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Evaluating Avian Use of Cover Crops in the Corn Belt

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Evaluating Avian use of Cover Crops in the Corn Belt

By

Megan Figura

A thesis submitted in partial fulfillment of the requirements for the

Master of Science

Major in Wildlife and Fisheries Sciences

Specialization in Wildlife Science

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THESIS ACCEPTANCE PAGE

Megan Figura

This thesis is approved as a creditable and independent investigation by a candidate for

the master's degree and is acceptable for meeting the thesis requirements for this degree.

Acceptance of this does not imply that the conclusions reached by the candidate are

necessarily the conclusions of the major department.

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It is my earnest wish that the work contained in this thesis will positively contribute to the field of avian conservation; but perhaps the most valuable, to me at

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Abstract

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The tallgrass prairie of North America has changed drastically since colonial settlement, with up to 99% of this region converted for agriculture and other land-uses. Concurrent with grassland conversion, grassland birds have experienced the most extreme, consistent, and widespread population declines of any avian guild. Agricultural lands in the U.S. Midwest were able to provide adequate habitat for several bird species until the 1950's; however, altered and intensified management practices have degraded much of remaining suitable habitat and undermined ecosystem functions. Consequently, many grassland birds have been identified as Species of Greatest Conservation Need (SGCN) in many State Wildlife Action Plans. The use of cover crops is a re-emerging practice in sustainable agriculture that may have beneficial impacts on grassland birds. Cover crops are widely accepted to have numerous agricultural benefits, but the potential to provide further ecological benefits, such as provision of habitat and resources for avifauna, are poorly understood. The goal of my study was to evaluate the use of cover crops by avifauna relative to perennial cover and fallow/row crop systems in southeastern Iowa, where cover crops and avifauna were suitably abundant. I accomplished this using two approaches: 1) I evaluated and compared community compositions among field types using permutational analysis of variance (PERMANOVA) and non-metric multidimensional scaling (NMDS) diagnostics and 2) I measured and quantified various microhabitat characteristics and used these to model variation in 4 avian population

metrics (i.e., total avian abundance, total avian richness, abundance of avian SGCN, and richness of SGCN). I documented 66 bird species during 2019 and 2021, 22 of which are considered Iowa SGCN. Results of PERMANOVA and NMDS suggested cover crops harbored a bird community that overlapped in abundance and richness with row crops but tended to be broader and support some species otherwise absent in croplands; however, cover crops did not completely overlap with habitat provided by perennial cover. Of the variables intended to predict avian population metrics, coverage by rye, litter residue cover, and litter depth appeared to be most influential. For example, a model of total avian abundance suggested a 10% increase in the mean area covered by litter predicted an increase of 9.4 birds. Management practices that promote litter accumulation and structure in cover cropped fields, such as later termination times, may improve cover crop fields for grassland bird species. My results may be used to develop planning tools and management strategies for resource managers, farmers, and biologists, and, when integrated with other sustainable agricultural practices, promote the conservation of imperiled birds.

Introduction

Over the past 50 years, grassland birds have experienced more extreme, consistent, and geographically widespread declines than any other avian guild (Knopf 1994, Samson and Knopf 1994, Vickery et al. 1999, Askins et al. 2007, Herse et al., 2017). In North America alone, grassland birds have shown the greatest magnitude (>700) and proportional (~53%) losses relative to any other breeding bird biome (Sauer et al. 2015, Rosenberg et al. 2019). Grassland birds are declining more than any other group of North American species (Samson and Knopf 1994). Management that promotes abundance and productivity of grassland birds has been described as a modern conservation priority (Cox et al. 2014).

Breeding Bird Survey (BBS) data collected from 1966-2002 indicated that 74% of grassland bird species were declining significantly, whereas only 3 species had significantly positive population trends (Sauer et al. 2003). A similar report using BBS data by Butcher and Niven (2007) identified >20 bird species with losses of >50% of individuals since the 1950's. Sauer and Link (2011) estimated that grassland obligate bird populations in North America collectively declined 37.0% from 1968 to 2008. Analyses of BBS data from 1966-2015 by Sauer et al. (2017) further reported 15 species (54%; n = 28) with significant negative declines during this period, with individual trends ranging from -0.6 to -3.2%/year and a further 4 species exhibited nonsignificant negative declines ranging from -0.4 to -0.9%/year. Additional analysis of BBS data collected during 2005-2015 indicated a reduced number of species with significant negative population trends (n = 5), ranging from -1.2 to -4.2%/year and another 9 species with nonsignificant negative trends from -0.1 to -1.9%/year (Sauer et al. 2017).

Numerous factors collectively contribute to declines in grassland bird populations. The principal threat to grassland birds is loss and modification of the habitats they rely upon (Knopf and Samson 1994, Murray et al. 2008, Shahan et al. 2017, Rosenburg et al. 2019). Temperate grasslands are the most modified and least protected of any terrestrial biome (Gayton 1990, Hoekstra et al. 2005, Glaser et al. 2012, Herse et al. 2018, Igl et al. 2018). Many grassland birds are either facultative or obligate specialists; obligate grassland specialists are exclusively adapted to grasslands and rely on them for all stages of their life cycle (e.g., breeding, migration, and wintering habitat) and make no use of other habitats, whereas facultative grassland specialists rely predominantly on grasslands but are able to take advantage of a wider array of habitat types (Vickery and Herkert 1999). The fates of grassland birds are inextricably intertwined with grasslands and the loss and modification of their critical habitat will be mirrored by consequential declines in these species (Correll et al. 2018).

In addition to their status as habitat specialists, many other factors contribute to grassland bird population declines and complicate management efforts that seek to promote their abundances and productivity. Grassland bird species respond differentially to aspects of vegetation structure (Dejong 2001, Greer 2009, Davis et al. 2021), meaning that management that aims to enhance habitat for some birds may result in neutral or even detrimental impacts on other grassland species. Modern grassland landscapes are highly fragmented (Bakker et al. 2002, Askins et al. 2007, Shahan et al. 2017, Herse et al. 2020) and many grassland birds are sensitive to either edge effects (Winter et al. 2000, Herse et al. 2018) or area of available habitat patches (Bock et al. 1999, Ribic et al. 2009), or as it appears is often the case, both (Herse et al. 2020). The culmination of

which has undoubtedly contributed to sharp population declines following changes in land use. Furthermore, many of these species are sensitive to the composition of the landscape surrounding otherwise suitable patches and landscape context influences their habitat selection and use (Bock et al. 1999, Murray et al. 2008, Herse et al. 2017). Grassland bird populations are also negatively influenced by extensive brood parasitism (Patten et al. 2006), nest depredation (Martin 1993, Ardizzone and Norment 1999), and habitat degradation due to encroachment of woody vegetation and introduced plant species (Bakker and Higgins 2009, Greer et al. 2016), all of which are associated with increased habitat fragmentation.

Historically, grasslands were the largest contiguous ecosystem and dominant vegetative province in North America (Knopf and Samson 1994, Herkert et al. 1996). Modern grasslands have been reduced to less than 40% of their original range (Comer et al. 2018) and ongoing conversion continues at a rate of almost 1% annually (Fields and Barnes 2019). Tallgrass prairie was historically the largest of the North American grassland ecoregions, spanning ~ 69 million ha; today it is estimated that >97% of its historic extent has been converted to other land uses, primarily agriculture (Knopf and Samson 1994, Hoekstra et al. 2005, Herse et al. 2020). Private ownership predominates remaining North American grasslands with ~84% privately owned, whereas <4% receives federal protection (Askins et al. 2007, Comer et al. 2018). The majority of remaining grasslands are now working lands, land that is put to human use (e.g., pastures, hayfields, croplands) and no longer consisting of native grasses (Askins et al. 2007, Herse et al. 2020).

At the heart of the historic extent of tallgrass prairie is now a region denoted as the Corn Belt, a district in the U.S. Midwest distinguished for its high productivity of cash crops. Producers in the Corn Belt provide more than one third of the global supply of corn and are the world's largest source of soybeans (Aronsohn 2019). In the absence of true grasslands, grassland birds in the Corn Belt region have come to rely on the artificial habitat associated with working lands (Askins et al. 2007). Agricultural lands were generally able to provide adequate breeding habitat for birds until the 1950's (Koford and Best 1995, Vickery and Herkert 1999, Norment 2002), when diversified farming was common and agricultural landscapes consisted of complex mosaics of grazing lands, croplands, and pastures. Altered management practices, such as agricultural intensification, shortened cutting rotations for hay, homogenization of croplands, increased use of pesticides, and removal of natural field edges have fragmented or degraded much of remaining suitable wildlife habitat (Norment 2002, Reinking et al. 2009). Such changes in land use can have significant impacts on biological diversity and other ecosystem functions (e.g., water quality, soil conditions, pollinator resources; Vickery et al. 1999, Wade et al. 2008).

Loss and modification of grasslands due to agricultural intensification is widely cited as a key driver in declines of grassland birds (Knopf and Samson 1994, Vickery et al. 1999, Murray et al. 2008, Shahan 2017). A major concern for conservation biologists is the increased potential for extinction of species that rely upon grasslands (Samson and Knopf 1994). Management of remining grasslands and working lands that is beneficial to birds is critical for the preservation of imperiled grassland species and of high priority to conservationists (Askins et al. 2007, Cox et al. 2014). Agricultural systems and practices

that promote biodiversity in a landscape dominated by human use are crucial (Blann 2006). Because remaining grasslands are predominantly privately owned, the development of strategies that inform and motivate landowners to implement "bird-friendly" practices is essential (Ciuzio et al. 2013), and with a robust understanding of the habitat needs of birds of conservation concern it should be possible to support their populations on agricultural lands (Askins et al. 2007).

Few agricultural practices truly benefit wildlife; but those that are most advantageous will be methods that tend to simulate more natural habitat conditions, such as supplementation of additional vegetative structure in croplands (Wilcoxen et al. 2018). A re-emerging practice in sustainable agriculture is the use of cover crops, which are defined as non-row crops planted in the off-season, typically fall, thereby shortening the fallow period and supplementing a period of plant growth and vegetative cover beyond the conventional cropping season (Reeves 1994, LaCanne 2017). The primary goal of planting cover crops is for maintenance and improvement of soil health in croplands; however, they have been extensively researched for their various other agricultural benefits. The potential for cover crops further providing ecological benefits, such as supplementation of habitat for wildlife, is lesser known and of great interest to resource managers.

In the U.S., cover crops have been heavily promoted and incorporated into many state nutrient reduction programs as a result of Mississippi/Gulf of Mexico Watershed Nutrient Taskforce 2008 Action Plan and the Environmental Protection Agency's guidance memorandum on water quality issues (Costello et al. 2012). The 2012 Census of Agriculture reported >1.1 million ha of cover crops planted annually in Iowa, Illinois,

and Indiana and projected the continued expansion of cover crop adoption (USDA NASS 2012). In addition to improving soil health, cover crops have been accepted to benefit agriculture in numerous ways, including increased soil organic matter, improved nutrient cycling, reduced soil erosion, water infiltration, reduced nutrient run-off, soil aeration, and weed and pest suppression (White and Barbercheck, 2017). Secondary benefits to the planting of cover crops include reduced need for fertilizers, herbicides, and pesticides (SARE 2012). While use of cover crops may initially result in economic losses, with continued use for 3-5 years, the agricultural advantages of cover crops may culminate in improved yield and profits to farmers (Myers et al. 2019). Moreover, a study by Schipanski et al. (2014) suggested that cover crops could improve 8 of 11 identified ecosystem services (i.e., functions of the environment that benefit humans) without negatively influencing crop yields.

