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Nesting Success of Dickcissel (Spiza americana) and Non-breeding Grassland Bird Use of Northwest Arkansas' Remnant and Restored Tallgrass Prairies

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Nesting Success of Dickcissel (*Spiza americana*) and Non-breeding Grassland Bird Use of Northwest Arkansas' Remnant and Restored Tallgrass Prairies

A thesis submitted in partial fulfillment
of the requirements for the degree of
Master of Science in Biology

by

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Abstract

Dwindling populations of North American grassland birds are linked to habitat loss. Tallgrass prairie only covers 3% of its pre-settlement-era range. Small-scale restoration projects attempt to increase acreage for prairie avifauna, and while some breeding grassland species are present at these sites, nesting success and non-breeding use are still largely unknown. Both life history aspects are required for effective grassland bird conservation. My first objective was to access nest success of the Dickcissel (*Spiza americana*) at two remnant and two restored tallgrass prairies in Northwest Arkansas. From May-August 2017 and 2018, I found 114 nests that I monitored to determine ultimate fate. I selected vegetative characteristics collected at nest and random sites combined with site-level variables to inform a logistic exposure model. Mean nest success was 8.5%, which varied by site but appeared unaffected by restoration status. Excluding predator presence, the most important predictors of nest success were site size size and brood parasitism. Prairie Kingsnake (*Lampropeltis calligaster*) was the primary nest predator. My second objective was to identify field site characteristics and vegetation use by non-breeding grassland-obligate birds in two remnant and three restored tallgrass prairies in Northwest Arkansas. Between September 2017 and May 2018, I tallied 44 species. Only eight species were grassland-obligate, but this assemblage accounted for about half of all detections. Grassland-obligate diversity was similar across seasons and between sites, except for a small isolated restored prairie which hosted much lower diversity. Dickcissel and Sedge Wren (*Cistothorus platensis*) detection rates increased significantly with distance to woody edge, indicating area sensitivity. Some species used primarily grass and forb, others utilized burned areas, and Le Conte's Sparrow (*Ammodramus lecontei*) frequented shrubs. Northwest Arkansas' remnant and restored tallgrass prairies are valuable for nesting Dickcissel and a suite of non-breeding

grassland birds. Special consideration for maintaining large parcels with fire that are distant from paved roads should be given for breeding Dickcissel. Acquiring large parcels and maintaining a shifting vegetation mosaic while retaining some woody vegetation could satisfy diverse habitat preferences for non-breeding grassland avifauna.

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Introduction

The main threat to wildlife populations globally is habitat loss and degradation (Bender et al. 1998). North American grassland birds are not an exception, as their populations have plummeted since the 1960s (Sauer et al. 2017). The conservation status of grassland birds, the most imperiled terrestrial bird assemblage in the continent (Sauer and Link 2011), is largely attributed to a decline in prairies. In particular, the central and eastern regions of North America have lost over 97% of tallgrass prairies since the pre-settlement era (DeLuca and Zabinski 2011). Remaining fragments may not be ideal for North American grassland birds. For example, many species are known to be area-sensitive on the breeding grounds, requiring large patches (Ribic et al. 2009a). Most grassland fragments are small and thus may not be occupied by some species. Roads also fragment the landscape and grassland birds often distance themselves from roads (Forman 2002, Sliwinski and Koper 2012). Nests are prone to higher brood parasitism and depredation rates in small fragments (Johnson and Igl 2001, Herkert et al. 2003), as woody encroachment facilitates access for edge habitat brood parasites and predators (Winter et al. 2000, Jensen and Finck 2004, Klug et al. 2010). Indeed, nest predation is the primary cause of nest failure in grassland birds (Pietz and Granfors 2000, Klug 2005, Ribic et al. 2012).

Some land managers are trying to reverse the downward trend in bird populations via completion of small-scale restoration projects. By sowing native plants, controlling exotic invasive plants, and creating disturbance regimes such as fire and grazing, managers attempt to mimic natural prairie conditions. These recreated habitats could play an integral role in connecting habitat patches in a fragmented landscape (Haddad et al. 2015). Restored prairies, many of which have been created through the United States Department of Agriculture’s Conservation Reserve Program (CRP), have attracted grassland birds, several of which are denser in restored sites than remnant sites (Johnson and Schwartz 1993, McCoy et al. 1999, Fletcher and Koford 2002, Ribic et al. 2009b). But high abundance does not confirm a viable population. Birds could be initially attracted to a site that yields low reproductive success (Pulliam 1988). Such population sinks occur for Dickcissel (*Spiza americana*) in abandoned fields and land restored via CRP in the United States (Zimmerman 1984, McCoy et al. 1999).

Nesting success is defined by the percentage of nests in a given area that produce at least one fledgling, and this can be a crucial measurement determining a restoration project's success. Management of restored prairies should focus on improving nesting success for grassland birds. Yet many CRP properties contain mostly non-native grasses and receive no prescribed burns (Vandever et al. 2002). Prescribed patch-burning and absence of non-native grasses have benefited nesting grassland birds (Fondell and Ball 2004, Churchwell et al. 2008, Monroe et al. 2016, Maresh Nelson et al. 2018), so identifying their influence on nesting success in non-CRP restored prairies is crucial. However, obtaining vegetative composition comparable to remnant tallgrass prairies has proven difficult in restoration projects, which may be disadvantageous for nesting prairie avifauna. More bare ground and taller and denser grass are characteristic of restored sites (Fletcher and Koford 2002, Ammann and Nyberg 2005), and vegetative characteristics are important for nest success of some grassland bird species (Fondell and Ball 2004, Berkeley et al. 2007, Ribic et al. 2012, Monroe et al. 2016). Yet it is unknown how these variables contribute to nesting success in remnant and reconstructed prairies.

Despite the fact that most North American grassland species are migratory (Herkert et al. 1996), habitat use of grassland birds during fall and spring migration and winter is understudied. Cryptic plumages, furtive behavior, and difficult weather conditions make surveying difficult (Fletcher et al. 2000). However, the notion of winter-limited species (Sherry and Holmes 1993) is growing in interest among avian biologists. Some forest-obligate species are negatively affected by deforestation on their overwintering grounds (Taylor and Stutchbury 2015), and winter habitat quality has carry-over consequences in the breeding season (Norris et al. 2004). Research on habitat use by non-breeding grassland birds has demonstrated the importance of patch size (Robertson et al. 2011), woody vegetation, (Pulliam and Mills 1977, Igl and Ballard 1999), and habitat heterogeneity (Baldwin et al. 2007, Hovick et al. 2014). To my knowledge, no studies have examined differences in non-breeding bird assemblages between remnant and restored tallgrass prairies.

My objectives were to determine variables that contribute to nest success of Dickcissel and to document grassland bird habitat association during the non-breeding season. In Chapter 1, I visited two remnant and two restored tallgrass prairies (size range: 6.5-27.5 ha) in Northwest Arkansas from May-August 2017 and 2018 to determine nesting success in the Dickcissel, a declining migratory grassland songbird (Sauer et al. 2017). Observing female behavioral clues led volunteers and I to discover nests, which I monitored until the nest was empty. I measured vegetation type composition and visual obstruction at nest sites paired with random locations. I also installed video surveillance cameras on a subset of nests to document nest predators. Here, I provide insight into the role of small remnant and restored tallgrass prairies for nesting Dickcissel by identifying variables that best predict nest success.

In my second chapter, I describe site selection and vegetation associations of non-breeding grassland avian assemblages in Northwest Arkansas. Between September 2017-May 2018, I conducted transect surveys at two remnant and three restored tallgrass prairies. I used these data to examine variation in species richness, diversity, detection rates, and vegetation association among sites and seasons, with particular focus on five grassland-obligate species. I also used simple linear regression to examine relationships between detection rates of focal species and distance to nearest woody edge. This research expands on knowledge of grassland bird occupancy in remnant and restored tallgrass prairies during the migratory and overwintering seasons.

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Chapter I

Dickcissel (*Spiza americana*) Nesting Success and Vegetation Characteristics in Northwest Arkansas' Remnant and Restored Tallgrass Prairies

Abstract

Of all North American bird-habitat guilds, prairie birds have declined the most drastically. One of the main reasons is extensive habitat loss. To increase acreage for grassland species, managers are restoring prairies by converting previously farmed land into prairies. Breeding bird surveys and nest success from studies on Conservation Reserve Program (CRP) restorations have been encouraging for some grassland species. However, nesting birds rely on particular vegetative characteristics to rear their offspring, and these variables may differ between remnant and restored prairies. The goal of this study was to evaluate the influence of vegetative and local landscape variables on nest success of a locally abundant but declining grassland species, the Dickcissel (*Spiza americana*). I located and monitored 114 nests at two restored and two remnant prairies in Northwest Arkansas in 2017 and 2018, and deployed surveillance cameras at 10 nests to identify nest predators. I measured vegetation composition and visibility at nests and paired random sites to assess variables influencing nest site selection. Principal components analysis revealed that, on average, Dickcissel nests were located in areas of lower but more variable visibility and lower bare ground/litter cover compared to random plots. A logistic exposure model of daily survival rate showed that mean nest success was 8.5%, but success varied by site (range: 0.9-19.5%) and appeared unaffected by vegetative characteristics, distance to woody edge or paved road, or nest height. Excluding predator presence, the most important predictors of nest success were site size and brood parasitism. Prairie Kingsnake (*Lampropeltis calligaster*) was a prominent predator. Special consideration for large sites and acquisition of new sites away from high-traffic roads should be given to enhance nesting success of this declining grassland songbird.

Introduction

Habitat loss and fragmentation are major threats to wildlife populations worldwide (Bender et al. 1998) and can result in drastic population declines (Dolman and Sutherland 1995). North American grassland birds, which are considered the continent's most imperiled terrestrial bird assemblage (Knopf 1994, Sauer and Link 2011), are no exception. The plight of grassland birds is often attributed to loss and/or degradation of native prairies, especially in the central and eastern regions of North America, where 97% of tallgrass prairies have been destroyed since the pre-settlement era (DeLuca and Zabinski 2011).

In prairie fragments that remain, degradation plays several detrimental roles in grassland bird recovery. Many breeding North American grassland bird species are known to be area-sensitive (Ribic et al. 2009a). Therefore, grassland fragments below a minimum size threshold may be devoid of certain species, even when the vegetative structure present is suitable (Herkert 1994). Roads also fragment the landscape, and some grassland bird species have shown avoidance of this edge type (Forman et al. 2002, Sliwinski and Koper 2012), perhaps due to traffic noise pollution interfering with singing birds (Brumm 2004). Nests in small fragments are prone to higher brood parasitism and predation rates (Johnson and Igl 2001, Herkert et al. 2003), as woody encroachment facilitates nest access for brood parasites and predators that prefer edge habitats (Winter et al. 2000, Jensen and Finck 2004, Klug et al. 2010a). The Brown-headed Cowbird (*Molothrus ater*) is pervasive in agricultural and suburban settings (Lowther 1993) and can negatively affect reproductive success in grassland birds (Winter 1999, Fondell and Ball 2004, Maresh Nelson et al. 2018), preferentially targeting nests placed higher off the ground (Fleischer 1986). Snakes are primary predators of bird nests in southern North America, particularly in early successional habitats (Thompson and Burhans 2003, DeGregorio et al.

2014). The Rat Snake (*Pantherophis obsolete* complex) is a frequent bird nest opportunist (Degregorio et al. 2014) that prefers woody edges compared to forest interiors (Blouin-Demers and Weatherhead 2001), and more often forages in the canopy (DeGregorio et al. 2014). Understanding nest predator ecology is especially important because nest predation has been identified as the main cause of nest failure in grassland birds (Pietz and Granfors 2000, Klug 2005, Ribic et al. 2012), whose nest predation rates generally surpass those of forest bird species (Martin 1993). In areas where perceived predation risk is high, some passerines decrease clutch size or forgo a second nesting attempt (Scheuerlein et al. 2001, Thomson et al. 2006), which also negatively affects reproductive success.

To reverse the trend of prairie loss, some landowners are recreating prairies through small-scale restoration projects. Targeted parcels often include previously farmed land, such as row-crop fields, pastures, and planted hayfields. In 2016 alone, the United States Department of Agriculture's Conservation Reserve Program (CRP) enrolled 204,000 ha from 34 states in its grassland category (Politsch 2016). By planting native seeds, controlling invasive exotic plants, and restoring disturbance regimes such as fire and grazing (to suppress forest succession), managers try to mimic remnant prairie conditions. Restoration may serve as an essential tool in maintaining habitat connectivity in our continent's fragmented prairie landscape (Haddad et al. 2015). Restored prairies have attracted some breeding grassland avifauna. For example, of 11 grassland-obligate species breeding in South Dakota (per United States Geological Survey's--from here on out, USGS--North American Breeding Bird Survey), eight were on average at least twice as dense in CRP fields than cropland (Johnson and Schwartz 1993). In Wisconsin, Bobolink (*Dolichonyx oryzivorus*) and Henslow's Sparrow (*Ammodramus henslowii*) appeared to favor CRP areas over remnant patches (Ribic et al. 2009b). McCoy et al. (1999) discovered that Missouri CRP grasslands were sources--where the population size is steady or increasing (Pulliam 1988)--for two declining grassland birds, Grasshopper Sparrow (*Ammodramus savannarum*) and Eastern Meadowlark (*Sturnella magna*). In Iowa, non-CRP restored prairies hosted greater densities of Grasshopper Sparrow and Savannah Sparrow (*Passerculus sandwichensis*) compared to remnant prairies (Fletcher and Koford 2002). Increases in abundance can be encouraging, but they may not be indicative of high nesting success.

Nesting success is defined by the percentage of nests in a given area that will produce at least one fledgling, which can be used as a measurement for determining a restoration project's success for breeding avifauna. A patch with relatively high bird density but low reproductive success is a sink; it attracts avifauna but fails to replace its population (Pulliam 1988). Sinks have been documented in Dickcissel (*Spiza americana*) in Kansas "old-fields" (assumed to be dormant hayfields or pastures; Zimmerman 1984) and in Missouri CRP grasslands (McCoy et al. 1999). It is unknown to what extent vegetative and local landscape variables contribute to nesting success in non-CRP restored prairies.

Management of restored prairies ranges from a hands-off approach, such as little to no burning and planting of exotic grass species in many CRP properties (Vandever et al. 2002) to a hands-on approach entailing rotational prescribed burns and exotic-invasive plant removal performed by restoration managers (Rowe 2010). The latter approach may bolster nesting success of grassland birds (Fondell and Ball 2004, Churchwell et al. 2008, Monroe et al. 2016, Maresh Nelson et al. 2018); however, even non-CRP restored tallgrass prairies may be missing some important nest site vegetative characteristics that birds select for shelter from weather (With and Webb 1993) and to conceal from predators (Götmark et al. 1995). Compared to remnant prairies, restored sites in Iowa had more bare ground and less vegetative cover (Fletcher and Koford 2002), and restored sites in Illinois had significantly taller and denser grass (Ammann and Nyberg 2005). Multiple vegetation variables have explained productivity of several grassland bird species (Fondell and Ball 2004, Berkeley et al. 2007, Ribic et al. 2012, Monroe et al. 2016), but no studies have examined vegetation and local landscape variables in the context of grassland bird nesting success in remnant and non-CRP restored tallgrass prairies.

The objective of this study was to identify vegetative and local landscape variables that affect daily nest survival rate in Dickcissel, a migratory grassland-obligate songbird (USGS North American Breeding Bird Survey) that is still relatively abundant in Northwest Arkansas (Neal 2009), despite a range-wide average population decline of 26% since 1960 (Sauer et al. 2017). I monitored Dickcissel nests at two remnant and two restored tallgrass prairies in Northwest Arkansas from May-August 2017 and 2018 to determine daily survival rate. I measured vegetation characteristics that could influence nest success at nest sites paired with random plots. I measured the burn interval and distance to the nearest paved road and woody edge to determine if nest success changed with increasing time and distance, respectively. Lastly, I installed video surveillance cameras at 10 nests to identify nest predators. I hypothesized that low nest placement, more forb cover, lower visibility, greater variation in visibility, and less bare ground/litter cover would increase nesting success. Increasing distance from edges (paved road, woody edge) and time since burn will increase nest success. Parasitized nests will have lower nest success rate than non-parasitized ones. I expected snake depredation to be the primary cause of nest failure on video monitored nests.

Methods

Study Sites

I selected two remnant and two restored prairie sites within Northwest Arkansas. Maps of each site are shown in Figure 1, and current and past management techniques by site are summarized in Table 1. Unless noted, grasses at each site are dominated by bluestem (*Andropogon* and *Schizachyrium* spp.), rye (*Elymus* spp.), and panicgrass (*Dichanthelium* spp), although not necessarily in that order. Average coverage by blackberry and other woody species (trees and roses) do not exceed 10% and 0.8% across sites, respectively.

