An evaluation of conservation efforts in Midwestern agricultural landscapes for birds

by

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A dissertation submitted to the graduate faculty

in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

Major: Wildlife Ecology

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Ames, Iowa

2023

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ACKNOWLEDGMENTS

There is a legion of individuals who have made this dissertation possible through mentoring, collaboration, and friendship. I give a deep, heartwarming thank you to my family for endless support throughout my undergraduate and graduate school ventures. I could not have reached the finish line without the love and support of Katie Fernald. Special thanks to my Plan of Study Committee – Dr. Lisa Schulte Moore, Dr. Bob Klaver, Dr. Anna Tucker, Dr. Adam Janke, and Dr. Jarad Niemi – for their consultation and investment in my professional development. My advisor, Dr. Lisa Schulte Moore, was patient and believed in me during the toughest of times. Dr. Bob Klaver's kindness was much needed at several junctions. My Plan of Study Committee at Tarleton State University, Dr. Heather Mathewson, Dr. Thomas Schwertner, and Dr. Jeff Breeden, made the path to a Ph.D. possible. Dr. John Quinn gave me my first research job at the University of Nebraska-Lincoln. I am grateful for NREM graduate students for their warmth and friendship during the last 5 years. They are, in no particular order: Bobby Cope, Ellen Audia, Caleb Krueger, Sarah Hoepfner, Matt Stephenson, Brett Kelly, Dylan Osterhaus, Sam Leberg, Marty Simonson, Nathan Tillotson, Kyle Olivencia, Andrew Annear, Brandt Boekhout, Tom Miles, Katie Miller, Lindsey Gapinski, Riggs Wilson, Ben West, Sally Carullo, Cait Campbell, Kelsey Wells, Rachel Saxon, Nicole Bosco, Seth Renner, Erik Griffen, Lewis Wiechmann, Tiffanie Stone, Andres Vargas, Zach Dienes, and EB Wiezen. Thank you to undergraduate researchers, Matt Thiesen, Drake Fehring, Aric Runge, Mary Kate Shaver, and Joseph McGovern for their steadfast work in the field and lab. I am thankful for the Eagle Grove Library and its friendly librarians for providing refuge between pheasant trap checks on cold winter days. And of course…thank you to my best friend Raven who has been my co-pilot for the last 7 years.

ABSTRACT

Loss of natural areas to agriculture has long been the largest threat to biodiversity. With a focus on habitat restoration for grassland birds, I evaluated three conservation practices—prairie strips, oxbow wetland restoration, and native grassland reconstruction—to inform their further integration into agricultural landscapes of the U.S. Midwest. I found that the density of 17 grassland birds species as a group was 2.6-fold higher in fields with prairie strips (3.65 birds/ha) than control fields without prairie strips (1.40 birds/ha). The largest increases in bird densities for treatment fields occurred between years 3 and 4 of establishment, congruent with typical shifts in management of prairie reconstructions. Large patch prairie, conventional cropland, and cropland with prairie strips had similar levels of species richness of spring vocalizing birds. I found higher species richness in bird communities associated with oxbow wetland restorations than nearby unrestored sites. In a multitaxa assesment of wildlife response to native grassland reconstruction compared to exotic, cool-season grasses, I documented minimal responses among wild bees, snakes, small mammals, and birds to increases in native plant cover during the first three years after restoration. Conservation practices within agricultural landscapes provide varying levels of benefits for grassland birds and other communities.

CHAPTER 1. GENERAL INTRODUCTION

Loss of natural to agriculture has long been the largest threat to biodiversity, including to birds (Green et al. 2005). Perhaps the most recognizable example of this threat is the effect of agriculture on grassland birds, i.e. species that breed in or depend heavily on large open landscapes dominated by grass and forbs. In North America, 80% of all grasslands have been lost through conversion to agriculture and only 1% of the original tallgrass prairie remains in most states and provinces (Knopf 1994, Samson and Knopf 1994). During the last several decades, agriculture in the Midwestern United States has intensified, increasing production and shifting towards monocultures that support fewer native species (Matson et al. 1997).

Since 1970, grassland birds in North America have experienced a 53% overall decline, the steepest reduction of any bird community (Rosenberg et al. 2019). In addition to habitat loss, fragmentation has been attributed to declines of many area-sensitive grassland species (Ribic et al. 2009). Some species have experienced precipitous decreases including Bobolinks (*Dolichonyx oryzivorus*) and Grasshopper Sparrows (*Ammodramus savannarum*), both of which prefer intact, contiguous patches of grassland (Bollinger and Gavin 1992, Davis et al. 2013, Herkert 2003, Samson 1980, Vickery et al. 1994). Even common, less area-sensitive species such as Dickcissels (*Spiza americana*) and Red-winged Blackbirds (*Agelaius poeniceus*) have declined 14% and 30% respectively (Pardieck et al. 2019).

Many of the “unproductive” landscape features that modern conventional agriculture has reduced – field margins, forest patches, hedgerows, etc. – can be important habitats for avian communities and the insects they rely on (Dennis and Fry 1992, Mineau and McLaughlin 1996). In Iowa, bird abundance is higher in strip-cover habitats (i.e. fencerows, shelterbelts, etc.) than any other agricultural land cover type (Best et al. 1995). Field margins, the strips of land between crops and field boundaries, often offer birds and insects more hospitable habitat than adjacent crop fields (Vickery et al. 2009). Hedgerows within agricultural landscapes have been found to increase the abundance and diversity of insects and birds that contribute crucial ecosystem services (Jobin et al. 2001, Morandin et al. 2014). Woodlots embedded within Midwestern agricultural settings have been found to benefit wintering resident birds (Doherty and Grubb 2000). Removal of these features has decreased the heterogeneity of agricultural landscapes and negatively influenced avian communities.

Conservation opportunities do exist within agricultural landscapes. For example, prairie strips offer a new opportunity to offset the negative impacts brought about by modern agriculture. The STRIPS (Science-based Trials of Rowcrops Integrated with Prairie Strips) project based at Iowa State University, is a long-term, interdisciplinary agricultural research project focused on understanding how strips of native prairie vegetation affect agriculture and the environment. The project team collaborates with landowners and farmers to strategically integrate prairie strips into working farms. Post-installation research focuses on monitoring the effects of strips on soil and water quality, biodiversity, social, and economic outcomes. In Phase I of the STRIPS project, scientists found that converting just 10% of a small watershed to prairie vegetation significantly reduced sediment, nitrogen, and phosphorus export in surface and subsurface runoff (Zhou et al. 2006, Helmers et al. 2012, Hernandez-Santana et al. 2013, Zhou et al. 2014, Schulte et al. 2017). Bird abundance, species richness, and diversity responded positively to the installations, though the experiment may have not reflected the Iowa landscape at large having been surrounded by restored grassland (Schulte et al. 2016).

Another opportunity for conservation is oxbow restoration. Oxbows lake and wetlands are effective at reducing the nitrate-nitrogen export from tile drainage systems (Fink and Mitsch

2007, Harrison et al. 2014), and restoring them where they have been degraded can help Midwestern states meet their nutrient reduction strategies, suggested under the United States Mississippi River/Gulf of Mexico 2008 Action Plan (MRGMWNTF 2008). In addition to their contributions to stream hydrology and water quality, oxbows provide critical habitat for several declining fish species including the federally endangered *Notropis topeka* (Topeka Shiner; Bakevich et al. 2013, Simpson et al. 2019). The U.S. Fish and Wildlife Service and The Nature Conservancy have completed over 100 oxbow restorations in the state (Kenney 2018). Increases in *N. topeka* populations following oxbow restoration in the Raccoon River and Boone River watersheds in central Iowa raised interest in oxbow restoration impacts on other wildlife species, including birds. Little to no research effort had been devoted to examining the effect of oxbow restoration on breeding birds beyond a species inventory in 2015, which documented 54 bird species using four restored oxbows along the Boone River (Harr 2015).

While small features can provide important resources for wildlife in agricultural landscape, larger restorations are also needed to conserve the full suite of biota dependent on natural sytems, including area-sensitive species. Seed availability, cost, and management objectives constrain the number of species used in large prairie restorations. Seed mix design is the largest driver of project costs and outcomes in prairie restoration (Larson et al. 2017, Phillips-Mao et al. 2015). Seed mixes with high grass-to-forb ratio are generally less expensive but produce grass-dominated stands with poor forb coverage (McCain et al. 2010, Valko et al. 2016) and little value as pollinator habitat (Hopwood 2008). Alternatively, seed mixes with high forb-to-grass ratio are expensive and susceptible to weed encroachment and soil erosion (Burke and Grime 1996). Little research has examined the influence of varying levels of seed mix diversity on wildlife impacts of prairie restoration at the scale of typical restorations within agricultural landscapes of the U.S. Midwest.

Dissertation Organization

This dissertation is organized into six chapters. This first chapter introduces the focus of this document. The second chapter examines the effect of prairie strip establishment on bird communities. The third chapter investigates the springtime bird community of common conservation land cover types and evaluates the use of autonomous recording units for springtime bird research. The fourth chapter examines breeding bird associations with oxbow restorations. The fifth investigates multitaxa wildlife response to native prairie reconstruction compared to exotic, cool-season grasses. The final chapter provides a general conclusion to the dissertation.

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CHAPTER 2. BIRD RESPONSE TO INTEGRATION OF PRAIRIE STRIPS WITHIN AGRICULTURAL LANDSCAPES

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Modified from a manuscript to be submitted to *Agriculture, Ecosystems, and Environment*

Abstract

Changing patterns in agricultural production over the last half century have had drastic impacts on native biodiversity. The continued decline of birds, with few exceptions, suggests agricultural intensification as a probable cause. In this study, we evaluated the response of birds to the establishment of prairie strips—a new practice composed of linear strips of reconstructed native perennial vegetation designed to conserve soil, water, and biodiversity—on commercial corn (*Zea mays* L.) and soybean (*Glycine max* [L.] Merr.) farms in Iowa, USA. We conducted bird point counts at 10 commercial farms that had fields (ranging 7.69 – 84.98 ha in extent) with prairie strips (treatment) and without prairie strips (control) during the breeding season (May – July) of 2015 – 2020. We compared bird richness, density, and diversity and examined the effects of local and landscape attributes on the entire bird community, the grassland bird community, and commonly-detected grassland species. On average, we detected 1.89-fold more birds and 1.24-fold more bird species in treatment fields compared to control fields. The density of 17 grassland birds species as a group was 2.60-fold higher in treatment fields (3.65 birds/ha) than control fields (1.40 birds/ha). We reported the largest increases in bird densities for treatment fields during the intial years of establishment, congruent with typical shifts in

management of prairie reconstructions. Species with the strongest response to prairie strip establishment were Red-winged Blackbird (*Agelaius phoeniceus*), Dickcissel (*Spiza americana*), and Common Yellowthroat (*Geothlypis trichas*). The age of the prairie strip was the most important and positively associated predictor of grassland bird, Red-winged Blackbird, and Dickcissel densities. Our results are consistent with studies that documented increase bird use of annual crop fields that include small patches of perennial or semi-natural vegetation. Although we documented increases in bird density, including some species of greatest conservation need such as Dickcissel and Eastern Meadowlark (*Sturnella magna*), prairie strips are unlikely to benefit area-sensitive grassland birds.

Introduction

Changing patterns in agricultural production over the last half century have had drastic impacts on native biodiversity (Sala et al. 2000, Hoekstra et al. 2004, Tsharntke et al. 2005). The continued decline of birds, with few exceptions, suggests agricultural intensification as a probable cause (Donald et al. 2006, Rosenberg et al. 2019). Habitat loss and pesticide use, which are commonly associated with intensification, are two important drivers of bird declines (Stanton et al. 2018). Removal of natural and semi-natural features, such as field margins, hedgerows, and forest patches, has decreased heterogeneity of agricultural landscapes and reduced important habitats for avian communities and the insects they sometimes rely on (Hunter 2005, Pryke and Samways 2014, Larsen-Gray and Loehle 2022).

Land conservation within agricultural landscapes reduces intensification and can provide important ecosystem services (Garibaldi et al. 2011, Garibaldi et al. 2020, Kremen et al. 2018, Smith et al. 2021). Yet, strategies for setting aside land are economically challenging due to high land costs and crop prices. The amount of land enrolled in the USDA Conservation Reserve

Program frequently ebbs and flows with commodity prices (Lark et al. 2015). Targeted conservation of smaller, less productive tracts of land can provide considerable ecological improvements while also reducing the amount of land removed from production (Schulte et al. 2006, Asbjornsen et al. 2014, Brandes et al. 2016).

A prominent hypothesis regarding landscape moderation of biodiversity patterns and processes is the Intermediate Landscape Complexity Hypothesis (ILHC), which states that effectiveness of local management for conserving biodiversity is higher in structurally simple (1-20% non-cropped area) rather than cleared (<1% non-cropped area) or complex (>20% non-cropped area) landscapes (Tschartnke et al. 2005; Fig. 2-1). The ILHC posits low effectiveness of local conservation efforts in cleared or extremely simple landscapes, which contain too few source populations to enable the success of local management. It also posits that structurally complex landscapes with a high proportion of non-crop area maintain high biodiversity and associated functioning (pest control, crop pollination, etc.) and therefore local conservation efforts do not result in a recognizable effect on species diversity. Many aspects of the ILCH remain understudied, and more examination of functional groups like birds will determine its utility beyond study of insect populations, which have been the focal taxa thus far.

In North America, grassland birds have declined more than any other avian community since 1970 (Rosenberg et al. 2019). Protection of large patches of grasslands is needed and would benefit many of the 32 grassland-obligate bird species in North America (Vickery et al. 1999). Grasslands greater than 1,000 ha in size have been found to increase reproductive success of grassland bird species (Herkert et al. 2013). Conversely, reproductive success may not vary significantly among small grassland patch sizes less than 150 ha (Walk et al. 2010).

In the midwestern U.S., as well as many other regions of the world, locations that were historically covered in extensive grassland are now under agricultural production. Given competing demands for and high cost of land in these regions, restoration and protection of smaller areas is more realistic (Chouinard et al. 2008, Tyndall et al. 2013). Land-sharing strategies that address resource concerns within agriculture fields may be a viewed more favorably amongst farmers and landowners than extensive reserve areas (Atwell et al. 2009, Arbuckle 2013, Arbuckle et al. 2015). Many but not all bird species respond strongly to small patches of habitat embedded in crop fields (Best et al. 1995, Berges et all. 2010, Conover et al. 2009).

The STRIPS (Science-based Trials of Rowcrops Integrated with Prairie Strips; www.prairiestrips.org) project based at Iowa State University, is a long-term, interdisciplinary agricultural research project designed to determine how strips of reconstructed diverse, native perennial vegetation (hereafter termed ‘prairie’) benefit agriculture and the environment. The project team collaborates with landowners and farmers to strategically integrate prairie strips into commercial farms. During initial trials at Neal Smith National Wildlife Refuge in central Iowa, bird abundance, species richness, and diversity responded positively to installations of prairie strips (Schulte et al. 2016). Treatments with multiple prairie strips had 1.53 – 2.88 times more birds, 1.53 – 2.13 times more bird species, and 1.40 – 1.98 times greater diversity compared to fields with 0% prairie. These patterns were driven by several species of grassland and shrubland nesters — specifically Eastern Kingbird (*Tyrannus tyrannus*), American Robin (*Turdus migratorius*), and Common Yellowthroat (*Geothlypis trichas*) — which were more abundant in treatment fields. Iowa species of greatest conservation need (SGCN; IDNR 2015) found in treatment fields included Grasshopper Sparrow (*Ammodramus savannarum*), Eastern

Meadowlark (*Sturnella magna*), Dickcissel (*Spiza americana*), and Field Sparrow (*Spizella pusilla*). Grasshopper Sparrows and Field Sparrows were found in greatest abundance during the initial establishment year while Dickcissels were more abundant in subsequent years. The community of birds using prairie strips changed in post-establishment years, with higher abundance, richness, and diversity in post-establishment years. Aside from SGCN species, Common Yellowthroat and American Goldfinch (*Spinus tristis*) were also more abundant post-establishment. Treatments with prairie strips have also been found to support 2.6 times more insect species and 3.5 times higher abundance of insect pollinators compared to all-crop controls (Schulte et al. 2017).

While results of this study suggest prairie strips as a cost-effective way to provide habitat for birds and other species within agricultural landscapes, the landscape matrix surrounding the experiment was not typical given its location within a National Wildlife Refuge: experimental catchments were surrounded by restored grassland rather than cropland (Schulte et al. 2016). Research conducted in landscapes more typical of Corn Belt agricultural systems would provide additional insight into the conservation value of prairie strips for grassland birds. The increasing number of farms across the Midwest with prairie strips offered an opportunity to evaluate how prairie strips function as wildlife habitat in landscapes dominated by commercial row crop production, and also how variation in the composition surrounding prairie strip affects their value as wildlife habitat according to the ILCH (Fig. 2-1). Farmer and farmland owner adoption of strips as a conservation practice increased since its inclusion in the U.S. Conservation Reserve Program in 2019 (USDA FSA 2019, 2022) allow this expanded investigation. Furthermore, an increasing number of farmers indicate they are willing to install strips (Arbuckle 2019, 2020), suggesting that additional study could be helpful in informing prairie strip field designs to potentially further benefit wildlife.

In this study, we extended the investigations of Schulte et al. (2016) to commercial farms across Iowa and also to test the ILCH. We examined the response of bird density and diversity to prairie strip installations and compared the effects of local and landscape attributes on the avian community. Our objectives were to: (1) compare overall bird and grassland bird richness, density, and diversity between crop fields with (treatment) and without (control) prairie strips, (2) determine which bird species respond most strongly to the addition of prairie strips, and (3) evaluate the effect of local and landscape vegetation attributes on grassland bird density in fields with prairie strips. We hypothesized: (a) treatment fields would have higher densities and diversity of grassland birds than control fields, (b) edge-adapted grassland species (e.g., Dickcissel, Sousa et al. 2022; Red-winged Blackbirds, *Agelaius phoeniceus*, Yasukawa and Searcy 2020) would respond more strongly than area-sensitive species (e.g., Grasshopper Sparrow, Vickery 2020) to prairie strip establishment, and (c) overall grassland bird community density would be highest in landscapes of intermediate complexity, as predicted by the ILCH (Tschartnke et al. 2005).

Methods

Study Area and Site Selection

We conducted this study on 10 commercial farms located in the state of Iowa (Fig. 2-2, Table 2-1). These farms comprised all of the known farms managed for commercial row-crop production of corn (*Zea mays* L.) and soybeans (*Glycine max* [L.] Merr.), as is common in the region, and with installations of prairie strips within 150-km distance of Ames, Iowa at the start of the study in 2015.

Each of the 10 farms included a paired comparison of control and treatment fields. Four farms included randomized assigment of control and treatment to fields (hereafter, ‘randomized farms); control and treatment fields were directly adjacent to each other at these farms. At the remaining six sites, the farmer and/or farmland owner chose to implement prairie strips on a specific field to address a resource constraint (specifically, soil erosion) and thus were unwilling to randomize treatment and control fields; the distance between control and treatment fields ranged 0 – 4.7 km at these non-randomized farms. In all cases, crop type and management were consistant among control and treatment fields at each site; topography and soil types were similar. Control fields ranged 14.1– 93.1 ha in size, averaging 36.1 ha with a standard deviation of 27.0 ha. Treatment fields ranged 7.7 – 85.0 ha in size, averaging 23.4 ha with a standard deviation of 22.9 ha. At each site, the amount of area sampled was constant between control and treatment fields for data comparatibility (Table 2-1). Land cover within 500 m of the center point of each field was similar but for the amount of prairie vegetation; treatment fields had, on average, 14.5% more prairie than control fields (Table A-1).

The climate of the study region is humid continental, with average statewide monthly temperature during the period of observation (May – July) of 26.4 degrees Celsius, and average monthly precipitation during this period of 12 cm (NOAA NWS, 2022). Landscapes are undulating with a maximum and minimum elevations of 128 – 602 m above sea level (USGS 2022). The region was dominated by commodity corn and soybean production (USDA 2017), with annual cropland comprising 60.9% of the study landscapes on average; the remainder on average was composed of cool-season grasses (17.3%), prairie (9.8%), woody vegetation (8.1%), developed areas (3.7%), and water (1.7%) (Table A-1). Land cover composition surrounding control and treatment fields was similar (Table A-1).

Bird Point Counts

In each control and treatment field, we placed between three and six point count stations spaced 200 meters apart in a staggered grid from a randomly selected starting point (Buckland et al. 2001). The number of point count stations was determined by field size, with smaller fields allocated fewer point count stations. We conducted BPCs three times per field per year between May 15 and July 15, coinciding with bird-breeding season in Iowa, for six years (2015 – 2020). Trained observers navigated to point count stations with a handheld GPS unit and began surveys as early as 15 min before sunrise and as late as one hour after sunrise. This period coincides with peak vocal activity in most songbirds (Robbins 1981, Robbins et al. 1989).

After arriving at a station, the observer remained stationary and silent for 2 min to allow birds to resume natural behavior. The observer then identified species, sex, and age (juvenile or adult) to each bird seen or heard during a 5-min survey period. Using a laser rangefinder, the observer also estimated the perpendicular distance to each individual bird detected. Exact distance estimations were not made for birds greater than 200 m from the observer. Surveys were not conducted during periods of rainfall or wind speeds exceeding 16 kmh (Manuwal and Carey 1991, Mikol 1980). Air temperature, wind speed, and percent cloud cover were recorded before and after surveys.

Land Use/Land Cover

We calculated local and landscape spatial covariates of study sites using interpretation and digitization of aerial imagery in ArcGIS (version 10.8.2, ESRI, Redlands, CA). Observers interepted digitized land cover from high-resolution digital aerial images provided by the U.S. Department of Agriculture National Agriculture Imagery Program (USDA NAIP 2020). Land cover attributes at each study site were classified into one of six land cover types — crop, developed, cool-season grass, prairie, woody, and water—and summarized within a 1-km radius of each survey grid (Table A-1). Uncertainties during digitalization were corrected by in-person, field-based verification of land cover. We created 500-m (local) and 1-km (landscape) buffers around each survey grid for estimating land cover metrics to account for variation in the scale of response among grassland bird species: some respond stronger to variation in local habitat characteristics while others respond to variation in landscape characteristics (Boscolo and Metzger 2009, Thompson et al. 2014, Shahan et al. 2017). Using the digitized landscapes, we calculated percentage cover of the six land cover types at the local and landscape scales, and summed the number of unique grass and prairie patches visible in satellite imagery. Grass cover consisted of lower diversity areas dominated by cool-season non-native grass species, usually smooth brome (*Bromus inermis* Leyss.), and typically was located in grassed waterways and on terraces in crop fields, filter strips around surface water, and roadside ditches. Prairie cover consisted of any diverse, reconstructed patches of grassland vegetation dominated by native grass and forb species (see English 2021 for a full description of plant cover). We used the number of grassy patches within 1 km of each survey grid as a metric for testing the ILCH. We used a minimum patch size of 10 m^2. Because of our focus on grassland birds, we considered increases in grassy patches to be increases in landscape complexity.

Statistical Analysis

Prior to all statistical analysis, we vetted the raw data for recording or transcription errors. Any updated erroneous or ambiguous data were corrected based on field data sheets. We then performed statistical analysis on cleaned data. We calculated species richness, Shannon's Index, and Simpson's Index as measures of diversity. Shannon's Index is a measurement of community heterogeneity and gives more weight to rare species (Shannon and Weaver 1949). Simpson's

Index incorporates proportional abundance or evenness of the community and gives more weight to common species (Simpson 1949). Bird species included in the grassland community were known obligate grassland users or found to prefer grassland habitat types according to Peterjohn and Sauer (1993). For density estimation, we used only territorial male detections detected by sound (Buckland et al. 2001, Newson et al. 2008). We sorted distances into 20-m bins from 0 to 100 m to remove potential bias of estimating distances (Buckland et al. 2001). We removed all detections beyond 100 m from species diversity and density analyses due to unreliable detection beyond that distance. Because 91% of Brown-headed Cowbirds (*Molothrus ater*) detections were flyovers, we compared the number of cowbirds detected per survey in control and treatment fields.

We first analyzed data from all sites to examine trends in bird communities. To evaluate a potential cause-effect relationship of prairie strips and birds, we used data from just the four randomized sites for a separate analysis. We used two-factor analysis of variance (ANOVA) with type III sum of squares to test for a difference in grassland bird species richness, density, and diversity between control and treatment fields and across years. We conducted this analysis using data from just the randomized sites and then for all sites. Following the paired-sample design character of our study, we used 'site' as a blocking variable to control for differences among farms. We followed with a Tukey HSD to examine pairwise differences between significant independent variables.

At all sites, we estimated yearly bird densities for both control and treatment fields using the package "Distance" (Miller et al. 2019) within R 4.1.2 (R Development Core Team 2021). We calculated naïve densities of the entire bird community and of grassland species that did not have 80 total detections. For the entire grassland community and for species with greater than 80 total detections, we used count modeling with a two-step bootstrap to account for imperfect detection and investigated the relationship between density and spatial covariates (Buckland et al. 2009, Rodriguez-Caro et al. 2017). Before modeling both detection functions and spatial models, we standardized all covariates and tested for correlations among covariates using Variance Inflation Factor (VIF) to avoid multicollinearity included in the same model. We removed highly correlated combinations ($|R| > 0.7$). First, we fit a detection function to the distance data to obtain a detection probability (Buckland et al. 2001). We evaluated the fit of the hazard rate, half-normal, and uniform key functions with and without cosine adjustments. We evaluated time of day, temperature, wind speed, and cloud cover as covariates to model heterogeneity in detection probabilities. We used an Akaike Information Criterion (AIC hereafter) framework and goodness-of-fit tests to determine the most appropriate detection probability model (Burnham et al. 2004).

After correcting counts for imperfect detection, we investigated patterns in grassland bird density using generalized linear mixed models with spatial covariates associated with the sampling area (Miller et al. 2013). We included only prairie strips sites during this stage. All models were tied to an *a priori* biological hypothesis aimed at explaining potential predictors of bird density (Table 2-2). After generating corrected counts, we used an all-subsets approach to construct a global model for predicting grassland community and species-level densities. The global model of each set was an additive model consisting of each covariate in the set. We tested the global model for zero inflation and overdispersion. We included site as a random effect in each model. We considered models within $\Delta AIC_c = 2.0$ to be competitive.

Results

Diversity and Density on Farms with Randomized Treatments

Across the four sites with randomized treatments, we made a total of 5,317 detections of 69 species over the six years of study (2015-2020). Of these species, we considered 16 to be grassland-obligate species (Table 2-3). The total number of birds observed per survey ranged from 1 to 52 individuals, and the total number of species ranged from 1 to 14 per survey.

Measures of diversity did not differ with treatment or year. We detected a similar number of species per survey in treatment fields (9.45, 95% confidence interval (CI): 8.30, 10.60 species) and control fields (9.04, 95% CI: 8.02, 10.06; treatment: $F=0.32$, $p=0.57$; year: $F=2.35$, $p=0.07$). Shannon's diversity index was similar in treatment fields (3.33) and control fields (3.28), with no effect of treatment ($F=1.06$, $p=0.31$) or year ($F=0.85$, $p=0.53$). We found the same pattern with Simpson's diversity index (treatment fields = 0.960; control fields = 0.958; treatment: $F=1.40$, $p=0.25$; year: $F=1.10$, $p=0.38$).

Bird density differed among treatment and control, with an average 1.52-fold increase in bird densities with prairie strips ($F=31.406$, $p<0.001$). We detected 4.21 birds/ha (95% CI: 3.19, 5.33 birds/ha) in treatment fields and 2.77 birds/ha (95% CI: 1.84, 3.70) in control fields. We further found an effect of treatment ($F=31.41$, $p<0.001$) and year ($F=8.16$, $p<0.001$) on the density of grassland birds with a 1.64-fold increase with prairie strips. Grassland bird density in crop fields with prairie strips trended upwards in years following the initial establishment of prairie strips but prairie strip age did not have a significant effect ($F=2.645$, $p=0.06$).

Diversity and Density on All Farms

Across all 10 farms surveyed during the six years of study (2015-2020), we made a total of 14,710 detections of 81 bird species. Of these species, we considered 17 to be grassland-obligate species (Table 2-3). The five most commonly detected species – Red-winged Blackbird, Dickcissel, Common Yellowthroat, Eastern Meadowlark, and Western Meadowlark – comprised 96.4% of all grassland bird detections and 97.3% of the difference in density between treatment and control fields. The total number of birds observed per survey ranged from 1 to 62 individuals, and the total number of species observed ranged from 1 to 14 species per survey.

Among species with greater than 80 detections, the naïve densities of Grasshopper Sparrows (control: 0.01 birds/ha; treatment: 0.05 birds/ha) and Sedge Wrens (control: 0.002 birds/ha; treatment: 0.04 birds/ha) were higher in treatment fields. Densities of Horned Lark (control: 0.02 birds/ha; treatment: 0.01 birds/ha) and Vesper Sparrow (control: 0.04 birds/ha; treatment: 0.02 birds/ha) were higher in control fields. Among other grassland species, Ring-necked Pheasants (*Phasianus colchicus*) and Upland Sandpipers (*Bartramia longicauda*) were most often detected at distances beyond 100 m.

Detections included 17 Iowa SGCN species (Table 2-3), of which, the most common were Dickcissel, Eastern Meadowlark, and Field Sparrow. Among non-grassland birds, the most common were Brown-headed Cowbird, Killdeer (*Charadrius vociferus*), and Song Sparrow (*Spiza melodia*). We detected 2.63 (95% CI: 2.01, 3.24) Brown-headed Cowbirds per survey on average in treatment fields and 2.97 (95% CI: 2.49, 3.45) cowbirds per survey in control fields.