Many plant species are used as cover crops, and all provide a variety of functions in addition to maintaining overall soil health. Major classes of cover crops include grasses, legumes, and brassicas. Grasses used as cover crops are typically chosen for their proficiency at scavenging nutrients in the soil, provision of root and topgrowth biomass, high carbon content, and ability to be used as forage for livestock and cattle, these include cereal rye (*Secale cereal*), oats (*Avena sativa*), winter wheat (*Triticum aestivum*), barley (*Hordeum vulgare*), and triticale (*Triticale hexaploide*). Due to the higher carbon content in grasses, they are slower to breakdown and provide a longer-lasting reside, promoting increased soil organic matter and longer-lasting weed suppression. Legumes, such as hairy vetch (*Vicia villosa*), crimson clover (*Trifolium incarnatum*), cowpeas (*Vigna unguiculata),* field peas (*Pisum sativum subsp. Arvense*), and medics (*Medicago*

spp.), form a symbiotic relationship with soil bacterium (e.g., *Rhizobium bacteria*), which fixes atmospheric nitrogen and provides a critical source of nitrogen for row crops. Legumes can also provide floral resources to pollinators and attract beneficial insects to crop fields. Brassica cover crops include species such as rapeseed (*Brassica napus*), field mustard (*Brassica rapa*), brown mustard (*Brassica juncea*), radishes (*Raphanus sativus*), and turnips (*Brassica rapa*). Brassicas have unique biofumigant properties making them well suited for suppressing soilborne pests and diseases; brassicas are also known for their rapid fall growth and deep-penetrating taproots that alleviate soil compaction (SARE 2012).

For the aforementioned reasons, cover crops represent one practice in sustainable agriculture and may be one method by which to mitigate some of the detrimental effects of intensive agriculture. Although cover crops are often promoted as being beneficial to wildlife, such as by supplementation of vegetative cover and simulation of more natural habitats, there has not yet been enough study to substantiate these claims. Extant research has suggested that cover crops may provide floral resources for pollinators (Ellis and Barbercheck 2015) and could enhance reptile abundance (Carpio et al. 2017). By reducing the fallow period in crop rotations, cover crops may also provide over-wintering habitat for beneficial arthropods (LaCanne 2017). Some research in Mediterranean olive groves in Spain (Castro-Caro et al. 2014, 2015) has suggested that cover crops may convey some benefits to birds in agroecosystems, however, cover crop use and management differs greatly in the U.S. Ultimately, research on grassland bird response to cover crops is severely lacking and further study is essential for the effective conservation of these imperiled species.

The purpose of my study was to evaluate the impact of cover crops on birds in agroecosystems, with a focus on those identified as Species of Greatest Conservation Need (SGCN) by the Iowa State Wildlife Action Plan (Iowa DNR 2015), relative to perennial grass cover and conventional fallow/row crop systems in order to identify management practices that most benefit threatened avifauna. My study took place in southeastern Iowa where both cover crops and grassland birds were suitably abundant. I conducted avian point count surveys, quantified vegetation and landscape characteristics, sampled soil characteristics and arthropods, and evaluated landscape attributes at multiple spatial extents. I used these data to estimate the relative influence of management, habitat characteristics, and other factors on habitat use by avifauna in this system. My research objectives were to: (1) evaluate and compare 4 avian population metrics (i.e., total specie abundance, total species richness, abundance of SGCN, and richness of SGCN) among cover types (i.e., cover crops, row crops, and perennial grassland); (2) measure and compare vegetation structure between cover types; (3) assess and compare arthropod diversity and biomass between cover types, and; (4) evaluate and compare soil health between cover types. Based on my results, I developed recommendations for best management practices to promote imperiled birds and species of conservation concern in cover cropped fields in the Corn Belt region. These recommendations will provide valuable tools for use in conservation planning and implementation by resource managers, farmers, and biologists.

Study Area

Historically, Iowa was dominated by 5.1 million ha of tallgrass prairie, but only 4,900 ha of these native grasslands remain in the state today (Hoch 2015). Iowa is a

leading agricultural producer in the U.S., with 12.3 million ha of croplands in 2021 (USDA NASS 2021b), most of which are used for corn and soybean production. Use of cover crops is still a growing practice, but between 2012 and 2017 planting of cover crops in the US increased by 50% with Iowa leading the charge with an increase of 156% in acreage of cover crops in the state (USDA NASS 2017). Survey data shows that Iowa farmers nearly doubled their use of cover crops between 2017 and 2019 with more than 1.25 million ha of cover crops for the 2020 crop year, accounting for 13.3% of the state's 9.3 million ha of croplands (Iowa Farm Bureau 2022). Cover crops are especially prevalent in the southeastern portion of the state where soils have lower organic matter and higher erodibility relative to other parts of the state (Wallander et al. 2021).

My research was conducted in and around Washington County, in southeastern Iowa, between 41˚56'00" N and 41˚12'00" N latitude and -91˚99'00" W and -91˚44'00" W longitude (Figure 1). At the time of this study, Washington County was one of the top producers for cover crops and had the highest cropland diversity in the state, making it a model landscape for providing insight into avian ecology on modern croplands with increased row crop diversification and cover crop adoption.

Methods

I conducted my field research during the spring and summer of 2019 and 2021; the COVID-19 pandemic prevented a field season in 2020. Additionally, logistical issues led to important limitations in the data collected in 2019; therefore, I only present summary statistics and related metrics from data collected in that year. All other analytical methods, results, and discussion in this Thesis focus on data collected in 2021. As all fields using cover crops within this study utilized exclusively cereal rye, all

reference to cover crops in the remainder of this Thesis refers to cereal rye unless stated otherwise. I collected field data May 1st through July 31st to coincide with the breeding season of grassland birds when avian detection is maximal.

Site Selection

        All lands in this study were privately owned and permission to conduct research on these properties was obtained by partner researchers at Iowa State University in 2018 (Shirley 2021). I classified fields by dominant land use type (by percent area) as: cover crop (i.e., fields that were seeded with cover crop the preceding fall), row crop (i.e., crop fields that were not treated with cover crops at any time), and perennial (fields primarily composed of grasses and not subject to active human use). Sample sizes of the three cover types were constrained by availability and to those which we were granted by landowners.

        Within each study field, I randomly selected 3-6 points proportional to field size to serve as survey stations throughout the season (Petit et al. 1995). Constraints on survey station selection included: 1) all stations were ≥100 m from field boundaries, edges (e.g., roads, tree belts), and structures (e.g., houses, grain elevators, hog barns) to avoid potential sources of disturbance, minimize bias, and retain integrity of relationship with the appropriate habitat (Petit et al. 1995, Ralph et al. 1997); and 2) selected points were ≥200 m from other stations to avoid oversampling (Petit et al. 1995, Ralph et al. 1997, Prescott and Murphy 1999). These survey stations were then used as fixed points from which I conducted avian point count surveys and monitored local vegetation throughout the season. Additionally, a station was placed at the approximate centroid of each field as the point of origin from which transects were drawn for arthropod and soil sampling.

Avian Surveys

Avian point count surveys were conducted by trained observers at each survey station. I chose the point count method for avian sampling because it is a standardized means for estimating populations and allows a reasonable compromise for quality of data given common limitations on time and labor (Butcher et al. 1993, Manley et al. 1993, Ralph 1993, Ralph et al. 1993). Upon arrival at a station, observers waited quietly for a period of 2 minutes prior to initiating the count to minimize the effects of disturbance due to traveling to and between stations (Kinkead 2016). Weather conditions were recorded for each survey, including temperature (C), wind speed (km/h), and approximate cloud cover. Surveys were not conducted in inclement weather unfavorable to bird detection, such as moderate or heavy rain, excessive wind, or conditions that otherwise impaired visibility or hearing (Ralph et al. 1993, Bakker and Troelstrup 1998). To further maximize detectability, I conducted point counts beginning at solar twilight (~0.5 h before sunrise) and ending 4.5 h after sunrise to coincide with maximum songbird activity (e.g., singing; Ralph et al. 1993, Murray et al. 2008). All species detected during a 5-minute survey period were recorded, as well as the mode by which they are first detected (e.g., calling, singing, or perched), and the estimated distance (m) from the observer upon detection (Ralph et al. 1993, 1997). Observers alternated between study fields over the duration of the season such that each field was surveyed approximately the same number of times by each observer to control for observer biases (Kinkead 2016).

Vegetation Monitoring

I evaluated the structure and characteristics of vegetation cover in each study field through periodic measurements taken concurrent with avian point counts, such that

vegetation data correspond with each avian survey. I collected and compiled these data to quantify microhabitat characteristics within and among study fields for subsequent use in general linear models (GLM) to explain observed variation in habitat selection and use by avifauna relative to my study objectives (Ralph et al. 1993). Specific vegetation measurements used in these models include effective leaf height (ELH) to the nearest 2.5 cm, litter depth (LD) to the nearest 1.0 cm, and estimated proportion of various cover types (i.e., bare ground, litter, forb, perennial, crop, and rye). Litter cover (i.e., litter residue) is defined in this study as dead plant material lying flat on the soil surface in some stage of decomposition (McCoy et al. 2019). As all row crops present in study fields were exclusively corn and soybeans, out of consideration of the scope and purpose of this study I collated both crops into a single variable category 'crop' to better isolate the relationship between row crops and avian populations. I estimated ELH using a Robel pole and running a meter stick down the length of the pole until it contacted the top of standing, live vegetation (excluding corn stubble or accumulated thatch) and recorded the height (Robel et al. 1970, Greer et al. 2016). Estimates of cover proportion and LD were taken using a 20 x 50 cm PVC frame quadrat (Daubenmire 1959, Higgins et al. 2012). I measured LD by inserting a meter stick within the quadrat until it came into contact with the substrate and recording the height at which the stick was obscured (Bakker and Higgins 2009). Finally, proportions of cover type were assessed by standing immediately adjacent and above the sampling quadrat and estimating proportional cover of major vegetative groups using a cover class scale commonly utilized in grasslands (Daubenmire 1959, Higgins et al. 2012). All vegetation measurements were taken 1 m away from the point in each cardinal direction and averaged to obtain one estimate per variable for the

station. These values were later included as covariates in the aforementioned GLM approach.

Arthropod Sampling

In accordance with my research objectives, I evaluated arthropod biomass and community structure within study plots as a measure of food abundance for avifauna and as a potential indicator of habitat quality (McIntyre and Thompson 2003, Atkinson et al. 2004, Pfrimmer 2017). Ground-dwelling arthropods were sampled using pitfall traps and collections were conducted on a monthly basis. I chose the pitfall trapping method because it was a passive sampling method that was also relatively insensitive to differences in foliage and represented the best compromise for quality of data given time and labor limitations (McIntyre and Thompson 2003).

Pitfall traps consisted of two transparent 255 ml plastic cups (10 cm depth and 7.6 cm diameter) nested together and placed in the ground so that the lips were flush with the soil (McIntyre and Thompson 2003). Traps were filled with ~60 ml 1:1 aqueous propylene glycol solution, which was chosen because it is an effective killing agent that is also largely non-toxic to vertebrates (McCravy and Willand 2007, McIntyre and Thompson 2003). I placed 10 pitfall traps at 5 m intervals along a linear transect originating from the centroid of each study field to control for edge effects. I chose this sampling intensity because it allowed for adequate spatial coverage in the core of the fields without oversampling or being subject to low trap independence (which results from high trap density; McIntyre and Thompson, 2003). Traps were left in the field for 72 h before collection, after which they were stored in sealed containers with 95% ethyl alcohol until transported to the lab for identification, drying, and weighing.