Chesney Prairie Natural Area (hereafter: Chesney Remnant) of Siloam Springs, Benton County (36.2186847, -94.4821322) is owned by the Arkansas Natural Heritage Commission and is the largest remnant tallgrass prairie in Northwest Arkansas. Sections of Chesney Remnant were burned on December 10, 2016 and October 19, 2017 during or immediately prior to the study period. Grasses are the main plant type here. Chesney Prairie also encompasses 6.5 ha of restored prairie (36.223201, -94.483551; hereafter: Chesney Restored). On July 10, 2017, Chesney Restored was completely mowed (during the study period), and it was completely burned on December 10 2016 (immediately prior to study period). Grasses and forbs (non-grass herbaceous plants) are approximately equal in proportion at Chesney Restored. Stump Prairie (hereafter: Stump) is a privately-owned remnant in Siloam Springs, Benton County (36.204320, -94.494699). The last burn occurred on October 19, 2017 (during the study period). Grasses cover the most ground at Stump. Woolsey Wet Prairie Sanctuary (hereafter, Woolsey is a wetland mitigation project owned by the City of Fayetteville, Washington County (36.0672676, -94.2335263). Immediately before and during the study, prescribed burns occurred at Woolsey on February 25, 2017 and March 8, 2018. Unlike all other field sites, forbs are the most abundant plant group at Woolsey. Dominant grasses include several sedge species (Genera *Carex* and *Cyperus*), bluestem, and panicgrass.

Study Species

The Dickcissel (*Spiza americana*) represents a declining grassland bird that remains locally abundant, allowing for an adequate sample size. This medium-sized songbird breeds in tallgrass prairies, pastures, and fields of the United States and spends the winter in Venezuela. The central/eastern Great Plains represent the core of the breeding range, where Dickcissel is among the most abundant of grassland avifauna (Temple 2002). While still “fairly common” in Northwest Arkansas (Neal 2009), its population has dwindled by 26% from 1960-2014 (Sauer et al. 2017). Males are polygamous, and consequently rarely devote time to rearing offspring (Temple 2002).

Dickcissel nests are open-cup and are constructed primarily with grasses nestled in a clump of grass, forbs, or a shrub, or on the ground in litter (Dechant et al. 2002). The mean clutch size is 4 (range = 3-6), with one egg added per day, and incubation beginning the penultimate or day the final egg is laid. Eggs hatch after an average of 12 days of incubation, then chicks fledge after an additional 9 days (Temple 2002). Thus, the Dickcissel development period lasts approximately 21 days.

This is a frequent host species for Brown-headed Cowbird (*Molothrus ater*), a brood parasite (Friedmann 1963). Brood parasitism rates (percentage of nests parasitized in a sample) for Dickcissel range from 0% in Illinois (Robinson et al. 2000) to 95% in Kansas (Elliott 1978). When laying eggs, the female cowbird typically removes one host egg for each parasitic egg laid (Friedmann 1963, Zimmerman 1983). Female Dickcissel may desert her nest during building and egg-laying stages due to parasitism attempts (Zimmerman 1982).

Sampling Methods

Between May and August in 2017 and 2018, volunteers and I used female behavioral clues to detect Dickcissel nests (Winter et al. 2003): a bird carrying something in her bill--such as grasses for nest-building or arthropod prey for chicks--or a bird repeating a chip note in agitation. Whenever I observed one of these events, I retreated further away where I could still observe her and waited until she returned to her nest. Less often, I noticed a female fly directly to a nest from a distance, and occasionally, I incidentally flushed a female off her nest while walking or discovered a nest without any previous clues. I searched for nests throughout the day, although I often avoided the hottest and most inactive time of the day (13h-18h). Upon finding a nest, I marked its GPS location, placed a small flag about 9 m north for future relocation (Winter et al. 2003), and recorded nest contents. Excluding initial observations of empty nests (during nest-building stage), I revisited each nest on average every 4.70 d (± 2.56 (SD); range = 1 - 14 d) until success (defined as ≥ 1 Dickcissel chick fledging) or failure.

For assignment of nest fate, I determined that a nest was successful only if I observed fledging (i.e. chick leaves the nest on its own) in person or via video. Given the close proximity of nesting Dickcissels to each other (3 m; this study), coupled with movement of offspring after leaving the nest (Rivers et al. 2003), and that grassland birds may bring food to the nest for up to 24 hours after a complete depredation event (Pietz and Granfors 2000), I did not use observations of fledglings in the general vicinity of a known nest or parental behavior as signs of nest success for empty nests. In the absence of broken eggshells, I did not assume that an empty disheveled nest was predated, as the nest's position may have shifted in a fledging event (Pietz and Granfors 2000). The exception would be if chicks were not old enough to fledge, prematurely or naturally. Similarly, a virtually untouched empty nest does not eliminate the possibility of depredation, as most grassland bird nest predators do not leave evidence of their presence (Pietz and Granfors 2000), especially snakes (Thompson and Burhans 2003). Misinterpretations of these clues may have led to an overestimation of fledgling counts in one Dickcissel study (Rivers et al. 2003).

I estimated expected fledging date using the mean development period, and either incubation initiation date (assumed to be the day of clutch completion), hatching date, or by estimating chick age (based on body size and feather development; Temple 2002). Nests that were empty before day 19 were considered failed.

Within typically 1-2 weeks after each nest was deemed inactive, I took vegetation measurements both at the nest site and a paired randomly-distributed plot (Higgins et al. 1996) generated using GIS (ESRI, ArcGIS ArcMap, version 10.6). Each plot consisted of five 1.50-m^2 sub-quadrats (Cain and Castro 1959). Data recorded included ocular estimate of percent vegetative cover (i.e. grass, forb, woody, bare soil, etc.; Higgins et al. 1996), mean litter depth, grass and forb species richness, and percent visual obstruction. Using a vegetation profile board (Nudds 1977), I counted the number of visible squares on a vegetation profile board (Nudds 1977) from a distance of 4.5 m to determine visual obstruction. I used Google Maps to measure the distance between each nest and the nearest woody edge, which was an area either within or on the perimeter of the property that contained a minimum of 3 shrubs/trees measuring at least 1.5 m in height, with crowns touching (loosely adopted from Jensen and Finck 2004).

Nest Video Surveillance

Without technological aid, it is rarely possible to determine the cause of nest failure (Pietz et al. 2012b). To identify nest predators, I installed miniature surveillance cameras (Renfrew and Ribic 2003). Generally, cameras placed on North American birds' nests in open habitats only minorly reduce predation rates compared to nests without cameras (Richardson et al. 2009), so misrepresentation of actual nest depredation was not a concern. The cameras were powered by 35 amp-hr car batteries, continuously recorded in infrared, and were on the lowest quality setting to maximize the recording time before switching batteries. I chose nests based on camera availability, age of nest, and achieving similar sample sizes of video-recorded nests at each site. I usually installed cameras on nests when the nest age was at least 16 d, as female Dickcissels often abandon a nest if there is such a disturbance earlier in the nest life (Mitchell et al. 2012). I placed cameras approximately 15 and 29 cm away from and 46 cm above the nest. I trimmed vegetation between the nest and camera to improve visibility while attempting to maintain protective cover. I left the nests alone until the predicted fledging day or until the battery needed to be replaced, whichever came earlier.

Data Analysis

I used principal component analysis (PCA) to identify differences in vegetation between nest and random plots and between and within sites. The PCs on the first two axes informed the model set for nest survival analysis. Loadings greater than 0.40 were considered important. To determine if PC1 and PC2 differed among sites or plot types (nest vs. random), I used ANOVA.

Daily survival rate (DSR) is the probability that a nest survives on a daily basis, whereas nest success is the percentage of nests fledging at least one chick. To calculate the latter, I used the formula DSR^21, where 21 is the average development period of Dickcissel (Temple 2002) and thus represents the probability that a nest survives through the entire nest cycle. To calculate DSR and to determine the importance of different continuous and categorical environmental factors on Dickcissel nest success, I created a generalized linear model using the logistic exposure method (Shaffer 2004). The method assumes that nests are correctly aged; knowing the exact time of success or failure is not required and staggered entry of nests is permissible

(Rotella et al. 2004). I developed the model using the *nestsurvival* package (Herzog 2013) in program R, version 3.5.2.

Some grassland birds have been documented fledging 2-3 d early in the presence of a perceived threat (e.g. Pietz et al. 2012a), including nest checks by humans (e.g. Pietz and Ganfors 2000, Mitchell et al. 2012, this study). Forced fledging can even occur after a nest check (Pietz and Ganfors 2000). Because of the possibility of early fledging, and because I did not check nests every day, there was uncertainty about the fate of some nests (n = 23). These nests contained chicks that were physiologically capable of fledging (7 d; this study), either at the mid-point date between nest checks or within two days of the final nest check. I dealt with this uncertainty by excluding the final nest check interval of the aforementioned nests from the model. I also removed nests from the analysis where: eggs or chicks were never observed (n = 9), only visited once (n = 8), and lacking vegetation data (n = 2). Nests that failed due to early camera installation (n = 10) were only used if there was at least one nest check prior to camera installation (n = 6); the remaining camera nests (n = 4) were not used because there were no intervals. Thus, 83 nests were used in DSR analysis.

First, I created a base model set consisting of 16 models that incorporated individual and additive effects of nest age, date between nest check intervals, and year, as these are well known to impact DSR (Grant et al. 2005). I also added site to the base model set to account for differences I did not explicitly measure at the site level (i.e. size and surrounding landscape). Incorporating time-based variables such as nest age, date, and year can improve survival estimates when included with variables of interest (Skagen and Yackel Adams 2012). I then determined relative variable importance on this balanced base model set by adding the Akaike information criterion (AIC) weight (ω) of each variable across models. Variables whose sums exceeded 0.50 were included in all models in a final *a priori* model set focused on evaluating support for the effects of the following variables of interest on DSR: brood parasitism, days since last burn, distance to woody edge, distance to paved road, nest height, PC1, and PC2 (the latter two variables contain vegetation data collected at nest sites). I selected the best predictor models by identifying those with the lowest AIC scores and used model averaging for models with a Δ AIC < 2.00 (Burnham and Anderson 2002) to generate DSR values.

Results

Summary Statistics

Field volunteers and I located 43 nests in 2017 and 71 nests in 2018, totaling 114 nests. Chesney Remnant contained the most nests (n = 49), followed by Woolsey (restored) (33), Stump (remnant) (19), and Chesney Restored (13). Most nests were discovered during the incubation stage (46). Brooding and nest building were also frequently encountered (34 and 32, respectively; Figure 2). Clutch sizes averaged lower at Stump and Woolsey (3.67 and 4.00, respectively) than Chesney Remnant and Chesney Restored (4.43 and 4.50, respectively; Table 2). In cases where incubation and 2-day old nestlings (or older) were observed, egg loss averaged 0.48 eggs/nest (n = 48).

Vegetative Characteristics

Variables in PC1 explained 36.6% of the overall observed variance in vegetation. Important loadings were less grass cover (-0.92), more forb/blackberry cover (0.92) and lower visibility (-0.58). Variables in PC2 explained 21.5% of variance, and included higher variation in visibility (0.63), lower visibility (-0.59), and less bare ground/litter cover (-0.44). Exploratory analysis of 2018 plots, which included several variables not consistently collected in 2017 (mean litter depth, blackberry cover, and forb species richness) revealed that forb richness was strongly positively correlated with forb cover, and thus, other PC1 variables. Although this may be an important explanatory variable, a small sample size prevented further analyses.

Factorial ANOVAs revealed that PC1 values differed among sites ($F_{3,220} = 36.15$, $p < 0.01$), but not plot types ($F_{1,220} = 1.41$, $p > 0.05$). There was no interaction between site and plot type on PC1 ($F_{3,220} = 1.79$, $p > 0.05$). Both site ($F_{3,220} = 20.15$, $p < 0.01$) and plot type ($F_{1,220} = 13.30$, $p < 0.01$) differed significantly on the PC2 axis, but there was no interaction ($F_{3,220} = 1.59$, $p > 0.05$). When comparing sites on the PC1 axis, Chesney Remnant and Stump tended to have higher grass cover, whereas Chesney Restored and Woolsey contained higher forb/blackberry cover (Figure 3). On the PC2 axis, Stump and Woolsey had higher visibility and more bare ground/litter cover, while Chesney Remnant and Chesney Restored had lower visibility, greater standard deviation (from here on out, variation) in visibility, and more woody cover (Figure 3). Nests tended to have lower, but more variable visibility than random locations and were associated with lower and bare ground/litter cover (Figure 4). The most selected nest substrates by site were as follows: grass at Chesney Remnant (40.82% used versus 66.60% availability), forb at Chesney Restored (50% used versus 53.3% availability), blackberry at Stump (47.37% used versus 4.09% availability), and woody vegetation at Woolsey (45.16% used versus 0.81% availability). “Used” refers to the percentage of nests placed in a vegetation type by site, whereas availability was derived from average coverage in random plots.

Nest Success

Of the 104 nests where contents were found, 6.7% were confirmed to have fledged at least one offspring, 58.7% completely failed, and fate was unknown for the remainder (34.6%). Causes of nest failure were unknown (75.4%), depredation (11.5%), abandonment caused by humans (9.8%), and abandonment for non-human reasons (3.3%). The brood parasitism rate was 11%, and all parasitized nests were split between Stump (n = 7) and Woolsey (n = 4). Parasitism rates at those sites were 37% and 15%, respectively.

In the base model, two of the four variables best predicted DSR: nest age + site. The top model was nest age + site ($\omega = 0.28$), followed by year + nest age + site ($\omega = 0.21$), then date + nest age + site ($\omega = 0.14$). Date, year, and date + year had lower weights than the null model. Because the only variables whose cumulative weights exceeded 0.50 were nest age and site, I included them in every model in the final DSR analysis

Nest age + site was the top model in the DSR analysis and had 17% of the weight, followed by nest age + site + days since burn ($\omega = 0.14$), then nest age + site + parasitism ($\omega = 0.12$; Table 3). The remaining 11 models each received between 3 and 8% of the weight, which consisted of nest height, woody edge distance, paved road distance, PC1, and PC2.

Across sites, mean estimated nest success was 8.5% (95% CI: 1.0-27.9%), with a mean DSR of 89.0% (95% CI: 79.9-94.3%, Table 2). Daily survival rate relative to nest age and site split into two groups (Figure 5A): Chesney Remnant and Woolsey had higher mean nest success (19.2 and 10.9%, respectively) whereas Chesney Restored and Stump had lower mean nest success (2.4 and 0.9%, respectively; Figure 5A). Nest age and DSR always shared a negative relationship. The effects of time since burn on mean DSR varied by site; DSR remained constant with respect to burn interval at Chesney Remnant and Woolsey, whereas at Chesney Restored and Stump, mean DSR appeared to drop slightly as time since burn increased (Figure 5B). Parasitized nests had marginally lower mean DSR compared to non-parasitized nests based on overlapping confidence intervals (3.8% versus 9.8%, Table 2, Figure 5C).

Nest Predators

Of the 10 video surveillance cameras deployed, nine predation events on 8/10 nests were identified across four field sites (Table 4). The most prevalent predator was the Prairie Kingsnake (*Lampropeltis calligaster*), comprising 50% of all predators and recorded at two field sites. The remaining were Rat Snake (one field site, 12.5%) and an unknown small animal (37.5%). Of the eight predated nests, 75% resulted in completely depredated nests, whereas the remaining 25% resulted in partially failed nests where at least one fledgling evaded predation.

Discussion

On average, Dickcissel nest sites contained lower but more variable visibility and less bare ground/litter cover compared to random locations. Dickcissel nesting success was low and was most strongly influenced by nest age and site. Days since burn and brood parasitism affected nest success to a lesser degree. Dickcissel nest vegetative characteristics, distance from woody edge and paved road, and height did not strongly influence nesting success, which leads me to reject hypotheses that those factors would have a strong effect on nesting success. Although days since burn was one of the top predictors of nest success, two sites shared a negative relationship with days since burn, whereas the other two sites appeared unaffected. Thus, I reject my hypothesis that increasing time since burn would increase nest success. As predicted, parasitized nests averaged lower nest success than non-parasitized nests. Finally, I my hypothesis that snakes would be the primary nest predator was supported, but small sample size prevented me from determining whether snake predation was related to woody edge proximity.

Nest Site Selection

My results support other literature on Dickcissel nest site selection. I found that Dickcissel tended to select nest sites that had lower but more variable visibility and less bare ground/litter cover compared to random plots. Lower visibility and less bare ground cover was found to be important in successful Dickcissel nests in Missouri (Winter 1999), and higher vegetative volume and less bare ground compared to surroundings were found at Dickcissel nests in Kansas (Hughes 1996). Both results are logical, as Dickcissel nests are often hidden in dense vegetation (Dechant et al. 2002), presumably for better concealment from predators. Nest density of passerine and non-passerine grassland birds shares a positive relationship with the availability of areas with high visual obstruction in Montana (Fondell and Ball 2004).

Dickcissel may benefit from increased availability of vegetation with low visibility, especially at Stump and Woolsey. Both sites averaged greater visibility in random plots, compared to Chesney Remnant and Chesney Restored, but nest placement in blackberry at Stump and tree saplings at Woolsey were disproportionately high compared to availability at each site. Due to the relative lack of dense vegetative structure and height at sites like Stump and Woolsey, branching plants such as blackberry and tree saplings may offer greater visual obstruction. Consistent with my results that Dickcissel selected nest sites with more variable visibility, Churchwell et al. (2008) found that Dickcissel nest success in Oklahoma increased and brood parasitism rates decreased with heterogeneity-based management (different successional stages and vegetation within each stage). Such management is regularly recommended for grassland bird conservation because it also enhances grassland bird species richness (Coppedge et al. 2008) and increases abundance of some invertebrate prey taxa (Engle et al. 2008).

