region. Because of uncertainty in the population responses for several of priority species, we suggest treatment effects may be best incorporated within a framework of adaptive management, where additional monitoring is expected to reduce uncertainty and improve learning over time (Lyons et al. 2008).

The effectiveness monitoring of Farm Bill conservation practices provides confidence to land managers and resource professionals, as well as increases accountability for the evidence-based management of natural resources in the public trust (Briske et al. 2017). The population responses can be used in population viability simulations to ask how much CRP is required to meet population targets for species of conservation need. In addition, the population responses can be used to understand the consequences of CRP enrollment and expiry on grassland birds in the PLJV region over time. For example, the results suggest changes in the enrollment or expiry of CRP may dramatically affect the population sizes of the northern bobwhite, ring-necked pheasant, Swainson’s hawk, Cassin’s sparrow, grasshopper sparrow, lark bunting, eastern meadowlark and western meadowlark. Finally the population responses to CRP can be used to set conservation priorities in the region (Wilson et al. 2009) to address the “what to do” and “where to do it” questions in conservation planning (Wilson et al. 2007). For example, systematic conservation planning (McBride et al. 2010) can be used to investigate tradeoffs involved with maximizing the population size of grassland birds, maximizing crop production and minimizing costs to private landowners to arrive at optimal solutions to the conservation of Great Plains agro-ecosystems (Behrman et al. 2015).

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Appendix A: Data Analysis

Distance Sampling Analysis

Density and Abundance Estimation

State process

We developed a zero-inflated N-mixture model (Royle 2004, Sillett et al. 2011) to estimate density and abundance for all strata and superstrata across all species with sufficient data. For a given species, the true occupancy state of point count location *k* in grid *j*, stratum *i*, and year *t* is distributed

$$z_{ijkt} \sim Bern(\psi_i).$$

The number of independent clusters of individuals, *N*, of a given species at point count location *k* in grid *j*, stratum *i*, and year *t* came from a Poisson distribution

$$N_{ijkt} \sim Poisson(\lambda_{ijt} \times z_{ijkt})$$

with mean λ_{ijt}. Abundances at all points within a grid came from a distribution with the same mean to account for the lack of independence between points, and we modeled λ as a function of time to estimate trend for each stratum:

$$\log(\lambda_{ijt}) = \alpha_i + r_i(t-1) + \varepsilon_j,$$

where α and r are stratum-specific intercepts and trends, respectively, and ε is a grid-specific random effect.

To avoid predicting species occurrence outside of observed ranges, we fixed ψ to 0 for all strata in which the species was never observed and used a prior informed by the observed proportion of grid-year combinations in a stratum in which the species was detected

$$logit(\psi_i) \sim Normal(\mu_{\psi_i}, \sigma_\psi^2),$$

where μ_{ψ_i} is the stratum-specific naïve occupancy and σ_ψ^2 is the annual variation in occupancy probabilities shared across strata. All other parameters had vague priors:

$$\boldsymbol{\alpha} \sim Normal(0,4),$$
$$\exp(\boldsymbol{r}) \sim Uniform(0.25,1.75),$$
$$\boldsymbol{\varepsilon} \sim Normal(0, \sigma_\varepsilon^2),$$

and

$$\sigma_\varepsilon^2 \sim Uniform(0,5).$$

We derived density, *D*, at the point count location as

$$D_{ijkt} = \frac{N_{ijkt} \times s}{A_c},$$

where A_c is the area of the point count circle (see *Observation process* section below) and *s* is the cluster size, which was sampled from the distribution

$$s \sim Gamma(k, \theta) + 1,$$

where *k* and θ were derived from the mean and variance of observed cluster sizes. We subtracted 1 from the mean when calculating *k* and θ and added 1 to the random variable to ensure cluster sizes were $\geq$ 1. We derived stratum-level density estimates by averaging all point-level density estimates within each stratum, and we took the area-weighted average of strata estimates to obtain superstrata estimates.

Observation process

We estimated the probability of detecting an independent cluster of individuals by fitting distance functions to the distance data collected during surveys (Buckland et al. 2001). We fit four detection models including: 1) half-normal constant [HN(.)], 2) hazard rate constant [Haz(.)], 3) half-normal year [HN(t)], and 4) hazard rate year [Haz(t)].