I processed arthropod samples in the lab by classifying the catch by Family for evaluation of abundance, richness, and community composition. To estimate dry-weight biomass, I dried samples in an oven for 48 h at 60 C and then weighed them to the nearest 0.01 mg (Taylor et al. 2006, Pfrimmer 2017).

Soil Sampling

I collected soil samples from study fields to evaluate possible indicators of ecosystem services and habitat quality for avifauna among row crops, cover crops, and perennial grassland. Samples were collected using protocol outlined by Ward Laboratories, Inc. (https://www.wardlab.com/submit-a-sample/soil-health-analysis/soil-health-sampling-procedure/) on a monthly basis concurrent with arthropod sampling and along the same 45 m transects. Soil samples were collected by extracting 10 soil cores (5 cm diameter and 15-20 cm depth) at 5 m spaced intervals. Collected soil cores were stored in plastic lined sample bags and kept in a cool place until they could be shipped to Ward Laboratories for processing. I elected to use the Haney soil health analysis package, which integrates biological and chemical factors to compute a soil-health score based on several soil health indicators (e.g., nutrient availability, soil respiration, organic carbon: organic nitrogen). The Haney test deliverables also include generalized cover crop recommendations for improving soil health (Gunderson, Ward Laboratories, Inc.). I used these data in GLM models to investigate potential differences in soil-health metrics between study plots with respect to various cover types and cropping practices.

Landscape Context

Grassland birds are highly sensitive to habitat characteristics at the landscape scale, such as land composition and land cover, and these play a large role in their habitat

selection and use (Bakker et al. 2002, Murray et al. 2008, Wolf et al. 2012, Greer et al.

2016). To investigate the influence of these variables on avifauna use, I derived land use

and composition data via geospatial layers from CropScape (USDA NASS 2021a), a

publicly available online database produced by the US Department of Agriculture

(https://nassgeodata.gmu.edu/CropScape/). I used these data to quantify area of habitat

types (e.g., cropland, grassland, woodland, wetland, developed, etc.) and index landscape

heterogeneity at multiple spatial extents (400 m, 800 m, 1600 m; Bakker et al. 2002,

Herse et al. 2017) from the centroid of each study field. I incorporated these indices in

later models to understand their relative influence on avian abundance and diversity.

Analyses

I conducted all statistical analyses on each of the 4 avian population metrics of

interest, abundances and richness of total species and abundances and richness of avian

SGCN (i.e., Species of Greatest Conservation Need as listed in the Iowa State Wildlife

Action Plan). I chose *a priori* to use a significance level of $\alpha = 0.10$ in order to balance

type I and II errors given restrictive sample sizes and because large alpha levels may be

more appropriate when assessing public land management practices on wildlife

populations where it is more important to detect a real effect than it is to avoid detecting a

false one (Stevens et al. 2003). Fixed radii are often applied in avian survey methods to

overcome distance-based detectability bias and to provide greater resolution for insight

into bird-habitat relationships, therefore I used only avian count data restricted to

detections ≤ 200 m in statistical analyses (Petit et al. 1995, Hutto 2016). Due to

limitations of the 2019 data, I calculated only summary statistics for these data and all

other analyses were performed only on the 2021 data. As stated previously, 2021 sample

sizes for each cover type (i.e., perennial, cover crop, and row crop) were largely

determined by landowner permissions, but further refined due to limitations in the data

set, e.g., incomplete survey data (Table 1). All analyses were carried out using RStudio

(Version 2022.7.1.554, RStudio, Inc., Boston, Massachusetts, USA).

I pooled data from avian surveys over the 2021 season within unique study fields

and compared incidence, relative abundance, species richness, and species diversity of

avifauna among fields and sought to explain variation in these dependent variables in

relation to cover type and various microhabitat metrics. Because search effort was

variable and a few unrecoverable error errors in data collection were found, I divided all

point count data by the number of surveys at a given field to produce counts relative to

effort or data availability. An assessment of normality of the data using Shapiro-Wilk's

test revealed the data did not conform to a normal distribution, so I used non-parametric

tests in subsequent analyses. I used a Kruskal-Wallis non-parametric ANOVA to test for

differences in avian abundance and richness among study fields by field treatment (i.e.,

row crop, cover crop, or perennial cover). If a Kruskal-Wallis test indicated a significant

treatment effect ($\alpha \leq 0.10$), I then evaluated differences among treatments using Dunn's

multiple comparison *post-hoc* test.

I used a non-parametric multivariate analysis of variance (PERMANOVA) with

1,000 permutations to evaluate whether avian communities were compositionally

different among field treatments. PERMANOVA test the null hypothesis using random

permutations of the data and produces pseudo *F*-statistics. I performed each

PERMANOVA twice: once using Jaccard's dissimilarity index and once using the Bray-

Curtis dissimilarity index. I chose to use both indices because they each captured slightly

different information about the community under investigation. Specifically, Jaccard's dissimilarity uses only incidence data to compare species that are shared or unique among communities' whereas Bray-Curtis accounts for and compares abundance data. Tests where the null hypothesis was rejected were further evaluated using the pairwise comparisons via the pairwise.adonis2 function (Martinez Arbizu 2017) to conduct multiple comparison tests *post-hoc*. I evaluated community compositions within treatments using visual inspection of non-metric multidimensional scaling (NMDS) ordinations using each Jaccard's and Bray-Curtis dissimilarity indices.

Because vegetation data were also found to be non-normally distributed following Shapiro-Wilk's test for normality, I analyzed vegetation characteristics using the Kruskal-Wallis non-parametric ANOVA and, when significant, conducted *post-hoc* pairwise comparison tests (Dunn's test) to determine treatment-specific differences. Further, I performed a Kruskal-Wallis test to determine if arthropod biomass differed significantly by treatment, in addition to PERMANOVA analysis of arthropod community composition to identify potential differences in food resources available to avifauna among treatments. Finally, I used Spearman rank correlation analyses to evaluate the strength of relationships between individual vegetative and arthropod variables relative to avian abundance and diversity. I also evaluated correlations between the Haney index of soil health and other study variables (i.e., avifauna population metrics and microhabitat characteristics) using Spearman rank correlation.

In the field, I noted that landscape heterogeneity varied greatly within and around fields; accordingly, I calculated landscape heterogeneity using Shannon's diversity index at multiple spatial extents (400 m, 800 m, & 1600 m) from the centroids of study fields to

quantitatively compare landscape context. Using these indices, I performed a Kruskal-Wallis non-parametric ANOVA to test for significant variation among treatments followed by a Dunn's *post-hoc* comparison when significance was found. The strength of relationship between these indices and avian population metrics (i.e., total abundance, total richness, SGCN abundance, and SGCN richness) were also tested using Spearman regression, and, where appropriate, later used in a global model to account for variation not explained by other predictor variables.

Lastly, I combined all aforementioned predictor variables into a single global model for each avian population metric of interest. I evaluated variables in the global model for multicollinearity using Spearman rank correlation ($P \geq 0.10$) and Variance Inflation Factor (VIF) diagnostics. I refined the models by removing independent variables that were not good predictors for the population metric of interest or that had VIF > 10 (Dormann et al. 2013, Morrissey and Ruxton 2018) unless it was not ecologically justifiable to do so. Finally, I evaluated Poisson and negative binomial variance structures using Akaike's Information Criterion adjusted for small sample size (AIC_C) and selected the distribution with the least value. The resultant models were then used to estimate the direction and magnitude of effect of each predictor variable's parameter estimate on the corresponding population metric. I interpreted parameter estimates and their estimates of uncertainty using 85% confidence intervals (Arnold 2010) as this was the most conservative approach with the objective being to identify ecological influences of habitat characteristics on avian populations.

I further explored my data for any other emergent patterns of interest. Because grassland obligate bird species are undergoing the most extreme, consistent, and

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geographically widespread decline of any other avian guild (Knopf 1994, Samson and Knopf 1994, Vickery et al. 1999, Askins et al. 2007, Herse et al., 2017), I determined to use a Kruskal-Wallis test to ascertain if there were a difference by treatment in grassland obligate abundance and richness. The resultant test statistic approached, but exceeded, the *a priori* significance level ($\alpha = 0.10$), therefore I conducted no further tests on grassland obligates, but note that further research is needed to understand agricultural influences on populations of grassland obligate bird species.

Results

A total of 4,985 birds among 66 species were detected during field seasons of 2019 and 2021, 22 of which were in my focal group of Iowa SGCN (Tables 2, 3). Tabular summaries for the years of 2019 and 2021 are presented separately due to all following analyses pertaining only to 2021 data as a consequence of inherent limitations in the 2019 data set. The 10 cumulative most abundant species from both years were red-winged blackbird (*Agelaius phoeniceus*; 24.2%), dickcissel (*Spiza americana*; 20.7%), eastern meadowlark (*Sturnella magna*; 10.6%), common yellowthroat (*Geothlypis trichas*; 6.6%), killdeer (*Charadrius vociferus*; 4.1%), barn swallow (*Hirundo rustica*; 3.4%), field sparrow (*Spizella pusilla*; 3.2%), American robin (*Turdus migratorius*; 3.1%), song sparrow (*Melospiza melodia*; 2.4%), and grasshopper sparrow (*Ammodramus savannarum*; 2.1%).

Observed avian distributions among cover types differed between field seasons 2019 and 2021. Data collected in the spring and summer of 2019 (Figures 2a, b) revealed both greater abundance and richness of both total species and SGCN in cover cropped fields relative to perennial or row cropped fields. Total abundance and richness of

avifauna in perennial fields exceeded row crops, as did SGCN abundance; however,

SGCN richness in row crop fields ($\bar{x}$ = 6.7, 95% CI [1.5, 11.8]) marginally exceeded that

in perennial cover ($\bar{x}$ = 6.0, 95% CI [3.4, 8.6]). Conversely, 2021 data (Figures 3a, b)

indicated the greatest abundance and richness of total species and of SGCN in perennial

fields; total species abundance and SGCN abundance and richness were next greatest in

cover crops and least in row crops, however, total richness in row cropped fields ($\bar{x}$ =

20.0, 95% CI [13.5, 26.5]) exceeded that documented in cover cropped fields ($\bar{x}$ = 15.8,

95% CI [11.5, 20.1]). All following results pertain only to the 2021 data set and

subsequent values are adjusted for variances in search effort.

Results of a non-parametric ANOVA (i.e., Kruskal-Wallis test) indicated

variation in abundance of total species detected among cover types to be significant (χ^2 =

5.8509, df = 2, P = 0.053) as was total species richness (χ^2 = 6.3984, df = 2, P = 0.0408).

Results of a *post-hoc* non-parametric pairwise multiple comparison test (i.e., Dunn's test)

indicated that observed overall abundance in perennial fields was significantly different

from observations in cover cropped fields (P = 0.0402) and row crops (P = 0.083) when α

= 0.10, but variation between cover crop and row crop systems was not significant (P =

1.0). Similarly, overall richness in perennial fields was significantly different from cover

crops (P = 0.021), however, row crops were not significantly different from perennial

cover (P = 0.743) or cover crops (P = 0.296). SGCN abundance differed significantly by

treatment (χ^2 = 8.6761, df = 2, P = 0.013) as did richness of SGCN (χ^2 = 7.3906, df = 2,

P = 0.025). Pairwise comparison revealed SGCN abundances in perennial fields and

cover crops differed significantly (P = 0.007) as did perennial and row crops (P = 0.067),

but no significant difference was detected between row crops and cover crops (P = 1.0)

Similarly, SGCN richness in perennial fields was significantly different from both cover crops ($P = 0.019$) and row crops ($P = 0.049$) but differences in SGCN richness were not significant between row crops and cover crops ($P = 1.0$).