Nest success

The top three models in the nest success analysis were nest age + site, nest age + site + days since burn, and nest age + site + parasitism. Patterns were most evident in relation to nest age and site. Mean Dickcissel nest DSR in this study (0.890) is comparable to reclaimed surface mines in Indiana (0.903; Galligan et al. 2006) and restored tallgrass prairies in Texas (0.895; Lituma et al. 2012). Daily survival rate was negatively related to nest age, which is expected because more nest (exposure) days increases the probability that a predator detects a nest.

Chesney Remnant and Woolsey had relatively high mean DSR (range: 0.900-0.924). These sites are both fairly large (27.5 and 18 ha, respectively), and increasing patch size has been shown to benefit Dickcissel nest success. Nest success in small prairie fragments in Missouri averaged only 9%, whereas nest success in large fragments nests averaged 31% (Winter 1996), and nest depredation rates of several grassland bird species decreased with increasing fragment size in Minnesota (Johnson and Temple 1990). Chesney Remnant also averaged higher visual obstruction by vegetation and greater variation in visual obstruction, which could both explain higher nest success there (Winter 1999, Churchwell et al. 2008).

Chesney Restored and Stump had significantly lower DSR (0.838 and 0.799, respectively), perhaps due to their relatively small size (6.5 ha each; Johnson and Temple 1990, Winter 1996). However, small sample sizes may complicate this trend. Nest depredation could have been particularly high at Chesney Restored, because restored sites may appeal to high densities of nest predators (Zimmerman 1984). The higher bare ground to vegetation ratio in restored grasslands (Fletcher and Koford 2002) may facilitate thermoregulation of snake predators (Blouin-Demers and Weatherhead 2001) and Chesney Restored had one of the highest average bare ground/litter cover (4.7%), preceded only by Stump (7.3%). Less bare ground cover increased Dickcissel nesting success in Missouri (Winter 1999). Chesney Restored had more bare ground/litter cover than another restored site, Woolsey (average: 2.8%). The difference in bare ground/litter cover could be attributed to the conservation management timeline. Chesney Restored's restoration began in 2015 (2-3 years before this study), whereas Woolsey's restoration started in 2008 (9-10 years before this study). Thus, Chesney Restored will likely require more time to produce more dense vegetation, particularly regenerating grasses (Lituma et al. 2012). Mundahl and Bosari (2016) also demonstrated that several grassland bird species were more abundant in older restorations than younger ones, perhaps alluding to vegetative differences between newer and older restorations. Therefore, it is possible that vegetative composition in Chesney Restored will become more favorable for Dickcissel nesting success.

There were also vegetation differences between sites. High grass cover was most prevalent at Chesney Remnant and Stump; grass was positively related to Dickcissel nest success in Missouri (Winter 1999). Woolsey had greater forb/blackberry cover; dense forb structure at the nest increased fledgling survivorship in Iowa and Nebraska (Berkeley et al. 2007). Neither plant type strongly influenced Dickcissel nest selection or nest success in this study, which may explain such variable results from Winter (1999) and Berkeley et al. (2007). One caveat in interpreting vegetation data is acknowledging that vegetation changes between nest site selection and brooding (Burhans and Thompson 1998). Because I waited to collect vegetation data until after nest termination to avoid stressing the birds, an important vegetative variable may have been missed. However, to obtain more accurate vegetation data, one would have to accept increasing the risk of nest abandonment (Winter et al. 2003).

Although burn interval was a top model in the nest success analysis, there was no consistent pattern across sites between burn interval and DSR. Chesney Remnant, Chesney

Restored, and Stump are burned on 3-yr rotations, yet none of them shared a similar DSR pattern in relation to days since burn. Woolsey, which is burned in its entirety each spring, was one of the most successful nest sites for Dickcissel. This counters Churchwell et al.'s (2008) findings that annual burning decreases Dickcissel nesting success compared to patch-burning, and counters Stochat et al.'s (2005a) results that Dickcissel nests are less numerous on burned grasslands compared to un-managed grasslands. Conversely, Johnson and Temple (1990) demonstrated that burns occurring in the past year resulted in lower nest depredation rates for several grassland bird species in Minnesota, yet abundance of reptilian and avian nest predators was highest in Oklahoman prairies burned in the previous 12 mo (Stochat et al. 2005b). The relationship between Dickcissel nest success and fire remains ambiguous in this study and in the literature. Since prescribed burning is an effective tool in suppressing woody vegetation succession in prairies (Briggs et al. 2002) yet can negatively affect Dickcissel nest success (Churchwell et al. 2008), it is recommended that grasslands are burned in a patch-mosaic pattern rather than complete burning over a 3 yr period (Dechant et al. 2002). This strategy still provides some cover for nesting Dickcissel.

Dickcissel nests that were parasitized averaged lower DSR than un-parasitized nests. Brown-headed Cowbird brood parasitism is well-known to decrease Dickcissel nest success (Elliot 1976, Fondell and Ball 2004, Maresh Nelson et al. 2018). Even nests containing cowbird chicks that were deemed successful typically fledged fewer Dickcissel offspring than non-parasitized nests (Winter 1999). Zimmerman (1983) found that 8.9% of females abandoned their nests during egg-laying and incubation periods due to parasitism. Reduced nest success due to brood parasitism could be caused by increased clutch sizes from added cowbird eggs. Larger clutch sizes increase incubation periods, thus increasing the number of exposure days to predators (Zimmerman 1983). Cowbird chicks also divert food resources from the host`s offspring, often resulting in the host parent removing its own weakened offspring from the nest (Friedmann 1963). The average parasitism rate of 11% in this study was relatively low, but within the known range of 0-95% for Dickcissel (Elliott 1978, Robinson et al. 2000). This wide variation is likely reflective of regional differences in Brown-headed Cowbird density (Basili 1997); Arkansas is considered an area of low to moderate cowbird density (Sauer et al. 2017). Brood parasitism only occurred at Stump and Woolsey. One similarity between these sites is that both share a border with a paved road. Brown-headed Cowbird has demonstrated decreasing abundance with increasing distance from roads (Sliwinski and Koper 2012), therefore acquiring land not adjacent to paved roads could help decrease rates of brood parasitism. While models containing distance to woody edge were not strongly supported, Brown-headed Cowbird is often associated with woody edge (Johnson and Igl 2001, Winter et al. 2000, Jensen and Finck 2004). Reducing woody edge could eliminate lookout perches for cowbirds (Gates and Gysel 1978), and thus lower brood parasitism rates.

Nest Predators

Almost all video-monitored nest failures (7/8) were due to predation, which has been documented as the primary cause of grassland passerine nest failure in numerous studies (Pietz and Granfors 2000, Klug 2005, Ribic et al. 2012). Snakes were the most prevalent nest predators, accounting for all identifiable predators (5). These results support other conclusions that snakes are the main nest predators of Dickcissel (Zimmerman 1984, Klug et al. 2010b). Prairie Kingsnake was most frequently observed consuming nest contents, followed by one Rat Snake observation. Although Rat Snake was one of the prominent predators captured on bird nest videos in early successional habitats in the eastern United States (DeGregorio et al. 2014), it was not the most frequent nest predator in this study. Prairie Kingsnake has never been documented as the prominent nest predator in video surveillance studies, although it has been documented as a sub-dominant field passerine nest predator to Rat Snake in Missouri (Thompson and Burhans 2003). However, Prairie Kingsnake is one of the most abundant large snakes at Woolsey and outnumbers Rat Snake (Baecher et al. 2018). Prairie Kingsnake may be more abundant in grassland habitats in Northwest Arkansas than grasslands in other regions, and thus a more important nest predator to prairie birds like the Dickcissel. Woody edge removal is often recommended as a means to decrease snake activity and/or abundance, consequently lowering nest depredation rates in species whose primary nest predators are snakes (Thompson and Ribic 2012). However, Prairie Kingsnake prefers grasslands and does not appear to frequent woody edges (Richardson et al. 2006), so woody edge removal may not be sufficient in lowering depredation rates by Prairie Kingsnake. Future studies focusing on identifying Dickcissel nest predators using more and higher quality nest cameras, along with estimating Prairie Kingsnake densities in prairie fragments are needed.

Conclusion

Dickcissel breeds in remnant and restored tallgrass prairies of Northwest Arkansas. Nest success varied by site but was not obviously related to restoration status, suggesting that restored prairies could be valuable additions to the landscape for Dickcissel. I found that nests sites were characterized by lower but more variable visibility and less bare ground/litter cover compared to available habitat. Prairie management should maintain these vegetative characteristics to attract nesting Dickcissel. Larger sites yielded higher nest success, so protecting larger prairie parcels could be crucial for Dickcissel nest success. Brown-headed Cowbird parasitism resulted in lower nest success than non-parasitized nests. Brood parasitism occurred at 2/4 sites, both of which lie immediately adjacent to paved roads that may have increased cowbird abundance. Therefore, habitat conservation efforts should focus on acquiring properties distant from paved roads. Prairie Kingsnake was the primary predator documented on nest cameras. Contemporary suggestions of reducing woody edges to lower nest depredation rates by snakes may not be adequate in managing Prairie Kingsnake, as this species does not associate with woody edges. Future studies should focus on identifying the role of prescribed burns on nest success and better defining the Dickcissel nest predator community from more sites in Northwest Arkansas.

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Tables and Figures

Table 1. Summary of field site management in Northwest Arkansas, USA. Chesney Remnant = CHRM, Chesney Restored = CHRS, Stump = STUM, and Woolsey = WOOL. For status, rem = remnant, and res = restored. “Planted” under previous land use refers to non-native cattle forage (tall fescue). Burn rotation refers to interval length between complete site burns. Under mow, herbicide, and woody removal, A = annual, H = historically (but not currently), and O = occasionally.

Site	Status	Size (ha)	Managed since	Prior land use	Native plantings	Burn (y)	Mow	Herbicide	Woody removal
CHRM	rem	27.5	2000, 2015	hayfield	no	3	no	yes, A	yes, A
CHRS	res	6.5	2015	bean field; plowed	yes	3	yes, A	yes, A	yes, A
STUM	rem	6.5	1998	hayfield	no	3	no	yes, O	yes, O
WOOL	res	18	2008	pasture; planted	some	1	no	yes, A	yes, A

Table 2. Reproductive parameters for Dickcissel in Northwest Arkansas (USA) remnant and restored tallgrass prairies. Clutch size only includes nests where full clutch sizes were known. Egg counts exclude Brown-headed Cowbird eggs (but include parasitized nests). Parasitized/non-parasitized nests only contain nests where contents were known (n = 104). Nest success is calculated by daily survival rate (DSR)^21, which is the number of Dickcissel development days (Temple 2002). Both DSR and nest success were derived from model averaging, using models with Δ AIC less than 2.00. Number of nests column refers to nests used in DSR analysis. All sites combined = ALL, Chesney Remnant = CHRM, Chesney Restored = CHRS, Stump = STUM, and Woolsey = WOOL.

Group	**Clutch size (n)**	**DSR (95% CI)**	**Nest success (95% CI)**	**Intervals**	**# of nests**	**Total # of nests**
ALL	4.17 ± 0.68 (64)	88.95 (79.92-94.32)	8.55 (0.90-29.26)	199	83	114
CHRM	4.43 ± 0.50 (28)	92.45 (86.45-95.94)	19.46 (4.70-41.85)	94	36	49
CHRS	4.50 ± 0.55 (6)	83.77 (63.55-94.06)	2.42 (0.01-27.65)	11	5	13
STUM	3.67 ± 0.98 (12)	79.94 (65.35-89.54)	0.91 (0.01-9.82)	37	18	19
WOOL	4.00 ± 0.49 (18)	90.01 (81.77-94.80)	10.98 (1.46-32.56)	57	24	33
Parasitized	3.30 ± 0.95 (10)	85.63 (74.74-92.42)	3.85 (0.22-19.08)	30	11	11
Non-parasitized	4.33 ± 0.48 (54)	89.54 (80.84-94.65)	9.82 (1.15-31.54)	169	72	93

Table 3. Akaike information criterion (AIC) model output table exploring predictor variables for Dickcissel daily survival rate in remnant and restored tallgrass prairies of Northwest Arkansas, USA. Shown are number of parameters (*K*), Δ AIC, and model weight (ω).

Model	***K***	**Δ AIC**	**ω**
nest age + site	5	0.00	0.17
nest age + site + days since burn	6	0.35	0.14
nest age + site + parasitism	6	0.72	0.12
nest age + site + nest height	6	1.55	0.08
nest age + site + woody edge distance	6	1.62	0.07
nest age + site + PC1	6	1.65	0.07
nest age + site + PC1 + days since burn	7	1.83	0.07
nest age + site + PC2	6	1.99	0.06
nest age + site + paved road distance	6	2.03	0.06
nest age + site + PC2 + days since burn	7	2.36	0.05
nest age + site + PC1 + nest height	7	2.84	0.04
nest age + site + PC1 + PC2	7	3.55	0.03
nest age + site + PC2 + nest height	7	3.59	0.03
nest age + site + PC1 + PC2 + days since burn	8	3.74	0.03

Table 4. List of 10 video monitored Dickcissel nests and their fates at four tallgrass prairies in Northwest Arkansas, USA. If a predator was present, it was identified to species (when possible) and age of the chicks when they were attacked (where days = d). Under nest outcome, F = failure. Chesney Remnant = CHRM, Chesney Restored = CHRS, Stump = STUM, and Woolsey = WOOL. Under predator, PKSN = Prairie Kingsnake, RASN = Rat Snake, and UK = unknown.

Site	**Final date**	**Nest outcome**	**Chick age**	**Predator**	**Start time**	**Duration (m)**
CHRM	2018-06-05	0% F	7 d	n/a	n/a	n/a
CHRS	2017-06-02	100% F	eggs	PKSN	14:53	14
CHRS	2018-06-06	60-80% F	7 d	UK	14:45	UK
STUM	2017-06-07	100% F	3 d	PKSN	11:24	3
STUM	2018-06-21	100% F	7 d	UK	08:08	27
STUM	2017-07-20	50% F	9 d	PKSN	13:15	4
STUM	2017-07-17	100% F	7 d	PKSN	10:14	3
STUM	2018-07-01	100% F	5 d	UK	14:48	52
WOOL	2017-06-18	100% F	3 d	RASN	19:01, 20:00	9, 19
WOOL	2017-06-25	0% F	9 d	n/a	n/a	n/a

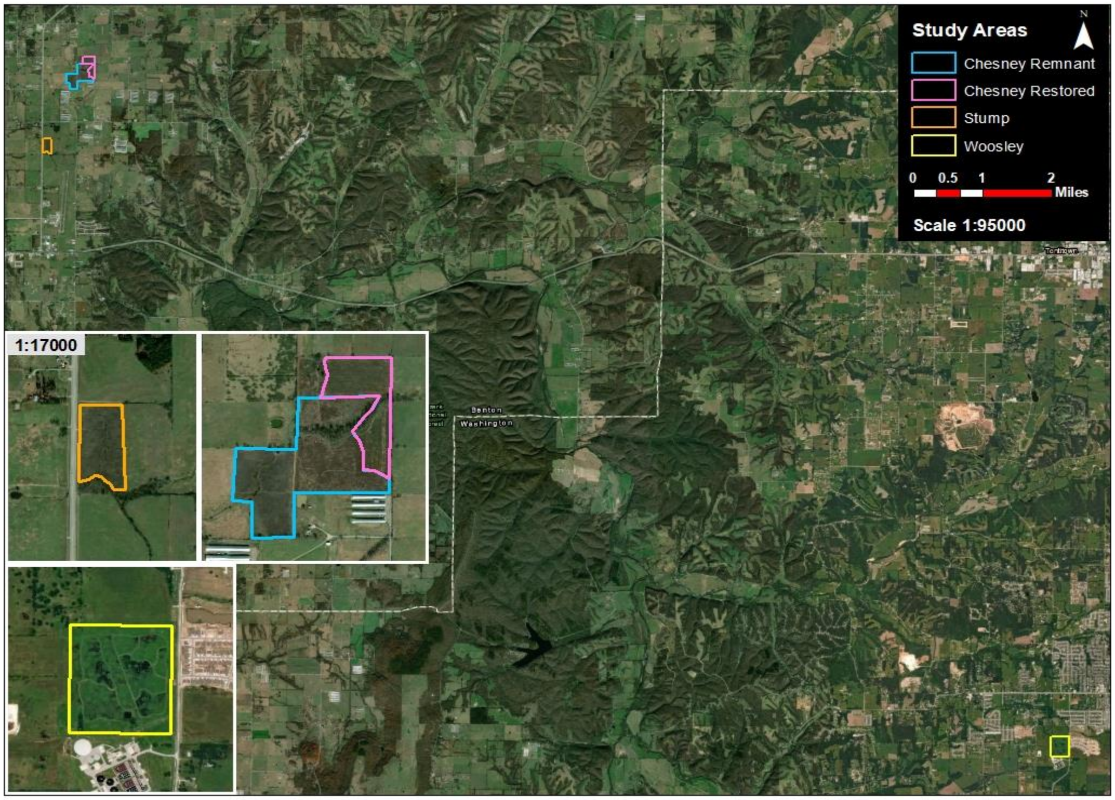

Figure 1. Map of study sites: Callie's Prairie (Fayetteville), Chesney Remnant, Chesney Restored, and Stump Prairies (Siloam Springs), and Woolsey Prairie (Fayetteville), Northwest Arkansas, USA. ESRI ArcGIS ArcMap, version 10.6.