Species richness trended higher in fields with prairie strips when considering all sites: we detected 1.24 times more species per survey in treatment fields (6.42, 95% CI: 6.02, 6.82 species) than control (5.16, 95% CI: 4.78, 5.54 species). The association between species richness and year was significant at the $\alpha = 0.01$ level ($F=5.74$, $p<0.001$) but the association with treatment was only significant at the $\alpha = 0.1$ level ($F=2.98$, $p=0.09$). Shannon's and Simpson's diversity indicies did not differ by treatment or year. Shannon's diversity for control sites was 3.30 and for treatment sites was 3.33 (treatment: $F=1.56$, $p=0.21$; year: $F=1.65$, $p=0.15$).

Simpson's diversity for control sites was 0.959 and treatment sites was 0.963 (treatment: F=1.82, p=0.18; year: F=1.70, p=0.15).

The trend toward a greater abundance of birds with prairie strips was consistent across all sites: on average, we found a 1.88-fold increase with prairie strips with 4.49 birds/ha (95% CI: 3.82, 5.16 birds/ha) in treatment fields and 2.38 birds/ha (95% CI: 1.97, 2.79 birds/ha) in control fields (Fig. 2-3, Table 2-4). We also found a strong response to the presence of prairie strips among grassland birds as a subset of the whole bird community, with a 2.61-fold higher density in treatment fields (F=145.64, p<0.001). Treatment fields averaged 3.65 grassland birds/ha (95% CI: 3.2, 4.1 birds/ha) compared to compared to 1.40 birds/ha (95% CI: 1.15, 1.65 birds/ha) in control fields (Fig. 2-3, Table 2-4).

The effect of year was also a significant (F=12.84, p<0.001). Grassland birds in crop fields with prairie strips trended upwards in the years following the initial establishment of prairie strips, with a notable increase from year 3 to year 4 (Fig. 2-4). There was a signficiant association between yearly changes in density and prairie strip establishment year (F=6.93, p<0.5). Pairwise comparions among establishment years revealed statistically-signficiant increases in grassland bird density between year 1 (1.64) and year 2 (3.83; p<0.05) and between year 3 (3.77) and year 4 (5.29; p<0.05).

Effect of Local and Landscape Attributes on Grassland Bird Detectability and Density for Fields with Prairie Strips

The hazard/cosine function, without additional covariates, provided the best fit for detection for grassland birds as a community on fields with prairie strips. Prairie strip age was the most competitive model for predicting grassland bird community density (Fig. 2-5A), and had a statistically signficiant positively association on density (β=0.656, 95% CI: 0.246, 1.066 birds/ha). The global model was also competitive (Table 2-5). The relationships among grassland bird density and local prairie cover (β=0.183, 95% CI: -1.590, 1.831 birds/ha), local grass cover (β=-0.821, 95% CI: -2.700, 1.058 birds/ha), local crop cover (β=-0.750, 95% CI: -2.677, 1.178 birds/ha), number of local prairie patches (β=-0.256, 95% CI: -1.499, 0.986 birds/ha), and the number of landscape grassy patches (β=0.243, 95% CI: -0.608, 1.094 birds/ha) were not significant and thus our assessment of the ILCH was inconclusive (Table 2-6).

We had an adequate number of detections to model detection proability for Red-winged Blackbird, Dickcissel, Common Yellowthroat, Eastern Meadowlark, and Western Meadowlark in each land cover type (Fig. 2-3), and include spatial covarariates for Red-winged Blackbirds, Dickcissels, and Common Yellowthroats. Each of these grassland species had higher densities in fields with prairie strips than in control fields.

Red-winged blackbird detection probability was best represented through the half/cosine detection function with temperature as a covariate. For predicting Red-winged Blackbird density, prairie strip age was the most competitive model and the global model was also competitive (Table 2-5). Prairie strip age was positively related to density (Fig. 2-5B) and the standardized regression coefficient for prairie strip age was statistically signficiant (Table 2-7; β=0.445, 95% CI: 0.196, 0.693).

Dickcissel detection probability was best represented with a hazard/cosine function and with temperature as a covariate. For predicting Dickcissel density, prairie strip age was the most competitive model and the global model was also competitive (Table 2-5). Prairie strip age was positively related to density (Fig. 2-5C) and the standardized regression coefficient for prairie strip age was statistically significant (Table 2-7; β=0.295, 95% CI: 0.100, 0.490). Local crop cover was also competitive but the relationship was not statistically significant (β=-0.080, 95% CI: -0.332, 0.173; Fig. 2-5D).

Common Yellowthroat detection probability was best represented through a half/cosine function with cloud cover as a covariate. For predicting Common Yellowthroat density, local crop cover was the only competitive model (Table 2-5; Fig. 5E), but the relationship was not statistically signficant (Table 2-7; β=-0.139, 95% CI: -0.295, 0.016).

Discussion

Our study investigated the bird response to the establishment of prairie strips on working farms across Iowa. Our objectives were to inform further refinement of agricultural conservation policies generally, and specifically related to the prairie strips practice by: (a) comparing overall bird and grassland bird density and diversity between fields with and without prairie strips, (b) determining which bird species respond most strongly to the addition of prairie strips , and (3) evaluating the effect of local and landscape vegetation attributes on grassland bird density in fields with prairie strips.

We found a strong positive response in grassland bird density to the establishment of prairie strips in corn and soybean fields (Fig. 2-3). We also documented a strong trend in increased density of grassland birds in post-establishment years with significant increases between years 1 and 2 and years 3 and 4 (Fig. 2-4). Prairie strip age was an important predictor of grassland bird density (Table 2-5).

Among grassland bird species, we found that Red-winged Blackbirds, Dickcissels, and Common Yellowthroats were most responsive to prairie strip establishment. Local crop cover was important for predicting Dickcissel and Common Yellowthroat densities. With the exception of Horned Larks and Vesper Sparrows, which were more common in control fields, all other grassland species trended toward having higher densities in fields with prairie strips. Horned

Larks and Vesper Sparrows prefer short, sparse vegetation such as crop reside and crop field edges (Beason 2020, Jones and Cornely 2020).

Our results were consistent in many ways with Schulte et al. (2016) and others who documented increased bird use of crop fields with perennial or semi-natural vegetation. In addititon to supporting increases in bird species richness and density, prairie strips also provide nesting habitat for grassland birds. Stephenson (2022) found higher nest densities and nest success in prairie strips compared to other available cover types on corn and soybean farms in Iowa.

Our model selection results suggested that grassland bird density on fields with prairie strips was most strongly influenced by prairie strip age and densities increased through time (Fig. 2-4). Grassland bird community density and assemblage are known to vary with development of vegetation. Schulte et al. (2016) documented stronger response among some grassland species, including Common Yellowthroat, Field Sparrow, and Dickcissel, during post-establishment years of prairie strips. In North Dakota, Fritcher et al (2004) found higher densities of Grasshopper Sparrow, Bobolink, and Dickcissel in less recently disturbed grasslands. Van Dyke et al. (2007) found delayed increases in density of Northern Bobwhite and Ring-necked Pheasant in recently-burned prairies.

We did not find support for the Intermediate Landscape Complexity Hypothesis. The metric that we used as an index of landscape complexity, the number of grassy patches within 1 km, was not competitive in any model set (Table 2-5). Though our study included sites in landscapes of varying complexity, the Iowa landscape has much less perennial vegetation than previous studies that have tested the ILCH (Tschartnke et al. 2005).

The results from our comparisons at randomized sites suggest a causal effect of prairie strip establishment on grassland bird density but not other community measures. Our results from comparisons at all study sites are correlation-based and provide limited inference into causal effects of prairie strips on bird communities. Larger sample sizes would provide more robust conclusions on the true effect of prairie strip establishment. We found prairie strip age to be an important predictor of both grassland community density and species density (Table 2-5). Though other models were competitive, confidence intervals of effect sizes for many overlapped 0.

Loss of natural habitats to agriculture has long been the largest threat to biodiversity, including to birds (Green et al. 2005). Since 1970, grassland birds in North America have experienced a 53% overall decline, the steepest reduction of any bird community (Rosenberg et al. 2019). Eighty percent of all grasslands in North America have been lost through conversion to agriculture and only 1% of the original tallgrass prairie remains in most states and provinces (Knopf 1994, Samson and Knopf 1994). During the last several decades, agriculture in the Midwestern United States has intensified, increasing production and shifting towards monocultures that support fewer natural species (Matson et al. 1997).

Prairie strips offer a new opportunity for offsetting the negative impacts brought about by modern agriculture. As a part of the Clean Lakes, Estuaries, and Rivers (CLEAR) Initiative in the 2018 farm bill, croplands converted to strips are eligible for enrollment in the Conservation Research Program (CRP) offered by the USDA Farm Service Agency. The new practice, CP43, allows flexibility in the location of prairie establishments on farms including the interior of fields, field perimeters, terrace channels, areas adjacent to waterways, and center-pivot irrigation corners (USDA 2019). As of September 30, 2022, there were at least 5,147 ha of CP43 in 14 states (USDA 2022).

Our study suggested that prairie strips expand bird use of agricultural landscapes, and appear to provide quality habitat, especially compared to farmland conservation features more typically used in the region, such as grass filter strips and terraces (Stephenson 2022). Over six years, we documented a positive effect of prairie strips on grassland bird density and positive relationship between prairie strips and species richness. However, prairie strips only seem to provide habitat for a subset of grassland birds; area-sensitive grassland birds, such as Bobolink (*Dolichonyx oryzivorus*) or Henslow's Sparrow (*Centronyx henslowii*), are uncommon in or absent from the farm fields we surveyed (Table 2-3). Despite positive responses in grassland bird density and species richness, larger grassland patches are likely needed to reverse grassland bird declines, especially for area-sensitive species.

Acknowledgements

This research was funded by the Leopold Center for Sustainable Agriculture (E2015-10), USDA Farm Service Agency (AG-3151-P-14-0065 and AG-3151-P-17-0108), and the U.S. Federal McIntire-Stennis program (IOW5354, IOW3617). Thank you to the farmers and land owners who allowed us access to their properties to conduct this research.

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Figures and Tables

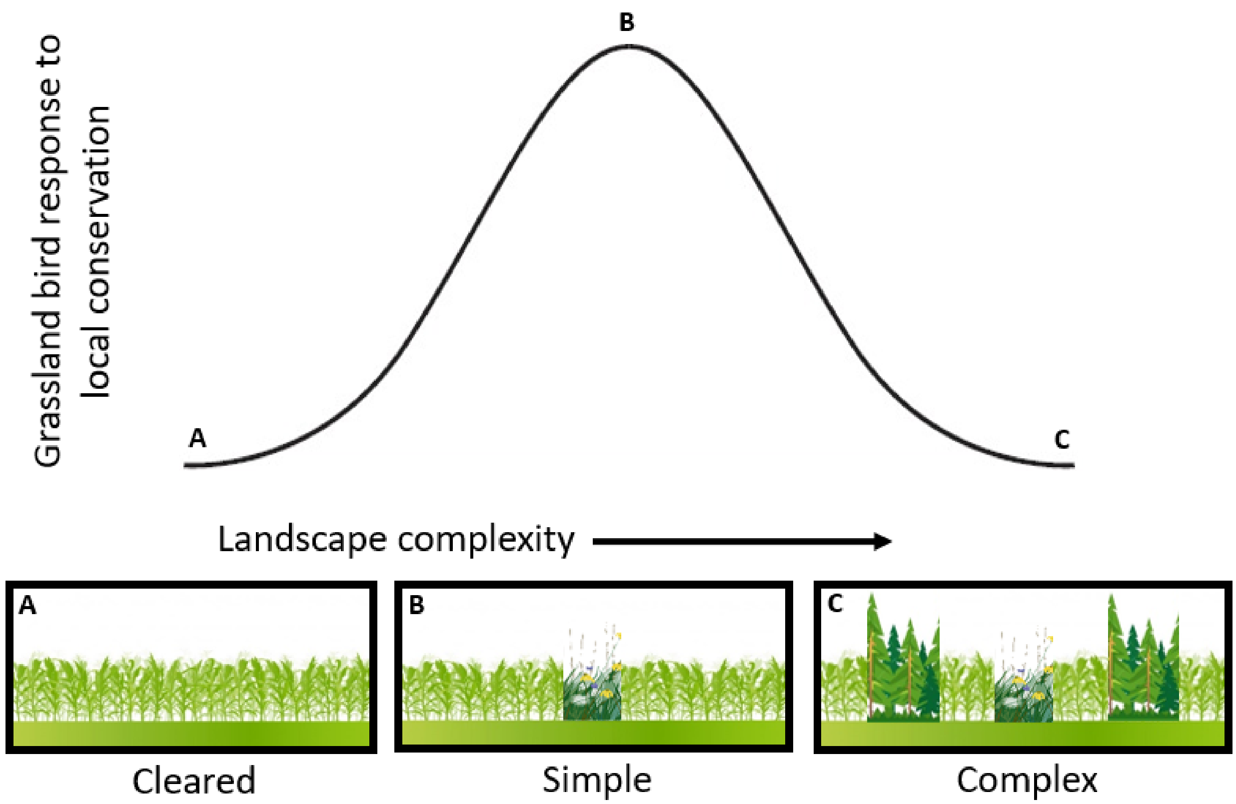


Figure 2-1. Hypothetical response of grassland birds to local conservation practices across a landscape complexity gradient. Modified from Tscharntke et al. (2012).

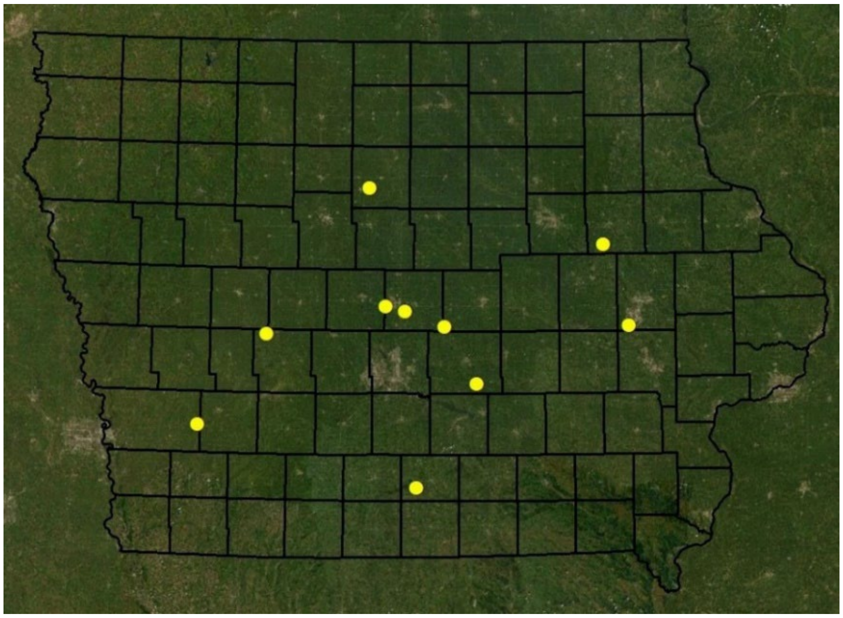


Figure 2-2. Location of commercial farms where bird point count surveys were conducted in Iowa, USA during 2015-2020. Aerial image of field with prairie strips in Wright County, Iowa, USA.

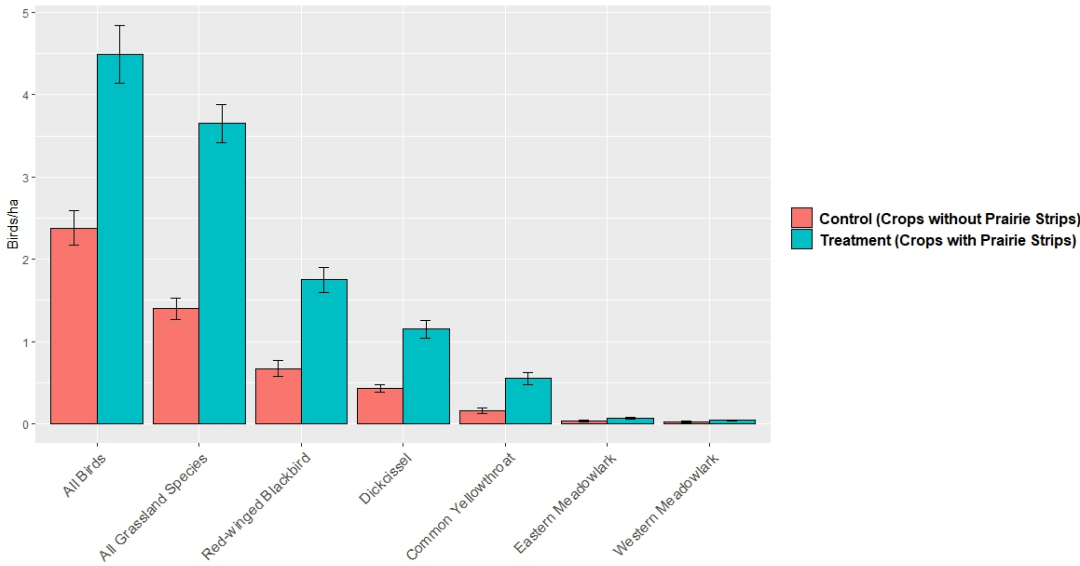


Figure 2-3. Mean densities of all bird species, all grassland bird species, and five commonly detected grassland species in commercial corn and soybean crop fields, with and without prairie strips, in Iowa, USA, 2015-2020. Error bars are standard error.

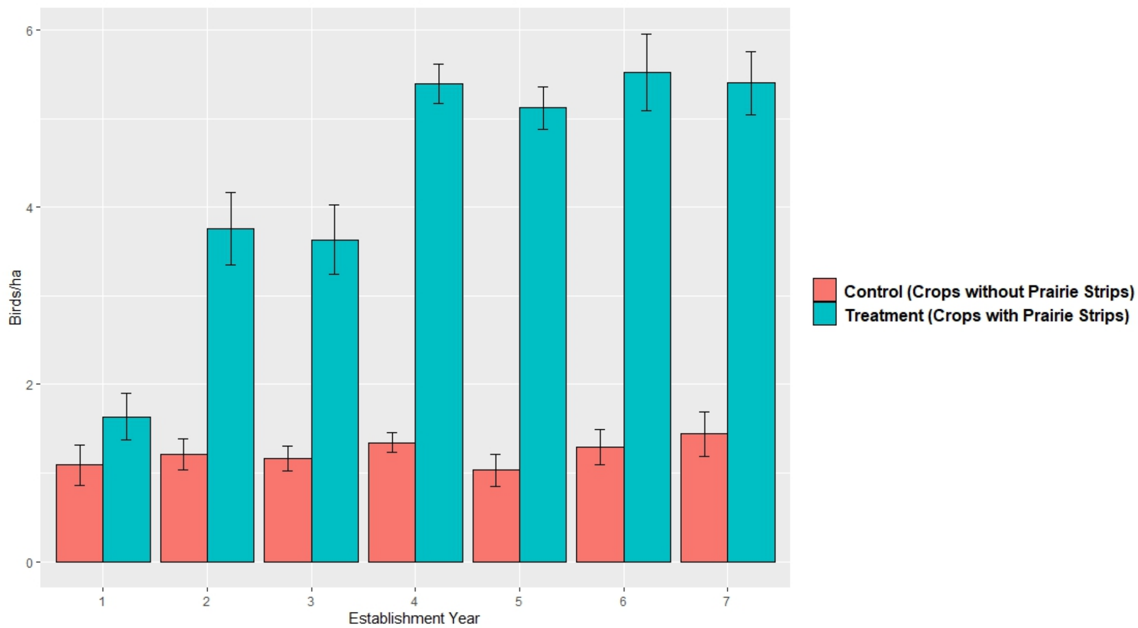
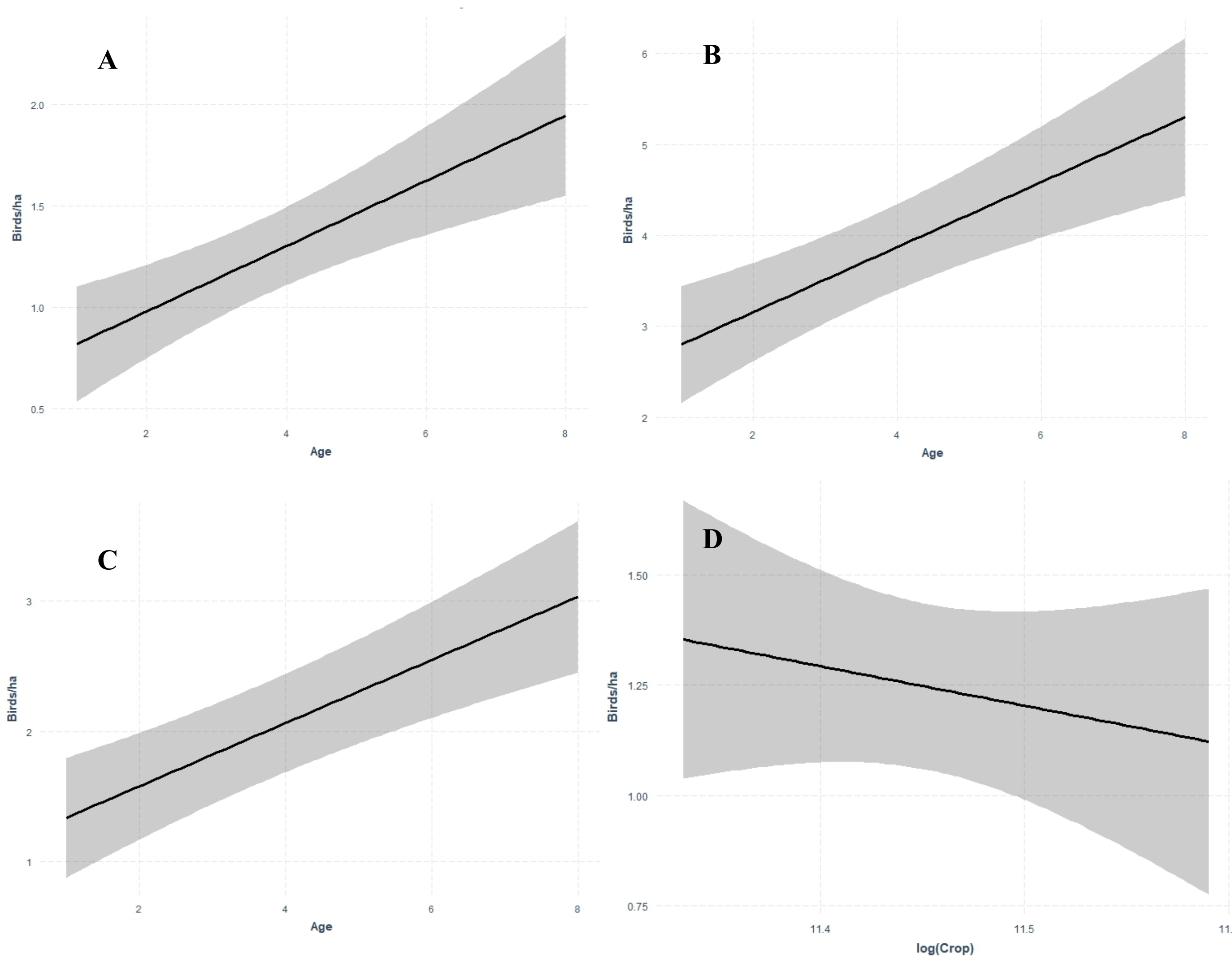


Figure 2-4. Mean densities of grassland birds in commercial corn and soybean crop fields, with and without prairie strips, in Iowa, USA, 2015-2020. Error bars are standard error.



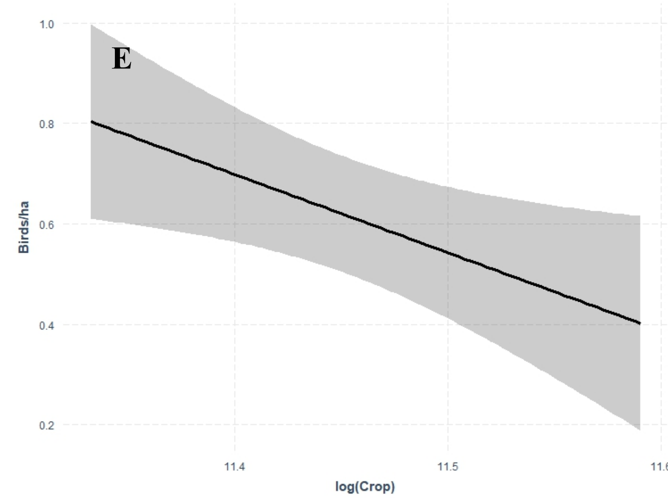


Figure 2-5. Linear plot illustrating relationships between grassland bird density and prairie strip age (A), Red-winged Blackbird density and prairie strip age (B), Dickcissel density and prairie strip age (C), Dickcissel density and local crop cover (D), and Common Yellowthroat density and local crop cover (E) in fields with prairie strips, May – July, 2015-2020. Plots are based on predictive models ranked by AIC_c. Shaded area around regression line represents 95% confidence intervals of the estimate.

Table 2-1. Commercial farms where bird point count surveys were conducted May – July, 2015 – 2020, in Iowa, USA. SD = standard deviation. Field area sampled was kept constant between control and treatment fields at each site. Detailed land cover characteristics included in Table A-1.

Site Name	County	Control Field Size (ha)	Treatment Field Size (ha)	Field Area Sampled (ha)	Area Restored to Prairie (ha)	Year Restored
ARM	Pottawattamie	28.16	8.09	18.84	0.77	2014
EIA*	Linn	24.78	20.23	25.12	2.27	2015
GUT	Story	26.55	25.50	18.84	2.14	2014
KAL	Jasper	14.11	11.33	25.12	1.29	2008
MCN*	Lucas	30.67	29.14	31.4	2.02	2014
RHO*	Marshall	23.12	12.95	18.84	1.05	2015
SMI	Wright	93.14	7.69	25.12	1.62	2015
SLO	Buchanan	78.93	84.98	25.12	1.82	2012
WHI	Guthrie	23.18	22.66	31.4	6.31	2015
WOR*	Story	17.89	11.74	18.84	0.85	2015
Mean	--	36.05	23.43	23.86	2.01	--
SD	--	26.98	22.88	4.69	1.60	--

*Randomized site

Table 2-2. Candidate set of variables used to predict the density of grassland birds in commerical row-crop fields (corn [*Zea mays* L.], and soybean [*Glycine max* (L.) Merr.]) with prairie strips. All land cover data were derived from U.S. Department of Agriculture National Agriculture Imagery Program (USDA NAIP 2020) and ground-truthed by field personnel.

Variable	**Definition and Data Source**
Landscape Grassy Patches	A landscape variable that is the summed number of grass and prairie patches within 1 km of survey grid.
Local Crop Cover	A local variable representing the proportion of land in corn or soybean crops within 500m of survey grid.
Local Grass Cover	A local variable representing the proportion of land covered in low diversity grass within 500m of survey grid.
Local Prairie Cover	A local variable representing the proportion of land covered in reconstructed prairie vegetation within 500m of survey grid.
Local Prairie Patches	A local variable representing the number of patches in reconstructed prairie vegetion within 500m of survey grid.
Local Water Cover	A local variable representing the proportion of land covered in water within 500m of survey grid.
Prairie Strip Age	A temporal variable that represents the number of years since prairie strips were established on treatment fields, as determined from the management records of cooperating farmers/farmland owners.

Table 2-3. Eighty-one species detected during bird point counts using unlimited distance surveys, including flyovers, in commercial row-crop fields (corn [*Zea mays* L.], and soybean [*Glycine max* (L.) Merr.]), without and with prairie strips, in Iowa, USA, 2015-2020.

Species	**Control (Crops without Prairie Strips)**	**Treatment (Crops with Prairie Strips)**
American Crow (*Corvus brachyrhynchos*)	64	48
American Goldfinch (*Spinus tristis*)	161	150
American Robin (*Turdus migratorius*)	296	199
Bald Eagle (*Haliaeetus leucocephalus*)*	-	3
Baltimore Oriole (*Icterus galbula*)	30	10
Barn Swallow (*Hirundo rustica*)	212	112
Black-capped Chickadee (*Poecile atricapillus*)	11	10
Belted Kingfisher (*Megaceryle alcyon*)	-	1
Bell's Vireo (*Vireo bellii*)*	1	-
Brown-headed Cowbird (*Molothrus ater*)	503	371
Blue Jay (*Cyanocitta cristata*)	39	33
Bobolink (*Dolichonyx oryzivorous*)*†	36	52
Brown Thrasher (*Toxostoma rufum*)	81	54
Blue-winged Teal (*Spatula discors*)	-	2
Canada Goose (*Branta canadensis*)	58	1
Carolina Wren (*Thryothorus ludovicianus*)	-	1
Cedar Waxwing (*Bombycilla cedrorum*)	37	3
Chipping Sparrow (*Spizella passerine*)	45	28
Chimney Swift (*Chaetura pelagica*)*	6	2
Cliff Swallow (*Petrochelidon pyrrhonota*)	2	5
Common Grackle (*Quiscalus quiscula*)	84	73
Cooper's Hawk (*Accipiter cooperii*)	1	1
Common Nighthawk (*Choreiles minor*)*	1	1
Common Yellowthroat (*Geothlypis trichas*)†	282	580
Dickcissel (*Spiza americana*)*†	734	1425
Downy Woodpecker (*Dryobates pubescens*)	2	1
Eastern Bluebird (*Sialia sialis*)	11	17
Eastern Kingbird (*Tyrannus tyrannus*)	26	22
Eastern Meadowlark (*Sturnella magna*)*†	261	355
Eastern Phoebe (*Sayornis phoebe*)	1	2
Eastern Towhee (*Pipilo erythropthalmus*)	-	2
Eastern Wood-peewee (*Contopus virens*)	10	2
Eurasian Collared-dove (*Streptopelia decaocto*)	4	4
European Starling (*Sturnus vulgaris*)	42	20
Field Sparrow (*Spizella pusilla*)*†	89	61
Great Blue Heron (*Ardea herodias*)	17	7
Great Egret (*Ardea alba*)	-	1
Great Crested Flycatcher (*Myiarchus crinitus*)	0	1
Gray Catbird (*Dumetella carolinensis*)	40	13
Grasshopper Sparrow (*Ammodramus savannarum*)*†	53	103
House Finch (*Haemorhous mexicanus*)	3	1
Horned Lark (*Eremophila alpestris*)†	99	38
House Sparrow (*Passer domesticus*)	72	17
House Wren (*Troglodytes aedon*)	86	52
Indigo Bunting (*Passerina cyanea*)	69	53

Table 2-3. Continued.