Avian communities among treatments were evaluated through permutational analysis of variance (PERMANOVA) with 1,000 permutations using 2 indices for dissimilarity. Results of a PERMANOVA using Jaccard's dissimilarity index signaled a significant difference in assemblages of total species by treatment among study fields (pseudo-$F_{2,20} = 2.64$, df = 2, $P = 0.001$) as did a PERMANOVA using the Bray-Curtis index (pseudo-$F_{2,20} = 2.41$, df = 2, $P = 0.008$). *Post-hoc* multiple pairwise comparisons using Jaccard's dissimilarity detected differences in avian community composition between perennial and cover crop fields ($P = 0.006$) and perennial and row crops ($P = 0.009$) but did not suggest a difference among cover crop and row crop treatments ($P = 1.0$). Similarly, *post-hoc* examination using Bray-Curtis' dissimilarity index indicated perennial cover differed significantly from cover crops ($P = 0.006$) and row crops ($P = 0.012$) but that cover crops and row crops did not significantly differ ($P = 1.0$). PERMANOVA results of SGCN assemblages using Bray-Curtis' and Jaccard's dissimilarity indices revealed observed variation to be significant among treatments (pseudo-$F_{2,20} = 2.97$, df = 2, $P = 0.004$) and (pseudo-$F_{2,20} = 4.43$, df = 2, $P = 0.001$) respectively. Following multiple pairwise comparisons, differences in SGCN diversity according to the Bray-Curtis index were significant between perennial and cover crops ($P = 0.003$) and perennial and row crops ($P = 0.009$) but not between cover crops and row crops ($P = 1.0$). Pairwise comparisons using Jaccard's incidence-based dissimilarity signaled significant differences between SGCN assemblages in perennial cover and cover

crops ($P = 0.003$) and perennial cover and row crops ($P = 0.009$) but not among cover

crops and row crops ($P = 1.0$). Inspection of NMDS ordination plots for both groups

using both indices (Figure 4) agreed with these preliminary findings by reflecting little to

no overlap between perennial cover and other treatments and a high degree of overlap

between cropland cover types. Additionally, in all plots the hull for cover crop

communities is more expansive than row crops, signaling that while cover crop fields

supported many or all of the same species as row crops, additional species are present in

cover crops not observed in row crops.

Percent vegetative cover of forbs was non-existent in row crop treatments and so

negligible in cover crops to the point it was considered a factor present only in perennial

cover. Non-parametric analysis of variance (i.e., Kruskal-Wallis test) of shared vegetative

characteristics among all treatments (i.e., percent cover of bare ground and litter, mean

ELH, and mean LD) (Figure 5) indicated that all variables were significantly different

among cover types ($\chi^2 > 4.61$, df $= 2$, $P < 0.10$) except mean percent litter cover ($\chi^2 =$

1.39, df $= 2$, $P = 0.499$). A Kruskal-Wallis test of percent cover of cash crop among cover

crop and row crop fields did not find a significant difference ($\chi^2 = 0.06$, df $= 2$, $P =$

0.808) (Figure 6). Ensuing *post-hoc* Dunn's tests indicated that percent cover of bare

ground was significantly different between perennial and cover crop ($P < 0.001$) and

perennial and row crop ($P = 0.056$) but not between cover crop and row crop treatments

($P = 0.61$); likewise, LD in perennial cover varied significantly from cover crop ($P <$

0.001) and row crop ($P = 0.036$) but not between cover crop and row crop treatments (P

$= 0.848$). ELH was determined to differ significantly between cover crop and row crop

treatments ($P = 0.065$) but not between either perennial and cover crop ($P = 0.102$) or

perennial and row crop ($P = 0.948$). ELH, however, was not endorsed by Spearman rank correlation to be a significant predictor of any avian metrics considered in this study (Table 4).

The relationship between availability and quality of food resources (i.e., arthropod biomass and arthropod richness) and avian metrics among cover types was first assessed using Spearman's rank correlation, which determined that arthropod biomass was a significant predictor and positively correlated with total avian abundance ($\rho = 0.42$, $P = 0.042$) and SGCN abundance ($\rho = 0.524$, $P = 0.009$) but was not a good predictor for either total avian richness ($P = 0.636$) or SGCN richness ($P = 0.618$) (Table 4). A Kruskal-Wallis test determined that mean arthropod biomass did not vary significantly by treatment ($\chi^2 = 0.38$, df = 2, $P = 0.827$). However, an evaluation of arthropod richness by family using Spearman's correlation coefficient did indicate a significant correlation with both overall avian abundance ($\rho = 0.608$, $P = 0.002$) and SGCN abundance ($\rho = 0.738$, $P < 0.001$), as well as with total avian richness ($\rho = 0.356$, $P = 0.088$) and SGCN richness ($\rho = 0.484$, $P = 0.017$). A Kruskal-Wallis test of arthropod richness by family among treatments determined variance was significant ($\chi^2 = 12.25$ df = 2, $P = 0.002$). *Post-hoc* examination (i.e., Dunn's test) suggested that arthropod richness in perennial fields significantly differed from other treatments, $P < 0.001$ for cover crops and $P = 0.04$ for row crops but found no such difference between cover crops and row crops (Figure 7). Results of a PERMANOVA of arthropod richness among treatments using Jaccard's dissimilarity index determined variance among treatments was significant (pseudo-$F_{2,20} = 2.12$, df = 2, $P = 0.007$) and pairwise comparisons suggested that perennial communities differed from cover crops ($P = 0.003$) and row crops ($P = 0.009$) but that crop treatments

did not differ ($P = 1.0$). Inspection of an NMDS ordination plot of arthropod communities among treatments using Jaccard's index (Figure 8) indicates that the community structure of arthropods present in perennial fields during this study was distinct from other treatments and that there is near total overlap of row crop and cover crop communities, however, the plot also indicates that cover crops support a larger diversity of arthropods than do row crops alone. Of all measured vegetative characteristics, only percent perennial cover ($\rho = 0.715$, $P < 0.001$) and litter depth ($\rho = 0.885$, $P < 0.001$) were determined to be positive predictors for arthropod richness, whereas percent litter cover had no relationship and all others were negatively related (Table 5). Mean Haney health score for soil was also found to be a significant yet negative predictor of arthropod richness in study fields ($\rho = -0.659$, $P < 0.001$).

Area of study fields was variable within and among cover types, ranging from 30 to 200 ha. Spearman regression reflected significant relationships between field area and total avian abundance ($\rho = 0.698$, $P < 0.001$) and SGCN abundance ($\rho = 0.649$, $P < 0.001$), however no significant relationship was detected between field area and richness of total species or SGCN. Landscape heterogeneity was quantified at multiple extents (400, 800, and 1600 m) from study fields' centroids. Further Spearman regressions indicated that landscape composition at any extent was not a good predictor for abundance of either total species or SGCN, however, each extent was found to be a good predictor for both total and SGCN avian richness (Table 4).

The final best-fitting model predicting total avian abundance (TAA) among cover crops, row crops, and perennial cover included predictor variables accounting for average proportions of ground cover by litter (Litter), perennial cover (Per), crops (Crop), and rye

(Rye), in addition to effective leaf height (ELH), litter depth (LD), Haney soil health score (Haney), arthropod richness (Arth), and field area (Hectares) (Table 6). Of these, Rye was positively associated with TAA and also had the greatest proportional influence on the dependent variable (β_{Rye} = 3.46, 85% CI [2.44, 4.47]); a 10% increase in mean area covered by rye was modeled to predict an increase in TAA of 2.7 additional individuals. Litter cover was found to be strongly positively associated with TAA (β_{Litter} = 1.43, 85% CI [0.66, 2.20]) and a 10% increase in mean area covered by litter in the model predicts an increase in TAA of 9.4 birds. Perennial cover was positively associated with TAA (β_{Per} = 1.16, 85% CI [0.53, 1.80]) and a 10% increase of which is predicted to lead to an increase in TAA of 3.0 birds. Percent ground cover by Crop was also a positive contributor to the model (β_{Crop} = 0.71, 85% CI [0.20, 1.22]) and a 10% increase is expected to cause a TAA increase of 4.3 birds. LD is positively correlated with TAA in the model (β_{LD} = 0.18, 85% CI [0.09, 0.28]) and predicts a 10% increase of which would result in 9.5 birds. ELH was found to be positively associated with TAA, however, the lower confidence limit approached zero (β_{ELH} = 0.03, 85% CI [2.44, 4.47]). Similarly, Hectares was modeled to have a positive influence, however its effect size was small and both limits near to zero ($\beta_{Hectares}$ = 0.003, 85% CI [0.001, 0.005]). Ten-percent increases in ELH or Hectares would be expected to lead to 3.5 and 3.9 birds respectively.

The final model predicting abundance of SGCN (SAB) within cover types of interest in this study included all the same variables as TAA but with differences in estimated effects (Table 6). Similar to the TAA model, Rye was positively associated and had the greatest proportional effect in the model (β_{Rye} = 4.91, 85% CI [3.76, 6.07]), a 10% increase in area covered by rye is predicted by the model to lead to an increase in

SAB of 1.7 birds of Species of Greatest Conservation Need. Percent area of Litter was also strongly positively associated with SAB ($\beta_{Litter} = 1.83$, 85% CI [0.92, 2.73]) and modeled to add 5.5 birds of SGCN following a 10% increase in area. Percent area covered by Per was positively influential in the model ($\beta_{Per} = 1.28$, 85% CI [0.56, 1.99]) and a 10% increase of which is expected to lead to an increase of 1.5 birds in SAB. Crop had a positive effect ($\beta_{Crop} = 1.31$, 85% CI [0.72, 1.90]) and predicted an additional 3.6 birds of SGCN following a 10% increase in the SAB model. LD and Arth had but positive effects in the model ($\beta_{LD} = 0.35$, 85% CI [0.25, 0.46]) and ($\beta_{Arth} = 0.02$, 85% CI [0.003, 0.05]) respectively, however, both were small and the lower confidence limit of Arth approached zero. Following a 10% increase in litter depth or arthropod richness, the model estimates increases in SAB of 8.6 and 2.5 birds respectively.

The final model predicting total avian species richness (TAR) among cover types included predictor variables signifying percent of area covered by Litter, Per, Crop, and Rye, as well as ELH, LD, Haney, Arth, and a predictor variable accounting for landscape composition within 800 m of sampled fields' centroids (Hi_800) (Table 6). Of these, Litter residue was positively associated with TAR ($\beta_{Litter} = 1.35$, 85% CI [0.23, 2.48]) and suggested an increase in TAR of 2.4 additional species for a 25% increase in average percent ground cover by litter in study fields. Model results also estimated that TAR was positively associated with ELH ($\beta_{ELH} = 0.06$, 85% CI [0.02, 0.09]), predicting an increase in TAR of 1.8 species for a 2.5 cm (i.e., 25%) increase in ELH. The parameter estimate for Hi_800 indicated a positive association with TAR, but the lower confidence limit was near to zero ($\beta_{Hi\_800} = 0.42$, 85% CI [0.01, 0.82]). Similarly, the parameter estimate for Haney soil index indicated a negative influence on TAR, but the effect was small and

equivocal because the upper confidence limit approached zero (β_{Haney} = -0.02, 85% CI [-0.04, -0.01]). Modeled changes in TAR following a 25% increase in landscape heterogeneity (Hi_800) or Haney are 1.9 and -4.1 species respectively.