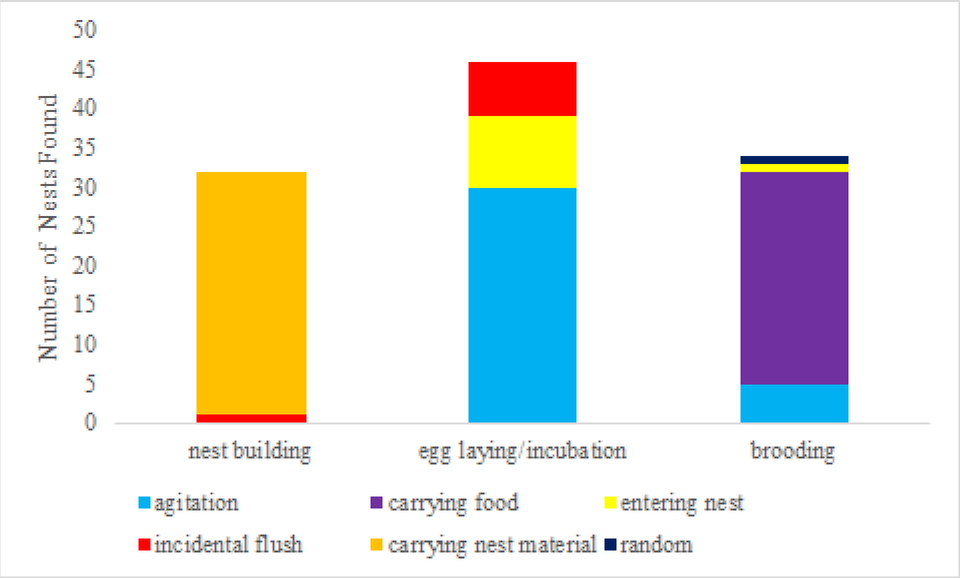

Figure 2. Number of active Dickcissel nests found in each nesting stage and behavior type used to locate nests totaled across sites and years (n = 112) in Northwest Arkansas, USA.

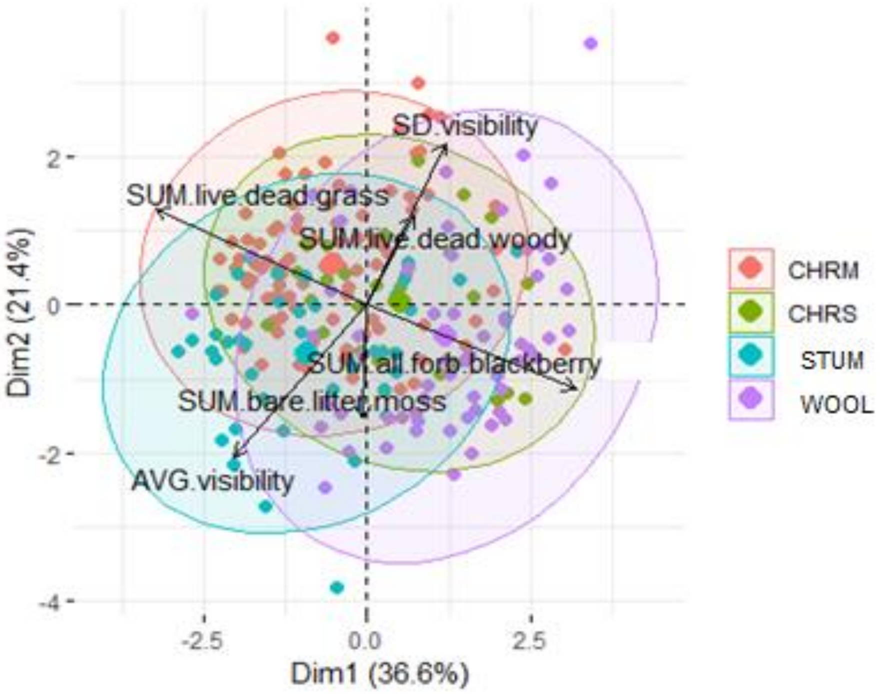

Figure 3. Principal components analysis average vegetation output for Dickcissel nests and random plots (combined) across years by site in remnant and restored tallgrass prairies of Northwest Arkansas, USA. Chesney Remnant = CHRM, Chesney Restored = CHRS, Stump = STUM, and Woolsey = WOOL.

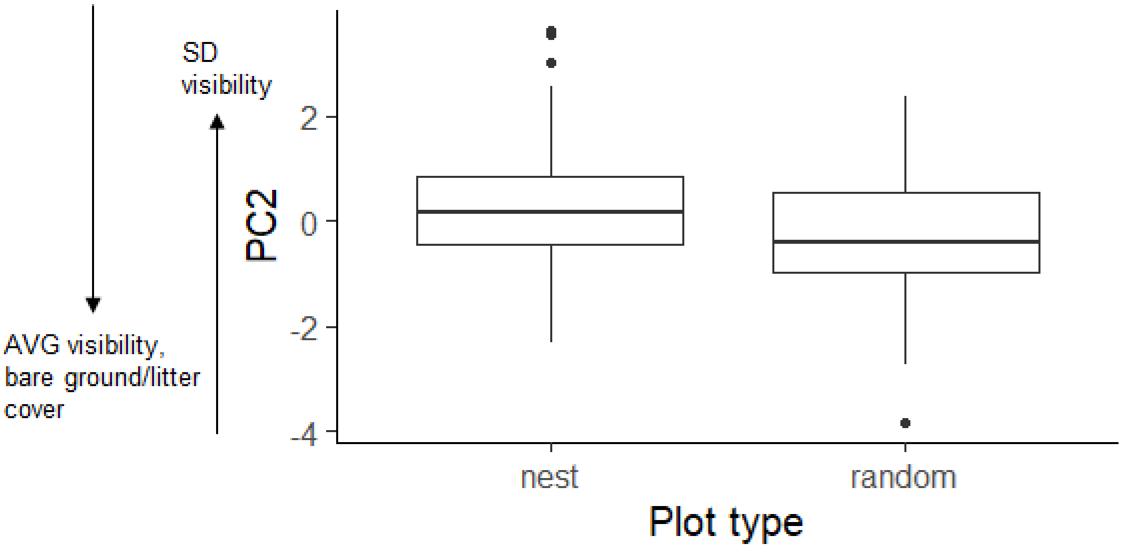

Figure 4. Box plot showing vegetative differences between nest and random plots on the PC2 axis in remnant and restored tallgrass prairies of Northwest Arkansas (ANOVA; F = 13.30, df = 1, $p < 0.001$).

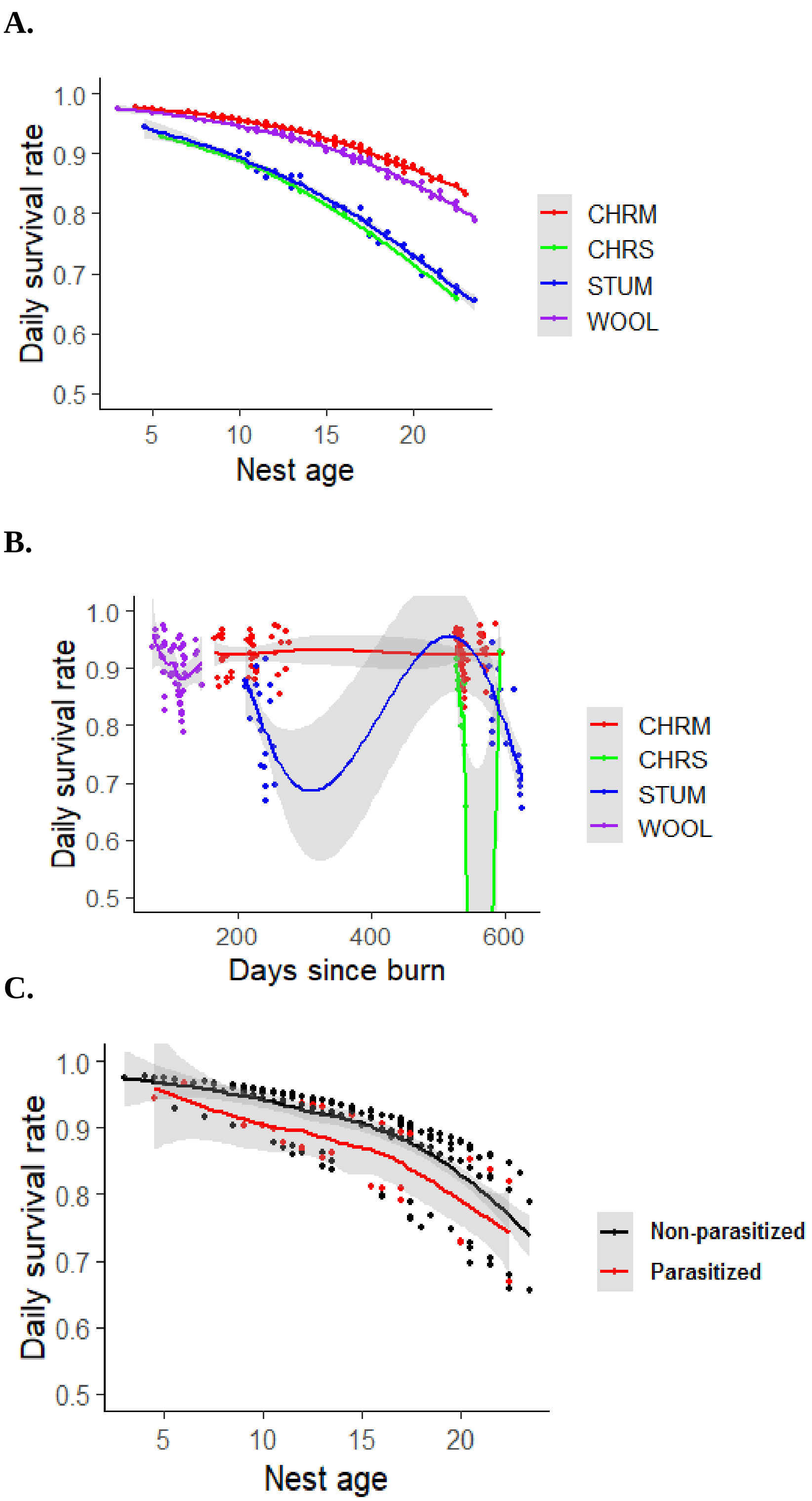

Figure 5. Variables (with 95% CI) as they relate to mean Dickcissel nest daily survival rate in remnant and restored tallgrass prairies of Northwest Arkansas, USA. **A.** Nest age by site. **B.** Days since most recent burn by site. **C.** Nest age by parasitism status. Chesney Remnant = CHRM, Chesney Restored = CHRS, Stump = STUM, and Woolsey = WOOL.

Chapter II

Diversity, Detection Rates, and Vegetation Use by Non-breeding Grassland Birds in Northwest Arkansas' Remnant and Restored Tallgrass Prairies

Abstract

North American grassland birds have experienced severe declines, due in part to extensive habitat loss. To mitigate declines, managers are restoring prairies by converting land previously used for farming into grasslands. Although several studies have focused on assessing quality of restored grasslands for breeding birds, fewer have expanded into the non-breeding season. I conducted transect surveys during the non-breeding season in two remnant and three restored tallgrass prairies in Northwest Arkansas and compared species richness, diversity, and detection rates of non-breeding grassland-obligate and non-grassland obligate birds among sites and seasons from September 2017-May 2018. Particular focus was given to five grassland-obligate species: Dickcissel (*Spiza americana*), Eastern Meadowlark (*Sturnella magna*), Le Conte's Sparrow (*Ammodramus leconteii*), Savannah Sparrow (*Passerculus sandwichensis*), and Sedge Wren (*Cistothorus platensis*). For these species, I also recorded habitat associations and correlated detection rates with distance to woody edge. Finally, I compared detection rates in burned and unburned transects following an autumnal prescribed burn at one site. I detected 1,455 individuals of 44 species, including eight grassland obligate species. Species richness and diversity of grassland-obligate birds was similar between sites, except Callie's Prairie, a small isolated restored prairie that had lower richness and diversity. Non-grassland obligate richness and diversity was highest at Woolsey Prairie, a large restored grassland interspersed with wetlands and wooded areas. An autumn prescribed burn resulted in a decrease of total grassland-obligate richness, but detection rate of non-grassland obligates was higher in recently burned transects than un-burned transects. Burned habitat made up a significant portion of habitat use for Eastern Meadowlark and Savannah Sparrow. Le Conte's Sparrow and Sedge Wren were almost entirely found in grass and forbs, and Dickcissel selected for woody vegetation.

Dickcissel and Sedge Wren detection rates were highest at the largest sites and were positively correlated with distance to woody edge, and thus may require large habitat patches. Le Conte's Sparrow favored sites containing woody cover, and Savannah Sparrow appeared attracted to a seed food source. My results demonstrate that non-breeding grassland-obligates use Northwest Arkansas' tallgrass prairies and that restored prairies can provide non-breeding habitat for grassland birds.

Introduction

Habitat loss and fragmentation are among the primary threats to wildlife populations globally (Bender et al. 1998, Dolman and Sutherland 1995). North America's most endangered terrestrial bird assemblage is grassland avifauna, which also suffers from habitat loss and fragmentation (Knopf 1994, Sauer and Link 2011). The decline of grassland birds is hardly surprising in the central and eastern North America, where 97% of tallgrass prairies have been destroyed since the pre-settlement era (Deluca and Zabinski 2011).

Remaining prairie fragments may not be suitable for grassland birds, but most research on this topic has taken place during the breeding season. Numerous species are known to be area-sensitive during the breeding season, requiring large patches of habitat (Ribic et al. 2009). Therefore, grassland fragments below a critical size may not suffice for certain bird species, even when the vegetative structure within a patch is suitable (Herkert 1994). Overall bird diversity and patch size share a positive relationship (Rahbek 1997), and grassland bird occurrence and richness during fall migration is best predicted by grassland patch size, indicating area-sensitivity outside of the breeding season (Robertson et al. 2011).

Another concern regarding fragmented prairies is predation risk. Breeding grassland bird communities are negatively affected by proximity to wooded edge (Beck et al. 2016), where nest predators are abundant (Klug et al. 2010). In temperate grasslands during the fall, winter, and spring, shrubs and trees serve as lookout perches for avian predators, such as hawks and shrikes (Macías-Duarte et al. 2017). Winter survivorship of Grasshopper Sparrow (*Ammodramus savannarum*) is lower in areas that contain shrubs (Macías-Duarte et al. 2017). Conversely, woody vegetation could act as thermal refuges from lethally cold temperatures (Hovick et al. 2014, Macías-Duarte et al. 2017). Grassland birds seem tolerant of woody areas during the non-breeding season (Igl and Ballard 1999) and may even compete for access to proximity to shrubs and trees; Savannah Sparrow (*Passerculus sandwichensis*) and Vesper Sparrow (*Pooecetes gramineus*) exhibit this behavior (Pulliam and Mills 1977). Since non-breeding grassland birds usually select habitat similar to breeding habitat (Hovick et al. 2014), birds should avoid migratory/wintering sites that have a high woody edge-to-area ratio, as several grassland-obligate species show negative associations with woody edges during the nesting season (Johnson and Igl 2001). Another predation risk comes from prescribed burns, which eliminate cover. Richness and detection of grassland-obligate birds has been found to increase with time since disturbance, such as fire (Hovick et al. 2014).

To increase grassland acreage, some landowners are recreating prairies through small-scale restoration projects involving previously farmed land. By planting native plant species, controlling exotic and/or invasive flora, and restoring disturbance regimes such as fire and grazing, managers try to mimic remnant prairie conditions. In North America’s fragmented prairie landscape, restoration may be an essential tool in habitat connectivity (Haddad et al. 2015). Restored prairies have attracted some grassland avifauna: Dickcissel (*Spiza americana*), Grasshopper Sparrow, Savannah Sparrow, Sedge Wren (*Cistothorus platensis*), and Western Meadowlark (*Sturnella neglecta*) have responded positively to prairie restoration during the breeding season (Fletcher and Koford 2002, Mundahl and Borsari 2016). Yet it is unknown to what degree these apparent attractions carry over into the non-breeding season.

Habitat use by grassland birds during the non-breeding seasons (fall and spring migration and winter) is understudied due to cryptic plumages, furtive behavior, and difficult weather conditions typical in those seasons (Fletcher et al. 2000). Recently, the “winter-limited species” theory has been growing in interest among avian biologists. The theory asserts that a species can be more threatened by winter habitat availability than breeding habitat availability (Sherry and Holmes 1993). For example, research on the Wood Thrush (*Hylocichla mustelina*) has shown that population growth is limited mainly by tropical deforestation in its winter range (Taylor and Stutchbury 2015). Furthermore, quality of winter habitat can have profound consequences for reproductive success: female American Redstarts (*Setophaga ruticilla*) that used “high quality” winter habitat raised more young that fledged earlier compared to those that occupied “low quality’ winter habitat (Norris et al. 2004). Considering that the majority of North American grassland birds are migratory and spend at least six months each year in migration and wintering seasons (Herkert et al. 1996), data on habitat use by prairie avifauna during the non-breeding season are needed for better informed conservation plans (Herkert and Knopf 1998).