Species	**Control (Crops without Prairie Strips)**	**Treatment (Crops with Prairie Strips)**
Killdeer (*Charadrius vociferus*)	340	304
Lark Sparrow (*Chondestes grammacus*)†	2	2
Least Flycatcher (*Empidonax minimus*)*	1	-
Mallard (*Anas platyrhynchos*)	11	7
Mourning Dove (*Zenaida macroura*)	108	93
Northern Bobwhite (*Colinus virgninianus*)*†	14	25
Northern Cardinal (*Cardinalis cardinalis*)	134	56
Northern Flicker (*Colaptes auratus*)	2	4
Northern Harrier (*Circus hudsonius*)*	-	1
Northern Rough-winged Swallow (*Stelgidopteryx serripennis*)	4	3
Rose-breasted Grosbeak (*Pheucticus ludovicianus*)	7	2
Ring-billed Gull (*Larus delawarensis*)	1	-
Red-bellied Woodpecker (*Melanerpes carolinus*)	29	11
Red-headed Woodpecker (*Melanerpes erythrocephalus*)*	10	10
Ring-necked Pheasant (*Phasianus colchicus*)†	243	215
Rock Pigeon (*Columba livia*)	17	6
Red-tailed Hawk (*Buteo jamaicensus*)	4	2
Ruby-throated Hummingbird (*Archilochus colubris*)	3	3
Red-winged Blackbird (*Agelaius phoeniceus*)†	1527	2378
Savannah Sparrow (*Passerculus sandwichensis*)†	12	-
Sedge Wren (*Cistothorus stellaris*)*†	12	91
Song Sparrow (*Melospiza melodia*)	161	223
Spotted Sandpiper (*Actitis macularius*)†	1	8
Tree Swallow (*Tachycineta bicolor*)	25	54
Turkey Vulture (*Cathartes aura*)	8	6
Upland Sandpiper (*Bartramia longicauda*)*†	43	69
Vesper Sparrow (*Pooecetes gramineus*)†	219	80
Warbling Vireo (*Vireo gilvus*)	6	6
White-breasted Nuthatch (*Sitta carolinensis*)	1	-
Western Meadowlark (*Sturnella neglecta*)†	182	223
Willow Flycatcher (*Empidonax traillii*)	2	-
Wild Turkey (*Meleagris gallopavo*)	5	4
Wilson's Phalarope (*Phalaropus tricolor*)*	-	1
Wood Duck (*Aix sponsa*)	-	1
Yellow-billed Cuckoo (*Coccyzus americanus*)*	2	3
Yellow Warbler (*Setophaga petechial*)	26	9

*Iowa species of greatest conservation need (IDNR 2015)
†Grassland Species

Table 2-4. Mean (standard error in parentheses) density of singing male birds per ha in commercial row-crop fields (corn [*Zea mays* L.], and soybean [*Glycine max* (L.) Merr.]) without and with prairie strips in Iowa, USA, 2015-2020.

Species	**Control (Crops without Prairie Strips)**	**Treatment (Crops with Prairie Strips)**
All Birds	2.38 (0.21)	4.49 (0.35)
All Grassland Birds	1.4 (0.13)	3.65 (0.23)
Common Yellowthroat	0.16 (0.03)	0.55 (.07)
Dickcissel	0.43 (0.04)	1.15 (0.11)
Eastern Meadowlark	0.03 (0.01)	0.07 (0.01)
Red-winged Blackbird	0.67 (0.09)	1.75 (0.15)
Western Meadowlark	0.02 (0.01)	0.04 (0.01)

Table 2-5. Model selection results estimating the influence of spatial variables in fields with prairie strips on the density of all grassland birds and the three most common species: Red-winged Blackbird, Dickcissel, and Common Yellowthroat. All models included site as a random effect. K = the number of variables (fixed and random) in each model; AIC = Akaike's Information Criterion; AIC_c = AIC corrected for small sample sizes; and w_i = Akaike weight.

Species	**Model**	***K***	**AIC_c**	**ΔAIC_c**	**w_i**
All Grassland Birds	Prairie Strip Age	4	202.54	0	0.352
	Global	10	203.38	0.847	0.230
	Local Crop Cover	4	204.22	4.848	0.151
	Local Prairie Cover	4	206.009	6.636	0.061
	Null	3	206.324	6.952	0.053
	Landscape Grassy Patches	4	206.966	7.593	0.038
	Local Prairie Cover + Local Grass Cover	5	207.001	7.628	0.037
	Local Grass Cover	4	207.017	7.644	0.037
	Local Prairie Patches	4	207.218	7.846	0.033
	Water	4	211.1865	11.813	0.004
Red-winged Blackbird	Prairie Strip Age	4	155.778	0	0.412008
	Global	10	156.8116	1.033632	0.245728
	Local Crop Cover	4	158.7326	2.954598	0.094042
	Local Grass Cover	4	159.3194	3.541378	0.07013
	Local Prairie Patches	4	159.885	4.107021	0.052854
	Landscape Grassy Patches	4	160.3581	4.580097	0.041721
	Null	3	160.4503	4.672352	0.03984
	Local Prairie Cover + Local Grass Cover	5	161.3939	5.615954	0.024855
	Local Prairie	4	162.2013	6.423297	0.0166
	Local Water	4	166.2227	10.44471	0.002223
Dickcissel	Prairie Strip Age	4	126.847	0	0.369
	Local Crop Cover	4	127.911	1.064	0.217
	Null	3	128.49	1.6435	0.162
	Local Prairie Cover	4	129.739	2.8923	0.087
	Landscape Grassy Patches	4	130.936	4.0889	0.048
	Local Prairie Patches	4	131.1	4.2535	0.044
	Local Grass Cover	4	131.495	4.6479	0.036
	Global	9	132.758	5.9111	0.019
	Local Prairie Cover + Local Grass Cover	5	132.887	6.0405	0.018
Common Yellowthroat	Local Crop Cover	4	81.492	0	0.621
	Null	3	83.602	2.109	0.216
	Landscape Grassy Patches	4	86.731	5.239	0.045
	Local Prairie Cover	4	86.786	5.294	0.044
	Local Prairie Patches	4	86.804	5.312	0.044
	Local Prairie Cover + Local Grass Cover	5	88.118	6.626	0.023
	Prairie Strip Age	4	90.515	9.023	0.007
	Global	9	98.649	17.157	0.001
	Local Grass Cover	4	131.49	50.001	0.001

Table 2-6. Standardized regression coefficients, 95% lower (LCI) and upper (UCI) confidence intervals, and p-values of global model of predictors of the density of all grassland birds in commercial row-crop fields (corn [*Zea mays* L.], and soybean [Glycine max (L.) Merr.]) with prairie strips.

Covariate	Estimate	LCI	UCI	*p*-value
Intercept	3.721	2.989	4.553	0.014
Prairie Strip Age	0.707	0.286	1.127	0.002
Local Crop Cover	-0.750	-2.677	1.179	0.452
Local Grass Cover	-0.821	-2.700	1.058	0.473
Local Prairie Cover	0.120	-1.590	1.831	0.847
Local Prairie Patches	-0.256	-1.499	0.986	0.811
Landscape Grassy Patches	-0.074	-1.352	1.203	0.920
Local Water	-0.493	-1.848	0.863	0.552

Table 2-7. Standardized regression coefficients, 95% lower (LCI) and upper (UCI) confidence intervals, and p-values for most competitive model predicting Red-winged Blackbird, Dickcissel, and Common Yellowthroat densities in commercial corn and soybean crop fields with prairie strips.

Species	Covariate	Estimate	LCL	UCL	*p*-value
Red-winged Blackbird	Intercept	1.980	1.477	2.483	0.000
	Prairie Strip Age	0.445	0.196	0.693	0.001
Dickcissel	Intercept	1.248	0.993	1.503	0.000
	Prairie Strip Age	0.295	0.100	0.490	0.006
Common Yellowthroat	Intercept	0.618	0.463	0.774	0.000
	Local Crop Cover	-0.139	-0.295	0.016	0.122

Appendix. Supplemental Tables.

Table A-1. Land cover characteristics within 500m of study field center point. SD = standard deviation.

Site Name	Field Type	%Crop	%Grass	%Prairie	%Woody	%Developed	%Water
ARM	Control	60	22	0	13	5	0
ARM*	Treatment	75	8	13	0	4	0
EIA	Control	80	12	8	0	0	0
EIA*	Treatment	40	28	21	6	5	0
GUT	Control	76	18	6	0	0	0
GUT	Treatment	73	9	18	0	0	0
KAL	Control	61	14	0	21	4	0
KAL	Treatment	51	32	12	0	5	0
MCN	Control	47	31	4	5	6	7
MCN	Treatment	62	12	8	2	0	16
RHO	Control	75	14	0	3	8	0
RHO*	Treatment	48	7	15	19	11	0
SMI	Control	77	23	0	26	0	0
SMI	Treatment	49	22	19	2	0	8
SLO	Control	59	21	0	12	8	0
SLO	Treatment	72	6	22	0	0	0
WHI	Control	76	10	0	10	4	0
WHI	Treatment	43	19	26	7	5	0
WOR	Control	56	23	7	9	5	0
WOR*	Treatment	38	14	16	27	3	2
Mean	Control	66.70	18.80	2.50	9.90	4.00	0.70
SD	Control	11.37	6.41	3.37	8.60	3.09	2.21
Mean	Treatment	55.10	15.70	17.00	6.30	3.30	2.60
SD	Treatment	14.22	9.18	5.31	9.35	3.53	5.34

*Randomized treatment field

CHAPTER 3. INVESTIGATION OF SPRINGTIME BIRD USE OF CORN BELT AGRICULTURAL LANDSCAPES USING AUTONOMOUS RECORDING UNITS

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Modified from a manuscript to be submitted to *Journal of Field Ornithology*

Abstract

Autonomous recording units (ARUs) emerged as a novel technology for avian acoustic monitoring in the 2000s. They have since been primarily used as a substitute for human observers during the breeding season; however, there is potential for use of ARUs in springtime soundscapes, such as to study spring departure or arrival of migratory birds and territory establishment of resident and spring arriving birds. We described springtime bird communities of agricultural landscapes, based on data collected between April 1 and May 15, 2015-2018, from ARUs located at 32 sites across 13 counties in Iowa, USA. We compared resident, wintering, and arriving bird communities across site types and analyzed trends in detection, departure and arrival times respectively for wintering and arriving bird species, and further examined springtime occupancy of five grassland species: common yellowthroat (*Geothlypis trichas*), field sparrow (*Spizella pusilla*), grasshopper sparrow (*Ammodramus savannarum*), savannah sparrow (*Passerculus sandwichensis*), and vesper sparrow (*Pooecetes gramineus*). We made 4,029 detections of 86 bird species, with an average detection of 11.6 species per ARU per day. The most frequent detections were composed of common farmland species, including red-winged blackbird (*Agelaius phoeniceus*), American robin (*Turdus migratorius*), brown-headed cowbird (*Molothrus ater*), ring-necked pheasant (*Phasianus colchicus*), and eastern meadowlark

(*Sturnella magna*); however, detections also included 18 species of greatest conservation need. Large patch prairies, conventional crops, and crops with prairie strips had higher per-survey species richness than crops with terraces. We found that, in addition to documenting species richness and departure/arrival times of migratory species, ARUs generated species-level detection probabilities similar to or higher than studies on breeding season occupancy of grassland bird species. Detection probabilities of our five focal species ranged from 0.36 – 0.89. Occupancy models revealed further significant springtime land cover associations for field sparrows, savannah sparrows, and vesper sparrows. We concluded that springtime deployment of ARUs can provide valuable information on bird communities and their dynamics.

Introduction

Autonomous recording units (ARUs) emerged as a novel technology for avian acoustic monitoring in the 2000s (Shonefield and Bayne 2017). ARUs have since been primarily used as a substitute for human observers in breeding bird surveys, as birds are more vocally active while breeding and are therefore more identifiable in audio recordings (e.g. Alquezar and Machado 2015, Furnas and Callas 2015, Perez-Gránados et al. 2018). ARUs have potential for a wider variety of innovative uses, however, such as during other times of year (Alquezar and Machado 2015, Shonfield and Bayne 2017).

There is potential for use of ARUs in springtime soundscapes, such as during spring departure or arrival of migratory birds, and territory establishment of resident and spring arriving birds. Buxton et al. (2016) used ARUs to examine shifts in arrival of migratory songbirds in Alaska and found bioacoustic indices to be useful for tracking arrival of songbirds. Other springtime studies include Sanders and Mennill (2014) and Colbert et al. (2015), which used ARUs to examine spring migratory movements and spring gobbling activity of wild turkey (*Meleagris gallopavo*), respectively. ARUs have also proven effective for studying ruffed grouse

(*Bonasa umbellus*) drumming behavior during April recordings in Minnesota (Deaux et al. 2020, Grinde et al. 2021). The use of ARUs in studies on spring migration and territory establishment, however, is still relatively unexplored. ARU studies could form a 'middle ground' augmenting broad-scale taxon-wide data collected through radar surveys with species-level information, and individual-scale data tracked through global positioning systems with community information. With climate change, land use change, and other potential disrupters affecting temporal and spatial patterns in biological activity, networks of ARUs could potentially be deployed in the spring to track species-level shifts in migration timing or spatial concentration or dispersion. Climate change is already thought to be the primary driver of shifts in spring arrival of birds (Mason 1995, Brown et al. 1999, Bradley et al. 1999, Crick 2004, Sparks et al. 2007, Swanson and Palmer 2009, Van Buskirk et al. 2009).

Breeding season ARU recordings are often used with occupancy modeling (e.g. Furnas and Callas 2015, Stiffler et al. 2018). Similarly, nearly all habitat-association studies that use occupancy modeling take place during the breeding season, with little attention given to non-breeding patterns. A major reason for this is the closure assumption of occupancy modeling, i.e. no changes in availability between survey periods (MacKenzie et al. 2002, Rota et al. 2009). However, Kendall et al. (2013) demonstrated that the closure assumption can be met by staggering arrival and departure times to avoid biasing occupancy estimates. Higher vocal activity near the beginning of the breeding season may also lead to higher detection rates. A study of Canadian forest birds found higher detection rates and higher species richness earlier in the breeding season (Ehnes et al. 2018). Springtime detection rates are largely unexplored in most systems but vocal activity may be high enough for accurate occupancy estimates.

We sought to evaluate the utility of ARUs for studying spring bird communities as part of a larger study seeking to understand grassland bird use of agricultural landscapes of the U.S. Corn Belt. Since 1970, grassland birds in North America have experienced a 53% overall decline, the steepest reduction of any bird community (Rosenberg et al. 2019). In North America, 80% of all grasslands have been lost through conversion to agriculture and only 1% of the original tallgrass prairie remains in most states and provinces (Knopf 1994, Samson and Knopf 1994). Corn Belt agriculture has intensified during the last several decades, with shifts toward monocultures that support fewer native species (Matson et al. 1997, Brown and Schulte 2011). Increasing agricultural efficiency has involved increased removal of natural field edges, expanding artificial drainage, tillage, use of pesticides, and early season mowing on croplands and high livestock stocking rates in pastures (Newton 1998, Brown and Schulte 2011). Many grassland-breeding species now rely on small, semi-natural grassy features embedded within agricultural landscapes. Some features have been found to increase bird use of crop fields including grass terraces (Hultquist and Best 2001), grass waterways (Bryan and Best 1991), field borders (Conover et al. 2009, Burger et al. 2010), and riparian buffers (Berges et al. 2010). These grassy features, usually dominated by exotic cool-season grasses, are unlikely to provide high quality breeding habitat to species that once relied on extensive tracts tallgrass prairies. Structurally different grasses, such as those found in on-farm features, have been shown to cause temporal and spatial shifts in breeding patterns of grassland birds (Anderson and Steidl 2020). Compared to forest birds, grassland species are more susceptible to local extinctions due to increasing temperatures; species occupying fragmented grasslands are at even higher risk (Jarzyna et al. 2016).

Migratory grassland birds are thus likely to be heavily affected by both climate change and habitat modification. Yet, little research has been devoted to their springtime habitat use. We sought to fill this gap by using ARUs to investigate the springtime bird community across agricultural landscapes in Iowa, a central U.S. Corn Belt state. Our specific objectives were to:

1. Describe springtime bird communities of agricultural landscapes;
2. Examine springtime detectability, occupancy, and variability in occurrence of five focal grassland species: common yellowthroat (*Geothlypis trichas)*, field sparrow (*Spizella pusilla)*, grasshopper sparrow (*Ammodramus savannarum)*, savannah sparrow (*Passerculus sandwichensis)*, and vesper sparrow (*Pooecetes gramineus)*;
3. Evaluate the utility of ARUs for springtime studies of avian habitat use.

We hypothesized that species richness would increase with the amount of grass cover at study sites given the Corn Belt region was historically, prior to EuroAmerican land use, dominated by grassland (Conner et al. 2001). Across our five focal species, based on published habitat associations (Birds of North America 2022), we predicted that common yellowthroat occupancy would increase with prairie cover and woody cover, field sparrow occupancy would increase with woody cover, grasshopper sparrow and savannah sparrow occupancy would increase with grass cover, and vesper sparrow occupancy would increase with crop cover. We also hypothesized that springtime species-level detection probabilities would be lower than those generated during breeding season studies.

Methods

Study Area

The study area was composed of 32 sites located across 13 counties in Iowa (Table A-1). Iowa is a central state within the US Corn Belt and its landscapes are dominated by annual row crop agricultural production, primarily for corn (*Zea mays* L.) and soybeans (*Glycine max* [L.] Merr.), which comprise 72.1% of the statewide land cover (Iowa State University Extension and Outreach 2016). Other common land cover types in include pasture (8.6%; mostly cool-season exotic grasses such as *Bromus inermis* Leyss), forest (6.5%), and developed (5.4%; Iowa State University Extension and Outreach 2016). The climate is humid continental, with cold winters and warm summers. Average statewide monthly temperature during the period of observations (April – May) of 22.5 degrees Celsius, and average monthly precipitation of 11.1 cm (NOAA NWS, 2022).

Study sites were chosen as a part of a broader investigation of bird use of agricultural landscapes (Stephenson 2022). Permission to access the land from the land manager was required for study. Sites were comprised of one of four types: (1) patches (38-102 ha) of reconstructed or restored prairie (hereafter, large patch prairies), (2) corn and soybean crops grown using conventional practices for the region and without substantial areas of conservation cover, (3) conventionally managed crops with terraces, and (4) conventionally managed crops with prairie strips. Crop fields with terraces included narrow berms installed to minimize soil erosion and covered in cool-season grasses. Crop fields with prairie strips included linear non-crop areas composed of diverse, reconstructed native prairie vegetation to improve biodiversity and provide multiple ecosystem services (Schulte et al. 2017). Perennial vegetation at sites was mostly dormant during the study period and crops were planted between mid-April and early-May.

Data Collection

For each ARU (Songmeter SM3, Wildlife Acoustics, Maynard, Massachusetts, USA), we generated a random point within a farm field and placed the unit in the nearest grassy feature or otherwise unfarmed area. Each unit was mounted ~1.5m above the ground on a steel fence post. ARUs were programmed to record daily for 1 hr beginning 15 min before sunrise and ending 45 minutes after sunrise. Acoustic data were routinely collected and stored for later analysis. We analyzed data collected from April 1-May 15, 2015-2018. We chose this period to coincide with the migratory season for grassland birds, and prior to when in-person observations through bird point counts typically begin in the study region. We provide a summary of workflow in Figure 3-1.

We analyzed each 60-min recording of the daily dawn chorus from each deployment location through the specified period, excluding days with excessive wind, rain, or other background noise. Of the 2,088 total available recordings, 348 were deemed usable with little noise interference. Due to low availability of recordings in some years, 299 of the surveys occurred in 2016 and 2018. We used an intermittent subsampling procedure generated using R statistical software R 4.1.2 (R Development Core Team 2021), listening to a random minute from each 5-minute segment of each 60-minute recording. For each recording, we recorded the common name of each species present with the ordinal number of each minute in which that species was detected (e.g., savannah sparrow in minutes 2, 8, 11, and 40). All species that could not be initially identified were checked by a secondary observer. We removed 111 unknown vocalizations, comprised mostly of distant calls that could not be identified to the species level, from the analysis. Of the 87 species we detected, we classified 44 species as spring arrivers, 34 as year-round residents, and nine as winterers according to the Iowa Ornithologists' Union (IOU 2020; Table 3-1).

We first compared species richness among site types. Because recording availability varied across sites, we used a two-way ANOVA to evaluate the effect of site type and Julian date on species richness. We then computed Tukey HSD to perform multiple pairwise-comparisons between the means of groups.

Temperature and wind speed are known to affect bird activity and thus detectability (Robbins 1981). We obtained mean daily temperature data from the nearest regional weather monitoring station (NOAA 2022). We were unable to obtain reliable historic wind speed data, but accounted for this factor's impact on observation by excluding days with excessive wind in recordings.

To examine environmental predictors of occupancy, we used aerial images provided by the National Agriculture Imagery Program (USDA NAIP 2020) to digitize land cover within 200 m of each ARU. We used field verification to resolve ambiguous land cover in aerial imagery. We calculated the land cover percentage of local environmental variables within 200 m of each ARU (Table 3-2). We chose 200 m based on the maximum detection distance of around 100 m for most grassland bird species, which was determined in-person breeding bird surveys at our sites (J.G., unpublished data). We used a six-class cover classification system: crop, grass, prairie, woody, developed, and water cover. We calculated the distance from each ARU to the nearest road, a variable that commonly thought to influence detectability of birds (Yip et al. 2017).

Occupancy Modeling

We evaluated species whose occurrence showed enough variability to allow successful model-fitting. We did not model the occurrence of common species, such as dickcissel (*Spiza americana*) and red-winged blackbird (*Agelaius phoenicius*), which were present at nearly all sites. Using detection histories of five focal species (common yellowthroat, field sparrow, grasshopper sparrow, savannah sparrow, and vesper sparrow), we created single-season occupancy models in R package ‘unmarked’ (Fiske and Chandler 2011). We considered a site to be each ARU deployment location in each year (hereafter “site-year”) and treated each day as a separate “site-visit.” We assumed independence among sites and years. In preparation for occupancy analysis of grassland songbirds, we classified each species as one of the following: year-round resident, winterer, or spring arriving. All focal species were considered spring arrivers. To avoid violating the closure assumption for occupancy modeling, we did not consider any of these species available for detection until either its first detection at a site or first detection at a nearby site if it was never detected during a specific year (MacKenzie et al. 2002, Kendall et al. 2013).

We log-transformed all covariates to reduce skewness and tested for collinearity among spatial covariates using variance inflation factor (VIF). We used temperature and distance to road to model detection of each focal species. Other variables commonly used to model detection such as observer and time of day did not vary in our study. After determining the best predictor of detection for each species, we then used spatial covariates to construct occupancy models. We created interaction models for each species based on known habitat associations and life history traits. Birds of the World (2022) was used as the definitive source for life history traits. In all models, we accounted for variation in survey effort at each site by offsetting each model’s regression by the number of surveys. We evaluated and ranked candidate models using Akaike information criterion adjusted for small sample size (AIC_c) and the associated Akaike weight, w_i (Burnham and Anderson 2002). We used model averaging and multimodel inference with $\Delta AIC_c < 2.0$ across all candidate models to estimate the effects of covariates on occupancy using package “AICcmodavg” (Mazerolle 2020) in R statistical software 4.1.2 (R Development Core

Team 2021). We reported parameter estimates and predicted occupancy rates with 85% confidence intervals as recommended for limited sets of *a priori* models (Arnold 2010).

Results

We made 4,029 detections of 86 bird species, with an average detection of 11.6 species per ARU per day. The most frequently detected species were red-winged blackbird (in 92% of recordings), American robin (*Turdus migratorius*; 86%), brown-headed cowbird (*Molothrus ater*; 78%), ring-necked pheasant (*Phasianus colchicus*; 72%), and eastern meadowlark (*Sturnella magna*; 59%). Eighteen of 87 species we detected are listed as Iowa species of greatest conservation need (SGCN; IDNR 2015). The mean last date of detection of wintering species was April 27th and the mean first date of detection of arriving species was April 30th.

Among site types, large patch prairies had the highest mean per-survey species richness at 13.55 ± 4.02 (standard deviation) followed by crops with prairie strips (11.99 ± 3.73), conventional crops (11.98 ± 4.02), and crops with terraces (9.96 ± 3.71; Fig. 3-2). Site type had a significant effect on species richness but not Julian date (Table 3-3). Among pairwise comparisons, species richness in crops with terraces was significantly less than conventional crops, crops with prairie strips, and large patch prairies ($p < 0.05$). All other pairwise differences were not statistically significant. Most birds were found in multiple site types, but dark-eyed junco (*Junco hyemalis*) and white-crowned sparrow (*Zonotrichia leucophrys*) were only detected at a control site with nearby woody cover; horned lark were detected in every site type but large patch prairie; swamp sparrows were only detected in a field with prairie strips in 2018; and Wilson's snipe (*Gallinago delicata*) were only detected at two sites. Several SGCN were detected during three or less fewer surveys. Notably, greater yellowlegs (*Tringa melanoleuca*) and northern bobwhites (*Colinus virgninianus*) were detected in every site type but large patch prairie.

The date of first detection of our five focal species varied (Table 3-4), and the number of occupied sites increased steadily throughout the study period (Fig. 3-3). All occupancy models met goodness-of-fit criteria and were unadjusted. Naive detection probabilities for our five focal species ranged from 0.36 – 0.89 (Table 3-4). The covariates for the top detection probability models for each focal species were: temperature for common yellowthroat, distance to road for field sparrow and vesper sparrow, and a constant (i.e. null) for grasshopper sparrow and savannah sparrow.

Species-level occupancy probabilities varied greatly among land cover types (Table 3-5). Spatial predictors of occupancy also differed (Table 3-6). Field sparrow occupancy was positively related to woody cover (Fig. 3-4B; β=2.19, 85% CI: $0.87 \leq \beta \leq 3.50$) and developed cover ($\beta$=1.32, 85% CI: $0.57 \leq \beta \leq 2.08$). Savannah sparrow occupancy was negatively related to woody cover (Fig. 3-4D; β=-1.70, 85% CI: $-2.74 \leq \beta \leq -0.66$). Vesper sparrow occupancy was negatively related to water cover (Fig. 3-3E; β=-0.73, 85% CI: $-1.29 \leq \beta \leq -0.19$) and woody cover ($\beta$=-0.65, 85% CI: $-1.27 \leq \beta \leq -0.02$). We did not find significant statistical relationships between land cover and the occupancy of either common yellowthroat or grasshopper sparrow. Prairie cover was the best predictor for common yellowthroat occupancy, but confidence intervals of beta estimates overlapped zero and were uninformative (Fig. 3-4A; β=2.71, 85% CI: $-0.69 \leq \beta \leq 6.11$). Crop cover was the best predictor of grasshopper sparrow occupancy, but again confidence intervals of beta estimates overlapped zero (Fig. 3-4C; β=-1.57, 85% CI: $-3.42 \leq \beta \leq 0.28$).

Discussion

We used the bird community of agricultural landscapes to examine the utility of ARUs for studies of springtime avian habitat use. We found that in addition to documenting species richness of springtime avian communities, ARUs generated species-level detection probabilities similar to or higher than studies on breeding season occupancy of grassland birds (Sidie-Slettedahl et al. 2015, West et al. 2016, Rigby and Johnson 2019, Vanausdall and Dinsmore 2020), though we conducted more surveys. Several focal migratory species showed significant trends in springtime habitat associations. Our study supports expanded use of ARUs and other acoustic devices in the examination of springtime bird communities.

We detected 87 total species across our study sites, including 44 species as spring arrivers, 34 as year-round residents, and nine as winterers, corresponding to 28% of species that regularly occur in the state throughout the year (IOU 2022). Among dates of first detections, we documented an eastern kingbird (*Tyrannus tyrannus*) on April 14, 2016. This is among the three earliest detections of the species in the state (IOU 2020). The detection occurred during an exceptionally warm period with statewide temperatures 3 – 5 degrees Celsius above average (NWS 2016).

We found a trend toward large patch prairies having the highest species richness; however, our hypothesis of increasing bird richness with increasing grassland cover was not statistically supported: springtime species richness was similar among site types we investigated with the exception of crops with terraces (Fig. 3-2). During the breeding season, terraces support fewer species and lower bird abundance than other grassy features (Hultquist and Best 2001). Terraces, by design, are placed in erosive portions of fields and we suspect the steep slopes of upland terraces reduce their value as cover for many species.