The final model for richness of SGCN (SRI) among treatments included all the same predictor variables as that for TAR but with lessened estimates and confidence limits (Table 6). Of all included predictors, only the estimate for LD had a confidence interval that did not include zero (β_{LD} = 0.21, 85% CI [0.04, 0.38]) and the model predicted a 25% increase of which would lead to an additional 1.0 SGCN species in the modeled population. The effect of ELH in the model is minor but likely positive, however the confidence interval for the effect approached zero (β_{ELH} = 0.06, 85% CI [-0.001, 0.12]) and remains equivocal.

Discussion

I evaluated the impact and potential benefits of cover crop implementation on 2 groups of avifauna, total avian species and SGCN focal species, in the Corn Belt Region of the U.S. Midwest. My objective was to quantify various land use and microhabitat characteristics to identify management practices that may enhance habitat use by avifauna in intensively managed agroecosystems. My results suggest that many principles for managing general avifauna in agroecosystems through cover crops may also a be applied to avian SGCN. However, some factors influenced the two groups in unique ways. Initial findings indicate management that aims to promote litter structure may be key to conservation of avifauna in croplands.

Mean abundances and richness of total species and SGCN documented in my study differed greatly between 2019 and 2021. This was likely due to 3 factors, the first

being that the number of study fields and surveys per fields were less in 2019 than 2021

due to constraints imposed by the limited time and labor that were remedied in the second

year. Related to the first factor, avian survey data in 2019 were recorded by a single

observer due to the aforementioned constraints, whereas 2021 avian data were recorded

by 3 observers who alternated rotationally among fields to minimize observer-based bias

(Kinkead 2016). Finally, the 2019 field season was characterized by extensive rainfall

and flooding, with my study area receiving 27.9 cm of rain in May-June compared to the

11.4 cm received in the same period in 2021 (IEM 2022). This likely influenced avian

behavior such that it reduced detection probabilities at time of surveys (O'Connor and

Hicks 1980) and precipitation-induced changes of vegetation structure (e.g., earlier

termination of cover crops, reduced crop growth) as well as increased disturbance from

farmers who required more attempts at planting due to wet conditions.

In 2021, all observed avian population metrics were greatest in perennial cover,

followed by cover crops, and least in row crops, with the exception that total species

richness observed in row crops ($\bar{x} = 20.0$) marginally exceeded that recorded in cover

crops ($\bar{x} = 15.8$). The reason for this occurrence may be that any apparent difference was

spurious and a function of sample size, as I was able to survey 12 cover crop fields but

only 4 row crop fields in 2021. Additionally, many avian detections in 3 of 4 row crop

fields appeared to be associated with tree belts on field edges that may have supported

bird species not otherwise found in agricultural fields. Examples of these included

woodland or generalist species (e.g., gray catbirds, *Dumetella carolinensis*, and blue jays,

Cyanocitta cristata) rather than grassland species that comprised the majority of SGCN.

Thus, the proximity of survey stations in row crop fields to tree belts likely influenced the

lower observed SGCN richness in row crops compared to cover crops

Community Comparisons

Community compositions of total avifauna differed among field types, but the

difference lay between perennial cover and both cropland treatments (i.e., cover crops

and row crops), whereas variation in total avifauna among cropland types was not

significantly different. However, inspection of the NMDS ordination plot (Figure 4a)

revealed that the avian community (total species) observed in row crop fields were nested

within the community observed within cover crop fields. Thus, the avian community in

row crop fields hosted many of the same species as cover crop fields, but the more

expansive hull of the cover crop community suggests this cover type supported a greater

array of species, including some not observed in row crops. Results of permutational

analysis of total species using the Bray-Curtis index were similar to those obtained using

Jaccard's; however, there was lesser overlap between cover crops and row crops and a

greater degree of overlap between cover crops and perennial cover relative to plots using

Jaccard's index. The Bray-Curtis index, unlike Jaccard's, is sensitive to differences in

abundance, therefore it may be surmised from the increased overlap between perennial

cover and cover crop hulls in the Bray-Curtis plot (Figure 4b) that increased similarity

was likely due to the same few highly abundant species shared between the two

treatments. Because perennial and row crop hulls were distinct, yet both overlapped

portions of the cover crop assemblage, it appears that cover crops support some, but not

all, avian species that may be lost when grasslands are converted to row crops. That is,

cover crops appeared to bolster and enhance the observed avian communities by

supporting greater abundances and diversities than may otherwise be present in typical fallow/row crop rotation systems.

Permutational analyses of exclusively SGCN communities similarly revealed that observed variation among treatments was significant, but was due to differences between perennial cover and cropland treatments rather than differences among cover crops and row crops. Likewise, ordination plots of SGCN communities (Figure 4c, d) reflected similar patterns as shown by plots of total species, where cover crops overlapped with both perennial and row crop hulls, yet perennial and row crops remained discreet. This agrees with the interpretation of the former plots, suggesting that fields treated with cover crops hosted a broader array and abundance of SGCN than would otherwise be present in agricultural fields. Thus, use of cover crops may help to support or recover some individuals and species that would otherwise be lost in the conversion from perennial cover to agricultural use.

Some highly abundant and ubiquitous species in my study included red-winged blackbirds, killdeer, eastern meadowlarks, dickcissels, and common yellowthroats; these 5 species accounted for 68% of all detections in 2021, yet abundance of these species in either cover crops or perennial fields ranged from 1.5- to 5-fold greater than that found in row crops. Species that were present in either perennial or cover crop fields but not in row crops included field sparrows, grasshopper sparrows, purple martins (*Progne subis*), Baltimore orioles (*Icterus galbula*), red-headed woodpeckers (*Melanerpes erythrocephalus*), brown thrashers (*Toxostoma rufum*), chipping sparrows (*Spizella passerina*), and tufted titmice (*Baeolophus bicolor*); of these species, 6 are identified as SGCN in Iowa. Only 2 Iowa SGCN, American kestrel (*Falco sparverius, n* = 2) and

northern flicker (*Colaptes auratus, n* = 1), were present in row crops but neither in perennial nor cover crop fields.

Results of permutational analysis and NMDS ordination plots (Figure 8) of arthropod communities among treatments followed the same patterns as avian communities. Assemblages observed in perennial cover were distinct from cropland treatments, but row crop assemblages were nested within the more expansive grouping of cover crops. These results suggest that cover crop treatment did not recover many arthropod taxa lost in the conversion from perennial cover, but they did enhance the communities present in agricultural lands. Arthropod biomass was significantly correlated with 2 metrics, total avian abundance and SGCN abundance; however, no statistical difference in mean biomass was detected among treatments. Arthropod richness was found to be significantly correlated with all 4 avian population metrics and to explain more observed variation in both total abundance and SGCN abundance than did arthropod biomass. Thus, I included arthropod richness and not arthropod biomass as a predictor variable in my models.

Mean arthropod richness in row crops did exceed the mean observed in cover crops, but this may have been due to differences in sample size, which was 3 times as great for cover crops relative to row crops. Additionally, several traps were lost due to local flooding or being dug up by animals, further reducing sample size. These losses may also reflect bias in the arthropod data resulting from more "failed" traps occurring in fields with less litter structure.

Model Inferences

Many independent variables in the TAA model were positive predictors for total

avian abundance. The only variable in the model that was negatively associated with

TAA was the Haney score, but the 85% confidence interval centered on zero, suggesting

the effect was equivocal. Likewise, the confidence interval for arthropod richness

included zero and the overall relationship was uncertain. Rye had the greatest effect of all

predictor variables I evaluated, with a coefficient of 3.5, which suggested that a 10%

increase in mean percent ground cover by rye would yield an increase of ~3 additional

birds in the population. This result may have been a product of several factors due to

cover crops, including reduction of percent bare ground, provision of ground cover by

litter residue, enhancement of litter depth, and by protracting the period in which

vegetation was present to provide perches and cover for foraging compared to typical

fallow/row crop rotations.

Habitat needs of grassland avifauna vary widely, for example, several species

respond negatively to area of bare ground (e.g., grasshopper sparrow and bobolink,

Dolichonyx oryzivorus) whereas others benefit from bare ground and preferentially nest

in habitats, including croplands, where bare ground is proportionally greater (e.g., horned

lark and upland sandpiper, *Bartramia longicauda*; Dejong 2001, Greer 2009, Greer et al.

2016). Rye was also associated with increased cover by litter residue, which 3 of the 4

models indicated was a positive predictor of avian presence in this study. Likewise,

presence of rye was associated with improved LD, a variable that was also substantially

associated with avian abundance in my models. Interestingly, a literature review by

Fisher and Davis (2010) determined that the 3 most consistent predictors of habitat use by

grassland birds were bare ground cover, litter depth, and vegetation height. These habitat characteristics may all be influenced by presence of rye or other cover crops, potentially benefitting avifauna in agricultural production systems. The positive association between rye and TAA may also be influenced by further potential benefits offered by cereal rye or other cover crop types, e.g., increased provision of invertebrate prey (Wilcoxen et al. 2018, Brennan 2020).

The variable with the second greatest influence in the TAA model was ground cover by litter residue. Litter cover affects grassland bird species differentially, a relationship that further varies with scale and landscape context (Davis et al. 2021). Species that prefer dense cover, especially during nesting, have been found to be positively associated with increased litter (Lokemoen and Beiser 1997). For example, previous research has indicated that presence of savannah sparrows (*Passerculus sandwichensis*) was positively associated with frequency of litter cover (Greer 2009), as were several other species, many of whom are identified as SGCN (Hyde and Campbell 2012). Likewise, a study in Iowa (Basore et al. 1986) found that birds selected nest sites in fields associated with more litter residue, but not necessarily increased cover height. Litter residue in croplands may be critical for birds by providing cover, food resources for birds, and food for insects eaten by birds (Lokemoen and Beiser 1997).

Model results also indicated that litter depth was a positive predictor of TAA. This result was partly due to the direct correlation between LD and frequency of cover by litter residue (i.e., litter that was deeper was usually litter that had wider spatial coverage) and may have been influenced by associations between deeper litter and perennial cover in study fields. Moreover, increased LD was associated with habitat structures where

vegetation was taller and more voluminous and bare ground was reduced (Igl et al. 2018). Deeper litter may also provide birds with increased cover, improved habitat for both foraging birds and prey insects, and potential nesting materials. As with other habitat variables in the model, grassland bird responses to LD tends to vary by species and region (Bakker et al. 2002, Greer 2009). Previous studies have suggested that bobolinks and sedge wrens (*Cistothorus stellaris*) are positively associated with LD in grasslands, but that LD has a negative influence on horned larks and upland sandpipers (Dejong 2001, Igl et al. 2018). Savannah sparrows (*Passerculus sandwichensis*), a grassland obligate species, are negatively associated with increased LD in tallgrass regions but not in mixed-prairie, where litter accumulates more slowly (Bakker and Higgins 2009).

Frequency of crop ground cover was suggested to be a positive contributor in the context of my models. Reasons for this relationship may include reduced area of bare ground and increased vegetative cover and vegetation profile (e.g., ELH). Finally, model results supported ELH and hectares as positive predictors of TAA, but their estimated coefficients were small relative to other variables in the model and their confidence intervals approached zero. ELH may not have been a stronger predictor due to the widely varying needs of grassland birds and because of the high correlation with Litter coverage (Lokemoen and Beiser 1997), which was found to be more influential in 3 of the 4 models in this study.