This study aimed to compare grassland bird species diversity, detection rates (rough abundance measurement), and habitat associations among restored and remnant tallgrass prairies in Northwest Arkansas. I conducted transect surveys at two remnant and three restored sites during the three non-breeding seasons (fall, winter, and spring) from September 2017-May 2018. I focused on five frequently-encountered grassland-obligate species (defined as dependent on grasslands by the North American Breeding Bird Survey, United States Geological Survey; hereafter: USGS): Dickcissel, Eastern Meadowlark (*Sturnella magna*), Le Conte’s Sparrow

(*Ammodramus leconteii*), Savannah Sparrow, and Sedge Wren. I calculated mean Shannon-Weiner diversity indices, mean richness, and detection rate seasonally and by site. I recorded vegetative associations of each focal species and correlated detection rates to woody edge proximity to explore how vegetation selection varied by species and site. Finally, I investigated how detection rates of grassland-obligate and non-grassland obligate species changed following an autumnal prescribed burn at one site.

I hypothesized that grassland-obligate and non-grassland obligate species richness, diversity, and detection rates would be highest at the largest sites with the most diverse habitats. I also predicted that distance to nearest woody edge would correlate to higher grassland-obligate detection rates and that vegetation use would vary by focal grassland species but be largely restricted to grass and forb. Finally, I predicted that an autumn burn would decrease total richness and detection rates of grassland-obligate species.

Methods

Study Sites

I selected two remnant and three restored tallgrass prairies all within Northwest Arkansas. Figure 1 shows site maps, and management history is listed in Table 1. Unless noted, grasses at each site are dominated by bluestem (*Andropogon* and *Schizachyrium* spp.), rye (*Elymus* spp.), and/or panicgrass (*Dichanthelium* spp.), although not always in that order. Average blackberry and other woody species (trees and roses) cover does not surpass 10% and 3% across sites, respectively.

Callie’s Prairie (hereafter: Callie’s) is a restored site owned by the City of Fayetteville (36.146485, -94.123681). Grass is the most prevalent and forb is the least abundant compared to all other field sites. A near-complete mowing event was completed on April 3, 2018 (during the study period) and approximately half of it was mowed on June 15, 2017 (immediately prior to the study period). Chesney Prairie Natural Area (hereafter: Chesney Remnant) is situated in Siloam Springs, Benton County (36.2186847, -94.4821322). It is the largest tallgrass prairie relic in Northwest Arkansas and is owned by the Arkansas Natural Heritage Commission. One third of the parcel was burned on October 19, 2017 (during the study period), and the other 2/3 were burned on December 10, 2016 (immediately prior to study period). Grasses are the main plant present. Chesney Prairie also covers 6.5 ha of restored prairie (36.223201, -94.483551; hereafter: Chesney Restored). It was the only site that was not disturbed during the study period, although it was mowed immediately prior to the study on July 10, 2017. Grasses and forbs (non-grass herbaceous plants) equally occupy the site. Stump Prairie (hereafter: Stump), a privately-owned remnant, is also located in Siloam Springs, Benton County (36.204320, -94.494699). The site was completely burned occurred on October 19, 2017 during the study period. Grass is the dominant plant type. Woolsey Wet Prairie Sanctuary (hereafter: Woolsey) is a wetland mitigation project owned by the City of Fayetteville, Washington County (36.0672676, -94.2335263). A complete burn took place on March 8, 2018 (during the study period) and February 25, 2017 (immediately prior to the study period). Forbs form the most abundant plant group at Woolsey, an anomaly among field sites. Dominant grasses include several sedge species (Genera *Carex* and *Cyperus*), bluestem, and panicgrass.

Sampling Techniques

Given the secretive behavior of birds outside of the breeding season, I selected a sampling protocol that would best detect them without depending on assistance from multiple surveyors. I followed Burnham et al. (1980) methodology for distance line-transect surveys. I haphazardly chose between three and seven transects at each site, with varying distance from woody edges. Individual transect lengths varied between 36 and 153 m. Pathways to start and end points were marked with flags. I minimized non-prairie habitat within transects and maintained a minimum buffer of 10 m from the next transect to avoid double-counting birds. I also varied the cardinal direction of transects.

I conducted surveys during three seasons in 2017-2018: fall (September 17-November 14), winter (December 1-February 26), and spring (March 13-May 21). During the high-turnover migratory seasons of fall and spring, I visited sites 1/wk, and during the overwintering season, I visited sites approximately 1/2 wk. The shortest period between surveys in the fall and spring was 4 d, and the longest period was 12 d. In the winter, the shortest and longest periods were 12 and 40 d, respectively. I systematically alternated the transects and sites with which I started and ended surveys to ensure I equally covered transects at each time of the morning. Surveys took place from 0.5-3.5 hr after sunrise (Igl and Ballard 1999). I followed the North American Breeding Bird Survey instructions (USGS) to identify ideal survey conditions, which excludes wind speeds exceeding 19 km/hr or any persistent precipitation.

For each survey, I noted date, start and end times, cloud cover, wind speed, temperature, transect direction, and species. Prior to starting a survey, I stood quietly 2 m back from the start flag for 30-60 s to allow birds to acclimate to my presence. During surveys, I slowly walked between flags along the transect and recorded birds that were perpendicularly within 35 m of the transect (survey area boundary was identified using a Nikon 8397 Aculon AL11 laser rangefinder); this was the smallest distance across all sites before reaching a woody edge. When a bird was sighted, I categorically described the vegetation within 1 m radius of the bird: burn, forb, grass, or woody. I only counted birds that were perched in the survey area during the surveys. Birds that were only flying overhead were excluded because it was difficult to determine if they were actually occupying prairie habitat. I did not include birds that were only detected before or after the survey period. Vocalizing individuals that I was confident were within the transect-survey area but were never seen were also counted.

Data Analyses

I defined a species as grassland-obligate as a species that requires prairie habitat (per North American Breeding Bird Survey's list of species-habitat groupings, USGS), which included the five focal grassland-obligate species: Dickcissel, Eastern Meadowlark, Le Conte's Sparrow, Savannah Sparrow, and Sedge Wren.

I examined variation in mean alpha diversity, or richness (number of species) among sites, by counting the number of unique grassland-obligate and non-grassland obligate species observed per survey across all transects within a site, and averaging richness across surveys for each site. I examined seasonal variation in species richness by tallying species observed in each survey week across all transects and sites and averaging richness across weeks within each season. To equalize effort among sites, I excluded 1-2 transects at each of the following sites: Callie's, Chesney Remnant, and Woolsey, so that total transect lengths ranged from 309 to 339 m among sites. I excluded birds that could not be identified to species. I assessed variation in richness across sites and seasons by visually examining means and associated standard errors and considered values to be different when standard errors were far from overlapping. I compared mean diversity of grassland-obligate, non-grassland obligate ("other"), and all bird species between sites and seasons using the Shannon-Wiener diversity index (H'; Shannon 1948):

$$H' = \sum_{i=1}^{S} (p_i)(\ln p_i)$$

where S is all sampled species (within a site across seasons, or within a season across sites), $i = 1$ refers to a single species, and p_i equals the proportion, or evenness, of one species out of all surveyed species within a site or season. I calculated mean H' by survey date at each site and averaged across dates for comparisons among sites. For seasonal comparisons, I combined data for all sites, calculated Shannon diversity for each survey week, and averaged across weeks within each season. As for richness, I excluded unknown species, omitted 1-2 transects from Callie's, Chesney Remnant, and Woolsey to equalize effort (309-339 across sites after omitting transects), and assessed differences based on strongly non-overlapping standard errors.

I calculated the mean detection rate of birds, standardized for effort (per 100 m), at each site by calculating the number of detections of all bird-habitat groups per transect-survey, then averaging across transects and seasons for each site. To examine seasonal abundance of each focal species, I averaged detections per transect-survey across transects and sites within a season. In addition to all birds identified to species, I included observations identified as *Ammodramus* sp., since all species in that genus are grassland-obligate; and for non-obligates, I included *Melospiza* sp. and *Spizella* sp. because species in those genera do not rely on prairies. I included all transects.

To examine vegetative associations for the five grassland-obligate focal species, I calculated the percentage of detections that occurred in each vegetation type (burn, grass, forb, and woody) for each species from all transects. I contrasted them to vegetation cover averaged from two sites' summer 2017 and 2018 random vegetation plots (Chesney Remnant n = 47, Woolsey n = 32; see Chapter 1). Each plot had five 1.50-m^2 sub-quadrats (Cain and Castro 1959), from which I visually estimated percent vegetative cover (i.e. grass, forb, woody, bare soil, etc.; Higgins et al. 1996). I evaluated weekly detection rates and total richness of each bird-habitat group between burned (n = 2) and un-burned (n = 5) transects in Chesney Remnant, focusing on the first four wk after a prescribed burn on 1/3 of the property on October 19, 2017.

I evaluated correlations with woody edge habitat using simple linear regressions of mean detections by transect of each focal species, regressed against distance to the nearest woody edge from the center of the transect. I included all transects in this analysis, and I defined "woody edge" as an area that contained a minimum of 3 shrubs or trees measuring at least 1.5 m in height, with crowns touching (loosely adopted from Jensen and Finck 2004). I used Google Maps to measure the distance between each transect's mid-point and the nearest woody edge.

Results

Summary Statistics

Cumulatively across seasons and sites, I conducted 598 transect-surveys and tallied 1,455 detections of 44 bird species (Tables 2A and 2B). Eight of those were grassland-obligate species (Table 2A), accounting for 51% of all detections across sites and seasons (which excludes an additional four species observed outside of surveying near or in the field sites). Unidentified sparrow genera (*Ammodramus* sp., *Melospiza* sp., and *Spizella* sp.) and unknown individuals (sparrows and passerines) accounted for 4% of all detections. The five most frequently encountered species comprised 62% of all sightings: Savannah Sparrow (29%), Swamp Sparrow (*Melospiza georgiana;* 11%), Dickcissel (9%), Le Conte's Sparrow (7%), and Song Sparrow (*M. melodia;* 6%).

Richness and Diversity

Cumulative (total across seasons) grassland-obligate species richness, from greatest to least, is shown by site: Chesney Remnant (n = 7 species), Woolsey-restored (6), Chesney Restored (5), and Callie's-restored and Stump-remnant (4). Cumulative non-grassland obligate species richness was as follows: Chesney Remnant (21), Woolsey (19), Callie's (16), Stump (9), and Chesney Restored (7). Mean grassland-obligate richness (species per survey-date) was similar across sites (range: 1.17-1.43), with the exception of Callie's (0.52; Figure 2A). Woolsey had both the highest mean non-grassland obligate and all species richness (2.52 and 3.70, respectively). The other sites' mean non-grassland obligate and all species richness ranged from 0.65-1.26 and 1.74-2.30, respectively. The lowest mean non-grassland obligate richness occurred at Chesney Restored. Mean grassland-obligate richness changed little between seasons, ranging from 2.80-2.89 (Figure 2B). Mean non-grassland obligate richness tended to be lower in winter, influencing a similar trend for all species.

Woolsey had the highest mean grassland-obligate bird diversity (*H'* = 0.25), which was similar to Chesney Remnant and Chesney Restored (0.21 and 0.20, respectively; Figure 3A). Callie's grassland-obligate diversity was 0. Woolsey also had the highest mean diversity of non-grassland obligate (0.75); the other sites ranged from 0.13-0.29, with Chesney Restored having the lowest. Mean overall diversity was highest at Woolsey as well (1.00), while other sites varied from 0.35-0.56; Stump had the lowest mean overall diversity. By season, mean grassland-obligate diversity was marginally higher in the fall and winter (0.60 and 0.58, respectively; Figure 3B), than the spring (0.41). As with richness, non-grassland obligate diversity tended to be lower (1.02) in the winter than fall and spring (1.29 and 1.22, respectively), influencing a similar pattern for overall diversity.

Detection Rates

Mean detection rates of grassland-obligate species were highest at Chesney Restored (3.37/100 m) and much lower at Callie's (0.50/100 m; Figure 4) than other sites. Mean non-grassland obligate detection rates were highest and lowest at Woolsey (3.25/100 m) and Chesney

Restored (0.52/100 m), respectively.

Highest mean detection rates for the five grassland-obligate focal species were as follows (Figure 5A): Dickcissel in Chesney Remnant and Chesney Restored (0.51 and 0.44/100 m, respectively), Eastern Meadowlark in Stump (0.33/100 m), Le Conte's Sparrow in Callie's and Woolsey (0.40 and 0.49/100 m, respectively), Savannah Sparrow in Chesney Restored (2.73/100 m), followed by Stump (1.15/100 m), and Sedge Wren in Chesney Remnant (0.37/100 m).

Mean detection rates also varied for each of the focal grassland-obligates by season. The following mean detection rates were the highest for each species (Figure 5B): Dickcissel in the spring (0.63/100 m), Eastern Meadowlark in the fall (0.17/100 m), Le Conte's Sparrow in winter (2.08/100 m), Savannah Sparrow in the winter (1.71/100 m), and Sedge Wren in the fall (0.24/100 m).

Vegetative Associations

Vegetation associations also varied by focal grassland-obligate species, except Le Conte's Sparrow and Sedge Wren which showed similar associations (Figure 6). Both species used primarily grass (64%) and forbs to a lesser degree (31-32%). Dickcissel observations were evenly split between forb, grass, and woody vegetation (29-37% each). Eastern Meadowlark appeared most often in burned areas (61%), followed by grass (22%), then forb (15%). Savannah Sparrow was most frequently encountered in forbs (45%), but also used burned areas extensively (33%), and sometimes occupied grassy habitats (16%).

At both Chesney Remnant and Woolsey, Dickcissel selected woody vegetation despite its relative scarcity. Vegetation selection by Le Conte's Sparrow and Sedge Wren mirrored availability (which was either grass or forbs, depending on site). Eastern Meadowlark and Savannah Sparrow predominantly occupied burned areas, which were unavailable in the summer

(Figures 7A and 7B). However, the remainder of Savannah Sparrow detections at Chesney Remnant reflected availability (more grass, less forbs) whereas it completely avoided grass at and selected for woody vegetation at Woolsey. Eastern Meadowlark selected for grass and against forb at Woolsey.

Woody Edge Associations

Out of the five focal species, two showed significant relationships with distance to woody edge: Dickcissel had the strongest positive correlation with distance to nearest woody edge (Figure 8A; $p = 0.0001$, $r^2 = 0.5101$). Sedge Wren also shared a positive correlation (Figure 8B; $p = 0.0139$, $r^2 = 0.2556$). Eastern Meadowlark (Figure 8C; $p = 0.3170$, $r^2 = 0.0477$), Le Conte's Sparrow (Figure 8D; $p = 0.4141$, $r^2 = 0.0320$), and Savannah Sparrow (Figure 8E; $p = 0.6282$, $r^2 = 0.0114$) demonstrated no correlation between detection rate and proximity to woody edge.

Effects of Autumnal Prescribed Burn

Mean detection rates for both burned and un-burned transects in Chesney Remnant varied widely, although non-grassland obligates appeared in greater numbers after the burn than grassland-obligates (Figure 9). Non-grassland obligates averaged 8.53-10.23/100 m in the burned transects three wk post-burn. During the same period, un-burned transects averaged 0.42-1.10/100 m of non-grassland obligates. However, grassland-obligate detection rates were generally similar between burned and un-burned transects for the first three weeks post-burn (ranges 0.68-1.70 and 0.36-2.54/100 m, respectively). No birds were seen on burned transects 4 wk after the burn. Total grassland-obligate richness in burned and un-burned plots was 2 and 5, respectively, and total non-grassland obligate richness in burned and un-burned plots was 5 and 4, respectively.

Discussion

Results did not support my hypothesis that grassland-obligate richness, diversity, and detection rates are highest at larger and habitat-diverse sites. Grassland-obligate richness and diversity were generally similar, with the exception of Callie’s, a small isolated restored prairie that had lower richness and diversity. However, high grassland-obligate detection rates were skewed by Savannah Sparrow at two small sites with low habitat diversity. Woolsey, a large habitat-diverse site, consistently yielded the highest richness, diversity, and detection rates for non-grassland obligate species. Distance from woody edge correlated to increased detection rates for two out of five grassland-obligate focal species, and vegetation associations were limited to grass and forbs in two out of five grassland-obligate focal species. Finally, I reject my hypothesis that an autumn burn decreases grassland-obligate detection rates, although it did decrease total grassland-obligate richness.

Site Comparisons

Mean grassland-obligate richness and diversity were comparable across sites, excluding Callie’s, a small, isolated habitat-diverse restored prairie. Generally, non-breeding grassland-obligate bird assemblages did not show patterns of site selection for characteristics like size and habitat diversity. However, other studies have suggested that size is an important factor in non-breeding grassland bird diversity: larger areas explained grassland-obligate richness during fall migration (Robertson et al. 2011). Limited site-level replication could have hidden broad patterns in this study, although some grassland-obligate species were detected at greater frequency in larger sites (see Focal Species section). Grassland birds may also be less sensitive to habitat availability in the non-breeding season. Several grassland sparrow species have been observed using (and sometimes competing for) wooded areas, a component absent from breeding habitat (Pulliam and Mills 1977, Igl and Ballard 1999).