We removed the furthest 10% of observed detection distances from the data set and binned the remaining detections into 10 evenly spaced distance classes. For half-normal functions, we calculated the detection probability, *p*, for each distance class, *l*, as:

$$p_l = \frac{2\pi \int_{c=b_l}^{c=b_{l+1}} \exp\left(-\left(\frac{c^2}{2\theta^2}\right)\right) c \, dc}{A_l},$$

where b_l and b_{l+1} are the cutpoints for *l*, θ is the half-normal shape parameter, and A_l is the area of *l*. Because of the lack of an analytical solution to the integral of the hazard rate function, we calculated *p* at the midpoint, *m*, of each distance class

$$p_l = 1 - \exp\left(-\left(\frac{m_l}{a}\right)^b\right).$$

To allow detection probabilities to vary by year, we sampled year-specific shape parameters from hyperdistributions:

$$\theta_t \sim Normal(\mu_\theta, \sigma_\theta^2),$$
$$a_t \sim Normal(\mu_a, \sigma_a^2),$$

and

$$b_t \sim Normal(\mu_b, \sigma_b^2),$$

with priors of

$$\mu_\theta \sim Unif(0{,}1000),$$
$$\mu_a \sim Unif(0{,}500),$$
$$\sigma_\theta, \sigma_a, \mu_b \sim Unif(0{,}100),$$

and

$$\sigma_b \sim Unif(0{,}25).$$

We then multiplied p_l by the proportional area of *l* to account for the probability that a cluster is within distance class *l* and obtain π_l, the probability a cluster is present within distance class *l* and is detected,

$$\pi_{lt} = \frac{p_{lt}A_l}{\sum_{l=1}^{L} A_l}.$$

We calculated the overall capture probability, p_{cap}, as

$$p_{cap} = \sum_{l=1}^{L} \pi_l,$$

and modeled the number of detections in each distance class at each point count location in year *t* as

$$\boldsymbol{y}_{\boldsymbol{ijkt}} \sim Multinom(\boldsymbol{\pi}_t, N_{ijkt}).$$

Detection model selection

To find the most parsimonious detection function while minimizing computing time, we fit detection-only models to the distance data, using the four model structures described above. We used the Watanabe-Akaike Information Criterion (WAIC; Watanabe 2010, Hooten and Hobbs 2015) to select the most parsimonious detection structure and then used that structure for detection probabilities in the full model to estimate density and abundance.

Superstratum trends

We developed a post-hoc approach to estimate trends for superstrata. Using the rolled-up estimates of density for superstratum, *i*, we fit a general linear model (GLM) to the samples from each Bayesian iteration, *m*,

$$\log\left(\widehat{D}_{itm}\right) \sim \alpha_{im} + r_{im}(t-1).$$

Fitting a GLM across iterations allowed us to incorporate uncertainty in trends due to uncertainty around density estimates, but it did not account for temporal variation. To incorporate this second form of variation, we sampled a random intercept ($\tilde{\alpha}_{im}$) and slope ($\tilde{r}_{im}$) for each iteration using the mean and standard error estimated using the GLM and made inference on the distribution of the resampled values,

$$\tilde{\alpha}_{im} \sim Normal(\mu_{\alpha_{im}}, SE_{\alpha_{im}})$$

and

$$\tilde{r}_{im} \sim Normal(\mu_{r_{im}}, SE_{r_{im}}).$$

Automated Analysis

We updated our analytical methods and are used Bayesian hierarchical models specifically designed for analysis of IMBCR data. We performed all data and output manipulation in R (R Core Team, 2019) and model fitting in JAGS (Plummer 2003, 2017) using the R package jagsUI (Kellner 2018). The R code called the raw data from the IMBCR Structured Query Language (SQL) server database and reformatted the data into a form usable with the JAGS code. We allowed the input of all data collected in a manner consistent with the IMBCR design to increase the number of detections available for estimating global detection rates for population density and site occupancy. The R code provided an automated framework for combining strata-level estimates of population density and site occupancy at multiple spatial scales, as well as estimating the standard deviations and credible intervals for the combined estimates.

We fit initial models to all species with at least 30 detections for density estimation and 10 detections for occupancy estimation. For density estimation, we fit the full model after determining whether there were enough detections based on results from the detection-only model fits. In some cases for both density and occupancy estimation, it was necessary to use a less parsimonious detection structure or simplified model structure to facilitate model convergence. We currently maintain version control of the automated analysis code in the Bird Conservancy repository (Atlassian Stash, version 3.6).