Most SGCN species with more than one detection were documented across all site types with the exception of greater yellowlegs and northern bobwhite, which were not found in large patch prairie. Greater yellowlegs are migratory during our study period but northern bobwhite are likely breeding and prefer mosaics of small patches of vegetation including grasslands and early successional vegetation (Brennan et al. 2020). During non-breeding seasons, Janke and Gates (2013) found that bobwhites selected early successional woody cover over grassland cover. Our large patch prairie sites contained little woody cover and were surrounded primarily by row crop fields.

Springtime occupancy varied among the five focal species we studied, and are consistent with previous studies on breeding habitat preferences (Fig. 3-3). Our hypotheses regarding field sparrow and vesper sparrow occupancy were supported but predictors of savannah sparrow occupancy differed from our expectations. Our analysis of the relationship of common yellowthroat and grasshopper sparrow occupancy according to land cover was inconclusive. Common yellowthroats prefer dense vegetation during breeding (Guzy and Ritchison 2020), but no study has quantified their habitat preference during spring migration. Grasshopper sparrows prefer grass-dominated fields and avoid crops (Vickery 2020). During the winter and spring, the species regularly occupies in weedy fields in the southeastern U.S. and co-occurs with savannah sparrows and song sparrows (*Melospiza melodia*; Dunning and Pulliam 1989).

We predicted that field sparrow occupancy would increase with woody cover. In this analysis, we found woody cover to be the best predictor and have a positive relationship with field sparrow occupancy. Other competitive models included water cover and developed cover (Table 3-6). Field sparrows prefer fields with a wealth of tree or shrub perches (Carey et al. 2020). During the winter and spring, field sparrows used abandoned agricultural fields and forest edges (Allaire and Fisher 1975). The species is more often found in less disturbed edge habitats in the eastern and southeastern U.S. (Marcus et al. 2000, Smith et al. 2005).

We predicted that savannah sparrow occupancy would increase with grass cover. We found woody cover to be the best predictor and have a negative relationship with savannah sparrow occupancy. An additive model of woody, water, and prairie cover, and a model including crop cover were also competitive (Table 3-6). Savannah sparrows prefer open country including grassy meadows, cultivated fields, and lightly grazed pastures and avoid areas with extensive woody cover (Wheelwright and Rising 2020). During the winter and spring, savannah sparrows are found in open fields, coastal marshes, and near surface water (Wheelwright and Rising 2020).

We predicted that vesper sparrow occupancy would increase with crop cover. We found water cover to be the best predictor of occupancy with a negative relationship. A model including woody cover was also competitive (Table 3-6). Vesper sparrows breed in dry, open areas with limited woody cover (Jones and Cornely 2020). During winter, vesper sparrows are found in grasslands, weedy fields, and savannahs (Howell and Webb 1995). In spring, they use pastures and weedy areas near fields and roadsides during migration (Jones and Cornerly 2020). Surprisingly, vesper sparrow occupancy was lower in conventional crops than field sparrow, grasshopper sparrow, and savannah sparrow occupancy (Table 3-5). This may be an artifact of site selection as sites with conventional crops often contained grass waterways.

While our study documented how ARUs can help quantitatively expand information on the spring natural history of bird species and allow deeper understanding of spatial and temporal pattern of occupancy, in-person surveys remain superior for studies on spring bird phenology. For example, in-person surveys generate higher detection rates than ARUs for secretive species (Sidie-Slettedahl et al. 2015). However, as with many ornithological studies, this study was conducted by academic researchers, whose occupations require presence on university campus during the non-summer months. Automated sampling techniques can help overcome these practical constraints to observing bird behavioral patterns, offering new knowledge about basic natural history and how it may be changing with their environments.

Our work was part of a larger project that also employed traditional methods such as bird point counts (see Chapter 2) and nest searches (Stephenson 2002) to study bird use of agricultural landscapes. This is the first study of non-breeding bird use of prairie strips, a conservation practice with multiple ecological benefits (Schulte et al. 2017). Combined with bird point counts conducted at the same locations during the breeding season (see Chapter 2), our springtime study of bird use of agricultural landscapes provides information on multi-season habitat associations of grassland birds. Still, wintering and migratory ecology of grassland birds are poorly studied (Vickery et al. 1999).

Given the limitations of ARUs, we employed several methods to ensure the robustness of our analyses. Since we could only be certain of the dates when birds began vocalizing, and not the true arrival dates of migrant birds, we did not start survey periods for occupancy analysis until after a site was known to be occupied by a species. Because acoustic recordings are commonly obstructed by ambient noise, particularly wind (Digby et al. 2013), we removed recordings with excess wind and analyzed selected recordings from days with ideal survey conditions. Being able to collect data over a large number of days, but remove surveys conducted under poor observational conditions was an advantage of long-term deployment of ARUs. Yet, it is still possible that ambient noise reduced our ability to detect some individuals. Limitations remain, however. We were unable to detect non-vocalizing individuals, which may have biased our estimates of richness, especially of wintering species, which may not increase vocalizations until reaching their breeding grounds and thus not be detected despite being present.

We manually analyzed recordings in this study. Advances in automated species recognition, now commonly used in simpler soundscapes, will likely increase the efficiency of processing large amounts of acoustic data (Priyadarshani et al. 2018). As of now, software used to generate spectrograms and edit sound are largely unable to parse species-level detections from breeding season recordings, which are often generated in complex soundscapes (Potamatis et al. 2014, Ulloa et al. 2016). Further development of software is needed for automated recognition to reduce processing time of large sets of complex field recordings and play an even larger role in ecological monitoring.

Conclusion

Soundscape ecology is a burgeoning field of research (Gasc et al. 2016), enabled by technological improvements in acoustic recorders (Servick 2014). ARUs allow researchers to easily repeat sampling, reduce observer bias and field time, and maintain a permanent record of surveys (Shonfield and Bayne 2017). We provided ecological information on grassland bird use of agricultural landscapes during spring, a period that remains vastly understudied for most birds. We conclude that springtime deployment of ARUs can provide worthwhile investigation into spring bird communities and their dynamics. The technology can provide an important tool in monitoring shifts in avian phenology in response to global climate change, a phenomenon that is already known to affect the spring arrival dates of migrant birds.

Acknowledgements

This research was funded by the Leopold Center for Sustainable Agriculture (E2015-10), USDA Farm Service Agency (AG-3151-P-14-0065 and AG-3151-P-17-0108), and the U.S. Federal McIntire-Stennis program (IOW5354, IOW3617). Thank you to the farmers and land owners who allowed us access to their properties to conduct research.

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Figures and Tables

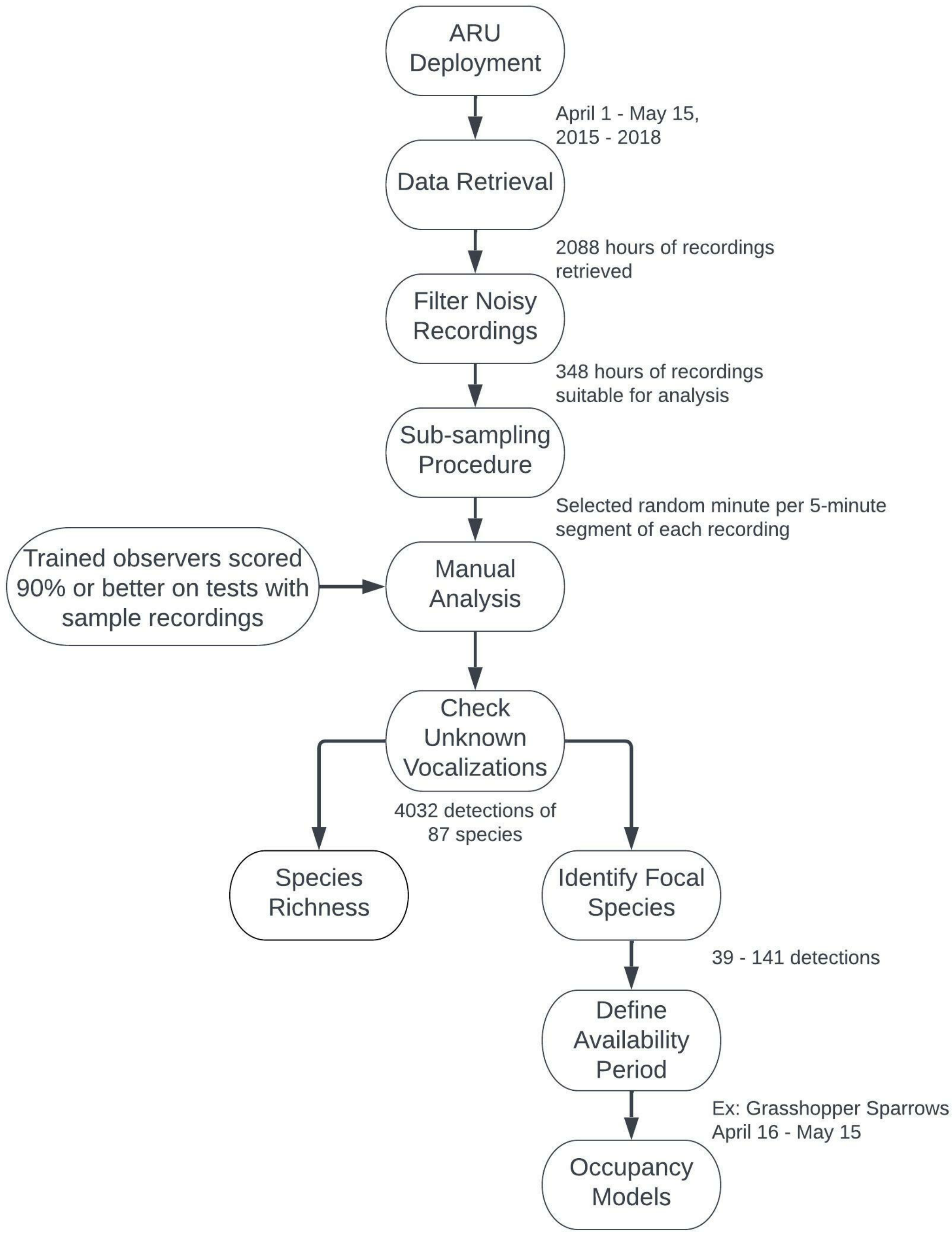

Figure 3-1. Summary of workflow for collecting, processing, and analyzing ARU recordings.

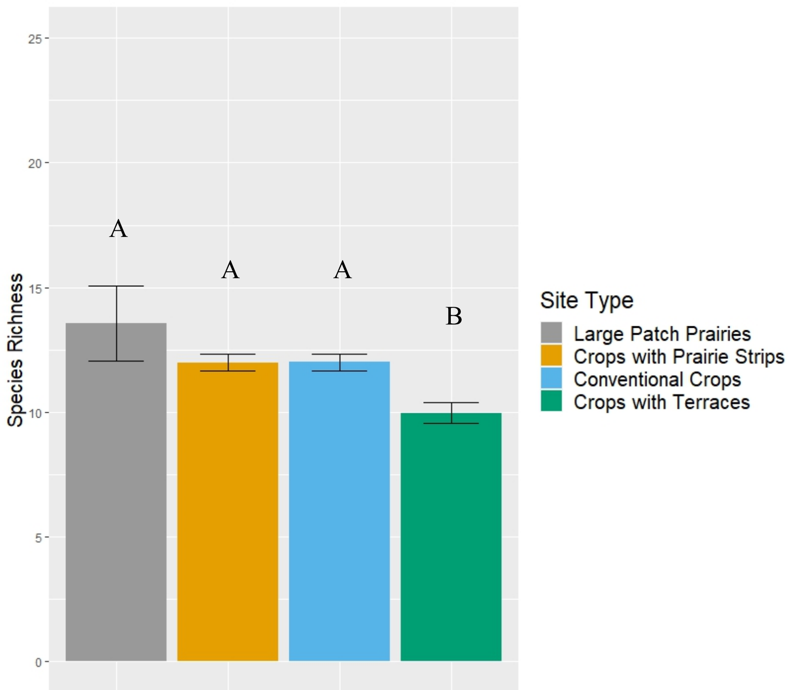

Figure 3-2. Mean per-survey species richness during audio recordings among site types. Error bars indicate standard error. Different letters indicate statistically significant differences between groups.

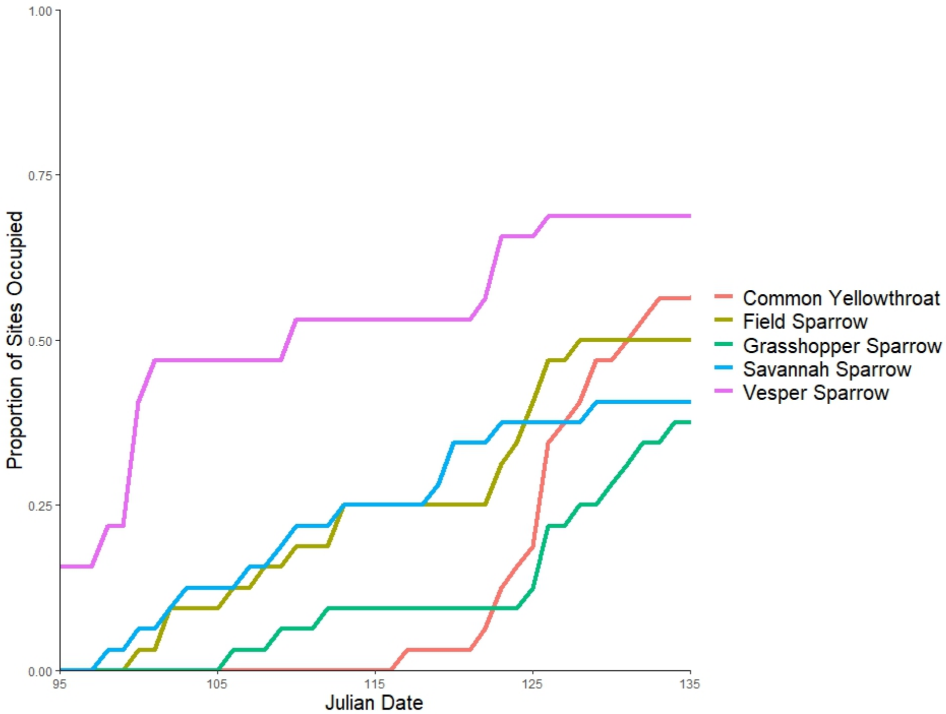

Figure 3-3. Proportion of sites occupied by five focal species across study period. Data were combined across years.

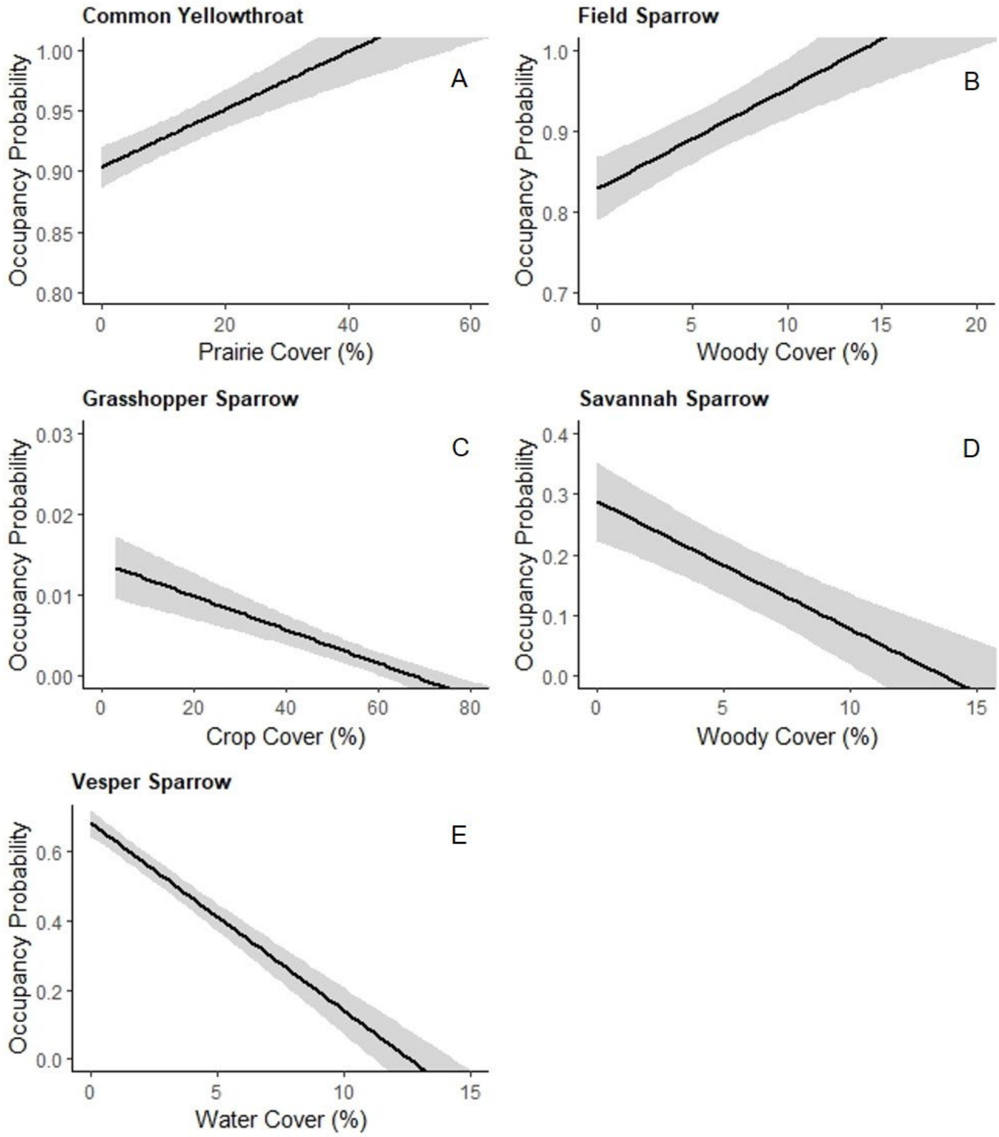

Figure 3-4. Model averaged predictions of occupancy probability of five focal species. Gray area represents 85% confidence limits for the linear model.

Table 3-1. Eighty-seven bird species detected during springtime autonomous recording unit (ARU) surveys in Iowa, 2015-2018. Migration classes based on IOU (2020) designations. Dashes indicate no detections.

Species	Migration Class	% Occurrence	Detections by Site Type				Availability Start Date†	Availability End Date†
			Large Patch Prairie	Conventional Crops	Crops with Terraces	Crops with Prairie Strips		
American Coot	Arriving	0.3	-	-	1	-	113	113
American Crow	Resident	35.2	14	34	43	31	92	136
American Goldfinch	Resident	18.9	10	29	6	17	95	135
American Robin	Resident	86.3	11	119	69	102	91	136
American Tree Sparrow	Wintering	1.1	-	-	-	4	100	110
Barred Owl	Resident	0.5	1	1	-	-	133	134
Baltimore Oriole	Arriving	3.6	1	3	3	2	125	136
Barn Swallow	Arriving	3.8	1	4	1	6	108	135
Black-capped Chickadee	Resident	1.4	-	2	1	-	97	130
Bell's Vireo*	Arriving	0.3	1	-	-	-	128	130
Blue Jay	Resident	33.1	10	65	8	30	92	135
Bobolink*	Arriving	4.6	6	1	2	4	123	134
Brown-headed Cowbird	Resident	78.1	10	107	59	98	91	136
Brown Thrasher	Arriving	33.1	10	50	15	41	98	136
Canada Goose	Resident	42.3	13	40	36	56	91	136
Cedar Waxwing	Resident	0.5	0	1	-	1	129	133
Chipping Sparrow	Arriving	6	1	14	2	4	98	133
Common Grackle	Resident	33.6	2	50	28	42	92	133
Common Nighthawk*	Arriving	0.5	1	1	-	-	133	134
Common Yellowthroat	Arriving	18	15	11	10	24	117	136
Dark-eyed Junco	Wintering	3.8	-	14	-	-	92	110
Dickcissel*	Arriving	9.8	4	6	10	10	122	136
Eastern Bluebird	Resident	2.2	2	4	1	1	103	132
Eastern Kingbird	Arriving	2.7	-	2	-	7	105	134

Table 3-1. Continued.

Species	Migration Class	% Occurrence	Detections by Site Type				Availability Start Date†	Availability End Date†
			Large Patch Prairie	Conventional Crops	Crops with Terraces	Crops with Prairie Strips		
Eastern Meadowlark*	Resident	59	14	77	48	61	91	136
Eastern Phoebe	Arriving	0.8	-	2	-	1	108	123
Eastern Towhee	Arriving	4.6	1	13	-	3	97	132
Eastern Wood-peewee	Arriving	0.3	1	-	-	-	125	130
Eurasian Collared-dove	Resident	5.7	3	4	7	5	97	135
European Starling	Resident	20.2	-	47	5	20	92	135
Field Sparrow*	Arriving	17.2	12	20	11	14	100	136
Great Blue Heron	Resident	1.1	2	1	1	-	124	135
Great Crested Flycatcher	Arriving	0.5	-	-	2	-	132	135
Golden-crowned Kinglet	Wintering	0.3	-	1	-	-	111	111
Great Horned Owl	Resident	0.3	-	0	-	1	103	103
Gray Catbird	Arriving	3	3	3	1	2	123	136
Grasshopper Sparrow*	Arriving	11.5	13	9	9	8	106	136
Greater Yellowlegs*	Arriving	2.2	-	3	1	4	98	117
Greater White-fronted Goose	Arriving	0.3	1	-	-	-	126	126
Harris's Sparrow	Wintering	3.6	-	10	-	3	93	133
Henslow's Sparrow*	Arriving	3	7	-	1	-	125	134
House Finch	Resident	3.3	2	9	-	1	92	131
Horned Lark	Resident	29	-	24	33	43	91	134
House Sparrow	Resident	10.1	-	25	-	10	94	130
House Wren	Arriving	4.1	2	4	3	4	114	136
Indigo Bunting	Arriving	2.2	-	4	2	1	125	136
Killdeer	Arriving	52.7	3	68	46	67	91	136
Lapland Longspur	Wintering	10.1	-	17	2	18	95	115
Lark Sparrow	Arriving	0.3	-	-	1	-	132	132
Lesser Yellowlegs*	Arriving	1.6	-	3	1	2	105	129

Table 3-1. Continued.

Species	Migration Class	% Occurrence	Detections by Site Type				Availability Start Date†	Availability End Date†
			Large Patch Prairie	Conventional Crops	Crops with Terraces	Crops with Prairie Strips		
Mall1ard	Resident	3	-	6	3	1	98	123
Mourning Dove	Resident	39.1	7	59	29	39	91	136
Northern Bobwhite*	Resident	3.8	-	2	3	6	112	136
Northern Cardinal	Resident	50.3	8	90	21	57	91	136
Northern Flicker	Resident	10.4	2	14	9	13	95	129
Northern Parula	Arriving	0.3	-	1	-	-	129	129
Northern Saw-whet Owl	Wintering	0.3	1	-	-	-	126	126
Purple Martin	Arriving	1.1	1	-	3	-	97	126
Rose-breasted Grosbeak	Arriving	1.4	-	1	2	2	121	136
Red-bellied Woodpecker	Resident	7.4	8	2	4	8	117	136
Red-headed Woodpecker*	Resident	2.2	2	-	1	5	108	132
Ring-necked Pheasant	Resident	71.6	16	90	61	81	91	136
Rusty Blackbird*	Wintering	1.6	-	3	-	3	94	117
Red-winged Blackbird	Resident	92.1	15	121	76	108	91	136
Sandhill Crane*	Arriving	0.3	-	-	-	1	99	99
Savannah Sparrow	Arriving	16.1	1	14	11	30	98	133
Sedge Wren*	Arriving	6.8	11	3	4	3	122	135
Sora	Arriving	1.6	2	1	2	-	123	135
Solitary Sandpiper*	Arriving	0.5	-	-	-	2	114	123
Song Sparrow	Resident	54.1	6	77	34	71	91	136
Spotted Sandpiper	Arriving	3	1	2	3	5	95	131
Swamp Sparrow	Resident	0.8	-	-	-	3	107	118
Tennessee Warbler	Arriving	0.3	-	-	-	1	131	131
Tree Swallow	Arriving	6	1	12	5	2	95	135
Trumpeter Swan*	Resident	0.3	-	-	-	1	115	115
Upland Sandpiper*	Arriving	2.2	-	3	-	4	117	133

Table 3-1. Continued.

Species	Migration Class	% Occurrence	Detections by Site Type				Availability Start Date†	Availability End Date†
			Large Patch Prairie	Conventional Crops	Crops with Terraces	Crops with Prairie Strips		
Vesper Sparrow	Arriving	39.3	5	47	27	62	95	136
Warbling Vireo	Arriving	0.8	-	-	2	1	126	136
White-crowned Sparrow	Wintering	1.1	-	4	-	-	93	126
Western Meadowlark	Resident	56.8	1	65	49	88	91	136
Wilson’s Snipe	Arriving	0.8	-	-	3	-	103	109
Wild Turkey	Resident	9.8	6	8	14	5	92	136
Wood Duck	Resident	1.1	-	1	1	2	103	135
White-throated Sparrow	Wintering	3	-	8	1	2	109	129
Yellow Warbler	Arriving	1.1	-	3	-	1	126	135
Yellow-rumped Warbler	Arriving	1.6	-	6	-	-	100	119

*Iowa species of greatest conservation need (IDNR 2015)
†Julian date

Table 3-2. Summary of mean land cover composition (standard deviation) surrounding autonomous recording unit (ARU) deployment sites in Iowa, USA, and distance from ARU to nearest road.

Site Type	Number Sites	% Crop	% Grass	% Prairie	% Woody	% Developed	% Water	Distance to road (m)
Large patch prairie	4	3.0 (4.2)	9.8 (4.9)	70.0 (21.1)	11.5 (9.7)	5.0 (5.8)	0.75 (1.5)	575.8 (311.4)
Crop fields with prairie strips	10	57.7 (15.1)	16.9 (11.9)	15.4 (7.6)	4.8 (8.2)	2.6 (2.3)	2.6 (5.3)	407.4 (184.0)
Crop fields with terraces	7	67.9 (11.5)	27.1 (13.5)	0 (0)	2.6 (4.5)	1.4 (2.6)	1.0 (2.6)	601.9 (420.9)
Conventional crop fields	11	66.8 (13.6)	21.6 (11.2)	0 (0)	7.2 (6.7)	3.2 (2.8)	1.2 (2.3)	299.55 (255.0)

Table 3-3. Two-way analysis of variance results for effect of land cover type and Julian date on species richness for birds during recorded during springtime using autonomous recording unit (ARU) surveys in Iowa, 2015-2018.

Effect	Df	Sum Sq.	Mean Sq.	F-value	p
Land Cover	3	338	112.69	7.192	0.000112
Julian Date	45	975	21.66	1.383	0.061429
Residuals	298	4669	15.67		

Table 3-4. Date of first detection based on springtime autonomous recording unit (ARU) surveys in Iowa, 2015-2018 and detection probabilities with standard errors (SE) of five focal bird species.

Species	Mean Date of First Detection	Detection probability (SE)
Common Yellowthroat	May 2	0.89 (0.05)
Field Sparrow	April 25	0.45 (0.05)
Grasshopper Sparrow	May 1	0.39 (0.07)
Savannah Sparrow	April 24	0.36 (0.05)
Vesper Sparrow	April 25	0.55 (0.03)

Table 3-5. Occupancy probabilities standard errors (SE) for five focal species across site types during springtime autonomous recording unit (ARU) surveys in Iowa, 2015-2018.

	Occupancy (SE)			
Species	Conventional Crops	Large Patch Grassland	Crops with Prairie Strips	Crops with Terraces
Common Yellowthroat	0.46 (0.14)	1.00 (0.00)	0.86 (0.11)	0.82 (0.19)
Field Sparrow	0.83 (0.12)	0.99 (0.01)	0.14 (0.09)	0.47 (0.22)
Grasshopper Sparrow	0.99 (0.05)	0.58 (0.42)	0.69 (0.24)	0.38 (0.18)
Savannah Sparrow	0.98 (0.39)	1.00 (0.00)	0.47 (0.16)	0.60 (0.25)
Vesper Sparrow	0.78 (0.20)	0.55 (0.39)	0.58 (0.13)	0.99 (0.01)

Table 3-6. Candidate model sets sorted by Akaike's Information Criterion with small sample adjustment (AIC_c) for five focal species.