Woolsey had the highest mean overall species richness and diversity, non-grassland obligate richness and diversity, and detection rates of all and non-grassland obligate species averaged across seasons. One major factor contributing to high overall diversity the large size of this site, which is the second largest of five field sites (18 ha), excluding surrounding hayfields that may also be occupied by birds. Avian richness is well known to increase with area (Rahbek 1997), including for southbound grassland migrants (Robertson et al. 2011). Woolsey also provides the highest habitat diversity of any site. Ponds and small wooded patches are interspersed within prairie habitat, which consequently attract birds that occupy those habitats. Like Woolsey, sites in southern Texas with a mix of grass and woody vegetation yielded higher grassland-obligate richness than habitats consisting of only grass or trees/shrubs (Igl and Ballard 1999), and sparrows wintering in grasslands may use woody areas for shelter (Pulliam and Mills 1977). Woolsey's complete springtime burn may have contributed to increased richness and diversity of both habitat-obligate bird groups. Post-burn Woolsey was exploited by a suite of opportunistic and populous birds (Otis et al. 2008, Wheelwright and Rising 2008, Vanderhoff et al. 2016), such as American Robin (*Turdus migratorius*), Mourning Dove (*Zenaida macroura*), and Savannah Sparrow, which were virtually absent from previous surveys at Woolsey. These birds were likely foraging on exposed seed and invertebrate prey (this study), or may have been easier to detect due to lack of cover, or a combination of both explanations. High detection rates of all and non-grassland obligate species may be explained by flocks of wetland emberizines, Swamp Sparrow and Song Sparrow (Mowbray 1997, Arcese et al. 2002), using dense forb patches for food and cover. Together they represented 43% of all Woolsey detections.

Chesney Restored had the highest mean grassland-obligate detection rate. The site was covered by white heath aster (*Symphyotrichum ericoides*), which may have been a reliable food supply. I regularly observed Savannah Sparrow consuming its seeds at Chesney Restored. Seed density and distribution are important characteristics of non-breeding grassland sparrow habitat (Grzybowski 1983, Titulaer et al. 2017). White heath aster could have acted as adequate springtime cover (Hovick et al. 2014) for Dickcissel and Savannah Sparrow at a time when Chesney Remnant, Stump, and Woolsey contained large areas of bare burned ground. These two generalist species form dense flocks where they occur (Temple 2002, Wheelwright and Rising 2008), thus accounting for the high grassland bird detection rate at Chesney Restored. Although the site had been mowed 2-10 mo before the survey period, this did not appear to have a negative effect on grassland-obligate richness and detections. These results counter findings by Hovick et al. (2014) showing that grassland-obligate richness and detections increase with increasing time since disturbance. I could not confirm that time since disturbance was an important factor in this study. Chesney Restored also contained low non-grassland obligate richness and detection rate. The small size (9 ha) and lack of habitat diversity resulted in few non-grassland obligates.

Callie's scored the lowest in mean grassland-obligate richness, diversity, and detection rate, and mean all species richness and detection rate. Factors affecting low grassland-obligate diversity, richness, and detection rate consist of small size (9 ha; Robertson et al. 2001, Ribic et al. 2009), surrounding landscape of forest and residential neighborhoods (Haddad et al. 2015), poor vegetative diversity (average cover of 79% grass and 11% forb; Robertson et al. 2001), and a near-complete mowing in April 2018. Previous studies have found that grassland-obligate richness and detection increases with increased time post-disturbance, resulting in greater vegetation height and density (Hovick et al. 2014), therefore the substantially shorter vegetation post-mow may not have provided suitable cover. Low all species richness and detection rate were unexpected given the availability of both prairie and wooded habitats at Callie's. However, common woody habitat species prefer to remain in a forest in lieu of crossing small openings (Desrochers and Hannon 1997). Woody habitat birds were seldom observed using scattered trees and shrubs within Callie's, instead staying in the adjacent forest.

Stump had among the lowest mean diversity for all species across seasons, which is not surprising given the complete burn early in the survey season (October 2017). Because it was entirely burned, no refuge existed throughout most of the migratory periods and the entire winter. Stump also had low habitat diversity; it consisted mostly of burned prairie, bisected by a narrow riparian zone. It appealed mainly to these three opportunistic species (Jackson and Jackson 2000, Otis et al. 2008, Wheelwright and Rising 2008), which summed to 78% of all detections at Stump after the burn: Killdeer (*Charadrius vociferus*), Mourning Dove, and Savannah Sparrow.

Seasonal Comparisons

Fall and winter yielded higher mean grassland-obligate diversity, although all seasons were similarly rich in grassland-obligate species. Thus, at least some individuals of most non-breeding grassland-obligates overwinter in Northwest Arkansas. At the site level, Woolsey supported high grassland-obligate richness and diversity, but was entirely burned in the spring. This event, combined with birds potentially leaving the wintering grounds between the end of winter season surveys and the start of spring season surveys, likely lowered evenness across sites in the spring. Non-grassland obligate species had higher mean richness and diversity during migratory seasons. Almost all of Northwest Arkansas' year-round species are non-grassland obligate, but more are summer residents or transient species (Neal 2009). All species diversity was approximately equal across seasons, whereas all species richness rose during fall and spring.

Savannah Sparrow

Savannah Sparrow accounted for 29% of all detections, making it the most frequently encountered of any species in this study. According to Fayetteville Christmas Bird Count (CBC) results from December 2017, winter 2017-2018 was an above-average detection year (1.42 versus 20-yr average of 0.91 birds/party hour; National Audubon Society 2018). Thus, detection rates during this study may not represent a typical winter for Savannah Sparrow. While country-wide it has declined on average by 49% from 1966- 2014 (Sauer et al. 2017), the Savannah Sparrow is still regarded as abundant across its continent-wide distribution (Wheelwright and Rising 2008). Igl and Ballard (1999) also found that Savannah Sparrow was one of the most abundant non-breeding grassland-obligate birds in southern Texas. Predictably, Savannah Sparrow was most often found in the winter; fall migration starts in late September/early October and spring migration ends sometime in May, and it is labeled as a "common winter sparrow" of Northwest Arkansas (Neal 2009).

Savannah Sparrow was detected most frequently at Chesney Restored, followed by Stump. Chesney Restored provided a probable food staple, white heath aster, which I often saw the sparrow eating. Stump may have exposed food sources from an October burn. These sites themselves are small, but they are embedded in a larger open landscape of hayfields and pastures. Therefore, it is possible that Savannah Sparrow is area-sensitive during the non-breeding season, which is mostly true during the breeding season (Herkert 1994, Vickery et al. 1994, Bollinger 1995, Renfrew and Ribic 2008, but see Johnson and Igl 2001, Bakker et al. 2002, Winter et al. 2006). It is unknown as to why Savannah Sparrow was more frequently found at Stump but not at other recently burned sites. Chesney Remnant and Woolsey also contained nearby open habitats (hayfields, pastures) that were burned during the study period.

Forb was the main vegetation type used by Savannah Sparrow, likely due to the preponderance of white heath aster at the site with the highest detection rate, Chesney Restored. Out of all detections at Chesney Restored, 79% came from forb patches. Indeed, other studies on wintering grassland sparrows concluded that seed density and preferred seed distribution influences sparrow density and occupancy, respectively (Grzybowski 1983, Titulaer et al. 2017). Burned habitat was another major component of vegetative associations across sites (33% of all Savannah Sparrow detections), which aligns with the species' status as a pioneer species in that it responds positively to disturbances (Andrews 2013), and its status as a generalist species, occupying a variety of open habitats (Wheelwright and Rising 2008). There was no correlation between Savannah Sparrow detection rate and distance to nearest woody edge, unlike its behavior in the breeding season (Bollinger 1995, Bakker et al. 2002), again highlighting its generalist tendencies in the non-breeding season.

Dickcissel

Dickcissel detections accounted for 9% of all bird detections across seasons, making it the third most frequently-detected species. Fayetteville CBC data are absent for this strictly neotropical migrant (Temple 2002). The conspicuous auditory and visual behavior of singing males make for easily-detectable birds. The high detection rate speaks to its relatively high abundance in Northwest Arkansas tallgrass prairies (Neal 2009), which is true throughout the breeding range of this declining species (Temple 2002, Sauer et al. 2017). Spring accounted for 97% of all Dickcissel detections, all occurring in only three survey-weeks. Results are similar to Igl and Ballard (1999). Given the prolific singing males and watching females build nests, most or all springtime observations were summer breeding residents. The start of the fall survey period was mid-September, which aligns with the end of Dickcissel migration in Northwest

Arkansas (Neal 2009). Consequently, southbound birds were mostly missed in my study.

Dickcissel was detected most frequently at Chesney Remnant and Chesney Restored. Both sites are located within an open landscape without nearby paved roads. Breeding grassland birds distance themselves from high-traffic roadways (Forman et al. 2002), plausibly to minimize energy required to sing louder than traffic noise (Brumm 2004). Dickcissel is known to be area-sensitive in the breeding season, more likely occupying large open patches over small ones (Winter 1998, Bakker et al. 2002). Hayfields and pastures that surround Chesney Remnant and Chesney Restored could provide additional habitat. Area-sensitivity in this study is further supported by the significant positive relationship between distance to nearest woody edge and Dickcissel detection rate. Nest density is low within 50 m of a woody edge (Jensen and Finck 2004, but see Patten et al. 2006), potentially in response to snakes (nest predator) and Brown-headed Cowbirds (*Molothrus ater*, brood parasite), which are prevalent along woody edges (Jensen and Finck 2004, Klug et al. 2010). My results concur with other findings that Dickcissel avoids woody edges.

Of all the grassland-obligate focal species, Dickcissel seemed the least selective of vegetation types. Recorded associations were evenly split between forb, grass, and woody vegetation. This lack of selection may be attributed to its ability to adapt to secondary habitats (Temple 2002). At Chesney Remnant and Woolsey, Dickcissel selected for woody vegetation, which was likely skewed by males procuring prominent singing perches.

Le Conte’s Sparrow

Le Conte’s Sparrow was the fourth most frequently-encountered species in this study (7% of all detections). Its high detection rate (mean range: 0.40-0.49 birds/100 m) is similar to results in southern Texas (Heath et al. 2008). While this is encouraging given its average nation-wide decline of 73% between 1960 and 2014 (Sauer et al. 2017), it is imperative to acknowledge that Le Conte’s Sparrow exhibits strong fluctuations in abundance in Northwest Arkansas (Neal 2009). The winter of 2017-2018 in particular was an exceptional year, reflected by a record-high Fayetteville CBC detection rate of 0.55 birds/party hour in December 2017 (compare to 20-yr average of 0.03 birds/party hour; National Audubon Society 2018). Not surprisingly, Le Conte’s Sparrow was detected with the highest frequency in the winter. Le Conte’s Sparrow does not stay for the entire duration of fall and spring migration, but regularly overwinters in Northwest Arkansas (Neal 2009).

Intriguingly, Le Conte’s Sparrow detection rates were highest at Callie’s and Woolsey. These restored sites are dramatically different both in size (9 ha and 18 ha, respectively), dominant vegetation types (grass and forb, respectively), and surrounding landscape (forest/suburban lots versus hayfields/suburban lots). Because it was regularly observed at Callie’s, Le Conte's Sparrow is apparently not area-sensitive during the non-breeding season, unlike the breeding season, when it only occurs in areas exceeding a certain size (Johnson and Igl 2001).

One similarity between Callie’s and Woolsey is the presence of woody vegetation within the property, which may be an important determinant in Le Conte's Sparrow site occupation and/or density. Woody vegetation is largely absent at other un-burned sites. Although it was rarely found in woody vegetation within transects (4% of all detections), the survey design focused on prairie habitat and thus mostly excluded woody areas. It is possible that Le Conte’s Sparrow seeks cover in shrubs and trees; the species has been observed using areas with woody vegetation (Igl and Ballard 1999, Baldwin et al. 2007, this study). Pulliam and Mills (1977) suggest that woody vegetation could be an important characteristic of wintering grassland sparrow habitat by providing cover. Yet the presence of shrubs in winter territories *decreased* the survivorship of Grasshopper Sparrow, a close relative (Macías-Duarte et al. 2017). Regression analyses demonstrated no correlation between detection rate and nearby wooded edge. Le Conte's Sparrow generally selected vegetation based on availability at a given site, the majority of which was grass and forbs. Monroe and O'Connell (2014) found a weak negative relationship between grass cover and Le Conte's Sparrow detections. However, this study suggests that both grass and forbs are similarly important.

Interestingly, this study's results on Le Conte's Sparrow occurrence after disturbances were drastically different from those found elsewhere (Baldwin et al. 2007, Hovick et al. 2014, but see Monroe and O'Connell 2014). The two former studies found that this species almost exclusively selected sites that have not undergone perturbation in a minimum of two years. However, my study showed that one of two sites with the highest detection rate of Le Conte's Sparrow was Woolsey, which has been burned annually in the spring for at least a decade. The other site, Callie's, underwent mowing in March 2018. After that event, several individuals still remained. Some management strategies, like annual burning and mowing, do not appear to have a significant impact on Le Conte's Sparrow occurrence during the non-breeding season.

Eastern Meadowlark

Out of the relatively few Eastern Meadowlark observations, most detections came from Stump. The species probably seized fauna and flora exposed from the October fire; 84% of all autumnal Eastern Meadowlark detections at Stump were in burned habitat. Two out of three site burns happened in fall, which likely contributed to this generalist's high detection rate in the fall (despite its status as a year-round resident; Neal 2009). Similar to Savannah Sparrow, even though Chesney Remnant and Woolsey also received prescribed burns that covered more area, Eastern Meadowlark did not use those sites as much. Adjacent pastures and hayfields were present at Chesney Remnant, Stump, and Woolsey, which could have provided additional habitat for this area-sensitive breeder (Herkert 1994, Vickery et al. 1994, Renfrew 2002, Renfrew and Ribic 2008). Why Stump was favored over Chesney Remnant and Woolsey is unknown. It is counterintuitive considering the proximity of Highway 59 and the species' adversity to road traffic during the summer (Forman et al. 2002), but this feature may not be important outside of the breeding season.

Eastern Meadowlark was typically associated with burned areas, totaling 61% of all Eastern Meadowlark detections across seasons. Such opportunistic use was expected given its status as a grassland habitat generalist (Fuhlendorf et al. 2006, Pillsbury et al. 2011). However, higher visibility in burned plots may have also contributed to a higher detection rate. Again illustrating its habitat generalist behavior, Eastern Meadowlark demonstrated no relationship with distance to nearest woody edge. This pattern extends into the breeding season (Bollinger 1995, Patten et al. 2006).

Despite its year-round "common resident" status in Northwest Arkansas (Neal 2009), high year-round abundance in the region (Jaster et al. 2012), and high density in southern Texas during the non-breeding season (Igl and Ballard 1999), I encountered relatively few Eastern Meadowlark. I noticed that it is more inconspicuous in the non-breeding season, and therefore transect surveys are likely under-detecting this species. Winter 2017-2018 appeared to be an average year compared to the 20-yr average detection rate for this songbird on the Fayetteville CBC (National Audubon Society 2018). Understanding non-breeding requirements of Eastern Meadowlark is especially crucial because across the country, this species has suffered an average population decline of 89% between 1966 and 2014 (Sauer et al. 2017).

Sedge Wren

Sedge Wren was detected in low numbers overall yet tied for a record-high detection rate on the Fayetteville CBC in December 2017 (0.14 birds/party hour in 2017 versus 20-yr average of 0.04 birds/party hour; National Audubon Society 2018). Project Prairie Birds (Texas Parks and Wildlife Department 1998) methodology was deployed in the Heath et al. (2008) study that showed Sedge Wren as one of the most abundant grassland-obligate species in the winter in southern Texas. This methodology includes coordinated flushing by three surveyors, so intensive surveying may be required to detect this secretive bird (Herkert et al. 2001).

Most Sedge Wren detections occurred in Chesney Remnant, while fewer birds were found in all other sites (except Stump, which had zero sightings) in the fall and winter. The large size of Chesney Remnant, coupled with a significant positive correlation between detection rate and distance to nearest woody edge, suggests that Sedge Wren is area-sensitive and avoids woody edges in the non-breeding season. Area sensitivity (Johnson and Igl 2001, Bakker et al. 2002, but also see Johnson and Igl 2001) and woody edge avoidance (Bakker et al. 2002) have also been demonstrated in the breeding season.

Autumn yielded the highest detection rate, likely resulting from birds moving south from late season breeding sites in bordering states (Herkert et al. 2001). No breeding evidence was confirmed in this study. Paucity of spring observations may be explained by a different northbound migratory pathway, few stopovers between over-wintering and breeding grounds, or high winter mortality rates. The lack of winter sightings was expected, given its “uncommon to rare” status in winter (Neal 2009).

Vegetation use mirrored availability at Chesney Remnant and Woolsey, suggesting that Sedge Wren selects what is available (which is almost exclusively grass and forbs). Thick grass and forbs appear to be especially important (Baldwin et al. 2007, Heath et al. 2008).