The final model aimed at predicting SAB included all the same predictor variables as the TAA model; however, the variables with the greatest influence and estimated magnitude of effect on SGCN abundance differed from those of the TAA model. Similar to indications from the TAA model, the most influential predictor in the SAB model was

frequency of rye cover. The coefficient for Rye (β_{Rye} = 4.91) in the SAB model was greater than in the TAA model (β_{Rye} = 3.46). These results indicated that a 10% increase in mean rye cover in croplands is predicted to add an additional 2 birds that are SGCN to the local population. The predicted increase in SAB seems small relative to the magnitude of the coefficient, but this was chiefly due to mean frequency of rye coverage being negligible in many cover crop fields by the time of study beginning in early May. Later termination would prolong the duration of time in which living, standing rye is present and augment litter residue left by rye after termination. Notably, in the field I observed that the last cover crop field to be terminated also had the tallest rye and supported a higher abundance of birds, especially dickcissels, a species positively associated with ELH (Dejong 2001, Greer 2009). Following cover crop termination, this field continued to be one of the most productive in avian and arthropod abundance during the course of the study.

Similar to previous results, ground cover by crops and litter residue as well as LD were positively associated with SAB, likely due to the same mechanisms as with TAA. Interestingly, the model indicated that crop cover (β_{Crop} = 1.31) was marginally more influential than perennial cover (β_{Per} = 1.28) in predicting abundance of avian SGCN in agricultural production systems. Previous studies (Igl et al. 2018) have suggested that vegetative composition is less influential on grassland bird habitat use than vegetative structure, and it may be that crop structure provides some traits that are preferred by grassland birds. For example, increased ELH in corn fields relative to idle grasslands may provide higher perches for species positively associated with tall vegetation, such as dickcissels (Dejong 2001, Greer 2009). Unlike the TAA model, arthropod richness was a

significant predictor with a positive association in the SAB model, but the estimated effect was small and both confidence limits near to zero.

Many of the predictor variables included in the final model intended to predict TAR were shared by the abundance models and likely operated by the same or similar mechanisms. Of the variables included in the final model intended to predict TAR, the mean proportion of litter cover appeared to have the greatest influence on the dependent variable (i.e., total richness). As with abundance, this was likely due to provision of habitat for foraging birds and prey insects and potential nesting materials (Lokemoen and Beiser 1997, Wilcoxen 2017). The variable with the next greatest effect was landscape heterogeneity at the 800 m scale from fields' centroids. This result was expected because increased diversity of habitats can support increased diversity of birds; however, this is not always the case for grassland obligate avifauna. ELH was also positively associated with TAR. In addition to providing cover and perches and its positive association with litter residue and LD, increased ELH may support further invertebrate prey species not sampled by pitfall traps in my study. However, not all grassland birds share the same habitat needs and some may be negatively influenced by increased ELH (e.g., grasshopper sparrows; Eggebo 2001).

The only predictor variable in the TAR model with a negative coefficient and confidence interval that did not include zero was the Haney soil health index. The parameter estimates for the Haney score in each of the other models were also negative, but the effects were never substantial and confidence intervals near to zero. The negative relationship between Haney score and arthropod richness seems counterintuitive, implying that one or more soil characteristics measured in the Haney test may be

detrimental to arthropod communities and, by extension, prey resources for birds. However, the Haney score includes estimates of many nutrients and chemicals measured from soil samples that may have been influenced by exogenous fertilizer inputs or moisture at time of sampling (Spencer and Hahn 2017), which would obscure the true nature of the relationships in question. Moreover, it may be that the Haney soil health test was not the best indicator of soil conditions as perceived by these organisms and a better measure – more ecologically supported than production agriculturally based – may be needed to characterize the relationships between birds, arthropods, and soil in production agricultural systems.

The only predictor variable in the SRI model with a confidence interval that did not include zero was LD, and its estimated effect size was small ($\beta_{LD} = 0.21$). Overall, both richness models, (i.e., TAR and SRI) had weaker predictive power than did models of abundance (i.e., TAA and SAB). This may be intuitively explained by the widely varying habitat needs of grassland birds (Bakker et al. 2002, Murray et al. 2008, Ribic et al. 2009). This supposition was further supported by the fact the SRI model performed poorly compared to the TAR model, and a large proportion of the SGCN in the SRI model were obligate grassland species.

Management Recommendations

Habitat needs of grassland birds vary greatly by both species and region, as does the spatial scale at which they respond to habitat (Bakker et al. 2002, Ribic et al. 2009, Greer et al. 2016, Davis et al. 2021). Further, many of these species are sensitive to the context of the surrounding landscape and this influences their habitat selection and use (Bakker et al. 2002, Ribic et al. 2009, Shahan et al. 2017). Proper management of

working lands for the preservation of imperiled grassland birds is a modern conservation priority, and this should be achievable with a sound understanding of their habitat needs (Askins et al. 2007, Cox et al. 2014). In all cases, model estimates for predicting grassland bird habitat selection and use may be improved with further study, replication, and larger sample sizes. Considerations in future studies should include area sensitivity of different grassland bird species and effects of landscape fragmentation (Bock et al. 1999, Bakker et al. 2002, Murray et al. 2008, Ribic et al. 2009). Information about reproductive success in these systems would provide insights to support management decisions and associated demographic consequences (Blake et al. 2021).

Many factors need to be considered when evaluating cover crop management with respect to benefits to birds, the most influential of which and potentially most manageable may be timing of termination of cover crops (Wilcoxen et al. 2018). In each of my models, litter residue and litter depth were strong predictors of avian abundance and richness; indeed, litter depth was the only variable confidently associated with richness of SGCN. Later termination times of cover crops would allow for more accumulation of biomass leading to increased litter residue and depth; whereas earlier termination times may lead to more rapid decomposition of litter (Reeves 1994). Later termination of cover crops has been suggested to be associated with increased activity of beneficial arthropods; thereby enhancing food resources for birds (Campos 2021). A study similar to mine suggested that delayed termination of cover crops may improve nest success and mitigate potential negative consequences to late breeders in these systems (Wilcoxen et al. 2018). Results of my study corroborated that later termination was positively associated with improved abundances of total avifauna and avian SGCN.

Although delaying cover crop termination appears to broadly benefit birds, research is still needed to evaluate cover crop termination dates to identify specific time-periods that would maximally benefit birds. Wilcoxen et al. (2018) reported that if cover crop termination is not carefully timed with the breeding phenology of the local bird community, it could lead to individuals initiating nests that are subsequently destroyed or abandoned when the cover crop is terminated. This scenario could lead to wasted energy expenditure, increase risk of individual mortality, and may result in cover crops becoming an ecological trap. Gallman (2020) evaluated use of cover crops by nesting waterfowl and found that most nest mortalities were due to mechanical disturbance during cover crop termination and row crop planting. He surmised that later termination dates may instead attract more birds to nest in cover crops only to have their nests subsequently destroyed by farming practices – a scenario that suggested an ecological trap (Gallman 2020). Wilcoxen et al. (2018) suggested that termination after 1 May could be most beneficial, in contrast to the recommended termination date of prior to 1 May recommended by Gallman (2020). An important distinction when interpreting these contradictory findings is that Wilcoxen et al.'s (2018) primary focus was on passerines and birds of highest conservation need, whereas Gallman (2020) focused on waterfowl. Habitat needs of these avian guilds differ considerably, thus, cover crop management practices that benefit one group may be neutral or even detrimental to the other. Further study is needed to evaluate the impacts of cover crop termination dates, on avifauna in general and avian SGCN in particular. It may be that termination times that are maximally beneficial to avifauna will vary not just regionally but also by the composition of the immediate avian community.

The type of cover crop used may also differentially impact birds, but this topic has received little study. One report found that fields planted with cereal rye supported the greatest abundances relative to other cover crop types evaluated (Wilcoxen et al. 2018). It was speculated that the structure of cereal rye (i.e., density and height) may be more attractive to birds and may harbor more arthropod prey (Wilcoxen et al. 2018). Additionally, a study that evaluated winter cover crops in California suggested that birds that preferred feeding on arthropods and weed seed/fruit occurred most often in cereal rye and legume-rye cover crops relative to other types (Brennan 2020). Many plant species may be agriculturally appropriate for use as cover crops, but initial findings support cereal rye as likely being beneficial to birds in agroecosystems. However, further evaluation in needed to ascertain the relationships of cover crop species and avifauna.

Use of cover crops alone will not reverse the declines experienced by grassland birds in agriculturally dominated landscapes. Other sustainable agricultural practices should be integrated with cover crop use to maximize benefits and best promote conservation of threatened avifauna. One such practice may be no-till farming, which is thought to benefit grassland birds in highly fragmented landscapes (Dejong 2001) and has been found to augment bird diversity, nest density, and nest success relative to tilled fields (Higgins 1977, Basore et al. 1986, Duebbert and Kantrud 1987, VanBeek et al. 2014). Further, fields with minimum or no-till practices often have more litter build-up (Lokemoen and Beiser 1997), which my results suggested may be a crucial factor in avian habitat selection and use in agricultural landscapes.

One of the many potential benefits to farmers of cover crops is the option of providing forage for grazing cattle. Rotational grazing has been promoted as a

management tool that may be beneficial to grassland birds in agroecosystems by enhancing heterogeneity of vegetation structure through mimicry of historic natural disturbances (Temple et al. 1999, Fuhlendorf 2006, Vold et al. 2019, Buckley et al. 2022). Rotational grazing of cover crops may be a practice beneficial to both farmers and birds, however, grassland bird responses to grazing are not unilateral (Bleho et al. 2014, Ahlering and Merkord 2016, Davis et al. 2021) and study will be needed to determine whether grazing truly enhances cover crops as bird habitat or if it could lead to mismatches resulting in an ecological trap. Other sustainable agricultural practices that could be integrated with cover crops use to benefit birds include conversion to organic farming (Dejong 2001) and incorporation of beetle banks (Thomas 2001) and grassed waterways (Bryan and Best 1991) in crop fields.

Conclusion

Agricultural practices that are most advantageous to wildlife will tend to stimulate more natural habitat; thus, supplementation of additional vegetative structure in croplands may offer a way to improve habitat suitability to wildlife in agricultural landscapes (Wilcoxen et al. 2018). Use of cover crops in agriculture is a means of prolonging the period in which vegetative growth is present relative to typical cropping rotations, thereby providing additional cover in which birds may perch or forage during migration and the start of breeding season. Moreover, my results suggest that cover crops provide additional benefits to avifauna in agricultural production fields through augmentation of habitat available to birds even past termination of the cover crop. Three of 4 avian population metrics I examined (i.e., total abundance and abundance and richness of SGCN) were more positively associated with fields that had been treated with cover

crops relative to row crops with no such treatment. Results of modeling suggested this was primarily due to enhanced litter cover and depth and reduction of bare ground; additionally, total avian abundance and richness were positively associated with increased ELH in cover crop fields. Cover crops and their modification of cropland habitat may be beneficial to arthropod richness, whereby improving resources available to birds in these systems (Gómez et al. 2018, Wilcoxen 2017, Wilcoxen et al. 2018); however, I acknowledge that some results from my study were limited and equivocal, replication, and ideally expansion, of this work is needed to accurately characterize the influence of cover crops on arthropods as a food resource for birds in croplands.

Because fields treated with cover crops supported greater bird abundance and diversity than row crops, even recovering some grassland species otherwise absent from croplands, cover crops did indeed preserve a portion of the avian community otherwise lost in croplands but did not promote the same community observed in grasslands. Not all SGCN that I documented were observed in either cropland type (e.g., Henslow's sparrow, *Ammodramus henslowii*, and sedge wren); however, many SGCN that were present in at least one crop field were more prevalent in cover crops than row crops, with only a few exceptions (e.g., dickcissels and horned larks) and these differences were marginal. Delayed termination times of cover crops may offer one way by which cover crops can be managed to benefit avifauna and avian SGCN. It is imperative, however, that we continue to investigate how these species use cover cropped lands if we are to effectively manage Species of Greatest Conservation Need. Future efforts should seek to: 1) better understand the habitat requirements of individual species; 2) identify species not appearing to benefit from cover crops; and; 3) determine ways their individual habitat

needs may be addressed through combined management of cover crops and other sustainable agricultural practices (e.g., no-till and organic farming). This will be likely be a challenging process given the divergent habitat needs of grassland birds and their current imperiled status, but when achieved it will be greatly rewarding as the beauty and song of avian populations are restored in their historic territories, and with them all the ecosystem services they provide.