Species	Model	*K*	AIC_c	ΔAIC_c	w_i
Common Yellowthroat	*p*(temp) Ψ(prairie)	4	86.81	0.00	0.53
	p(temp) Ψ(water+prairie)	5	88.86	2.05	0.19
	p(.) Ψ(.)	2	89.77	2.95	0.12
	p(temp) Ψ(grass)	4	92.04	5.22	0.03
	p(temp) Ψ(crop)	4	92.44	5.62	0.03
	p(temp) Ψ(developed)	4	93.34	6.52	0.02
	p(temp) Ψ(water)	4	93.82	7.01	0.02
	p(temp) Ψ(woody)	4	93.86	7.04	0.02
	p(temp) Ψ(crop+grass+water+prairie)	7	93.93	7.12	0.02
	p(temp) Ψ(global)	9	100.11	13.30	0.01
Field Sparrow	*p*(distroad) Ψ(woody)	4	197.59	0.00	0.32
	p(distroad) Ψ(water)	4	198.26	0.67	0.23
	p(distroad) Ψ(developed)	4	198.99	1.39	0.16
	p(distroad) Ψ(global)	9	199.33	1.73	0.13
	p(distroad) Ψ(woody+prairie+developed)	6	200.88	3.29	0.06
	p(distroad) Ψ(woody*prairie)	6	201.37	3.77	0.05
	p(distroad) Ψ(crop)	4	204.69	7.10	0.01
	p(.) Ψ(.)	2	205.58	7.98	0.01
	p(distroad) Ψ(prairie)	4	206.82	9.22	0.01
	p(distroad) Ψ(grass)	4	207.97	10.37	0.01
Grasshopper Sparrow	*p*(.) Ψ(crop)	3	123.84	0.00	0.20
	p(.) Ψ(.)	2	124.03	0.19	0.18
	p(.) Ψ(grass)	3	124.15	0.31	0.17
	p(.) Ψ(distroad*grass)	5	124.82	0.98	0.12
	p(.) Ψ(prairie)	3	126.00	2.16	0.07
	p(.)_psi_praigrass	4	126.04	2.19	0.07
	p(.) Ψ(woody)	3	126.25	2.40	0.06
	p(.) Ψ(developed)	3	126.36	2.51	0.06
	p(.) Ψ(water)	3	126.36	2.52	0.05
	p(.)_psi_woodxgrass	5	128.05	4.21	0.02
	p(.) Ψ(global)	10	136.32	12.48	0.01
Savannah Sparrow	*p*(.) Ψ(woody)	3	223.07	0.00	0.39
	p(.) Ψ(woody+water+prairie)	5	224.2	1.21	0.21
	p(.) Ψ(crop)	3	224.67	1.60	0.17
	p(.) Ψ(developed)	3	227.08	4.01	0.05
	p(.) Ψ(prairie)	3	227.41	4.34	0.04
	p(.) Ψ(.)	2	228.14	5.06	0.03

Table 3-6. Continued.

Species	Model	K	AIC_c	ΔAIC_c	w_i
	p(.)Ψ(distroad+crop+woody+prairie)	6	228.35	5.28	0.03
	p(.) Ψ(global)	9	228.36	5.29	0.03
	p(.) Ψ(crop+water+prairie)	5	228.73	5.66	0.02
	p(.) Ψ(water)	3	229.70	6.62	0.01
	p(.) Ψ(grass)	3	230.25	7.18	0.01
Vesper Sparrow	p(distroad) Ψ(water)	4	348.31	0.00	0.45
	p(distroad) Ψ(woody)	4	350.22	1.91	0.17
	p(distroad) Ψ(crop)	4	352.01	3.68	0.07
	p(distroad) Ψ(developed)	4	352.32	4.01	0.06
	p(distroad) Ψ(prairie)	4	352.48	4.16	0.05
	p(distroad) Ψ(grass)	4	352.48	4.17	0.05
	p(distroad) Ψ(grass*water)	6	352.56	4.24	0.05
	p distroad) Ψ(crop+woody)	5	352.64	4.323	0.05
	p(distroad) Ψ(crop+woody+prairie)	6	354.58	6.266	0.02
	p(.) Ψ(global)	8	383.16	34.84	0.01
	p(distroad) Ψ(.)	2	383.86	35.54	0.01

Appendix. Supplemental Tables.

Table A-1. Study site locations and land cover characteristics.

Farm	County	Site Type	% Crop	% Grass	% Prairie	% Woody	% Developed	% Water	Distance to Road (m)
ARM	Pottawattamie	Crops with prairie strips	75	8	13	0	4	0	173
ARM		Crops with terraces	74	26	0	0	0	0	429
ARM		Conventional crops	60	22	0	13	5	0	274
DMW	Dallas	Crops with prairie strips	74	7	11	4	4	0	610
EIA	Linn	Crops with prairie strips	40	28	21	6	5	0	236
EIA		Crops with terraces	82	11	0	0	7	0	279
EIA		Conventional crops	88	12	0	0	0	0	97
GUT	Story	Crops with prairie strips	73	9	18	0	0	0	616
GUT		Conventional crops	82	18	0	0	0	0	540
JUD	Carroll	Crops with terraces	59	41	0	0	0	0	418
JUD		Conventional crops	44	50	0	2	0	4	736
JUD		Large patch prairie	0	6	84	7	0	3	738
JUD		Large patch prairie	0	5	87	8	0	0	817
KAL	Jasper	Conventional crops	61	14	0	21	4	0	135
KAL		Crops with prairie strips	51	32	12	0	5	0	208
MCN	Lucas	Crops with prairie strips	62	12	8	2	0	16	421
MCN		Conventional crops	51	31	0	5	6	7	144
SLO	Buchanan	Crops with prairie strips	72	6	22	0	0	0	506
SLO		Crops with terraces	71	29	0	0	0	0	1427
SLO		Conventional crops	59	21	0	12	8	0	47
SME	Webster	Conventional crops	74	14	0	7	3	2	267
SME		Crops with terraces	58	21	0	11	3	7	924
SMI	Wright	Crops with prairie strips	49	22	19	2	0	8	395

Appendix A-1. Continued.

Farm	County	Site Type	% Crop	% Grass	% Prairie	% Woody	% Developed	% Water	Distance to Road (m)
SMI		Conventional crops	77	23	0	26	0	0	748
WAT	Page	Large patch prairie	9	13	41	5	11	0	124
WAT		Crops with terraces	79	14	0	7	0	0	388
WHI	Guthrie	Crops with terraces	52	48	0	0	0	0	348
WHI		Large patch prairie	3	15	68	5	9	0	624
WHI		Conventional crops	76	10	0	10	4	0	141
WHI		Crops with prairie strips	43	19	26	7	5	0	147
WOR	Story	Crops with prairie strips	38	14	16	27	3	2	563
WOR		Conventional crops	63	23	0	9	5	0	166

CHAPTER 4. THE EFFECT OF OXBOW RESTORATION ON BREEDING BIRDS IN AN AGRICULTURAL LANDSCAPE

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Modified from a manuscript published in *Ecological Restoration*

Introduction

Oxbow lakes and wetlands are regarded as among the most biologically diverse aquatic systems in the world (Ward 1988). They are formed when the meander of a river is cut off through sedimentation, leaving an isolated, curved body of water in the former channel (Wohlman and Leopold 1957). As a feature of river floodplains, oxbows tend to have high biodiversity and ecological function compared to neighboring areas because of their structural complexity and the accumulation of organic matter over time (Hillman et al. 1986).

Oxbows and their functions have been lost from many agricultural areas. Stream channelization and removal of adjacent riparian vegetation drastically altered historical stream hydrology and resulted in refilling of oxbows (Schumm et al. 1984). Runoff from vast agricultural systems has also altered the composition and biological value of existing oxbows (Wren et al. 2008). In states like Iowa, the majority of waterways have been channelized to make more room for agricultural production. Yet, after over a century of removal, oxbows are now being restored to provide crucial ecological functions in agricultural landscapes.

Oxbows are effective in reducing the nitrate-nitrogen export from tile drainage systems (Fink and Mitsch 2007, Harrison et al. 2014), and help Midwestern states meet their nutrient reduction strategies, mandated under the United States Mississippi River/Gulf of Mexico 2008 Action Plan (MRGMWNTF 2008). In addition to their contributions to stream hydrology and water quality, oxbows provide critical habitat for several declining fish species including the federally endangered *Notropislo topeka* (Topeka Shiner; Bakevich et al. 2013, Simpson et al. 2019). In Iowa, oxbow restoration efforts have been primarily focused on Topeka Shiner habitat. The U.S. Fish and Wildlife Service and The Nature Conservancy have completed over 100 oxbow restorations in the state (Kenney 2018).

Increases in *N. topeka* populations following oxbow restoration in the Raccoon River and Boone River watersheds in central Iowa raised interest in oxbow restoration impacts on other wildlife species, including birds. Little to no research effort had been devoted to examining the effect of oxbow lake restoration on breeding birds beyond a species inventory in 2015, which documented 54 bird species using four restored oxbows along the Boone River (Harr 2015).

Our objective was to provide a description of the bird species that use restored oxbows in central Iowa. Our objectives were to examine differences in bird communities and species richness between restored and unrestored sites. We hypothesized that restored oxbows would report higher richness of breeding birds than nearby unrestored locations in the same watershed. We also hypothesized that wetland and forest breeding birds, which could use the marsh and riparian vegetation adjacent to oxbows, respectively, would comprise the majority of the bird community.

Methods

Our study was conducted in the Boone River watershed in Hamilton County, Iowa in partnership with The Nature Conservancy. The landscape around this portion of the Boone River was largely dedicated to corn and soybean production. Prior to data collection, The Nature Conservancy oversaw the restoration of three oxbow lakes in 2013. All sites were located within 100m of White Fox Creek north of Webster City, Iowa. In 2016 and 2017, we deployed an autonomous recording unit (ARU) at each of the three restored oxbows and three nearby unrestored sites along White Fox Creek. Distances between each restored and nearby unrestored site ranged from 0.7 km to 1.2 km. At restored sites, we placed ARUs within 50m of the oxbow. Immediate surrounding land cover of the six sites included riparian woodland vegetation, grazed pasture grasses, and ungrazed cool-season grasses.

We programmed ARUs to record simultaneously during 30 minutes of dawn bird chorus starting 15 minutes prior to sunrise each day May 15 – July 15 of both years. After retrieving acoustic data, we used a random subsampling procedure to select recordings to analyze. First, we generated a random day to analyze for each week of the study period and then sub-sampled a random minute from each 5-minute segment of each random recording. During analysis of recordings, a trained listener recorded the first detection of each species and made note of any atypical or unidentified vocalizations to be reviewed by a second listener. We reviewed all unknown and unusual vocalizations before statistical analyses. We assumed a similar detection range among sites. Sampling distance captured by ARUs depend on a myriad of factors, including topography and the vocal characteristics of birds, but are generally shorter than that of human observers (Schonfield and Bayne 2017). We removed recordings with excess wind, rain, or other loud background noise from consideration. Some recordings were lost due to equipment malfunction but no sites were surveyed less fewer than seven times during the eight-week study period of each year. Species were sorted into the following breeding guild classifications according to Peterjohn and Sauer (1993): (1) grassland; (2) forest; (3) shrubland; (4) wetland; and (5) generalist. We used a Shapiro-Wilk test of normality and a Welch's t-test to evaluate differences between the number of species detected at restored and unrestored sites. We conducted a total of 86 surveys; 42 at restored sites and 44 at unrestored sites.

Results and Discussion

We detected 80 total bird species across all surveys; 71 species at restored sites and 58 species at unrestored sites (Table 4-1). We detected more species per survey at restored sites (Welch's t-test; t = -4.81, $p = 0.00006$; Fig. 4-1). Contrary to our prediction, restored sites had a higher percentage of shrubland nesting species (25%) than unrestored sites (11%). These included *Toxostoma rufum* (Brown Thrasher) and *Dumetella carolinensis* (Gray Catbird), which were both detected at higher rates at restored sites. The most common species detected at restored sites were *Melospiza melodia* (Song Sparrow), *Geothlypis trichas* (Common Yellowthroat*)*, and *Phasianus colchicus* (Ring-necked Pheasant). Notable species that were only detected at restored sites included *Ammodramus savannarum* (Grasshopper Sparrow), *Cistothorus palustrisi* (Marsh Wren), and *Actitis macularius* (Spotted Sandpiper). By comparison, unrestored sites had a higher percentage forest and generalist species, and the most common species detected were *Corvus brachyrhynchos* (American Crow), *Turdus migratorius* (American Robin), and *Troglodytes aedon* (House Wren).

We provide a baseline study of bird communities at restored and unrestored oxbows in central Iowa. Our use of ARUs allowed for more extensive investigation into breeding bird use of oxbows than previous in-person surveys, and we detected 17 more species than Harr (2015).

Six species of greatest conservation need (IDNR 2015) – *Dolichonyx oryzivorous* (Bobolink), *Spizella pusilla* (Field Sparrow), *A. savannarum*, *Colinus virginianus* (Northern Bobwhite), *Cistothorus stellaris* (Sedge Wren), and *Coccyzus americanus* (Yellow-billed Cuckoo) – were detected at higher rates at restored sites than unrestored sites. Depending on the species, bird response to oxbow restoration was partially due to the creation of an aquatic environment, changes in the surrounding terrestrial environment, or both. The higher prominence of shrubland birds around restored sites might be a result of vegetation succession since restoration, and might not be stable in time without continued management.

Oxbow lakes and wetlands are dynamic systems, transformed by disturbance, ecological succession, and surrounding land use. In addition to improving an array of other ecological functions, oxbow restorations appear to be an effective strategy for expanding breeding bird habitat in agricultural landscapes. As oxbow environments change through time, we expect their associated bird communities to similarly transition. Oxbow restoration offers an opportunity to integrate small conservation features into landscapes dominated by agricultural production.

Acknowledgements

We thank Karen Wilke and The Nature Conservancy for their support in project coordination and data collection. Funding for this project was provided by the USDA McIntire-Stennis Program (IOW05534).

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Figures and Tables

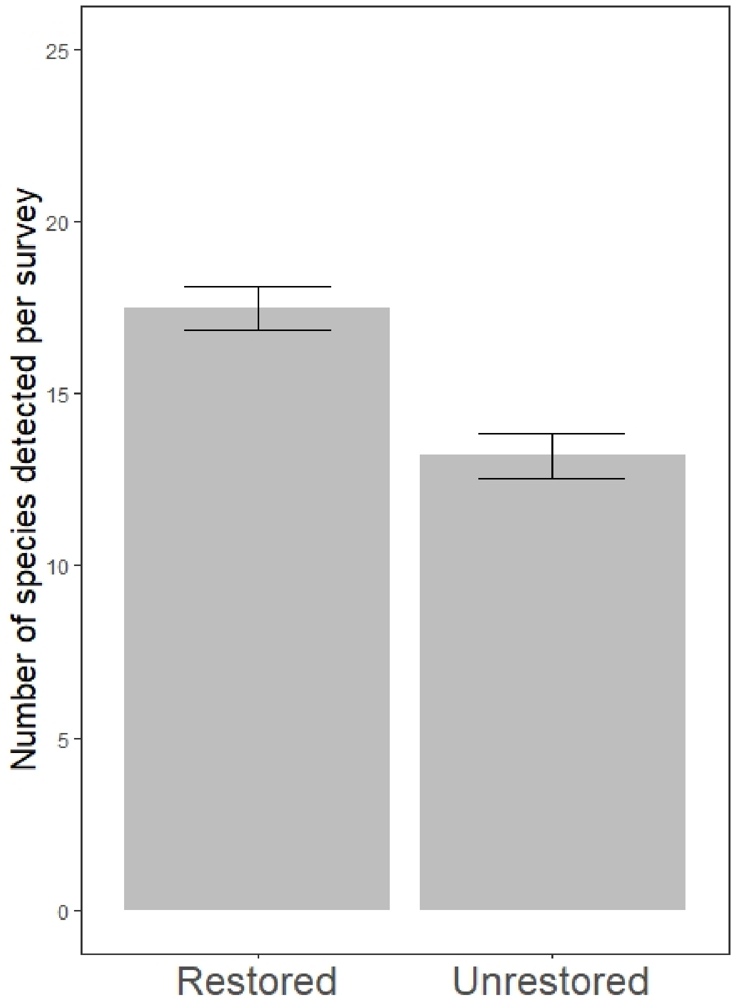

Figure 4-1. The mean number of bird species detected during ARU surveys at restored and unrestored sites. Error bars indicate one standard deviation.

Table 4-1. Species and guild designations of bird detected at restored oxbow lakes and nearby unrestored sites in Hamilton County, Iowa.

Guild	Species	Restored	Unrestored
Forest	*Setophaga ruticilla* (American Redstart)	X	
	Icterus albula (Baltimore Oriole)	X	X
	Poecile atricapilus (Black-capped Chickadee)	X	X
	Cyanocitta cristata (Blue Jay)	X	X
	Polioptila caerulea (Blue-gray Gnatcatcher)	X	
	Thryothorus ludovicianus (Carolina Wren)	X	X
	Bombycilla cedrorum (Cedar Waxwing)	X	X
	Spizella passerina (Chipping Sparrow)	X	X
	Picoides pubescens (Downy Woodpecker)		X
	Syornis phoebe (Eastern Phoebe)	X	
	Contopus virens (Eastern Wood-peewee)	X	X
	Myiarchus crinitus (Great-crested Flycatcher)	X	X
	Leuconotopicus villosus (Hairy Woodpecker)	X	X
	Troglodytes aedon (House Wren)	X	X
	Empidonax minimus (Least Flycatcher)*	X	X
	Leiothlypis ruficapilla (Nashville Warbler)		X
	Colaptes auratus (Northern Flicker)	X	X
	Dryocopus pileatus (Pileated Woodpecker)	X	
	Pheucticus ludovicianus (Rose-breasted Grosbeak)	X	X
	Melanerpes carolinus (Red-bellied Woodpecker)	X	X
	Melanerpes erythrocephalus (Red-headed Woodpecker)*	X	X
	Vireo olivaceus (Red-eyed Vireo)		X
	Piranga olivacea (Scarlet Tanager)	X	X
	Piranga rubra (Summer Tanager)		X
	Catharus ustulatus (Swainson's Thrush)		X
	Leiothlypis peregrina (Tennessee Warbler)	X	X
	Tachycineta bicolor (Tree Swallow)	X	X
	Sitta carolinensis (White-breasted Nuthatch)	X	X
	Meleagris gallopavo (Wild Turkey)	X	
	Aix sponsa (Wood Duck)	X	X
	Hylocichla mustelina (Wood Thrush)*		X
	Coccyzus americanus (Yellow-billed Cuckoo)*	X	X
	Empidonax flaviventris (Yellow-bellied Flycatcher)	X	
	Sphyrapicus varius (Yellow-bellied Sapsucker)	X	
	Icteria virens (Yellow-breasted Chat)*		X
	Setophaga dominica (Yellow-throated Warbler)		X
Generalist	*Corvus brachyrhynchos* (American Crow)	X	X
	Turdus migratorius (American Robin)	X	X
	Hirundo rustica (Barn Swallow)	X	X
	Molothrus ater (Brown-headed Cowbird)	X	X
	Chaetura pelagica (Chimney Swift)*	X	X
	Quiscalus quiscula (Common Grackle)	X	X
	Sialia sialis (Eastern Bluebird)	X	X

Table 4-1. Continued.

Guild	Species	Restored	Unrestored
Generalist	*Streptopelia decaocto* (Eurasian Collared-Dove)	X	
	Sturnus vulgaris (European Starling)	X	X
	Haemorhous mexicanus (House Finch)	X	
	Passer domesticus (House Sparrow)		X
	Zenaida macroura (Mourning Dove)	X	X
	Melospiza melodia (Song Sparrow)	X	X
Grassland	*Dolichonyx oryzivorus* (Bobolink)*	X	X
	Geothlypis trichas (Common Yellowthroat)	X	X
	Spiza americana (Dickcissel)*	X	X
	Sturnella magna (Eastern Meadowlark)*	X	X
	Spizella pusilla (Field Sparrow)*	X	
	Ammodramus savannarum (Grasshopper Sparrow)*	X	
	Colinus virginianus (Northern Bobwhite)*	X	
	Phasianus colchicus (Ring-necked Pheasant)	X	X
	Agelaius phoeniceus (Red-winged Blackbird)	X	X
	Cistothorus stellaris (Sedge Wren)*	X	X
	Pooecetes gramineus (Vesper Sparrow)	X	
	Sturnella neglecta (Western Meadowlark)	X	
Shrubland	*Empidonax alnorum* (Alder Flycatcher)	X	X
	Spinus tristis (American Goldfinch)	X	X
	Passerina caerulea (Blue Grosbeak)	X	
	Toxostoma rufum (Brown Thrasher)	X	X
	Tyrannus tyrannus (Eastern Kingbird)	X	
	Pipilo erythrophthalmus (Eastern Towhee)	X	X
	Dumetella carolinensis (Gray Catbird)	X	X
	Passerina cyanea (Indigo Bunting)	X	X
	Cardinalis cardinalis (Northern Cardinal)	X	X
	Vireo gilvus (Warbling Vireo)	X	
	Setophaga petechia (Yellow Warbler)	X	
Wetland	*Megaceryle alcyon* (Belted Kingfisher)	X	X
	Branta canadensis (Canada Goose)	X	X
	Charadrius vociferus (Killdeer)	X	X
	Anas platyrhynchos (Mallard)	X	
	Cistothorus palustris (Marsh Wren)	X	
	Parkesia noveboracensis (Northern Waterthrush)	X	
	Actitis macularius (Spotted Sandpiper)	X	
	Melospiza georgiana (Swamp Sparrow)		X

*Iowa species of greatest conservation need (IDNR 2015)

CHAPTER 5. INITIAL MULTITAXA RESPONSE TO NATIVE GRASSLAND RECONSTRUCTION COMPARED TO EXOTIC, COOL-SEASON, AGRICULTURAL GRASSES

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Modified from a manuscript to be submitted to *Ecological Restoration*

Abstract

Integration of native grassland vegetation, even in small amounts, into agricultural landscapes provides a promising approach for balancing the needs of native biodiversity with the needs of agricultural production. Little research has examined the influence of varying levels of grassland diversity on wildlife at the scale of typical reconstructions in the Midwestern U.S. Assessment of wildlife communities could provide valuable insight into tradeoffs associated with native origin, diversity, and cost of grassland reconstruction. We established an experiment at a farm in northern Missouri, USA, in February, 2018 to determine the response of multiple biodiversity taxa to three levels grassland plant diversity. The experiment used a randomized complete block design, where individual fields were grouped to one of three blocks based on proximity and similar land-use history and treatment type was randomly assigned. From May to August in 2018-2020, we surveyed plant, pollinator, snake, small mammal, and breeding bird communities in 14 treatment fields seeded to either a 15 species low diversity seed mix or a 31 species high diversity seed mix, as well as six control fields that consisted of a pre-existing mix of predominantly exotic, cool-season agricultural grasses common in the study region (i.e., tall

fescue [*Festuca arundinacea*], smooth brome [*Bromus inermis*]). We examined biodiversity response to treatment and time since restoration. Our results indicated a large increase in native perennial vegetation between years two and three of study in both the low and high diversity prairie treatments. In low diversity fields, grass coverage increased from 19% in 2018 to 40% in 2020 and forb coverage increased from 39% in 2018 to 46% in 2020. In high diversity fields, grass coverage increased from 17% in 2018 to 39% in 2020 and forb coverage increased from 44% in 2018 to 50% in 2020. Increases in native grass and forb cover did not result in an increase in wild bee, snake, small mammal, or bird richness or abundance within the first three years of restoration. We expect wildlife communities will exhibit stronger associations with diversity treatments as prairie vegetation becomes established in subsequent years.

Introduction

Agricultural land cover comprises nearly half of the global land base (Ellis and Ramankutty 2008). With rising global population and changing diets, demand for agricultural products is expected to grow in coming decades (Godfray et al. 2010, Naylor 2011). The Millennium Ecosystem Assessment established that increases in production have historically equated to declines in the remaining suites of ecosystem services, including habitat for biodiversity (MEA 2005). Given this situation, effective means for balancing agricultural production with other needs—or blurring the lines between production and conservation—are sorely needed (Foley et al. 2005; Schulte et al. 2006).

The integration of diverse, native, perennial vegetation, even in small amounts, into agricultural landscapes is a needed component of broader conservation approach, that also includes large reserve areas, to sustain native biodiversity (Garibaldi et al. 2020). In the Midwestern U.S., where grassland wildlife communities continue to decline, various agencies, organizations, and producers have attempted to conserve biodiversity and restore ecosystems services by integrating reconstructed native prairie vegetation within croplands to the historical land cover. Seed availability, cost, and management objectives constrain the number of species used in prairie restorations. Seed mix design is the largest driver of project costs and outcomes in prairie restoration (Larson et al. 2017, Phillips-Mao et al. 2015). Seed mixes with high grass-to-forb ratio are generally less expensive but produce grass-dominated stands with poor forb coverage (McCain et al. 2010, Valko et al. 2016) with little value as pollinator habitat (Hopwood 2008). Alternatively, seed mixes with high forb-to-grass ratio are expensive and susceptible to weed encroachment and soil erosion (Burke and Grime 1996).

Little research has examined the influence of varying levels of seed mix diversity on wildlife impacts of prairie restoration at the scale of typical restorations within agricultural landscapes of the U.S. Midwest. Investigations of biological response to management of small, often plot-level (<100 m^2) restorations include research at Cedar Creek Ecosystem Science Reserve (Symstad et al. 1999), Kellogg Biological Station (Robertson and Hamilton 2015), and the Wisconsin Arboretum (Rooney and Leach 2010). Studies at Konza Prairie Biological Station include replicated experimental work over more extensive areas (>10 ha) in a landscape dominated by grassland instead of crops (Verheijen et al. 2022). Post-restoration monitoring of ecosystem communities could provide valuable insight into the tradeoffs of high and low diversity seed mixes, given the high cost of restoration in this region (Tyndall et al. 2013).

Monitoring efforts in restored prairies are often focused on plant communities and the invertebrates that are likely to respond soon after. However, examining the response of other taxa such as snakes, small mammals, and birds may provide a more complete understanding to early restoration efforts. Grassland bird responses to restoration efforts are complex. In Iowa, grassland bird communities are similar in restored prairies and remnant prairies, except for species that prefer more open vegetation of restorations, such as grasshopper sparrow (*Ammodramus savanarum*) and savannah sparrows (*Passerculus sandwichensis*; Fletcher and Koford 2002). In Kansas, restored prairies have fewer bird species and lower diversity than haylands and pastures due to dominance of a few bird species (Rahmig et al. 2008). Although bird response to local restoration efforts vary, heterogeneity of vegetation over landscapes is also a key driver in grassland bird communities (Hovick et al. 2015).

Small mammals, which play an important role in grassland community structure and functioning (Burke et al. 2020), have been found to initially respond negatively to large prairie restoration efforts due to alteration of soil and vegetation, followed by recoveries (Stone 2007). Higher forb diversity and frequent prescribed fire in restored prairies are likely to increase small mammal abundance (Glass and Eicholz 2021). Responses are likely to vary by species, though, as voles tend to associated with grass-dominated areas (Howe and Lane 2004) and mice prefer recently-disturbed patches with high production of forb seeds (Matlack and Kaufman 2001).

Reptile, primarily snakes in the Midwestern U.S., response to prairie restoration is understudied, compounded by a lack of information on natural history (Dodd 1993). A limited body of research indicates that snakes respond positively to increases in local woody cover in restored prairies (Martino et al. 2012, Glass and Eicholz 2022). Woody cover allows for easier predator avoidance and thermoregulation (Webb and Shine 1997). Forb cover and diversity may also influence snake communities in prairies, due to changes in prey communities. Larger bodied snakes such as kingsnakes and ratsnakes rely on rodents as prey (Jenkins et al. 2001, Trauth and McAllister 1995), while smaller snakes like garters prey on insects (Durso et al. 2021).

Wild bee communities are often a central focus of prairie restoration. Compared to unrestored areas, the diversity of the wild bee community has been found to increase with prairie restoration (Tonietto and Larkin 2018, Kordbacheh et al. 2020, Sexton and Emery 2020). Forb coverage and diversity is a primary concern of most restoration practices for wild bees. Floral communities are important drivers of bee community dynamics (Biesmeijer et al. 2006, Lane et al. 2020) and floral community differences are linked to bee community composition (Denning and Foster 2018). High diversity seed mixes may result in increased wild bee diversity and abundance.

We present initial, Year 1 through 3 (2018-2020), results from the Ruckman Farm Diversity Experiment, which is located in northwest Missouri, USA. Our goal with this experiment was to understand biodiversity response to prairie reconstruction with different levels of seed mix diversity, with the broader goal of informing more extensive prairie reconstruction and ecological restoration efforts within the study region. Prairie reconstruction in the region is currently pursued by multiple public (e.g., Missouri Department of Conservation, U.S. Department of Agriculture Natural Resources Conservation Service, U.S. Fish and Wildlife Service), non-profit (e.g., Environmental Defense Fund, Missouri Prairie Foundation, The Nature Conservancy), and private organizations (Roeslein Alternative Energy, Smithfield Foods), working independently or in partnership, and also by private individuals. In particular, a new project called Horizon II, led by Roeslein Alternative Energy and funded in part by the USDA Climate-smart Commodities Partnership Program, seeks to dramatically expand prairie reconstruction in the Midwestern region by 12 million ha by 2050 to address joint concerns about greenhouse gas emissions, soil loss, water quality degradation, flooding, and biodiversity loss associated with agriculture in the region. While, in the short-term, prairie reconstruction through this project will be funded by a federal grant, the intent in the long-term is to establish biofuel and ecosystem service markets that, when financial incentives are layered, can complement existing agricultural markets for agricultural commodities (e.g., corn, soybean, beef, swine). Based on prior ecological research, we hypothesized that biodiversity measures would be higher in fields restored to native vegetation than unrestored fields and increase with plant diversity. Specifically, we predicted species richness and measures of abundance of wildlife taxa would be higher in prairie treatments compared to the control, the high diversity treatment compared to the low diversity treatment, and would increase over time in the prairie treatments. In addition to informing future prairie reconstructions, data from the Ruckman Farm Diversity Experiment will be used to inform landscape and watershed modeling efforts, such as presented in Audia et al. (2022).

Methods

Study Area

The Ruckman Farm Diversity Experiment was established at a swine farm owned by Smithfield Foods, located 15 km north of the city of Albany in Gentry County, Missouri, USA. The region has a temperate climate with an average annual temperature range of -5 −32 ^{0}C and 96 cm average annual precipitation (U.S. Climate Data 2022). Precipitation typically falls April – October. The topography is undulating, with elevations ranging 201 − 298 m above sea level. Soils have loam surface layers with dense subsoils that are primarily clay loam. In addition to infrastructure required for swine production, the site also included large tracts of exotic cool-season grass hayfields and pastures dominated by tall fescue (*Festuca arundinacea*) and smooth brome (*Bromus inermis*), as well as scattered woodlands in riparian areas or on steeper slopes (9-14%). Grass fields were regularly used for cattle grazing and swine manure application on a near annual basis prior to establishing the experiment. The surrounding landscape was similar in vegetation composition and also included occasional row crop fields.