Response to an Autumn Burn

The majority (84%) of birds detected in burned sections of Chesney Remnant in October and November 2017 were non-grassland obligates and primarily generalists that formed flocks (Jackson and Jackson 2000, Otis et al. 2008): American Pipit (*Anthus rubescens*), Killdeer (generalist), and Mourning Dove (generalist) comprised 79% of all detections in recently burned habitat. The two commonest non-grassland obligate species in un-burned habitat--Song Sparrow and Swamp Sparrow--accounted for only 26% of all un-burned detections. Mourning Dove and Killdeer regularly used winter burns in Texas (Reynolds and Krausman 1998). Mean detections were always higher in burned than un-burned areas, which is consistent with Reynolds and Krausman’s results (1998). Bird flocks can better exploit this exposed landscape by using flock vigilance to monitor for predators (Grzybowski 1983), and the three commonest species in burned transects often flocked together. Flocking birds were seldom seen in non-burned transects. Total non-obligate richness was similar in burned and unburned transects, which supports Reynolds and Krausman (1998).

Mean grassland-obligate detection rates did not vary strongly between burned and un-burned transects. Only two grassland-obligates utilized burned areas: Eastern Meadowlark and Savannah Sparrow. Both have been described as generalists (Wheelwright and Rising 2008, Jaster et al. 2012), and therefore are most likely to occupy disturbed areas compared to prairie specialists. Savannah Sparrow is especially known to respond positively to disturbances (Andrews 2013). Total grassland-obligate richness was 60% lower in un-burned than burned plots, supporting the idea that more grassland-obligate species select vegetative cover over areas of bare ground. Classic examples include Le Conte's Sparrow and Sedge Wren (Reynolds and Krausman 1998), neither of which were observed in burned transects across all sites.

Temporally, the burned transects appeared to reach a limit of attraction (likely food; this study). From mid-November (four wk after burn) to mid-January, I detected 0 birds in burned areas. Even after mid-January, non-grassland obligate detection rates never reached the numbers in the first three wk immediately post-burn.

Conclusion

Tallgrass prairies in Northwest Arkansas support diverse migratory and overwintering grassland-obligate bird species. During June and July of 2017 and 2018, I only encountered three species, whereas from September 2017-May 2018, I detected 12 species. My study contributes to a limited body of knowledge on North American non-breeding grassland bird occurrence and habitat use, specifically that some non-breeding grassland-obligates were area-sensitive and associated with a variety of vegetation types. Conservation efforts should concentrate on maintaining a shifting mosaic of vegetation (created by fire) and protecting large habitat patches, which should satisfy diverse vegetation and area requirements of non-breeding grassland bird species. My focus on remnant and restored tallgrass prairies in Northwest Arkansas demonstrated that restored sites are used similarly to remnant sites by grassland-obligates. Therefore, restored tallgrass prairies could act as a substitute for lost remnants, but further research is needed on food availability and mortality rates to fully understand habitat quality of restored grasslands. Longer term studies should also be conducted to confirm patterns observed in this short-term project. However, conservation efforts should be made with diverse transient and overwintering grassland birds in mind in this region of the continent.

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Tables and Figures

Table 1. Summary of field site management in Northwest Arkansas, USA. Chesney Remnant = CHRM, Chesney Restored = CHRS, Stump = STUM, and Woolsey = WOOL. For status, rem = remnant, and res = restored. “Planted” under previous land use refers to non-native cattle forage (tall fescue). Burn rotation refers to interval length between complete site burns. Under mow, herbicide, and woody removal, A = annual, H = historically (but not currently), and O = occasionally.

Site	**Status**	**Size (ha)**	**Managed since**	**Prior land use**	**Native plantings**	**Burn (y)**	**Mow**	**Herbicide**	**Woody removal**
CALL	res	9	2009	pasture; planted	yes	3-5	yes, A	yes, A	yes, H
CHRM	rem	27.5	2000, 2015	hayfield	no	3	no	yes, A	yes, A
CHRS	res	6.5	2015	bean field; plowed	yes	3	yes, A	yes, A	yes, A
STUM	rem	6.5	1998	hayfield	no	3	no	yes, O	yes, O
WOOL	res	18	2008	pasture; planted	some	1	no	yes, A	yes, A

Table 2A. List of eight grassland-obligate species (per USGS Breeding Bird Survey) observed on surveys during all seasons, cumulative detections, and number of sites in which each species was detected. Codes that have higher Partners in Flight (PIF) values represent species of greater conservation concern.

Species	**PIF**	**Detections**	**Sites**
Northern Harrier (*Circus cyaneus*)	11	2	1
Short-eared Owl (*Asio flammeus*)	12	1	1
Sedge Wren (*Cistothorus stellaris*)	7	58	4
Vesper Sparrow (*Pooecetes gramineus*)	11	3	2
Savannah Sparrow (*Passerculus sandwichensis*)	8	427	5
Le Conte’s Sparrow (*Ammodramus lecontei*)	13	97	4
Dickcissel (*Spiza americana*)	11	129	4
Eastern Meadowlark (*Sturnella magna*)	11	42	5

Table 2B. List of 36 non-grassland obligate species (per USGS Breeding Bird Survey) observed on surveys during all seasons, cumulative detections, and number of sites in which each species was detected. Codes that have higher Partners in Flight (PIF) values represent species of greater conservation concern.

Species	**PIF**	**Detections**	**Sites**
Northern Bobwhite (*Colinus virginianus*)	12	18	2
Mourning Dove (*Zenaida macroura*)	6	59	4
Sora (*Porzana carolina*)	9	1	1
Killdeer (*Charadrius vociferous*)	9	66	2
Wilson’s Snipe (*Gallinago delicata*)	9	7	2
Red-bellied Woodpecker (*Melanerpes carolinus*)	7	2	1
Northern Flicker (*Colpates auratus*)	9	10	3
Eastern Phoebe (*Sayornis phoebe*)	8	12	2
Eastern Kingbird (*Tyrannus tyrannus*)	11	1	1
Scissor-tailed Flycatcher (*Tyrannus forficatus*)	11	18	4
Blue Jay (*Cyanocitta cristata*)	8	1	1
Tree Swallow (*Tachycineta bicolor*)	10	6	1
House Wren (*Troglodytes aedon*)	5	3	2
Marsh Wren (*Cistothorus palustris*)	7	8	3
Eastern Bluebird (*Sialis sialis*)	7	20	1
American Robin (*Turdus migratorius*)	5	59	4
Brown Thrasher (*Toxostoma rufum*)	10	1	1
American Pipit (*Anthus rubescens*)	8	27	2
American Goldfinch (*Spinus tristis*)	7	3	1
Lapland Longspur (*Calcarius lapponicus*)	7	2	1
Orange-crowned Warbler (*Vermivora celata*)	9	1	1
Common Yellowthroat (*Geothlypis trichas*)	9	10	4
Palm Warbler (*Setophaga palmarum*)	11	4	2
Yellow-rumped Warbler (*Setophaga coronata*)	6	8	1
Clay-colored Sparrow (*Spizella pullida*)	10	5	1
Field Sparrow (*Spizella pusilla*)	12	2	1
Song Sparrow (*Melospiza melodia*)	8	92	4
Lincoln’s Sparrow (*Melospiza lincolnii*)	9	19	5
Swamp Sparrow (*Melospiza georgiana*)	6	161	3
White-crowned Sparrow (*Zonotrichia leucophrys*)	8	14	2
Dark-eyed Junco (*Junco hyemalis*)	8	11	2
Northern Cardinal (*Cardinalis cardinalis*)	5	3	2
Red-winged Blackbird (*Agelaius phoeniceus*)	8	34	2
Rusty Blackbird (*Euphagus carolinus*)	12	6	1
Common Grackle (*Quiscalus quiscula*)	9	1	1
Brown-headed Cowbird (*Molothrus ater*)	7	1	1

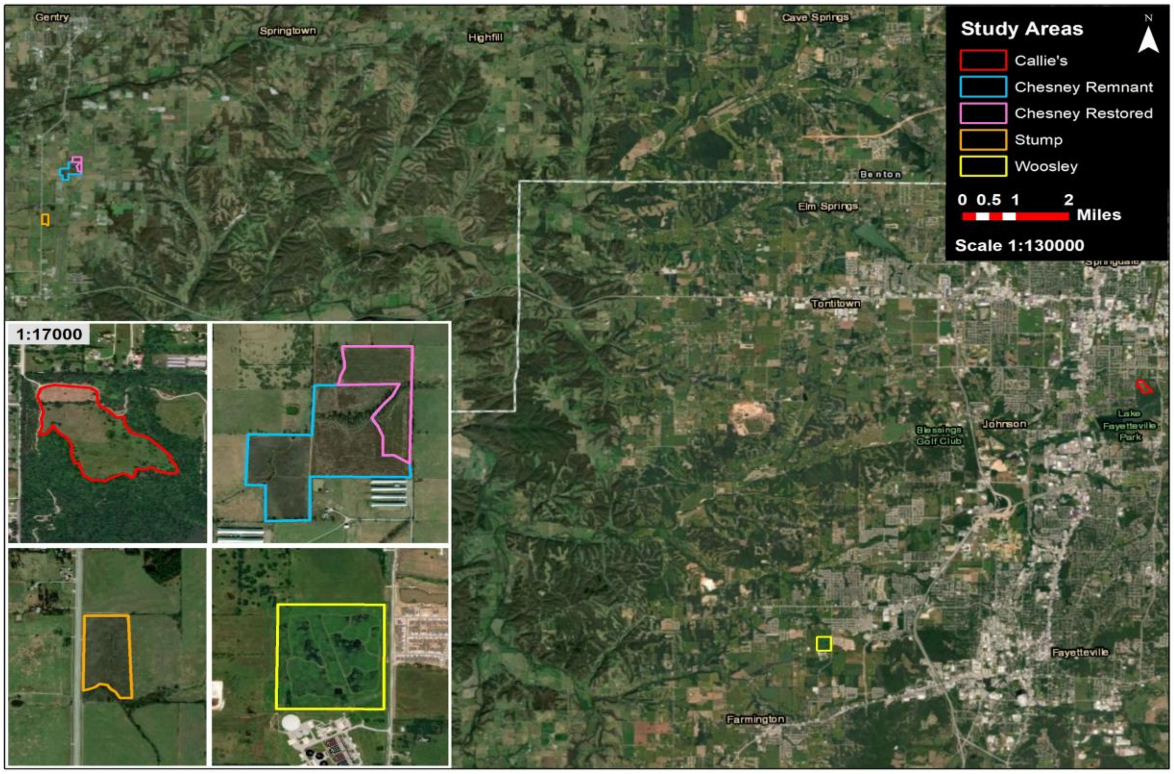


Figure 1. Map of study sites: Callie's Prairie (Fayetteville), Chesney Remnant, Chesney Restored, and Stump Prairies (Siloam Springs), and Woolsey Prairie (Fayetteville), Northwest Arkansas, USA. ESRI ArcGIS ArcMap, version 10.6.

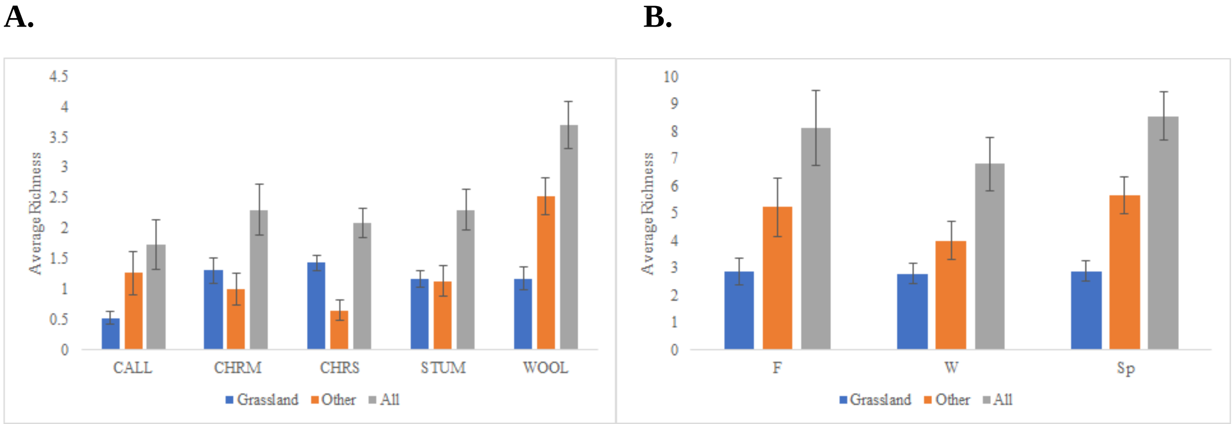


Figure 2. Mean species richness per survey for grassland-obligates (n = 7 species), non-obligates ("other," n = 34), and all species (n = 41), **A.** by site averaged across surveys, and **B.** by season with all sites combined and averaged by survey-week. Surveys occurred from September 2017-May 2018. Error bars show ± 1 SE. CALL = Callie's, CHRM = Chesney Remnant, CHRS = Chesney Restored, STUM = Stump, and WOOL = Woolsey. F = fall, W = winter, and Sp = spring.

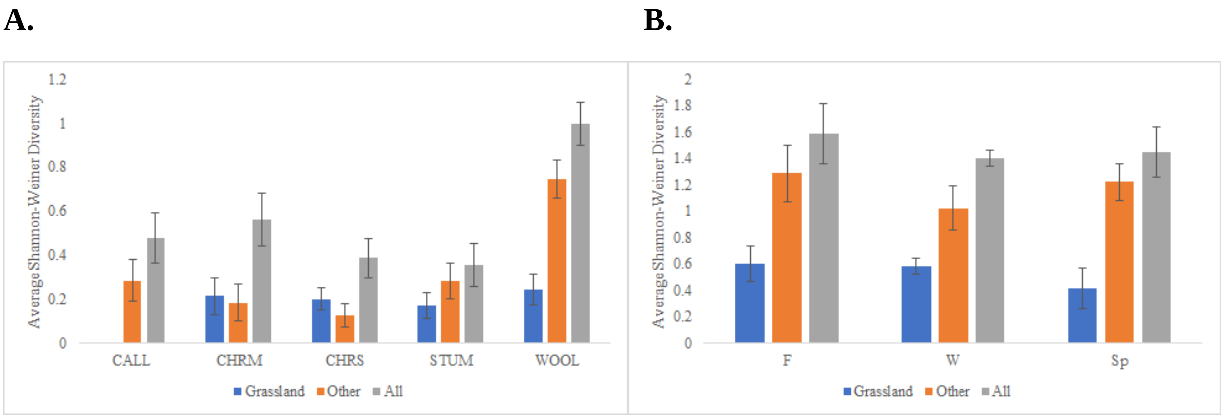


Figure 3. Mean Shannon-Weiner diversity (*H'*) per survey for grassland-obligate (n = 7 species), non-obligate ("other," n = 34), and all species (n = 41), **A.** by site averaged across surveys, and **B.** by season with all sites combined and averaged by survey-week. Surveys occurred from September 2017-May 2018. Error bars show ± 1 SE. CALL = Callie's, CHRM = Chesney Remnant, CHRS = Chesney Restored, STUM = Stump, and WOOL = Woolsey. F = fall, W = winter, and Sp = spring.

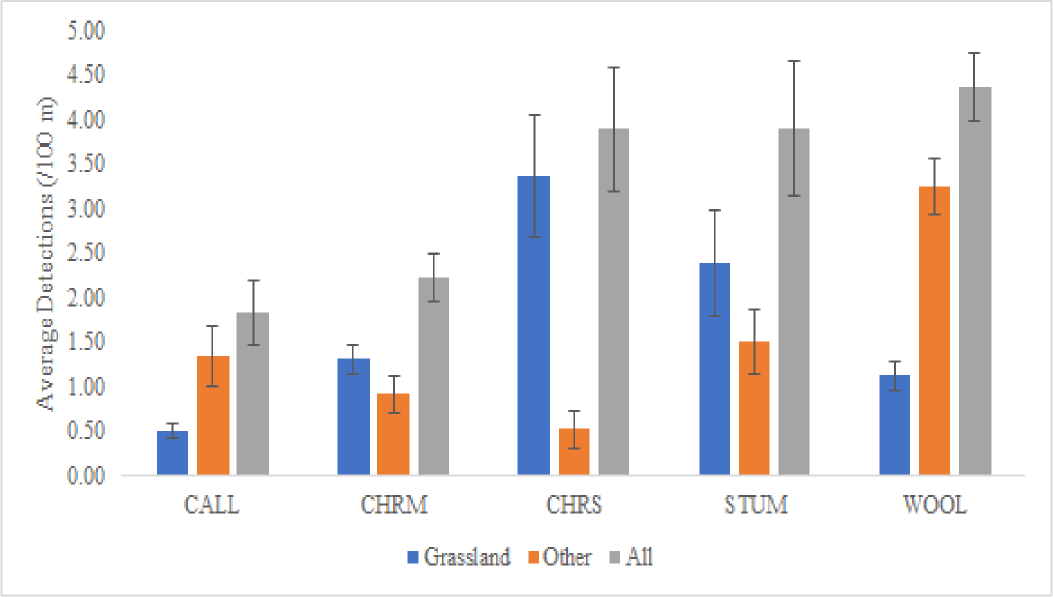


Figure 4. Mean total detections (per 100 meters per transect) for grassland-obligate, non-grassland obligate (other), and all species at each site, averaged across seasons. Surveys occurred from September 2017-May 2018. Error bars show ± 1 SE. CALL = Callie's, CHRM = Chesney Remnant, CHRS = Chesney Restored, STUM = Stump, and WOOL = Woolsey.