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Appendix

Figure 1. Visualization of 2021 study fields by cover type (i.e., perennial cover, cover crops, and row crops) in Washington and Henry Counties of southeastern Iowa, where avian habitat selection was evaluated in agroecosystems.
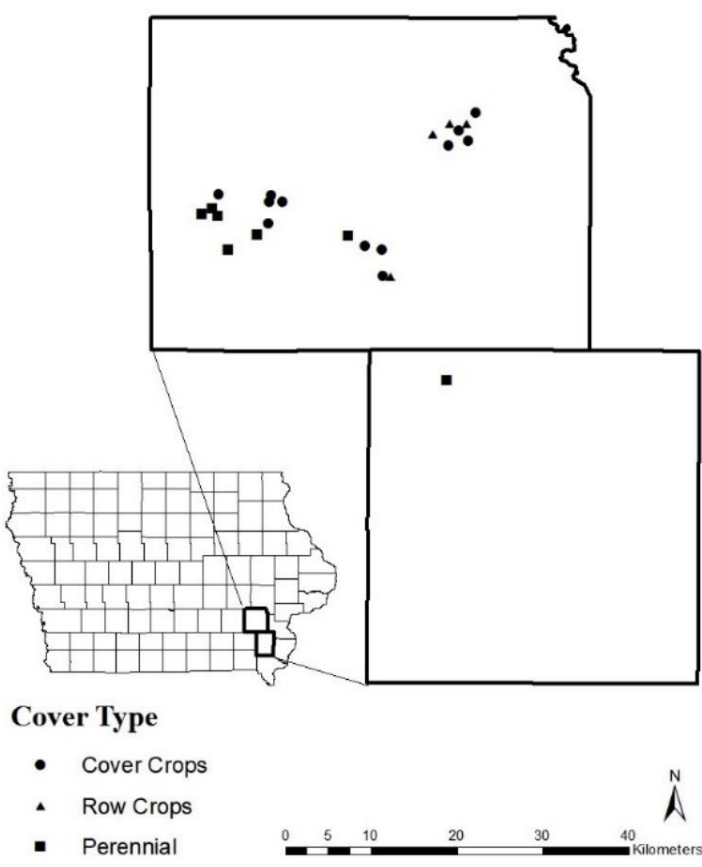
Table 1. Final sample sizes of each cover type (i.e., perennial cover, cover crops, and row crops) surveyed in 2021 as determined by landowner permissions, labor allowances, and available survey data.

	Survey 1 ($n = 16$)	Survey 2 ($n = 22$)	Survey 3 ($n = 22$)	Survey 4 ($n = 21$)	Survey 5 ($n = 19$)	Survey 6 ($n = 22$)
Cover Crop	7	12	12	12	10	11
Row Crop	3	4	4	4	4	4
Perennial	6	6	6	5	5	7

Table 2. Number of avian Species of Greatest Conservation Need (SGCN) detected and proportion of observations by cover type (i.e., perennial cover, cover crops, and row crops) in southeastern Iowa in the 2019 field season.

| Species	n	Cover Crop ($n = 5$)	Row Crop ($n = 3$)	Perennial ($n = 4$)
Northern Bobwhite *Colinus virginianus*	3	1.00		
Upland Sandpiper *Bartramia longicauda*	9		1.00	
Red-headed Woodpecker *Melanerpes erythrocephalus*	3	0.33		0.67
Northern Flicker *Colaptes auratus*	2	1.00		
Eastern Wood-pewee *Contopus virens*	8	0.75		
Eastern Kingbird *Tyrannus tyrannus*	5	0.60	0.40	0.25
Horned Lark *Eremophila alpestris*	22	0.55	0.45	
Purple Martin *Progne subis*	13	0.38	0.62	
Bank Swallow *Riparia riparia*	6	0.67		0.33
Sedge Wren *Cistothorus stellaris*	9	0.33	0.33	0.33
Common Yellowthroat *Geothlypis trichas*	66	0.33	0.09	0.48
Field Sparrow *Spizella pusilla*	38	0.58	0.11	0.32
Grasshopper Sparrow *Ammodramus savannarum*	10	0.30	0.20	0.50
Dickcissel *Spiza americana*	113	0.49	0.27	0.25
Eastern Meadowlark *Sturnella magna*	82	0.39	0.32	0.29

Table 3. Number of avian Species of Greatest Conservation Need (SGCN) detected and proportion of observations by cover type (i.e., perennial cover, cover crops, and row crops) in southeastern Iowa in the 2021 field season.

Species	*n*	Cover Crop (*n* = 12)	Row Crop (*n* = 4)	Perennial (*n* = 7)
Northern Bobwhite *Colinus virginianus*	5		0.20	0.80
Upland Sandpiper *Bartramia longicauda*	27	0.96	0.04	
Chimney Swift *Chaetura pelagica*	15			1.00
Red-headed Woodpecker *Melanerpes erythrocephalus*	6	0.67		0.33
Northern Flicker *Colaptes auratus*	1		1.00	
American Kestrel *Falco sparverius*	2		1.00	
Eastern Wood-pewee *Contopus virens*	19	0.47	0.21	0.32
Eastern Kingbird *Tyrannus tyrannus*	22	0.23	0.05	0.73
Horned Lark *Eremophila alpestris*	62	0.69	0.31	
Purple Martin *Progne subis*	4	0.50		0.50
Bank Swallow *Riparia riparia*	1			1.00
Sedge Wren *Cistothorus stellaris*	9			1.00
Wood Thrush *Hylocichla mustelina*	4	1.00		
Brown Thrasher *Toxostoma rufum*	8	0.50		0.50
Common Yellowthroat *Geothlypis trichas*	262	0.35	0.09	0.56
Field Sparrow *Spizella pusilla*	122	0.18		0.82
Grasshopper Sparrow	93	0.02		0.98

Ammodramus savannarum				
Henslow's Sparrow	30			1.00
Ammodramus henslowii				
Dickcissel	921	0.51	0.21	0.28
Spiza americana				
Bobolink	7	0.14		0.86
Dolichonyx oryzivorus				
Eastern Meadowlark	444	0.43	0.11	0.46
Sturnella magna				
Baltimore Oriole	3	0.67		0.33
Icterus galbula				

Figure 2. Summaries of mean observations of measured avian population metrics (i.e., total abundance, abundance of SGCN, total richness, and richness of SGCN) with standard error in 2019 in southeastern Iowa.
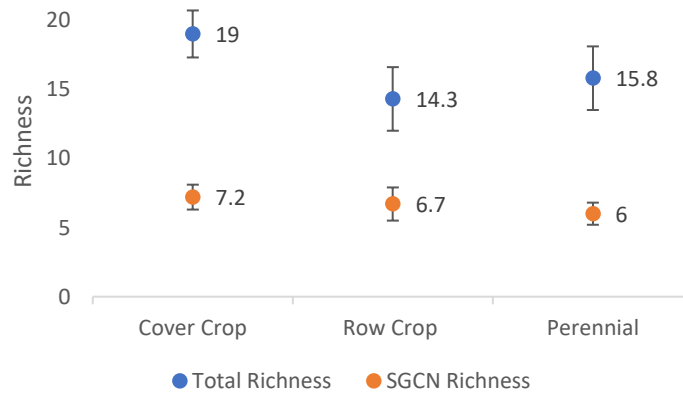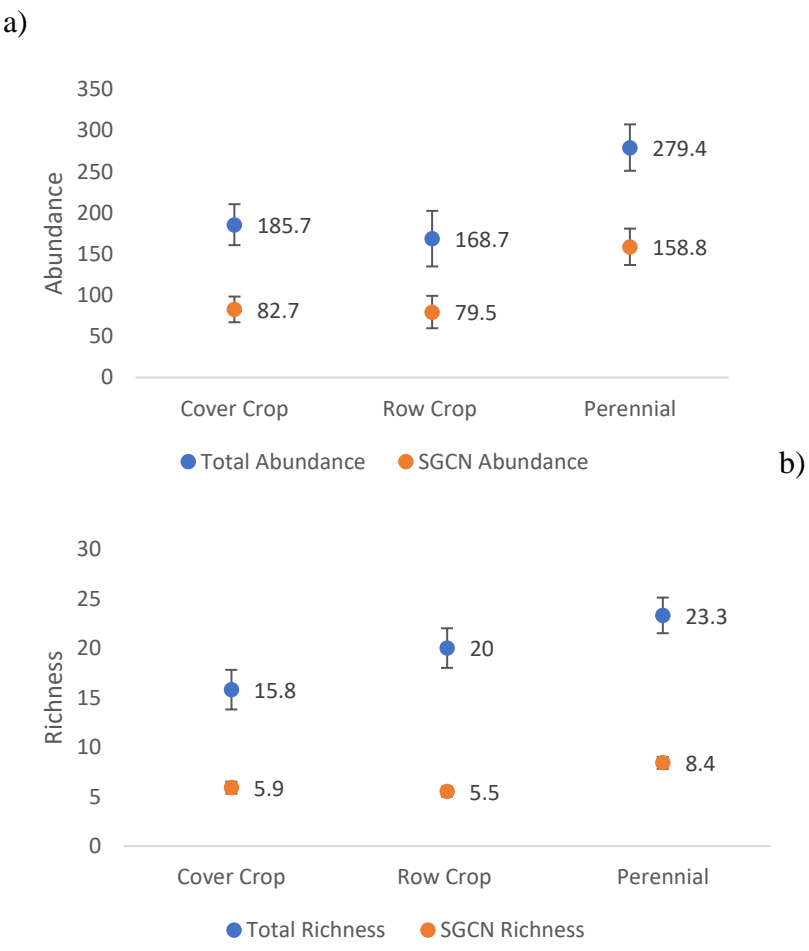
a)
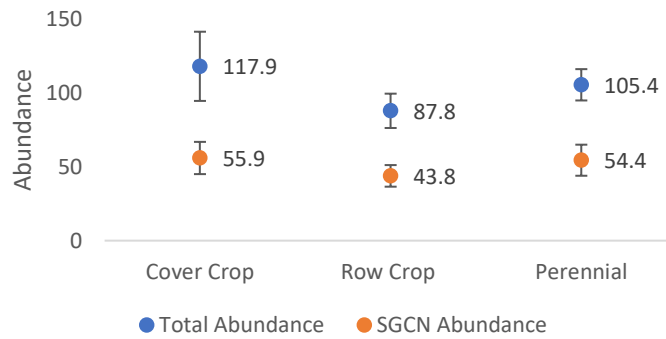
b)

Figure 3. Summaries of mean observations of measured avian population metrics (i.e., total abundance, abundance of SGCN, total richness, and richness of SGCN) with standard error in 2021 in southeastern Iowa.

a)

b)

Table 4. Correlation coefficients of study variables with observed avian population metrics (i.e., total abundance, SGCN abundance, total richness, and SGCN richness) where significance was determined as produced by Spearman rank regression using survey data collected in southeastern Iowa in 2021.