Experimental Design

We selected 20 distinct fields, between 1.3 ha and 7.8 ha in size, with different management histories for prairie restoration and monitoring. We used a randomized complete block study design to split fields into three blocks of similar spatial proximity and historical management (Fig. 5-1a). We then split each block into two sets of fields for planned every-other-year harvest in the future. We used seed mix diversity as a random split-factor to determine the treatment type for each field (Fig. 5-1b). Eight treatment field were seeded to a "high diversity" mix of 31 native grass and forb species, and six fields were seeded to a "low diversity" mix of 15 grass, forb, and legume species (Table 5-1, Table 5-2). The remaining six fields were left in the existing exotic cool-season grasses to serve as control fields for comparison.

Prairie Restoration

We worked with Roeslein Alternative Energy, a land restoration company, to established the experimental prairie vegetation on 14 treatment fields. The pre-existing fescue-brome mix in treatment fields was terminated in October 2017 when fields were sprayed with glyphosate herbicide and disked to even out the topography and prepare for planting (Table A-1). From February 12 – 16, 2018, treatment fields were seeded to native prairie species using a Great Plains seed drill (Great Plains Ag, Salina, Kansas) according to the experimental design. Seeds were purchased from Pure Air Natives, Inc., Wentzville, Missouri. Seed mixtures were designed based on multiple criteria: likelihood of having more than one forb species in bloom throughout the growing season, likelihood of establishment success based on the experience of local restoration professionals, availability, and cost (Table 5-1, Table 5-2). Post-restoration management included strategic mowing and frequent spot spraying of noxious weeds (Table A-1), including Canada Thistle (*Cirsium arvense*) and Wild Parsnip (*Pastinaca sativa*).

Biodiversity Monitoring

We surveyed multi-taxa response, including plant, pollinator, snake, small mammal, and breeding bird, between May and August, 2018-2020, in control and treatment fields. We used the Daubenmire method (Daubenmire 1959, Hirsch et al. 2003) to determine the composition of plant communities. We generated 12 random points in each field to place Daubenmire quadrats during the first week of August of each year. In each quadrat, we measured vegetation height, species composition and percent coverage, and noted the number of flowering forb and milkweed individuals. In total, we conducted 648 plant surveys. We identified unknown plant species using Bryson and DeFelice (2010).

To examine the pollinator community response to restoration, we conducted 24-hour bee bowl surveys (Droege 2012, Gill and O'Neal 2015). At five random locations within each field, we deployed fluorescent bee bowls filled with soapy water once per month, June – August, 2018 – 2020. We did not deploy bee bowls during periods of rainfall or wind speeds exceeding 16 kmh. After collecting bowls, we used morphological characteristics to identify all wild bees under a microscope. Bees were identified to the lowest taxonomic unit possible, and stored on Iowa State University campus for later confirmation of identification. We identified specimens using the Discover Life Key (Ascher and Pickering 2012). A representative specimen of each taxonomic unit was archived in the collections of the Department of Plant Pathology, Entomology, and Microbiology at Iowa State University. In total, we completed 680 bee bowl station-days.

We conducted coverboard surveys to monitor the snake and small mammal response to restoration (Grant et al. 1992, Joppa et al. 2010). In April of 2018, we randomly placed 201 plywood coverboards across study fields. From May – August, 2018 – 2020, we flipped each board twice per month and identified any snake or small mammal underneath. We confirmed species identification using various field guides (Reid 2006, Powell et al. 2016). In total, we conducted 3,961 coverboard surveys.

To investigate bird community response, we conducted 5-min bird point-count (BPC) surveys three times each year with distance sampling at randomly-generated locations in each field (Buckland et al. 2001, Rosenstock et al. 2002). BPC survey locations were unchanged throughout the study period. We did not conduct BPC surveys in fields 13 and 14 (Fig. 5-1) because we were unable to survey a 100-m radius point count station within these fields without substantial overlap of adjacent woody areas. The number of point-count stations per field varied between one and three depending on field size and shape. Stations within fields were a minimum of 100m apart. After arriving at a station, the observer remained stationary and silent for 2 min to allow birds to resume natural behavior. The observer then identified species, sex, and age (juvenile or adult) to each bird seen or heard during a 5-min survey period. Using a laser rangefinder, the observer also estimated the perpendicular distance to each individual bird detected. Exact distance estimations were not made for birds greater than 100 m from the observer. Surveys were not conducted during periods of rainfall or wind speeds exceeding 16 kmh (Manuwal and Carey 1991, Mikol 1980). Air temperature, wind speed, and percent cloud cover were recorded before and after surveys.

In 2020, we discontinued all monitoring in fields 12-14 (Fig. 5-1) due to frequent management in attempts to control thistle and red clover (*Trifolium pratense*) invasions.

Statistical Methods

Data were checked for quality assurance prior to statistical analyses. We standardized all dependent variables. We used a Shapiro-Wilk test of normality and conducted ANOVA to test for differences in the native plant communities of each treatment. We conducted two-way ANOVA to compare the effects of year and treatment on pollinator species richness and abundance, snake and small mammal detection rates, bird abundance and richness. We followed each ANOVA with a Tukey HSD to examine pairwise differences between significant independent variables. All statistical analyses were performed in R 4.2.2 (R Development Core Team 2022).

For analysis of data on bird response, we used distance sampling models to estimate detection probability and abundance of the grassland community and individual species (Buckland et al. 2001). We used only territorial male detections (Buckland et al. 2001, Newson et al. 2008). We sorted distances into 20-m bins from 0 to 100 m to remove potential bias of estimating distances (Buckland et al. 2001). We removed all detections beyond 100 m from abundance and richness analyses due to unreliable detection beyond that distance. We used package "Distance" (Miller et al. 2019) to evaluate the fit of the hazard rate, half-normal, and uniform key functions with and without cosine adjustments. Detection functions use the fall-off in detections as distance away from the observer increases to model detection probability (Buckland et al. 2001). We evaluated time of day, temperature, wind speed, and cloud cover as covariates to model heterogeneity in detection probabilities. We used an Akaike Information Criterion (AIC hereafter) framework and goodness-of-fit tests to determine the most appropriate detection probability model (Burnham et al. 2009).

Results

Plants

The plant community differed significantly among treatment types and years (Table 5-3). Both grass and forb/legume cover were significantly different between control and treatment fields in all years (Table 5-4); furthermore, the cover of native species was consistently higher for the diversity treatments and increased over time (Fig. 5-2, Fig. 5-3). There were no significant differences in plant species richness (Fig. 5-3) or plant cover by plant functional (Table 5-4) group between low diversity and high diversity treatments in any years.

Vegetation in control fields was relatively stable throughout three years of monitoring, consisting primarily of fescue and brome grasses with lesser amounts of smartweed (*Persicaria spp.*), horse nettle (*Solanum carolinense*), and wild lettuce (*Lactuca virosa*) (Fig. 5-2; Table A-2). Vegetation in both low and high diversity treatments shifted significantly among years, transitioning from annual forb species in 2018 toward dominance by native perennial species in 2019 and 2020 (Fig. 5-2; Table 5-3). In low diversity fields, native grass coverage increased from 19% in 2018 to 40% in 2020 and forb coverage increased from 39% in 2018 to 46% in 2020. By 2020, the five most common species were common ragweed (*Ambrosia artemisiifolia*), switchgrass (*Panicum virgatum*), wild bergamot (*Monarda fistulosa*), showy partridge pea (*Chamaecrista fasciculata*), and foxtail (*Setaria spp.*; Table A-2). In high diversity fields, native grass cover increased from 17% in 2018 to 39% in 2020 and forb cover increased from 44% in 2018 to 50% in 2020. By 2020, the five most common species were switchgrass, common ragweed, pale purple coneflower (*Echinacea pallida*), false sunflower (*Heliopsis* helianthoides), and showy partridge pea (Table A-2). The following 12 plant species were never observed during surveys within the first three years of evaluation despite their inclusion in seed mixes: blue wild indigo, butterfly milkweed, common mountain mint, Indiangrass, leadplant, New England aster, purple prairie clover, showy tick trefoil, stiff goldenrod, Virginia wildrye, white prairie clover, and yellow wingstem.

Bees

We collected 4,728 individual bees of at least 71 unique taxa across the three years of study (Table A-3). Seven taxa comprised over 90 percent of the sample: *Lasioglossum spp.* (58.3%; specimens of the genus *Lasioglossum* could not be identified to the species level using morphological characteristics), *Augochlorella aurata* (8.8%), *Halictus ligatus* (8.2%), *Agapostemon texanus* (5.0%), *Agapostemon virescens* (4.6%), *Melissodes bimaculatus* (3.7%), and *Augochlora pura* (2.6%). Year had a significant effect on both bee species richness and bee abundance (Table 5-3). Treatment alone did not have a significant effect but the interaction between year and treatment was significant, with greater diversity and abundance across study years (Table 5-3). Despite significant differences, there were no clear trends in bee response to this experiment (Fig. 5-3).

Some species were not found in any treatment types. *Andrena erythronii*, *Ceratina dulpa*, *Eucera hamata*, *halictus tripartitis, Megachile parallela, Melissodes denticulatus Melissodes subillatus,* and *Triepeolus cressonii* were only found in the high diversity treatments; *Andrena commoda*, *A. geranii, A. nivalis, Colletes latitarsis, Hoplitis spoliata* and *Ptilothrix bombiformis* were only found in the low diversity treatments; *Ceratina calcarata*, *C. mikmaqi*, *Halictus rubicundus*, and *Melissodes boltoniae* were found in the diversity treatments but not the controls; and *Agapostemon sericeus, Andrena barbara*, *A. wilmattae, Megachile frugalis, Melissodes menuachus, M. niveus, Osmia lignaria, Peponapis pruinosa,* and *Sphecodes pimpinellae* were only found in the control fields.

Snakes

We detected 699 snakes of nine species during coverboard surveys, including three Missouri state-listed SGCNs (MODOC 2015): Great Plains ratsnake (*Elaphe guttata emoryi*), lined snake (*Tropidoclonion lineatum*), and plains gartersnake (*Thamnophis radix*; Table A-4). We captured one eastern yellow-bellied racer (*Coluber constrictor flaviventris*) and one Great plains ratsnake during the study period. Detection rates for snakes, including different species, increased significantly over time (Fig. 5-3; Table 5-3). Species richness was marginally higher (*p*-value = 0.068) in diversity treatments compared to the control, but the diversity treatments did not differ from each other (*p*-value = 0.491; Fig. 5-3).

Small Mammals

We detected 879 small mammals of six taxa during coverboard surveys (Table A-5). All taxa are commonly found in a variety of habitats and none of the species are Missouri SGCNs (MODOC 2015). Mammal species richness varied by year, and there was also a significant year by treatment interaction, but no difference in detection rates (Table 5-3). Mice (*Peromyscus* spp.) were the most common mammalian taxa detected. In treatment fields, detection rates of mice decreased each year (Table A-5). The overall pattern of response of the small mammal community was ambiguous (Fig. 5-3).

Birds

We made 5,088 detections of 67 bird species (Table A-6), 14 of which we considered grassland species (Peterjohn and Sauer 1993) and 11 are Missouri SGCNs (MODOC 2015). The most frequently detected species included Red-winged Blackbirds (*Agelaius phoeniceus*; 15.2% of detections), Dickcissels (*Spiza americana;* 11.9%), Common Yellowthroats (*Geothlypis tricas*; 7.9%), Brown-headed Cowbirds (*Molothrus ater*; 6.8%), and Eastern Meadowlark (*Sturnella magna*; 5.4%). We found a significant effect of year for both bird species richness and abundance, and a significant effect for abundance by treatment (Table 5-3). Among pairwise comparisons, bird abundance in low diversity fields was significantly greater than control fields

(p-value = 0.029) but not high diversity fields (p-value = 0.129; Fig. 5-3).

Grassland birds collectively comprised 47.6% of all breeding bird detections, and species richness differed by year and their abundance differed across treatments (Fig. 5-3; Table 5-3). Pairwise comparisons among treatments revealed a statistically significant difference in grassland bird abundance between low diversity (8.53) and control fields (6.94; $p<0.05$) but not high diversity (7.21) and control fields. Among grassland species, Red-winged Blackbird and Dickcissel were more abundant in diversity treatment fields than control fields (Table A-6). Area sensitive species, such as the Grasshopper Sparrow, showed no trends toward any treatment (Table A-6).

Discussion

In our study, native plant cover increased through time in diversity treatment fields while exotic plant cover decreased, as expected. We documented increasing native grass, forb, and legume cover in both low diversity and high diversity prairie restoration treatments across the first three years of experimentation. Most notably among differences was higher native forb and legume cover in high diversity fields than low diversity fields. Meissen et al. (2019) found similar trends in plant community composition when comparing three seed mix types in Iowa. We conducted vegetation surveys in August, which may have led to a bias toward late-blooming species in our data. Though we included early-blooming species in our seed mixes, previous research has attributed underrepresentation of early phenology species in restored prairies to use of seed from fall bulk seed harvests (Carter and Blair 2012).

Our hypothesis of higher wild bee richness and abundance in diversity treatments was not supported (Fig. 5-3, Table 5-3). We collected a significantly higher number of bee specimens in 2018 than 2019 and 2020. We suspect lower forb cover in 2018 led to higher conspicuousness of bee bowls, and thus greater effectiveness in capturing bees. In 2019 and 2020 as forb cover increased in diversity treatments, and we qualitatively observed that bee bowls became less attractive due to other nearby foraging opportunities. While a strong relationship between bee community measures and forb cover did not hold for this study as in others (Kwaiser and Hendrix 2007, Hopwood 2008, Kordbacheh et al. 2020), this may have been due to the timing of our vegetation sampling, which only occurred in August of each year and was not coincident with earlier bee bowl surveys. Previous studies have found a strong relationship between bee and forb communities in prairies when sampling was conducted simultaneously (Kwaiser and Hendrix 2007, Hopwood 2008).

Our hypothesis of higher snake species richness and abundance in diversity treatment fields was not supported. We recorded a strong effect for year for both snake species richness and detection rate (Table 5-3), and a marginally significant effect of treatment on species richness, with a trend toward higher in the high diversity treatment (Fig. 5-3). A majority of our snake species detections were common or plains garter snakes, with diets primarily composted of insects. Glass and Eicholz (2022) found a negative relationship between forb cover and snake abundance. They encountered more large-bodied snakes than we did; thus, the negative relationship may have been due to the tendency of voles to associate with grass-dominated areas. We also observed richer and more abundant snake communities in 2019 and 2020 than 2018 (Fig. 5-3), which we expect was due to a lag in snake use of cover boards after deployment. A similar response to coverboard age was found in salamanders (Hesed 2012).

Our hypotheses of higher small mammal detection rates in diversity treatments and across time were not supported. While there was a significant effect of year on species richness (Table 5-3), our overall results were ambiguous (Fig. 5-3). After 2018, small mammal richness and abundance were similar or higher in control fields than diversity treatments. Previous work on the relationship between plant richness and small mammal abundance provided mixed results but Arlettaz et al. (2010) found higher abundance in wildflower strips than other cover types. Variance in small mammal communities likely occurs at much larger spatial scales than in our study. Glass and Eichholz (2021) found small mammal abundance in restored prairies in Illinois was largely governed by differences in habitat structure at the landscape scale. Small mammal communities were relatively stable in diversity treatments in our study. Contrarily, Stone (2007) found that small mammal use of prairies decreased after initial restoration practices due to alteration of soil and vegetation and then recovered three to five years post-restoration.

Our hypothesis of higher grassland bird abundance in diversity treatment was supported but only for low diversity fields compared to control (Fig. 5-3). Contrary to expectation, however, we did not find an effect for bird species richness. The effect of year was significant of both metrics. Red-winged Blackbirds and Dickcissels were the most prominent grassland species in all treatment types. Both had higher abundance in diversity treatment fields than control fields, likely a result of increased structure provided by prairie plants compared to fescue-brome vegetation in control fields. We attribute high variation of Red-winged Blackbird abundance estimates in low diversity fields to the tendency of low diversity fields to be close to water at our study site. Red-winged Blackbirds frequently associate with surface water and wetlands (Yasukawa and Searcy 2020). Dickcissels are obligate grassland specialists and prefer open grasslands with dense cover (Sousa et al. 2022), such as those found in treatment fields.

Our findings indicate a potential lag in biodiversity response to restoration of native vegetation in agricultural landscapes. Overall, we observed a strong response of the plant community to prairie restoration, but ambiguous to negligible differences in the response of multiple wildlife taxa to the initial establishment of native vegetation. Contrary to our expectations based on ecological theory and the published literature, as reviewed in the introduction, increases in native grass and forb cover over the initial years following establishment did not result in a clear increases in wild bee, snake, small mammal, or bird species richness or abundance. We expect wildlife communities will exhibit stronger associations with diversity treatments in subsequent years as native vegetation becomes more fully established. The significance of the year effect across many response measures, and ambiguity in the pattern of response by treatment indicate that a longer-term period of data collection is needed.

Acknowledgements

This project was funded by the U.S. Fish and Wildlife Service under Cooperative Agreement Award F16AC00155, with additional financial assistance from the Department of Natural Resource Ecology and Management at Iowa State University. Smithfield Foods and Roeslein Alternative Energy respectively hosted the experiment and established the native prairie vegetation. We thank Dr. Jarad Niemi for assisting with the experimental design and Dr. Matt O'Neal for informing bee sampling design. We thank Dakota McKune, Drake Fehring, and Mary Kate Shaver for their assistance in the field, and Kate Borchardt for assistance with wild bee identification.

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Figures and Tables

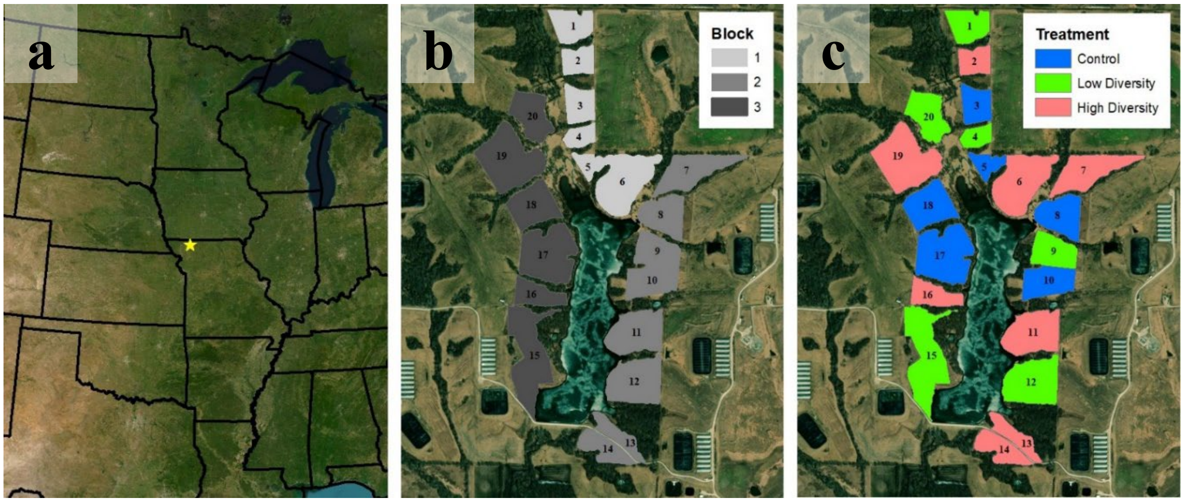

Figure 5-1. The Ruckman Farm Diversity Experiment is (a) located within the U.S. Midwest. (b) 20 fields were grouped into three blocks based on spatial proximity and historical management. Fields surround a 21-ha lake. (c) Treatment type was randomized within three blocks.

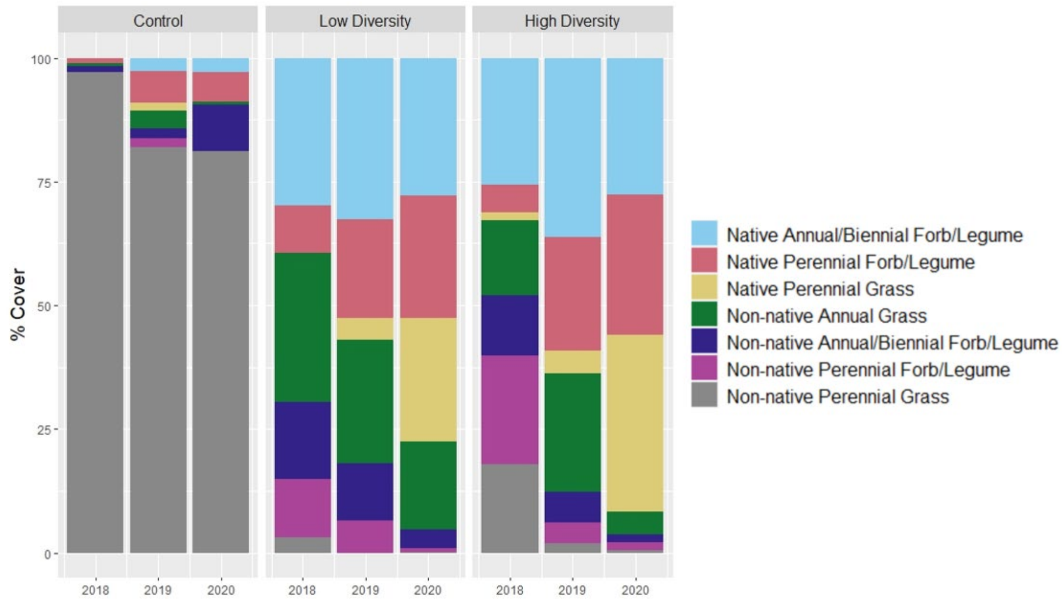

Figure 5-2: Plant community composition of control, low diversity, and high diversity fields in the Ruckman Farm Diversity Experiment in northwest Missouri, USA, 2018-2020.

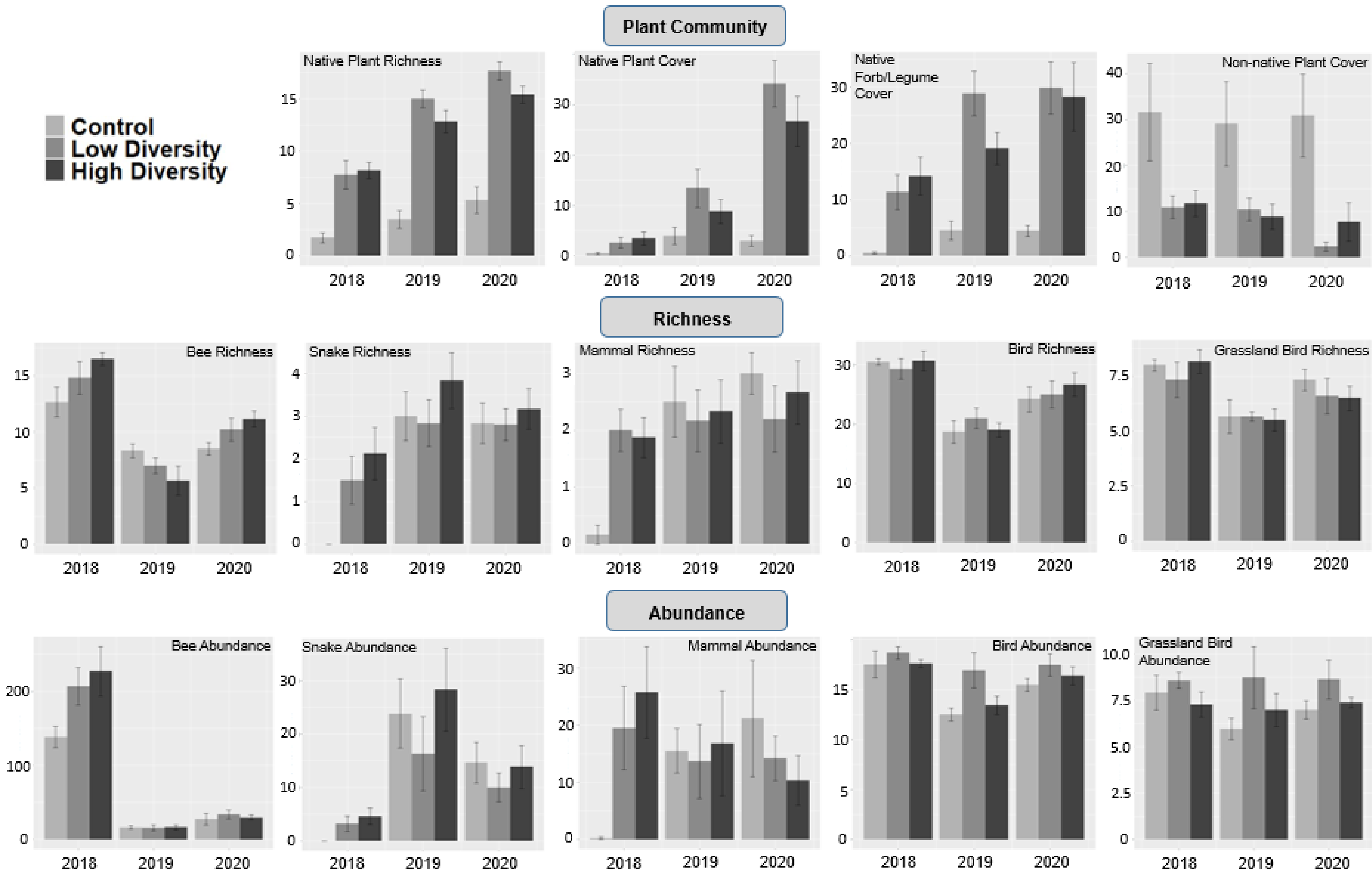

Figure 5-3. Treatment effects on measures of biodiversity in the Ruckman Farm Diversity Experiment, 2018-2020. For measures of biodiversity, bars depict means over a sampling period. Letters denote statistical differences among treatments. Error bars show standard errors.

Table 5-1. Forb and legume species composition of low and high diversity prairie seed mixes used in the Ruckman Farm Diversity Experiment in northwest Missouri, USA.

Common Name	Scientific Name	Planting Rate							
		Low Diversity Mix				High Diversity Mix			
		seeds/g	g/m^2	% by weight	% by seed	seeds/g	g/m^2	% by weight	% by seed
Black-eyed Susan	*Rudbeckia hirta*	3,880	0.020	6%	33%	3,880	0.007	2%	12%
Blue Vervain	*Verbena hastata*	4,410	0.003	1%	6%	4,410	0.003	1%	7%
Blue Wild Indigo	*Baptista australis*	-	-	-	-	1,600	0.008	2%	<1%
Butterfly Milkweed	*Asclepias tuberosa*	-	-	-	-	3,500	0.007	2%	<1%
Common Evening Primrose	*Oenothera biennis*	-	-	-	-	55,000	0.003	1%	3%
Common Milkweed	*Asclepias syriaca*	140	0.007	2%	<1%	140	0.007	2%	<1%
Common Mountain Mint	*Pycnanthemum virginianum*	11,685	0.002	1%	8%	11,685	0.002	<1%	4%
Compassplant	*Silphium lacinatum*	230	0.007	2%	<1%	230	0.012	4%	<1%
False Sunflower	*Heliposis helianthoides*	230	0.042	13%	4%	230	0.038	10%	4%
Foxglove Beard Tongue	*Penstemon digitalis*	4,056	0.007	2%	12%	4056	0.003	1%	6%
Golden Alexander	*Zizia aurea*	423	0.003	1%	1%	423	0.003	1%	1%
Gray Goldenrod	*Solidago nemoralis*	-	-	-	-	8,465	0.002	1%	6%
Gray-headed Coneflower	*Ratibida pinnata*	890	0.007	2%	3%	890	0.007	2%	3%
Illinois Bundleflower	*Desmanthus illinoensis*	-	-	-	-	4,888	0.028	8%	2%
Lanceleaf Coreopsis	*Coreopsis lanceolota*	440	0.002	16%	10%	440	0.021	6%	4%
Lead Plant	*Amorpha canescens*	-	-	-	-	600	0.014	4%	4%

Table 5-1. Continued.

Common Name	Scientific Name	Planting Rate							
		Low Diversity Mix				High Diversity Mix			
		seeds/g	g/m^2	% by weight	% by seed	seeds/g	g/m^2	% by weight	% by seed
New England Aster	*Aster novaeangliae*	2,680	0.034	1%	4%	2,680	0.002	1%	2%
Pale Purple Coneflower	*Echinacea pallida*	-	-	-	-	176	0.007	2%	1%
Partridge Pea	*Cassia fasciculate*	134	0.004	33%	6%	134	0.049	15%	3%
Plains Coreopsis	*Coreopsis tinctoria*	-	-	-	-	3,086	0.007	2%	9%
Purple Coneflower	*Echinacea purpurea*	232	0.042	13%	4%	232	0.028	8%	3%
Purple Prairie Clover	*Dalea purpurea*	-	-	-	-	705	0.014	4%	4%
Rosinweed	*Silphium integrifolium*	140	0.017	5%	1%	140	0.018	5%	1%
Showy Tick Trefoil	*Desmodium canadense*	-	-	-	-	158	0.007	2%	<1%
Stiff Goldenrod	*Solidago rigida*	-	-	-	-	1,622	0.003	1%	2%
Sweet Black-eyed Susan	*Rudbeckia subtomentosa*	-	-	-	-	1,622	0.007	2%	5%
Western Ironweed	*Vernonia baldwinii*	-	-	-	-	846	0.007	2%	3%
White Prairie Clover	*Dalea candida*	-	-	-	-	925	0.007	3%	4%
Wild Bergamot	*Monarda fistulosa*	2,750	0.007	2%	8%	2,750	0.035	1%	4%
Yellow Wingstem	*Verbesina helanthoides*	-	-	-	-	494	0.011	3%	2%

Table 5-2. Grass species composition of low and high diversity prairie seed mixes used in the Ruckman Farm Diversity Experiment in northwest Missouri, USA.

Common Name	*Scientific Name*	g/m^2
Big Bluestem	*Andropogon gerardii*	0.112
Canada Wildrye	*Elymus canadensis*	0.056
Indiangrass	*Sorghastrum nutans*	0.112
Little Bluestem	*Schizachyrium scoparium*	0.084
Switchgrass	*Panicum virgatum*	0.084
Virginia Wildrye	*Elymus virginicus*	0.056

Table 5-3. Two-way analysis of variance results for effect of treatment and year on multiple biodiversity response variables.

	Species Richness					Abundance Measure*				
	df	Sum Sq.	Mean Sq.	*F*-value	*p*	df	Sum Sq.	Mean Sq.	*F*-value	*p*
Plants: All										
Treatment	2	2018.7	1009.4	61.521	9.99e^{-14}	[redacted]				
Year	2	384.7	192.4	11.724	7.68e^{-05}					
Treatment*Year	4	52.8	13.2	0.805	0.528					
Residuals	46	754.7	16.4							
Plants: Native										
Treatment	2	1213.7	606.9	78.584	1.45e^{-15}	2	6675	3338	21.314	2.81e^{-07}
Year	2	621.2	310.6	40.218	7.95e^{-11}	2	14036	7018	44.816	1.58de^{-11}
Treatment*Year	4	75.8	18.9	2.453	0.591	4	6220	1555	9.931	7.06e^{-06}
Residuals	46	355.2	7.7			46	7203	157		
Plants: Forbs and Legumes										
Treatment	2	1745.5	872.7	58.377	2.39e^{-13}	2	5357	2678.5	19.713	6.56e^{-07}
Year	2	284.3	142.1	9.507	3.50 $^{e-04}$	2	558	279.1	2.054	0.140
Treatment*Year	4	97.9	24.5	1.637	0.181	4	167	41.7	0.307	0.872
Residuals	46	687.7	15.0			46	6250	135.9		
Bees										
Treatment	2	31.3	15.67	2.817	0.070	2	18269	9135	4.581	0.0153
Year	2	587.6	293.80	42.800	1.22e^{-12}	2	370209	185104	92.82	2.0e^{-16}
Treatment*Year	4	75.0	18.75	3.370	0.017	4	18075	4519	2.266	0.0765
Residuals	46	242.0	5.76			46	91734	1994		
Snakes										
Treatment	2	9.75	4.875	2.857	0.068	2	220	110.1	0.877	0.423
Year	2	43.83	21.916	12.841	3.71^{e-05}	2	3963	1981.4	15.783	6.04e^{-06}
Treatment*Year	4	7.44	1.859	1.089	0.373	4	264	65.9	0.525	0.718
Residuals	46	78.51	1.707			46	5775	125.5		
Mammals										
Treatment	2	1.25	0.626	0.469	0.629	2	362	181.2	0.614	0.545
Year	2	16.41	8.205	6.136	0.004	2	35	17.5	0.059	0.942
Treatment*Year	4	13.37	3.344	2.501	0.055	4	2135	533.7	1.810	0.143
Residuals	46	61.51	1.337			46	13566	294.9		
Birds: All										
Treatment	2	9.3	4.7	0.264	0.769	2	60.25	30.13	5.018	0.012
Year	2	1015.6	507.8	28.657	1.07^{e-08}	2	120.61	60.31	10.045	2.00 $^{e-04}$
Treatment*Year	4	35.4	8.8	0.499	0.736	4	19.69	4.92	0.820	0.519
Residuals	44	779.7	17.7			44	264.16	6.00		
Birds: Grassland										
Treatment	2	1.96	0.982	0.501	0.610	2	28.67	14.336	3.232	0.049
Year	2	44.53	22.267	11.344	1.06^{e-04}	2	4.53	2.263	0.510	0.604
Treatment*Year	4	2.95	0.736	0.375	0.825	4	7.51	1.877	0.423	0.791
Residuals	44	86.37	1.963			44	195.15	4.435		

* The specific measure of abundance varies by taxon: percent cover for plants; number of individuals collected in each field for wild bees; detection rate (number of detections per 100 cover board flips) for snakes and mice; and number of birds detected per survey for birds (see Methods for details).

Table 5-4. Contrasts among treatments and years of the percent cover of grasses and forbs and legumes. LD = low diversity, HD = high diversity, and CI = confidence interval.

		Grasses		Forbs and Legumes	
Year	Treatment Comparison	Mean (95% CI)	*p*	Mean (95% CI)	*p*
2018	LD - Control	-36.86 (-47.99, -25.73)	0.00	34.85 (23.63,46.06)	0.00
2018	HD - Control	-39.36 (-49.77, -28.95)	0.00	39.98 (29.49, 50.46)	0.00
2018	HD - LD	-2.50 (-12.91, 7.91)	0.99	5.13 (-5.36, 15.62)	0.84
2019	LD - Control	-45.48 (-57.22, -33.76)	0.00	37.72 (25.90, 49.53)	0.00
2019	HD - Control	-47.72 (-58.86, -36.59)	0.00	38.86 (27.65, 50.07)	0.00
2019	HD - LD	-2.24 (-13.97, 9.49)	0.99	1.14 (-10.67, 12.96)	0.99
2020	LD - Control	-31.21 (-42.89, -19.53)	0.00	27.28 (15.51, 39.04)	0.00
2020	HD - Control	-31.50 (-42.63, -20.37)	0.00	31.39 (20.18, 42.60)	0.00
2020	HD - LD	-0.29 (-11.97, 11.39)	1.00	4.11 (-7.65, 15.87)	0.97

Table 5-5. Contrasts among treatments and years for wild bee species richness and abundance. LD = low diversity, HD = high diversity, and CI = confidence interval.

		Bee species richness		Bee abundance	
Year	Treatment Comparison	Mean (95% CI)	*p*	Mean (95% CI)	*p*
2018	LD - Control	2.17 (-2.26, 6.59)	0.80	68.67 (-15.22-152.56)	0.2
2018	HD - Control	3.83 (-0.31, 7.98)	0.09	89.04 (10.57-167.51)	0.01
2018	HD - LD	1.67 (-2.48, 5.81)	0.92	20.38 (-58.10-98.85)	0.99
2019	LD - Control	-1.33 (-5.76, 3.09)	0.98	-0.67 (-84.55-83.22)	1.00
2019	HD - Control	-2.67 (-7.09, 1.76)	0.57	-1.67 (-85.55-82.22)	1.00
2019	HD - LD	1.33 (-3.09, 5.65)	0.98	-1.00 (-84.89-82.89)	1.00
2020	LD - Control	1.70 (-2.95, 6.35)	0.95	6.30 (-81.68-94.28)	0.99
2020	HD - Control	2.67 (-1.76, 7.09)	0.57	2.33 (-81.56-86.22)	1.00
2020	HD - LD	0.97 (-3.68, 5.61)	0.99	-3.97 (-91.95-84.02)	1.00

Appendix. Supplemental Tables.

Table A-1. Management log for the Ruckman Farm Diversity Experiment located in northwest Missouri, USA, 2017-2020. Date are presented in international format.

Date	Management activity			
		Control	Low Diversity	High Diversity
2016-08-15	Hayed all experimental fields	X	X	X
2017-04-17	Applied herbicide (4.8 L/ha Glyphosate + 0.29 L/ha Imazapic + Cornbelt surfactant) to treatment fields		X	X
2017-08-17	Mowed all treatment fields		X	X
2017-09-25	Applied herbicide (3.5 L/ha Glyphosate + 1 pint/acre 2,4-D + ammonium sulfate) to treatment fields		X	X
2018-02-02	Seeded native species using Great Plains seed drills (Great Plains Ag, Salina, Kansas) with planting width of either 3.1 or 4.6 and depth set to 0.64 cm		X	X
2018-07-19	Mowed all treatment fields to 25 cm height		X	X
2019-05-15	Mowed all experimental fields to 25 cm height	X	X	X
2019-06-17	Mowed select treatment fields to control non-native species		X	X

Table A-2. Life-history group (LHG) and mean percent cover ± standard deviation of plant species by treatment using the Daubenmeier survey method at the Ruckman Farm Diversity Experiment located in northwest Missouri, USA, 2017-2020. LHGs include NPG = native perennial grass, NAG = native annual grass, XPG = non-native perennial grass, XAG = nonnative annual grass, NPF = native perennial forb, NBF = native biennial forb, NAF = native annual forb, XPF = non-native perennial forb, XBF = non-native biennial forb, XAF = non-native annual, NAL = Native Annual Legume, XPL = non-native perennial legume, XAL = non-native annual legume.

Common name	Scientific name	LHG	Mean percent cover ± standard deviation by treatment								
			Control			Low Diversity			High Diversity		
			2018	2019	2020	2018	2019	2020	2018	2019	2020
Alfalfa	*Medicago sativa*	XPL	-	-	-	0.07 (0.59)	-	-	1.84 (7.29)	-	-
Barnyardgrass	*Echinochloa crus-galli*	XAG	-	-	-	5.49 (17.20)	1.74 (7.32)	-	1.82 (8.37)	0.62 (3.02)	0.07 (0.59)
Big Bluestem	*Andropogon gerardi*	NPG	-	-	-	-	1.88 (4.85)	2.92 (8.98)	-	1.81 (6.07)	5.92 (13.59)
Birdsfoot Trefoil	*Lotus corniculatus*	XPF	-	-	0.07 (0.59)	0.56 (2.73)	1.06 (8.96)	0.33 (2.58)	4.24 (10.81)	0.35 (1.75)	0.97 (5.35)
Black Medic	*Medicago lupulina*	XPF	-	-	-	0.35 (1.53)	-	-	1.51 (7.98)	-	0.07 (0.59)
Blackeyed-Susan	*Rudbeckia hirta*	NBF	-	-	0.07 (0.59)	9.47 (16.44)	3.07 (6.68)	2.50 (10.58)	2.66 (5.98)	2.99 (6.03)	1.86 (6.16)
Blue Vervain	*Verbena hastata*	NPF	-	-	-	-	1.67 (6.77)	-	-	1.25 (7.16)	-
Broadleaf Plaintain	*Plantago major*	XPF	-	-	-	-	-	-	-	-	0.14 (1.18)
Brome spp.	*Bromus*	XPG	52.61 (23.84)	67.01 (25.20)	32.01 (31.59)	-	-	-	0.10 (1.02)	-	-
Bull Thistle	*Cirsium vulgare*	XBF	-	0.56 (3.31)	-	3.89 (15.52)	-	-	1.20 (10.25)	-	-
Burdock spp.	*Arctium*	XBF	-	-	-	-	0.49 (2.93)	-	-	-	-
Bush Clover spp.	*Lespedeza*	-	-	-	-	-	-	-	2.14 (9.02)	-	-
Canada Goldenrod	*Solidago canadensis*	NPF	0.07 (0.59)	-	0.62 (3.75)	-	0.49 (3.58)	0.50 (2.87)	-	-	-
Canada Thistle	*Cirsium arvense*	XPF	-	1.46 (5.40)	-	-	0.83 (4.11)	-	0.31 (3.06)	-	-

Table A-2. Continued.

Common name	Scientific name	LHG	Mean percent cover ± standard deviation by treatment								
			Control			Control			Control		
			2018	2018	2018	2018	2018	2018	2018	2018	2018
Canada Wild Rye	*Elymus canadensis*	NPG	-	0.14 (1.18)	-	-	-	-	-	-	-
Chicory	*Cichorium intybus*	XPF	-	-	-	-	0.14 (1.18)	-	-	-	-
Clammy Groundcherry	*Physalis heterophylla*	NPF	-	-	-	-	-	-	0.03 (0.31)	-	-
Common Blackberry	*Rubus allegheniensis*	NPF	-	-	0.14 (1.18)	-	0.14 (0.83)	1.15 (6.69)	-	-	0.01 (0.12)
Common Dandelion	*Taraxacum officinale*	XPF	-	-	-	1.18 (4.63)	-	0.17 (1.29)	3.23 (13.06)	-	-
Common Evening-Primrose	*Oenothera biennis*	NBF	-	-	-	-	0.14 (1.18)	1.87 (7.05)	0.16 (0.87)	0.90 (3.29)	2.31 (6.62)
Common Milkweed	*Asclepias syriaca*	NPF	0.14 (0.72)	-	-	0.56 (2.45)	-	-	-	-	0.07 (0.59)
Common Mullein	*Verbascum thapsus*	XPF	-	-	-	0.62 (3.75)	0.56 (3.31)	-	0.21 (1.24)	0.42 (1.83)	-
Common Pokeweed	*Phytolacca americana*	NPF	0.21 (1.77)	-	-	-	-	-	-	-	-
Common Ragweed	*Ambrosia artemisiifolia*	NAF	-	0.14 (1.18)	0.06 (0.33)	1.04 (4.74)	6.04 (11.93)	15.40 (21.00)	4.90 (14.20)	14.93 (23.08)	16.24 (25.50)
Common Sunflower	*Helianthus annuus*	NAF	-	-	-	0.35 (1.75)	0.14 (1.18)	-	0.10 (0.72)	-	-
Compass Plant	*Silphium laciniatum*	NPF	-	-	-	-	-	-	0.05 (0.51)	-	0.49 (2.40)
Crabgrass spp.	*Digitaria*	XAG	-	-	-	0.62 (3.13)	-	-	0.62 (4.25)	-	-
Curly Dock	*Rumex crispus*	XPF	0.14 (1.18)	0.14 (1.18)	-	2.57 (7.60)	0.07 (0.59)	0.08 (0.65)	2.71 (9.17)	1.81 (5.39)	0.42 (1.63)
Daisy Fleabane	*Erigeron strigosus*	NAF	0.14 (1.18)	0.56 (3.71)	0.03 (0.24)	0.49 (1.91)	7.04 (13.08)	0.33 (1.56)	0.05 (0.51)	4.17 (8.26)	0.28 (1.15)
Deptford Pink	*Dianthus armeria*	XAF	-	-	-	0.14 (0.83)	1.32 (3.75)	-	-	0.42 (1.63)	-
Dock spp.	*Rumex*	-	-	0.21 (1.77)	0.35 (2.95)	1.18 (4.86)	0.21 (1.31)	-	1.04 (5.62)	-	-

Table A-2. Continued.

Common name	Scientific name	LHG	Mean percent cover ± standard deviation by treatment								
			Control			Control			Control		
			2018	2018	2018	2018	2018	2018	2018	2018	2018
False Nutsedge	*Cyperus strigosus*	NPG	-	0.07 (0.59)	0.07 (0.59)	0.76 (3.91)	0.76 (2.87)	0.33 (1.56)	0.31 (1.89)	1.74 (5.94)	0.14 (0.83)
False Sunflower	*Heliopsis helianthoides*	NPF	-	-	-	2.50 (6.50)	2.29 (5.43)	5.58 (10.95)	1.04 (3.07)	2.92 (5.09)	4.32 (7.79)
Fescue spp.	*Festuca*	XPG	40.62 (23.68)	14.65 (18.92)	50.42 (29.83)	-	-	-	-	-	-
Field Bindweed	*Convolvulus arvensis*	XPF	-	-	-	0.42 (2.18)	-	-	-	-	-
Field Pennycress	*Thlaspi arvense*	XAF	0.03 (0.24)	-	0.07 (0.59)	-	-	0.03 (0.26)	-	X	0.35 (1.75)
Fireweed	*Chamerion angustifolium*	NPF	-	-	-	-	-	-	-	-	0.22 (1.28)
Foxglove Beardtongue	*Penstemon digitalis*	NPF	-	-	-	-	-	0.70 (3.12)	-	-	1.04 (4.36)
Foxtail spp.	*Setaria*	XAG	0.56 (4.71)	3.61 (7.08)	0.62 (3.45)	15.75 (23.30)	16.39 (19.68)	15.88 (26.43)	7.04 (14.90)	22.78 (20.00)	4.54 (12.83)
Giant Ragweed	*Ambrosia trifida*	NAF	-	-	-	-	-	0.67 (2.15)	0.73 (6.20)	1.46 (7.76)	3.10 (6.67)
Golden Alexander	*Zizia aurera*	NPF	-	-	-	-	-	0.17 (1.29)	-	-	-
Goldenrod spp.	*Solidago*	-	-	0.28 (2.36)	0.35 (2.11)	0.00 (0.00)	0.42 (3.54)	0.33 (1.56)	-	0.62 (3.24)	-
Goosegrass	*Eleusine indica*	XAG	-	-	-	-	-	-	1.41 (8.07)	-	-
Gray-headed Coneflower	*Ratibida pinnata*	NPF	-	-	-	0.07 (0.59)	0.83 (3.25)	4.50 (9.01)	0.02 (0.20)	2.01 (5.08)	2.47 (7.21)
Hedge Bindweed	*Calystegia sepium*	NPF	0.21 (1.77)	-	-	-	-	-	-	-	-
Hemp Dogbane	*Apocynum cannabinum*	NPF	-	1.74 (10.69)	0.21 (1.31)	-	-	0.08 (0.65)	0.10 (1.02)	-	0.14 (0.83)
Hoary Vervain	*Verbena stricta*	NPF	-	-	0.07 (0.59)	-	0.07 (0.59)	0.33 (2.03)	0.47 (2.52)	0.49 (2.25)	0.38 (1.95)
Honey Locust	*Gledistsia triacanthos*	-	-	-	0.03 (0.24)	-	-	0.07 (0.52)	0.02 (0.20)	0.14 (1.18)	0.62 (3.35)

Table A-2. Continued.

Common name	Scientific name	LHG	Mean percent cover ± standard deviation by treatment								
			Control			Control			Control		
			2018	2018	2018	2018	2018	2018	2018	2018	2018
Hop Sedge	*Carex lupulina*	NPG	-	-	-	-	-	-	-	-	0.56 (2.97)
Horsenettle	*Solanum carolinense*	NPF	0.07 (0.59)	3.33 (7.92)	2.15 (3.95)	0.07 (0.59)	0.15 (1.18)	0.63 (1.80)	0.14 (0.78)	0.83 (3.14)	0.26 (0.96)
Illinois Bundleflower	*Desmanthus illinoensis*	NPF	-	-	-	-	-	-	0.05 (0.51)	1.25 (3.82)	0.51 (1.91)
Ivyleaf Morningglory	*Ipomoea hederacea*	XAF	-	-	-	0.14 (1.18)	-	-	-	-	-
Kentucky Bluegrass	*Poa pratensis*	XPG	-	-	-	0.62 (5.30)	-	-	0.36 (1.95)	-	-
Kochia	*Kochia scoparia*	XAF	-	-	-	-	-	-	-	-	0.07 (0.59)
Lambsquarters spp.	*Chenopodium*	NAF	0.35 (2.42)	-	-	4.03 (9.74)	-	-	7.19 (19.00)	0.35 (2.11)	0.03 (0.24)
Little Bluestem	*Schizachyrium scoparium*	NPG	-	-	-	-	-	0.75 (3.42)	0.10 (0.72)	-	-
Looking Glass	*Brunnera macrophylla*	XPF	-	-	0.03 (0.24)	-	-	0.03 (0.26)	-	-	-
Mare's Tail	*Conyza canadensis*	NAF	-	0.28 (2.36)	2.68 (7.73)	3.12 (8.41)	2.08 (4.26)	0.90 (2.52)	4.53 (13.24)	7.64 (14.70)	0.65 (3.25)
Maximilian Sunflower	*Helianthus maximiliani*	NPF	-	-	-	0.14 (1.18)	-	-	0.21 (2.04)	-	-
Moth Mullein	*Verbascum blattaria*	XBF	-	-	-	-	-	-	-	0.07 (0.59)	-
Musk Thistle	*Carduus nutans*	XBF	-	-	0.07 (0.59)	-	-	-	-	-	0.07 (0.59)
Nettle spp.	*Urtica*	-	-	0.56 (2.15)	-	0.14 (1.18)	0.21 (1.77)	-	-	0.07 (0.59)	-
Nightshade spp.	*Solanaceae*	-	-	0.14 (1.18)	0.97 (3.18)	-	-	0.17 (1.29)	-	-	0.07 (0.59)
Oxeye Daisy	*Leucanthemum vulgare*	XPF	-	-	-	-	0.14 (1.18)	-	-	-	-
Pale Dock	*Rumex altissimus*	NPF	-	-	-	-	-	-	-	0.56 (4.71)	-

Table A-2. Continued.

Common name	Scientific name	LHG	Mean percent cover ± standard deviation by treatment								
			Control			Control			Control		
			2018	2018	2018	2018	2018	2018	2018	2018	2018
Pale Purple Coneflower	*Echinacea pallida*	NPF	-	-	-	-	1.11 (4.46)	2.87 (5.98)	-	1.88 (5.20)	4.70 (10.22)
Partridge Pea	*Chamaecrista fasciculata*	NAL	-	0.28 (2.36)	0.14 (0.83)	3.36 (9.15)	4.11 (8.48)	7.88 (15.16)	1.01 (3.45)	2.71 (5.17)	4.22 (7.42)
Pilewort	*Erechtites hieraciifolius*	NAF	-	1.39 (5.19)	-	-	-	-	-	-	-
Plains Coreopsis	*Coreopsis tinctoria*	NPF	-	-	-	-	-	-	1.04 (3.83)	2.50 (8.14)	1.12 (4.51)
Plantain spp.	*Plantago*	-	-	-	-	0.28 (1.43)	1.25 (4.00)	-	-	1.32 (3.02)	-
Poison Hemlock	*Conium maculatum*	XBF	-	-	-	-	-	0.08 (0.65)	-	-	-
Prairie Threeawn	*Aristida oligantha*	NAG	-	-	-	-	-	0.33 (2.58)	-	-	-
Prickly Lettuce	*Lactuca serriola*	XBF	-	-	1.04 (4.69)	0.14 (1.18)	0.56 (2.45)	0.75 (3.66)	0.05 (0.51)	1.32 (3.84)	0.07 (0.59)
Prostrate Pigweed	*Amaranthus blitoides*	NAF	-	-	-	0.28 (2.36)	-	-	-	-	-
Purple Lovegrass	*Eragrostis spectabilis*	NPG	-	-	-	-	-	-	0.99 (6.27)	-	-
Queen Ann's Lace	*Daucus carota*	XBF	-	0.14 (1.18)	-	-	0.15 (1.18)	0.62 (3.21)	0.68 (6.63)	0.07 (0.59)	-
Red Clover	*Trifolium pratense*	XBL	0.28 (1.65)	0.97 (5.85)	-	1.60 (5.92)	5.43 (14.16)	0.78 (2.74)	5.47 (12.56)	3.68 (9.75)	0.90 (4.31)
Reed Canary Grass	*Phalaris arundinacea*	NPG	-	1.39 (11.79)	-	-	0.35 (2.95)	0.67 (3.12)	-	-	-
Rigid Goldenrod	*Solidago rigida*	NPF	-	-	-	-	0.14 (1.18)	0.25 (1.94)	-	-	1.88 (7.19)
Sedge spp.	*Carex*	-	-	-	-	-	-	0.08 (0.65)	-	-	-
Sensitive Briar	*Mimosa quadrivalvis*	NPF	-	-	-	-	-	-	0.10 (1.02)	-	-
Silver Cinquefoil	*Potentilla argentea*	XPF	-	0.14 (1.18)	-	0.76 (4.87)	0.50 (2.08)	-	0.10 (1.02)	0.69 (2.42)	-

Table A-2. Continued.

Common name	Scientific name	LHG	Mean percent cover ± standard deviation by treatment								
			Control			Control			Control		
			2018	2018	2018	2018	2018	2018	2018	2018	2018
Smartweed spp.	*Persicaria*	-	3.61 (13.25)	5.62 (12.39)	3.82 (11.05)	15.69 (29.19)	8.54 (17.49)	1.08 (3.30)	13.54 (31.14)	5.76 (12.91)	1.15 (5.37)
Smooth Groundcherry	*Physalis longifolia*	NPF	-	-	-	1.14 (9.43)	-	-	-	-	-
Smooth Pigweed	*Amaranthus hybridus*	NAF	-	-	-	0.56 (3.20)	-	-	0.16 (1.14)	-	-
Sowthistle spp.	*Sonchus*	-	-	-	-	0.00 (0.00)	0.07 (0.59)	-	-	-	0.03 (0.24)
Spurge spp.	*Euphorbia*	-	-	-	-	0.14 (1.18)	-	-	0.89 (5.38)	-	-
St. John's Wort	*Hypericum perforatum*	XPF	-	-	-	-	0.08 (0.60)	-	-	0.76 (5.91)	-
Stinging Nettle	*Urtica dioica*	NPF	-	-	-	-	0.35 (2.11)	-	-	0.07 (0.59)	-
Stinkgrass	*Eragostis cilianensis*	XAG	-	-	-	-	-	-	0.21 (2.04)	-	-
Sulphur Cinquefoil	*Potentilla recta*	XPF	-	-	-	0.49 (2.25)	0.49 (2.54)	0.10 (0.57)	0.16 (1.53)	-	0.18 (0.86)
Swamp Agrimony	*Agrimonia parviflora*	NPF	-	0.21 (1.77)	0.76 (3.43)	0.00 (0.00)	-	-	-	-	-
Switchgrass	*Panicum virgatum*	NPG	-	-	-	-	0.83 (4.11)	22.33 (24.45)	-	2.71 (6.92)	31.50 (28.28)
Thistle Sp.	*Cirsium*	-	0.62 (4.27)	0.07 (0.59)	1.18 (5.78)	4.93 (14.33)	-	0.58 (2.78)	1.20 (6.80)	0.21 (1.01)	0.03 (0.24)
Tickseed Coreopsis	*Coreopsis tripteris*	NPF	-	-	-	-	2.64 (5.87)	1.43 (4.33)	-	2.36 (5.87)	2.94 (10.23)
Timothy Grass	*Phleum pratense*	XPG	-	-	-	2.22 (11.89)	-	0.03 (0.26)	12.92 (24.89)	1.81 (10.01)	0.56 (2.15)
Velvetleaf	*Abutilon theophrasti*	XAF	-	0.14 (1.18)	-	-	-	-	0.05 (0.51)	-	-
Venice Mallow	*Hibiscus trionum*	XAF	-	-	-	-	0.14 (1.18)	-	-	-	-
Virginia Pepperweed	*Lepidium virginicum*	NAF	-	-	-	0.07 (0.59)	1.18 (5.21)	-	-	0.69 (3.39)	0.07 (0.59)

Table A-2. Continued.

Common name	Scientific name	LHG	Mean percent cover ± standard deviation by treatment								
			Control			Control			Control		
			2018	2018	2018	2018	2018	2018	2018	2018	2018
Water Hemp	*Amaranthus rudis*	NAF	-	-	-	3.40 (9.74)	-	-	4.43 (13.74)	-	-
Western Ironweed	*Vernonia fasciculata*	NPF	-	0.49 (4.12)	-	-	-	0.17 (1.29)	-	-	-
Western Yarrow	*Achillea millefolium*	NPF	-	-	-	-	0.21 (1.31)	-	-	-	-
White Clover	*Trifolium repens*	XPL	-	-	-	0.90 (2.70)	0.90 (4.99)	0.32 (1.96)	1.35 (4.32)	0.21 (1.77)	-
White Heath Aster	*Symphyotrichum ericoides*	NPF	-	-	0.62 (3.35)	-	1.18 (5.84)	4.12 (10.60)	-	2.92 (9.11)	7.86 (15.01)
White Vervain	*Verbena urticifolia*	NPF	-	-	0.04 (0.35)	-	0.83 (5.24)	0.08 (0.65)	-	0.76 (2.87)	0.28 (1.37)
White Wild Indigo	*Baptisia lactea*	NPF	0.28 (2.36)	0.28 (1.65)	0.88 (4.65)	-	-	-	-	-	-
Wild Bergamot	*Monarda fistulosa*	NPF	-	-	-	-	2.01 (6.64)	6.40 (11.28)	-	2.01 (5.79)	6.24 (11.68)
Wild Grape	*Vitis riparia*	NPF	-	-	-	-	-	-	-	0.07 (0.59)	-
Wild Lettuce	*Lactuca virosa*	XBF	-	-	7.15 (10.31)	-	-	1.25 (4.66)	-	-	0.14 (0.83)
Wild Parsnip	*Pastinaca sativa*	XBF	0.07 (0.59)	0.07 (0.59)	-	0.35 (2.11)	0.28 (1.65)	-	0.10 (1.02)	0.14 (1.18)	-
Winged Loosestrife	*Lythrum alatum*	NPF	-	-	-	-	0.21 (1.77)	-	-	-	-
Yellow Nutsedge	*Cyperus esculentus*	NPG	-	-	-	-	0.21 (1.77)	-	-	-	-
Yellow Sweet Clover	*Melilotus officinalis*	XBL	-	-	-	-	-	-	-	0.14 (1.18)	-
Yellow Woodsorrel	*Oxalis stricta*	NPF	-	-	0.03 (0.24)	0.21 (1.31)	-	-	0.05 (0.51)	-	-

Table A-3. Mean abundance ± standard deviation of wild bee taxa observed per field by treatment using bee bowl surveys at the Ruckman Farm Diversity Experiment located in northwest Missouri, USA, 2017-2020.