Variable	Biomass	Crop	Rye	Per	Litter	Bare	LD	ELH
Total Abundance	0.421*			0.520*		-0.590**	0.644**	
Total Richness		-0.44*	0.447*					
SGCN Abundance	0.524*	-0.475*	0.615**			-0.727***	0.772***	
SGCN Richness		-0.458*	0.602**	-0.511*		-0.412	0.477*	

	Hi_1600	Hi_800	Hi_400	Hectares	Haney	Arth
				0.698***	-0.436*	0.608**
		0.532**	0.595**		-0.545**	0.356
				0.649***	-0.481*	0.738***
	0.385	0.545**	0.487*		-0.571**	0.484*
	0.427*					

Figure 4. Non-metric multidimensional scaling (NMDS) ordination plots of observations of avian communities grouped in convex hulls by cover type (i.e., perennial cover, cover crops, and row crops) using two measures for dissimilarity from survey data collected in southeastern Iowa in 2021. a) Total avian species evaluated using Jaccard's index for dissimilarity, b) total avian species evaluated using the Bray-Curtis index for dissimilarity, c) avian SGCN evaluated using Jaccard's index, and d) avian SGCN evaluated using the Bray-Curtis index.
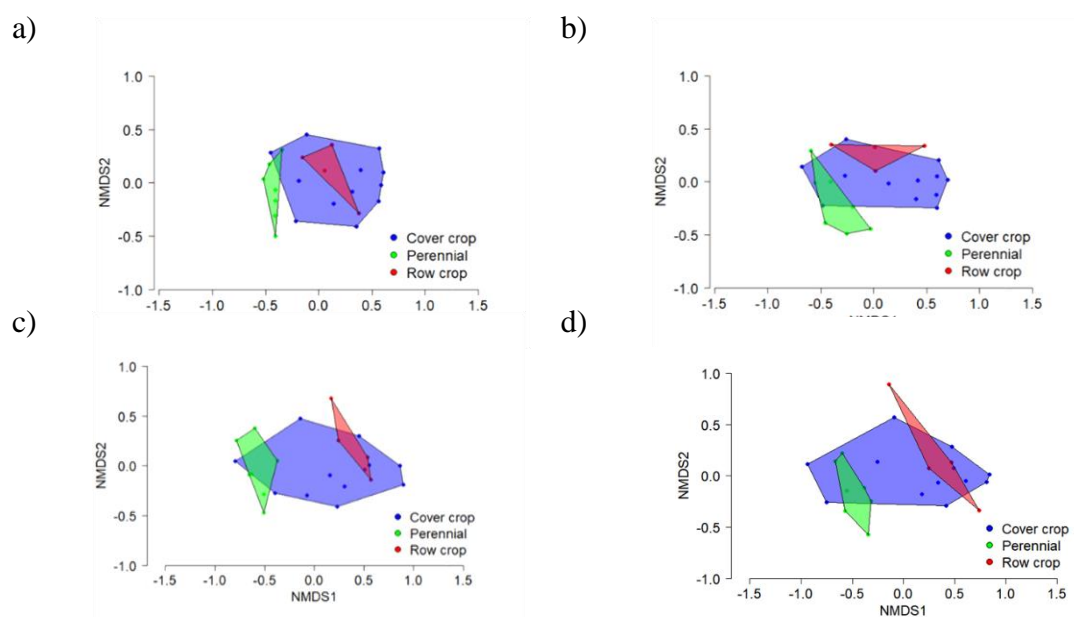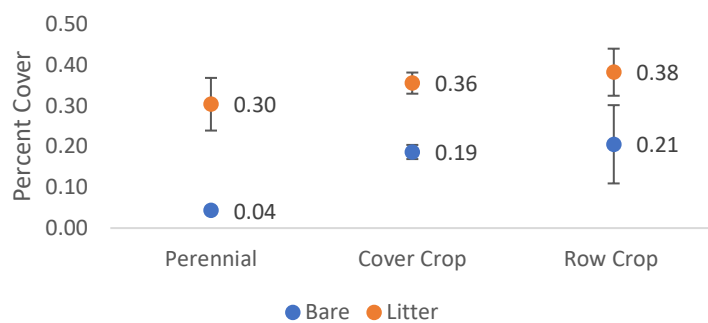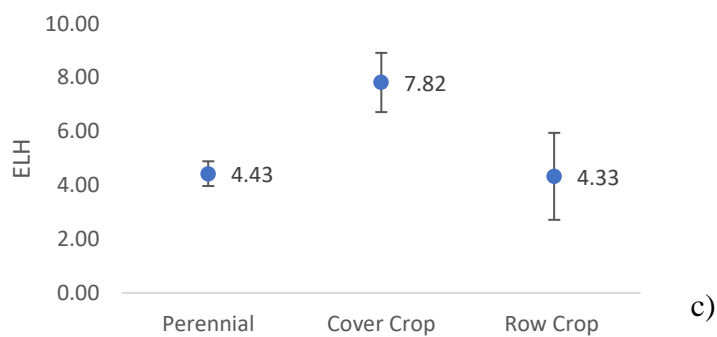
Figure 5. Mean comparisons with standard error for measured vegetative characteristics among perennial cover, cover crop, and row crop study fields in southeastern Iowa during 2021 field season.
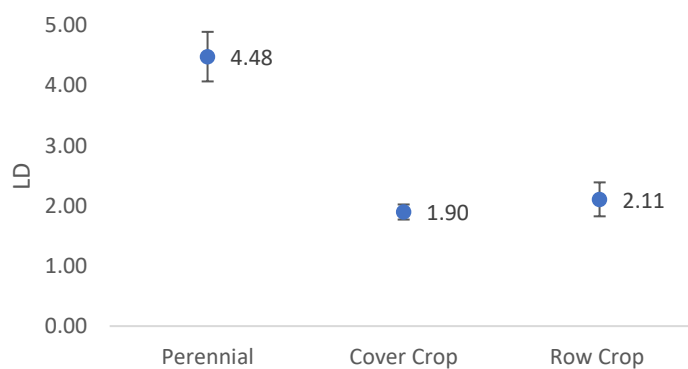
a)

b)

c)

Figure 6. Mean comparison with standard error of percent ground cover by cash crops among cover crop and row crop treatments in study fields in southeastern Iowa in field season of 2021.

Figure 7. Mean comparisons with standard error of arthropod richness (Arth) observed among cover types in southeastern Iowa in 2021.
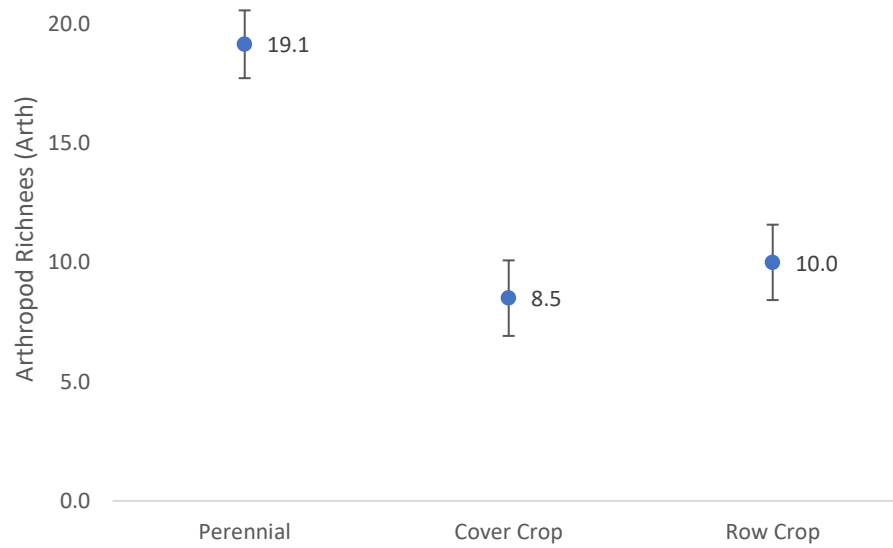
Figure 8. Non-metric multidimensional scaling (NMDS) ordination plot of observed arthropod richness (Arth) grouped in convex hulls by cover type (i.e., perennial cover, cover crops, and row crops) using Jaccard's Dissimilarity Index from survey data collected in southeastern Iowa in 2021.
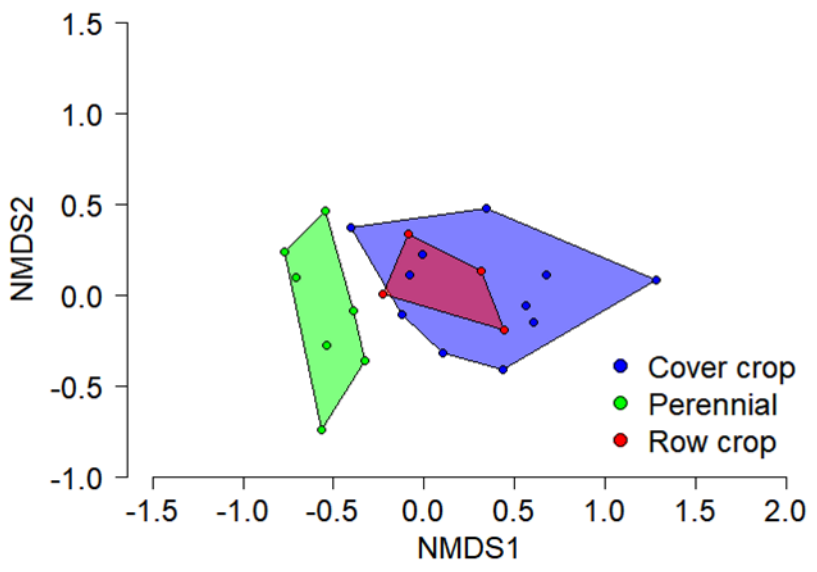
Table 5. Correlation coefficients of study variables with observed arthropod richness (Arth) where significance was determined as produced by Spearman rank regression using survey data collected in southeastern Iowa in 2021.

Haney	Crop	Rye	Per Litter	Bare	LD	ELH	Arthropod Richness
-0.659***	-0.661***	-0.388	0.715***	-0.829***	0.885***	-0.592**	

Table 6. Parameter estimates with 85% confidence intervals for influence of land use and microhabitat variables in modeled avian population metrics (i.e., total abundance, total richness, SGCN abundance, and SGCN richness) among cover crops, row crops, and perennial cover in study fields in Iowa for 2021.

	SGCN				Total Species			
	Richness		Abundance		Richness		Abundance	
	85% CI	β	85% CI	β	85% CI	β	85% CI	β
Litter	2.234 -1.228	0.503	2.731 0.924	1.828	2.478 0.228	1.353	2.200 0.661	1.430
Per	1.420 -1.105	0.158	1.988 0.562	1.275	1.362 -0.416	0.473	1.796 0.529	1.163
Crop	1.187 -0.641	0.273	1.900 0.720	1.310	0.977 -0.191	0.393	1.219 0.204	0.712
Rye	4.098 -1.047	1.526	6.071 3.756	4.913	1.426 -2.109	-0.341	4.472 2.441	3.457
ELH	0.118 -0.001	0.058	0.059 -0.008	0.026	0.094 0.018	0.056	0.058 0.003	0.031
LD	0.382 0.038	0.210	0.456 0.250	0.353	0.191 -0.047	0.072	0.275 0.091	0.183
Haney	0.009 -0.030	-0.010	0.006 -0.014	####	-0.007 -0.035	-0.021	0.002 -0.016	-0.007
Arth	0.043 -0.036	0.004	0.045 0.003	0.024	0.030 -0.024	0.003	0.025 -0.013	0.006
Hi_800	1.003 -0.238	0.383			0.823 0.007	0.415		
Hectares			0.003 -0.002	0.001			0.005 0.001	0.003