Scientific name	Mean abundance ± standard deviation by treatment								
	Control			Low Diversity			High Diversity		
	2018	2019	2020	2018	2019	2020	2018	2019	2020
Agapostemon sericeus	0.1 (0.0)	-	-	-	-	-	-	-	-
Agapostemon splendens	0.1 (0.0)	0.1 (0.0)	2.0 (0.8)	3.5 (0.7)	-	0.1 (0.0)	2.25 (0.9)	0.1 (0.0)	0.1 (0.0)
Agapostemon texanus	11.0 (6.1)	1.3 (0.5)	0.1 (0.0)	12.5 (12.4)	0.1 (0.0)	-	10.8 (12.8)	-	0.1 (0.0)
Agapostemon virescens	11.2 (8.8)	3.5 (1.0)	3.8 (2.4)	8.7 (8.2)	2.7 (1.5)	3.0 (0)	5.6 (1.5)	1.2 (0.6)	2.0 (0)
Andrena barbara	0.1 (0.0)	-	-	-	-	-	-	-	-
Andrena commoda	-	-	-	-	-	0.1 (0.0)	-	-	-
Andrena cressonii cressonii	-	-	-	-	-	-	-	-	0.1 (0.0)
Andrena evythronii	-	-	-	-	-	-	0.1 (0.0)	-	-
Andrena geranii	-	-	-	-	-	0.1 (0.0)	-	-	-
Andrena nivalis	-	-	-	-	0.1 (0.0)	-	-	-	-
Andrena wilmattae	-	0.1 (0.0)	-	-	-	-	-	-	-
Augochlora pura	2 (1.7)	0.1 (0.0)	1.2 (0.4)	5.6 (5.1)	0.1 (0.0)	3.3 (2.6)	5.4 (7.1)	3.0 (0.8)	3.3 (3.3)
Augochlorella aurata	9.5 (3.4)	3.2 (4.4)	4.5 (6.3)	12.5 (4.2)	3.3 (2.3)	5.4 (3.6)	18.4 (7.7)	3.0 (1.2)	4.2 (1.8)
Augochloropsis metallica	0.1 (0.0)	0.1 (0.0)	-	2 (0.8)	3.0 (0)	-	3.0 (1.4)	-	-
Bombus bimaculatus	-	0.1 (0.0)	-	0.1 (0.0)	-	-	1.3 (0.6)	1 (0.0)	-
Bombus griseocollis	0.1 (0.0)	-	-	-	0.1 (0.0)	0.1 (0.0)	0.1 (0.0)	-	0.1 (0.0)
Bombus impatiens	0.1 (0.0)	-	-	0.1 (0.0)	0.1 (0.0)	-	1.0 (0.0)	-	0.1 (0.0)
Bombus pensylvanicus	0.1 (0.0)	0.1 (0.0)	0.1 (0.0)	2.0 (1.4)	1.3 (0.6)	0.1 (0.0)	1.2 (0.4)	0.1 (0.0)	1.5 (0.7)
Calliopsis andreniformes	0.1 (0.0)	-	-	-	-	-	0.1 (0.0)	-	-
Ceratina calcarata	-	-	-	0.1 (0.0)	-	-	0.1 (0.0)	0.1 (0.0)	-
Ceratina dulpa	-	-	-	-	-	-	0.1 (0.0)	-	-
Ceratina mikmaqi	-	-	-	0.1 (0.0)	0.1 (0.0)	2.0 (1.4)	0.1 (0.0)	0.1 (0.0)	2.7 (1.5)
Colletes lattitarsis	-	-	-	0.1 (0.0)	-	-	-	-	-
Eucera hamata	-	-	-	-	-	-	0.1 (0.0)	-	-
Halictus confusus	0.1 (0.0)	-	-	4.1 (1.7)	-	2.0 (1.4)	2.4 (1.9)	-	-
Halictus ligatus	6.3 (3.8)	-	0.1 (0.0)	20.2 (14.4)	2.8 (2.4)	1.5 (0.7)	23.1 (19.5)	3.6 (2.6)	1.6 (0.9)
Halictus parallelus	-	1.3 (0.6)	0.1 (0.0)	1.7 (1.2)	1.5 (0.7)	1.7 (0.6)	1.2 (0.4)	0.1 (0.0)	0.1 (0.0)
Halictus rubicundus	-	-	-	1.5 (0.7)	-	-	-	-	0.1 (0.0)
Halictus tripartitus	-	-	-	-	-	-	2.5 (2.1)	0.1 (0.0)	-
Hoplitis spoliata	-	-	-	-	-	0.1 (0.0)	-	-	-
Hylaeus affinis	-	-	0.1 (0.0)	-	-	-	-	0.1 (0.0)	1.3 (0.6)
Hylaeus floridanus	2.0 (0.0)	-	-	-	-	0.1 (0.0)	0.1 (0.0)	-	-

Table A-3. Continued.

Scientific name	Mean abundance ± standard deviation by treatment								
	Control			Low Diversity			High Diversity		
	2018	2019	2020	2018	2019	2020	2018	2019	2020
Hylaeus messillae	-	-	0.1 (0.0)	0.1 (0.0)	-	0.1 (0.0)	-	-	0.1 (0.0)
Hylaeus modestus	-	-	-	0.1 (0.0)	0.1 (0.0)	0.1 (0.0)	1.3 (0.6)	-	0.1 (0.0)
Lasioglossum sp. (*dialictus*)	89.6 (28.5)	3.5 (2.8)	13.2 (15.1)	131.7 (44.4)	3.8 (4.1)	11.8 (7.6)	143.63 (86.7)	4.4 (2.1)	7.8 (2.3)
Lasioglossum sp. (*evylaeus*)	2.0 (1.0)	1.3 (0.6)	0.1 (0.0)	2.0 (0.8)	0.1 (0.0)	0.1 (0.0)	2.2 (1.6)	-	0.1 (0.0)
Megachile addena	-	-	0.1 (0.0)	-	-	-	-	-	0.1 (0.0)
Megachile brevis	0.1 (0.0)	-	-	-	-	-	-	-	0.1 (0.0)
Megachile campanulae	0.1 (0.0)	-	-	0.1 (0.0)	-	-	0.1 (0.0)	-	0.1 (0.0)
Megachile centucularis	-	-	-	-	-	-	0.1 (0.0)	-	0.1 (0.0)
Megachile frugalis	0.1 (0.0)	-	-	-	-	-	-	-	-
Megachile motivaga	0.1 (0.0)	0.1 (0.0)	-	0.1 (0.0)	-	0.1 (0.0)	0.1 (0.0)	0.1 (0.0)	0.1 (0.0)
Megachile paralella	-	-	-	-	-	-	0.1 (0.0)	-	-
Megachile pugnata	0.1 (0.0)	-	-	-	-	-	-	0.1 (0.0)	-
Megachile rotundata	-	-	0.1 (0.0)	-	-	-	-	-	-
Megachile texana	-	0.1 (0.0)	-	-	-	-	-	-	0.1 (0.0)
Melissodes agilis	-	-	-	-	-	-	0.1 (0.0)	-	-
Melissodes bimaculatus	2.7 (1.5)	1.3 (0.5)	3.0 (0.9)	1.8 (0.8)	3.3 (2.3)	5.0 (2.3)	4.4 (1.6)	5.4 (2.9)	5.7 (4.5)
Melissodes boltoniae	-	-	-	-	0.1 (0.0)	-	0.1 (0.0)	-	0.1 (0.0)
Melissodes communis	-	0.1 (0.0)	-	-	0.1 (0.0)	-	0.1 (0.0)	-	-
Melissodes comptoides	2.5 (1.3)	0.1 (0.0)	1.3 (0.5)	2.0 (1.4)	-	1.4 (0.5)	4.5 (4.1)	0.1 (0.0)	0.1 (0.0)
Mellisodes coreopsis	-	-	-	-	-	0.1 (0.0)	-	-	-
Melissodes denticulatus	-	-	-	-	-	-	0.1 (0.0)	-	-
Melissodes druriellus	0.1 (0.0)	0.1 (0.0)	-	-	-	0.1 (0.0)	0.1 (0.0)	0.1 (0.0)	-
Melissodes menuachus	0.1 (0.0)	-	-	-	-	-	-	-	-
Melissodes niveus	0.1 (0.0)	-	-	-	-	-	-	-	-
Melissodes subillatus	-	-	-	-	-	-	0.1 (0.0)	-	-
Melissodes trinodis	0.1 (0.0)	0.1 (0.0)	-	1.5 (0.7)	0.1 (0.0)	1.5 (0.7)	0.1 (0.0)	-	1.5 (0.7)
Osmia lignaria	-	-	0.1 (0.0)	-	-	-	-	-	-
Peponapis pruinosa	-	0.1 (0.0)	-	-	-	-	-	-	-
Ptilothrix bombiformes	-	-	-	-	0.1 (0.0)	-	-	-	-
Specodes pimpinellae	0.1 (0.0)	-	-	-	-	-	-	-	-
Svastra obliqua	-	-	0.1 (0.0)	-	-	-	-	-	0.1 (0.0)
Triepeolus cressonii	-	-	-	-	-	-	0.1 (0.0)	-	-
Xylocopa virginica	-	0.1 (0.0)	-	0.1 (0.0)	-	-	-	0.1 (0.0)	-

*Ascher and Pickering (2012) was used as the definitive taxonomic source for bee identification. * = first recording for Gentry County, Missouri.

Table A-4. Mean ± standard deviation of detection rates (number of detections per 100 coverboard flips) of snake taxa observed per field by treatment using cover board surveys at the Ruckman Farm Diversity Experiment located in northwest Missouri, USA, 2017-2020. * = Missouri species of greatest conservation need (MODOC 2015).

Common name	Scientific name	Mean number of detections (± standard deviation) by treatment								
		Control			Low Diversity			High Diversity		
		2018	2019	2020	2018	2019	2020	2018	2019	2020
Brown snake	*Storeria dekayi*	-	2.5 (3.0)	0.1 (0.0)	1.5 (0.7)	2.3 (1.2)	1.5 (0.7)	2.5 (2.1)	0.1 (0.0)	1.8 (0.5)
Common garter snake	*Thamnophis sirtalis*	-	13.3 (7.4)	10.8 (6.0)	1.7 (1.2)	12.7 (14.6)	7.8 (5.4)	0.1 (0.0)	16.1 (17.2)	9.7 (8.8)
Eastern yellow-bellied racer	*Coluber contrictor*	-	-	-	-	0.1 (0.0)	-	-	-	-
Great Plains rat snake*	*Elaphe guttata emoryi*	-	-	-	-	-	-	-	0.1 (0.0)	-
Lined snake*	*Tropidoclonion lineatum*	-	0.1 (0.0)	-	-	1.7 (1.2)	0.1 (0.0)	2.7 (1.5)	1.3 (0.5)	0.1 (0.0)
Plains garter snake*	*Thamnophis radix*	-	9.6 (4.8)	3.0 (2.9)	-	2.3 (1.9)	1.2 (0.4)	-	4.8 (4.4)	2.3 (1.3)
Prairie kingsnake	*Lampropeltis calligaster*	-	-	0.1 (0.0)	-	-	-	-	0.1 (0.0)	-
Prairie ringneck snake	*Diadophis punctatus*	-	0.1 (0.0)	0.1 (0.0)	-	-	0.1 (0.0)	0.1 (0.0)	3.3 (2.6)	2.3 (2.3)
Western ribbon snake	*Thamnophis proximus*	-	0.1 (0.0)	0.1 (0.0)	-	-	-	0.1 (0.0)	-	0.1 (0.0)

Table A-5. Mean ± standard deviation of detection rates (number of detections per 100 cover board flips) of mammal taxa observed per field by treatment using cover board surveys at the Ruckman Farm Diversity Experiment located in northwest Missouri, USA, 2017-2020.

Common name	Scientific name	Mean number detections (± standard deviation) by treatment								
		Control			Low Diversity			High Diversity		
		2018	2019	2020	2018	2019	2020	2018	2019	2020
Least shrew	*Cryptotis parva*	-	5.8 (5.5)	7.5 (5.9)	2.0 (1.4)	2.7 (2.8)	3.0 (2.8)	3.0 (2.6)	3.0 (2.8)	2.5 (1.7)
Mice	*Peromyscus* spp.	0.1 (0.0)	6.8 (4.3)	9.2 (10.3)	17.2 (16.6)	10.4 (11.3)	9.2 (6.8)	24.0 (21.9)	13.3 (10.0)	6.8 (6.1)
Northern short-tailed shrew	*Blarina brevicauda*	-	0.1 (0.0)	6.0 (5.6)	-	-	6.3 (1.5)	-	0.1 (0.0)	3.0 (2.5)
Vole	*Microtus* spp.	-	0.1 (0.0)	4.5 (3.7)	2.5 (1.9)	5.3 (7.5)	2.0 (1.4)	1.3 (0.6)	1.8 (1.3)	3.3 (4.0)

Table A-6. Mean ± standard deviation of bird species detections per survey during point count surveys at the Ruckman Farm Diversity Experiment located in northwest Missouri, USA, 2018-2020. ^ = grassland bird species (Peterjohn and Sauer 1993). * = Missouri species of greatest conservation need (MODOC 2015).

Common name	Scientific name	Mean detections (± standard deviation) by treatment								
		Control			Low diversity			High diversity		
		2018	2019	2020	2018	2019	2020	2018	2019	2020
American Crow	*Corvus brachyrhynchos*	0.02 (0.15)	-	-	0.06 (0.24)	0.12 (0.33)	0.08 (0.27)	0.13 (0.40	-	0.08 (0.28)
American Goldfinch	*Spinus tristis*	0.20 (0.46)	0.10 (0.31)	0.10 (0.41)	0.33 (0.66)	0.50 (0.76)	0.72 (1.06)	0.23 (0.51	0.23 (0.51)	0.65 (1.11)
American Robin	*Turdus migratorius*	0.32 (0.52)	0.10 (0.31)	0.41 (0.50)	0.48 (0.74)	0.42 (0.58)	0.72 (0.79)	0.37 (0.59)	0.27 (0.45)	0.19 (0.40)
Baltimore Oriole	*Icterus galbula*	0.30 (0.46)	0.25 (0.44)	0.55 (0.63)	0.21 (0.46)	0.35 (0.56)	0.40 (0.50)	0.23 (0.47)	0.23 (0.43)	0.30 (0.52)
Barn Swallow	*Hirundo rustica*	0.09 (0.47)	-	0.14 (0.58)	0.08 (0.28)	0.12 (0.43)	-	0.10 (0.30)	0.12 (0.33)	0.38 (1.11)
Bell's Vireo*	*Vireo bellii*	-	-	-	-	-	-	0.02 (0.14)	-	-
Belted Kingfisher	*Megaceryle alcyon*	-	-	-	-	-	-	-	-	0.3 (0.16)
Black-capped Chickadee	*Poecile atricapillus*	0.05 (0.21)	-	0.03 (0.19)	-	-	-	0.02 (0.13)	0.04 (0.20)	0.03 (0.16)
Blue Jay	*Cyanocitta cristata*	0.34 (0.48)	0.45 (0.60)	0.48 (0.57)	0.31 (0.55)	0.19 (0.41)	0.52 (0.77)	0.58 (0.75)	0.15 (0.37)	0.49 (0.69)
Bobolink^*	*Dolichonyx oryzivorus*	0.93 (1.21)	0.80 (1.51)	0.90 (0.97)	0.31 (0.51)	0.12 (0.33)	0.36 (0.49)	0.52 (0.87)	0.15 (0.46)	0.24 (0.49)
Brown-headed Cowbird	*Molothrus ater*	1.05 (1.10)	0.50 (0.76)	1.41 (1.18)	1.35 (1.37)	0.96 (0.72)	1.08 (1.07)	1.10 (1.17)	0.81 (0.94)	1.41 (0.89)
Brown Thrasher*	*Toxostoma rufum*	0.32 (0.56)	0.25 (0.44)	0.03 (0.19)	0.17 (0.38)	0.27 (0.45)	0.08 (0.28)	0.31 (0.51)	0.23 (0.43)	0.19 (0.40)
Canada Goose	*Branta canadensis*	-	-	0.3 (0.19)	-	-	-	-	-	0.16 (0.83)
Cedar Waxwing	*Bombycilla cedrorum*	0.05 (0.21)	-	0.21 (1.11)	0.08 (0.45)	-	-	-	-	-
Chipping Sparrow	*Spizella passerina*	-	-	-	-	-	-	0.06 (0.31)	-	-
Common Grackle	*Quiscalus quiscula*	0.07 (0.25)	-	0.03 (0.19)	0.08 (0.28)	0.08 (0.27)	0.08 (0.28)	0.04 (0.19)	0.08 (0.27)	0.03 (0.16)

Table A-6. Continued.

Common name	Scientific name	Mean detections (± standard deviation) by treatment								
		Control			Low diversity			High diversity		
		2018	2019	2020	2018	2019	2020	2018	2019	2020
Common Yellowthroat^	*Geothlypis trichas*	1.30 (0.95)	0.90 (0.72)	1.14 (0.74)	1.10 (0.90)	1.12 (0.99)	1.20 (0.91)	1.08 (0.86)	0.73 (0.78)	1.81 (1.13)
Dickcissel^*	*Spiza americana*	1.38 (1.19)	1.10 (0.79)	1.66 (0.81)	2.13 (1.28)	2.50 (0.99)	2.24 (0.83)	1.83 (1.45)	2.54 (1.27)	2.35 (1.25)
Downy Woodpecker	*Picoides pubescens*	0.02 (0.15)	0.05 (0.22)	-	0.02 (0.14)	0.08 (0.27)	0.08 (0.28)	0.02 (0.14)	-	0.03 (0.16)
Eastern Bluebird	*Sialia sialis*	0.14 (0.35)	0.05 (0.22)	-	0.15 (0.36)	0.31 (0.47)	0.08 (0.28)	0.13 (0.32)	0.27 (0.45)	0.05 (0.23)
Eastern Kingbird	*Tyrannus tyrannus*	0.43 (0.62)	0.20 (0.41)	0.34 (0.55)	0.40 (0.84)	0.23 (0.51)	0.12 (0.44)	0.38 (0.69)	0.12 (0.33)	0.08 (0.28)
Eastern Meadowlark^*	*Sturnella magna*	1.18 (0.99)	0.70 (0.73	0.62 (0.68)	1.42 (1.03)	0.62 (0.57)	0.48 (0.59)	1.27 (1.03)	0.38 (0.75)	0.43 (0.55)
Eastern Phoebe	*Sayornis phoebe*	0.04 (0.21)	-	-	-	-	-	0.04 (0.19)		
Eastern Towhee	*Pipilo erythrophthalmus*	0.07 (0.25)	-	0.14 (0.35)	0.08 (0.35)	0.15 (0.37)	0.24 (0.44)	0.04 (0.19)	0.04 (0.20)	0.16 (0.37)
Eastern Wood-peewee*	*Contopus virens*	0.28 (0.42)	0.15 (0.37)	0.24 (0.44)	0.31 (0.51)	0.23 (0.43)	0.12 (0.33)	0.19 (0.39)	-	0.27 (0.45)
Field Sparrow^	*Spizella pusilla*	0.30 (0.51)	0.05 (0.22)	0.31 (0.54)	0.17 (0.43)	0.23 (0.51)	0.24 (0.44)	0.17 (0.38)	0.27 (0.53)	0.35 (0.54)
Grasshopper Sparrow^*	*Ammodramus savannarum*	0.52 (0.66)	0.35 (0.49)	0.14 (0.35)	0.29 (0.58)	0.23 (0.51)	0.24 (0.44)	0.42 (0.54)	0.23 (0.43)	0.14 (0.35)
Gray Catbird	*Dumetella carolinensis*	0.66 (0.71)	0.55 (0.88)	0.41 (0.63)	0.48 (0.65)	0.54 (0.71)	0.44 (0.51)	0.65 (0.74)	0.31 (0.55)	0.35 (0.54)
Great-crested Flycatcher	*Myiarchus crinitus*	0.05 (0.21)	-	-	0.04 (0.20)	-	-	-	0.04 (0.20)	-
Hairy Woodpecker	*Picoides villosus*	0.05 (0.21)	-	0.03 (0.19)	0.04 (0.20)	-	-	0.02 (0.14)	-	0.03 (0.16)
Henslow's Sparrow^*	*Ammodramus henslowii*	-	-	-	-	-	-	-	0.02 (0.14)	-
House Sparrow	*Passer domesticus*	-	-	-	-	-	-	0.02 (0.14)	-	-
House Wren	*Troglodytes aedon*	0.75 (0.78)	0.45 (0.51)	0.62 (0.62)	0.83 (0.75)	0.58 (0.50)	0.76 (0.72)	1.10 (1.18)	0.54 (0.86)	0.43 (0.60)

Table A-6. Continued.

Common name	Scientific name	Mean detections (± standard deviation) by treatment								
		Control			Low diversity			High diversity		
		2018	2019	2020	2018	2019	2020	2018	2019	2020
Indigo Bunting	*Passerina cyanea*	0.34 (0.57)	0.40 (0.50)	0.17 (0.47)	0.85 (0.77)	0.42 (0.58)	0.32 (0.48)	0.77 (0.70)	0.54 (0.58)	0.14 (0.35)
Killdeer	*Charadrius vociferus*	0.09 (0.29)	0.15 (0.37)	-	0.15 (0.41)	0.04 (0.20)	0.08 (0.28)	0.12 (0.38)	-	-
Lark Sparrow^	*Chondestes grammacus*	0.02 (0.15)	-	-	0.04 (0.20)	-	-	0.06 (0.24)	-	-
Least Flycatcher	*Empidonax minimus*	-	-	-	-	-	0.04 (0.20)	-	-	0.03 (0.16)
Mallard	*Anas platyrhynchos*	-	-	0.03 (0.19)	-	-	-	-	-	-
Mourning Dove	*Zenaida macroura*	0.55 (0.85)	0.60 (0.82)	0.28 (0.45)	0.71 (1.24)	0.42 (0.58)	0.44 (0.65)	0.40 (0.69)	0.38 (0.57)	0.27 (0.51)
Northern Bobwhite^*	*Colinus virginianus*	-	-	-	0.08 (0.28)	-	0.04 (0.20)	0.02 (0.14)	-	0.05 (0.23)
Northern Cardinal	*Cardinalis cardinalis*	0.45 (0.73)	0.15 (0.37)	0.10 (0.41)	0.35 (0.56)	0.04 (0.20)	0.28 (0.46)	0.38 (0.60)	-	0.14 (0.35)
Northern Flicker	*Colaptes auratus*	0.05 (0.21)	0.15 (0.37)	0.17 (0.38)	0.08 (0.28)	0.08 (0.27)	0.20 (0.40)	0.06 (0.24)	0.04 (0.20)	0.08 (0.27)
Orchard Oriole	*Icterus spurius*	0.05 (0.21)	0.05 (0.22)	0.07 (0.26)	-	-	-	0.04 (0.19)	0.08 (0.27)	0.08 (0.28)
Red-bellied Woodpecker	*Melanerpes carolinus*	0.23 (0.48)	0.15 (0.37)	0.31 (0.47)	0.15 (0.36)	0.35 (0.56)	0.32 (0.48)	0.25 (0.48)	0.15 (0.37)	0.30 (0.46)
Red-headed Woodpecker*	*Melanerpes erythrocephalus*	0.20 (0.41)	0.15 (0.36)	0.21 (0.41)	0.19 (0.45)	0.19 (0.45)	0.28 (0.46)	0.23 (0.43)	0.23 (0.42)	0.11 (0.31)
Red-tailed Hawk	*Buteo jamaicensis*	-	-	-	-	-	0.12 (0.33)	-	-	0.09 (0.27)
Red-winged Blackbird^	*Agelaius phoeniceus*	1.66 (1.58)	1.45 (0.94)	1.79 (1.83)	3.08 (1.74)	4.73 (4.77)	3.84 (2.88)	2.21 (1.72)	2.31 (2.29)	2.00 (1.65)
Ring-necked Pheasant^	*Phasianus colchicus*	0.05 (0.21)	0.05 (0.22)	0.18 (0.38)	0.02 (0.14)	-	0.12 (0.33)	0.12 (0.32)	-	0.19 (0.40)
Rose-breasted Grosbeak	*Pheucticus ludovicianus*	0.18 (0.39)	0.18 (0.39)	0.17 (0.38)	0.13 (0.33)	0.23 (0.43)	0.28 (0.46)	0.21 (0.50)	0.08 (0.27)	0.32 (0.47)
Ruby-throated Hummingbird	*Archilochus colubris*	-	-	0.03 (0.19)	-	-	-	-	-	-

Table A-6. Continued.

Common name	Scientific name	Mean detections (± standard deviation) by treatment								
		Control			Low diversity			High diversity		
		2018	2019	2020	2018	2019	2020	2018	2019	2020
Savannah Sparrow^	*Passerculus sandwichensis*	-	-	-	-	-	-	0.02 (0.14)	-	-
Sedge Wren^	*Cistothorus platensis*	0.34 (0.64)	0.70 (0.73)	0.07 (0.26)	0.04 (0.20)	-	-	0.02 (0.14)	0.12 (0.33)	-
Song Sparrow	*Melospiza melodia*	0.70 (0.67)	0.75 (0.55)	0.79 (0.62)	0.88 (0.73)	0.96 (0.87)	0.68 (0.56)	0.98 (0.87)	0.88 (0.77)	0.89 (0.61)
Tree Swallow	*Tachycineta bicolor*	0.07 (0.33)	-	-	-	-	-	0.10 (0.49)	-	0.03 (0.16)
Tufted Titmouse	*Baeolophus bicolor*	-	-	-	-	-	-	0.02 (0.14)	-	-
Turkey Vulture	*Cathartes aura*	-	-	0.31 (0.71)	-	-	0.04 (0.20)	-	-	0.22 (1.00)
Veery	*Catharus fuscescens*	-	-	-	0.02 (0.14)	-	-	-	-	-
Warbling Vireo	*Vireo gilvus*	0.02 (0.15)	0.05 (0.22)	0.24 (0.44)	0.08 (0.28)	0.23 (0.43)	0.12 (0.33)	0.02 (0.14)	0.08 (0.27)	0.14 (0.35)
Western Meadowlark^	*Sturnella neglecta*	-	-	0.07 (0.28)	0.04 (0.20)	-	-	-	-	-
White-breasted Nuthatch	*Sitta carolinensis*	0.22 (0.15)	-	-	0.02 (0.14)	-	-	-	-	-
Wild Turkey	*Meleagris gallopavo*	-	-	-	-	-	-	-	-	0.03 (0.16)
Wood Duck	*Aix sponsa*	-	-	-	-	-	-	-	-	0.03 (0.16)
Yellow-billed Cuckoo*	*Setophaga petechia*	0.25 (0.49)	0.20 (0.41)	0.21 (0.41)	0.19 (0.49)	0.19 (0.49)	0.32 (0.56)	0.13 (0.34)	0.23 (0.43)	0.11 (0.31)
Yellow Warbler	*Coccyzus americanus*	0.36 (0.72)	-	-	0.42 (0.64)	0.04 (0.19)	0.04 (0.20)	0.25 (0.52)	0.04 (0.19)	-

CHAPTER 6. GENERAL CONCLUSION

I sought to evaluate the effectiveness of conservation efforts in Midwestern agricultural landscapes for providing bird habitat. In investigating the bird response to establishment of prairie strips on working farms across Iowa, I revealed a strong association among grassland birds and prairie strips in corn and soybean fields. Grassland bird density especially increased 3 years after the establishment of prairie strips. Red-winged Blackbirds, Dickcissels, and Common Yellowthroats responded most strongly among grassland species. Overall this work combined with a companion study (Stephenson 2022) suggest prairie strips expanded and improved the habitat within agricultural landscapes of Iowa for nesting grassland birds that are not area-sensitive. While prairie strips are likely to help stem the loss of some grassland bird species and contribute to improved outcomes for soil, water, and pollinators (Schulte et al. 2017), larger grassland patches, on the order of 10s to 1000s of hectares, are likely needed to reverse the widespread declines in the overall grassland bird community, especially for area-sensitive species (Stephenson 2022).

My investigation of springtime bird communities in agricultural landscapes provided information on habitat associations and phenology of grassland birds. I concluded that springtime deployment of autonomous recording (ARUs) units provided unique investigation into spring bird communities and their dynamics. We found that in addition to documenting species richness of avian communities, ARUs generated species-level detection probabilities similar to or higher than studies on breeding season occupancy of birds. The technology provides an important tool, which could be used in monitoring shifts in avian phenology in response to global climate change (Buxton et al. 2016).

In a third study, I examined breeding bird associations with restored oxbows in north-central Iowa. More species were detected per survey at restored oxbow sites compared to nearby unrestored sites. The most common species detected at oxbow restorations included Song Sparrow, Common Yellowthroat, and Ring-necked Pheasant. Species of greatest conservation need such as Grasshopper Sparrow, Marsh Wren, and Spotted Sandpiper were also detected. We provided the first known quantitative survey of bird communities associated with oxbow restoration. In addition to assisting with flood control, improving water quality, and providing habitat for a variety of fish species including endangered Topeka shiner (Bakevich et al. 2013, Simpson et al. 2019), oxbow restoration appeared to be an effective strategy for expanding breeding bird use of agricultural landscapes.

I further evaluated the community response of multiple wildlife taxa to native grassland establishment at the scale of typical restorations in the U.S. Midwest, and compared the response communities associated with exotic, cool-season grasses typically found in agricultural landscapes of the region. Within the first three years of restoration, I documented minimal responses among wild bees, snakes, small mammals, and birds to increases in native plant cover in experimental prairie treatments. My findings are contrary to my expectations based on ecological theory and literature review, and indicate a potential lag in biodiversity response to restoration of native perennials in agricultural landscapes. I expect wildlife communities will exhibit stronger responses to native grassland establishment if the native plant community further outcompetes the non-native plant community in subsequent years.

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