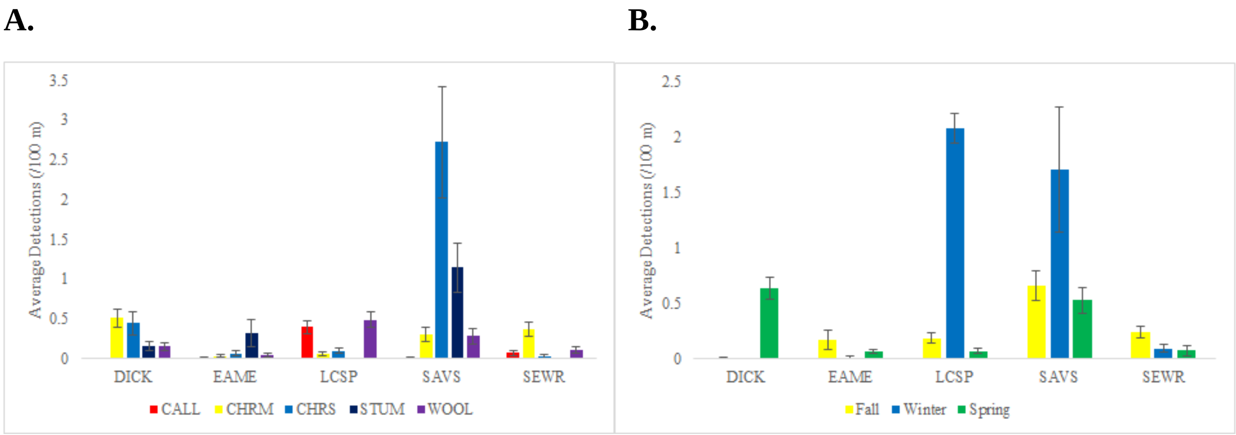


Figure 5. Mean total detections (per 100 meters per transect) of five grassland-obligate focal species **A.** by site (averaged across seasons) and **B.** by each season (averaged across sites). Surveys occurred from September 2017-May 2018. Error bars show ± 1 SE. DICK = Dickcissel, EAME = Eastern Meadowlark, LCSP = Le Conte's Sparrow, SAVS = Savannah Sparrow, and SEWR = Sedge Wren; CALL = Callie's, CHRM = Chesney Remnant, CHRS = Chesney Restored, STUM = Stump, and WOOL = Woolsey.

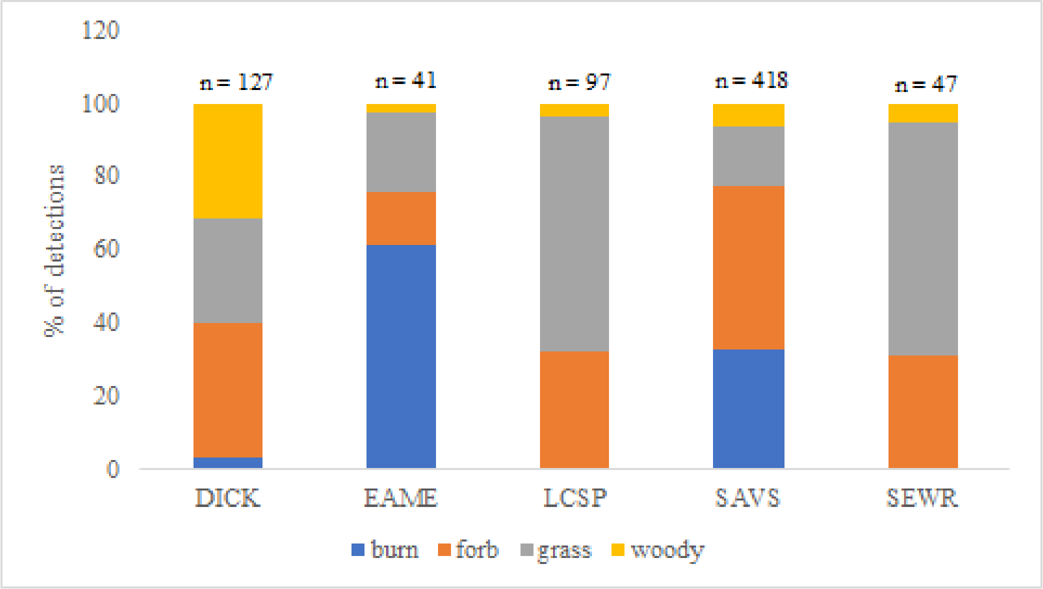


Figure 6. Recorded vegetation associations of five grassland-obligate focal species, cumulatively detected across all sites from September 2017-May 2018, where n represents the number of observations. DICK = Dickcissel, EAME = Eastern Meadowlark, LCSP = Le Conte's Sparrow, SAVS = Savannah Sparrow, and SEWR = Sedge Wren.

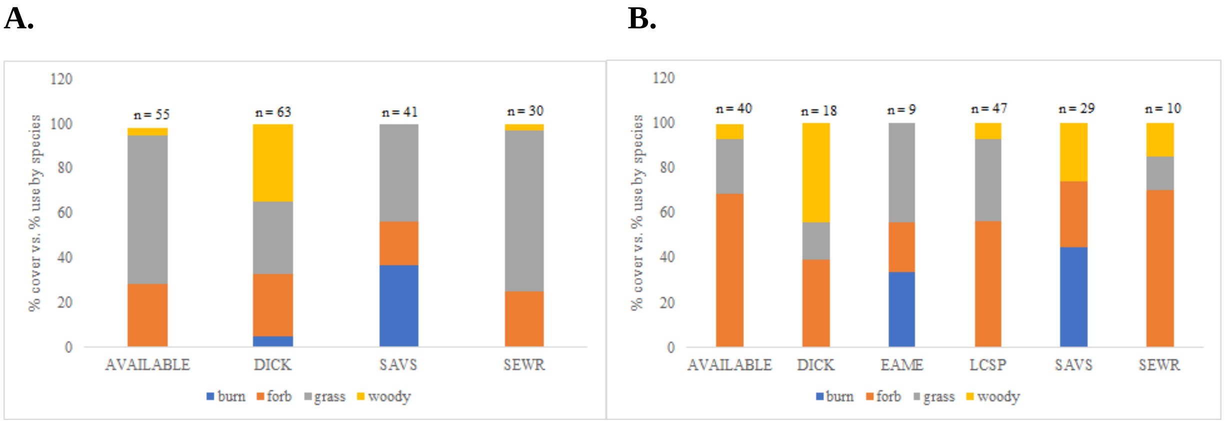
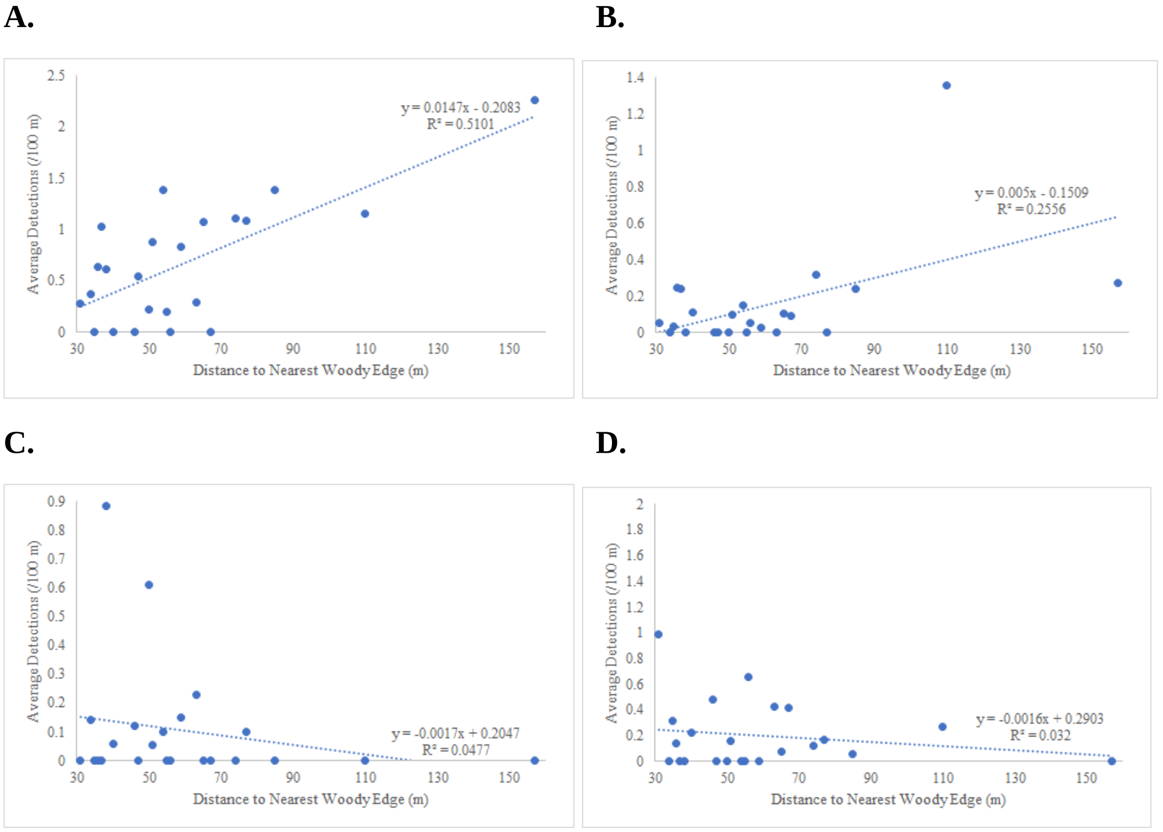


Figure 7. Average vegetation composition from random summer vegetation plots versus use for five grassland-obligate species, where n represents the number of observations. **A.** Dickcissel (DICK), Savannah Sparrow (SAVS), and Sedge Wren (SEWR) during all seasons at Chesney Remnant and **B.** Dickcissel (DICK), Eastern Meadowlark (EAME), Le Conte's Sparrow (LCSP), Savannah Sparrow (SAVS), and Sedge Wren (SEWR) during all seasons at Woolsey. Surveys occurred from September 2017-May 2018.



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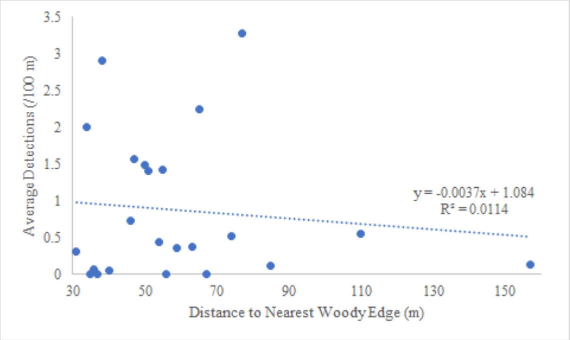

Figure 8. Linear regression of mean detections (per 100 m) and distance to nearest woody edge (m) at four sites across seasons; n = 23 transects. Surveys occurred from September 2017-May 2018, unless noted otherwise. **A.** Dickcissel (March-May 2018 only); $r^2 = 0.5101$, $p = 0.0001$. **B.** Sedge Wren; $r^2 = 0.2556$, $p = 0.0139$. **C.** Eastern Meadowlark; $r^2 = 0.0477$, $p = 0.3170$. **D.** Le Conte's Sparrow; $r^2 = 0.0320$, $p = 0.4141$. **E.** Savannah Sparrow; $r^2 = 0.0114$, $p = 0.6282$.

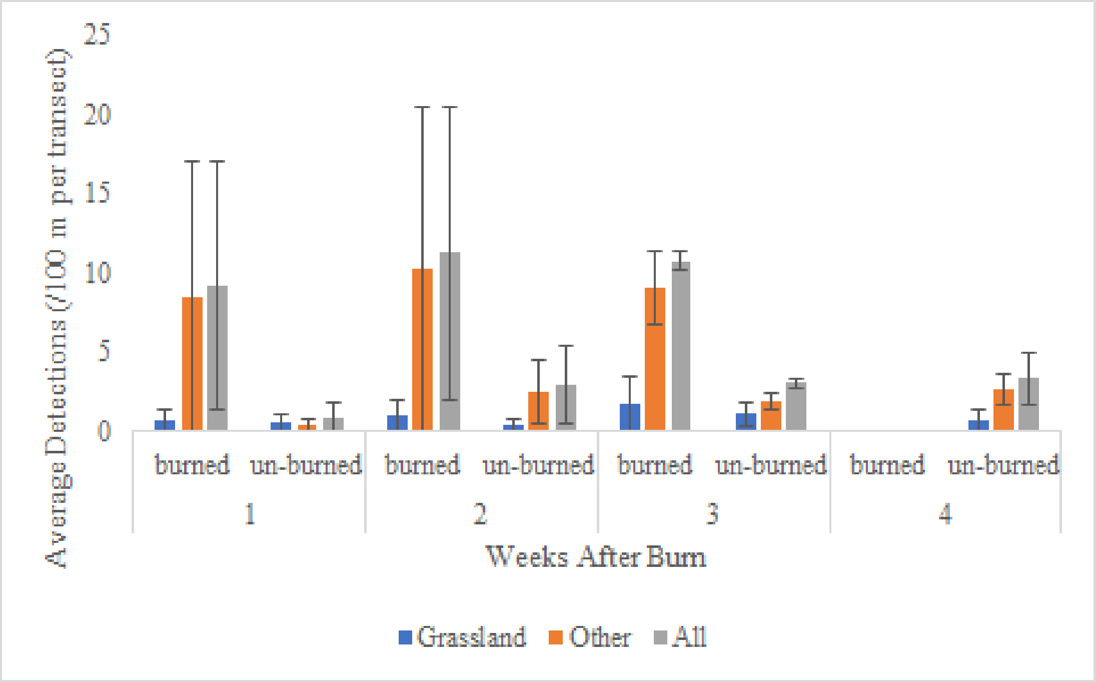

Figure 9. Mean detections (per 100 meters) of grassland-obligate and non-grassland obligate ("other") birds per transect during four weekly surveys post-burn in burned and un-burned sections within Chesney Remnant. Error bars show ± 1 SE. Burn was completed on 32% of parcel on October 19, 2017.

Conclusion

Grassland birds have declined more than any other terrestrial bird group in North America (Sauer et al. 2017). With less than 3% of tallgrass prairie left (DeLuca and Zabinski 2011), investigation of prairie avian ecology is essential to create effective conservation strategies. Severe habitat loss and fragmentation may be alleviated through habitat restoration (Haddad et al. 2015), which could be critical in protecting grassland birds. Understanding how breeding grassland birds fare reproductively and finding patterns of non-breeding grassland bird occupancy in reconstructed prairies is paramount to grassland bird conservation.

A handful of studies have shown that several grassland bird species are attracted to prairies restored through the United States Department of Agriculture's Conservation Reserve Program (CRP) during the breeding season (Johnson and Schwartz 1993, McCoy et al. 1999, Fletcher and Koford 2002, Ribic et al. 2009b), and have high nesting success for certain species (McCoy et al. 1999). However, CRP management may not be appropriately targeted at grassland bird conservation, as nesting avifauna benefit from patch-burning and removal of exotic flora (Fondell and Ball 2004, Churchwell et al. 2008, Monroe et al. 2016, Maresh Nelson et al. 2018). These management actions are typical in remnant tallgrass prairies but are seldom performed on CRP properties (Vandever et al. 2002). Yet even non-CRP restored grasslands may be lacking particular vegetative features that positively contribute to nesting success of grassland birds (Winter 1999, Fletcher and Koford 2002, Ammann and Nyberg 2005, Berkeley et al. 2007). It is important to identify the roles of vegetation and landscape variables in reconstructed grasslands for nesting prairie avifauna.

Most North American grassland birds spend a minimum of half the year during the non-breeding season (spring and fall migrations, and overwintering; Herkert et al. 1996), thus understanding their ecology is imperative to implement sound conservation plans (Herkert and Knopf 1998). The relatively few studies on non-breeding grassland avifauna suggested that some species may require large habitat patches (Robertson et al. 2011), some sparrows apparently tolerate woody vegetation (Pulliam and Mills 1977, Hovick et al. 2014), and creating a heterogeneous landscape via patch-burn management and increasing intervals between burns may promote high grassland-obligate bird species diversity (Hovick et al. 2014). However, it is unknown how these patterns apply to restored tallgrass prairies.

My research in Northwest Arkansas on nesting success of a locally abundant but declining neotropical migratory grassland songbird, the Dickcissel (*Spiza americana*; Temple 2002, Neal 2009, Sauer et al. 2017) in relic and reconstructed grasslands managed with prescribed fire and non-native plant control demonstrated that nesting success was not related to restoration status. Excluding nest predator presence, site size (positive relationship) and brood parasitism (negative relationship) were the best predictors of nest success. Large sites and acquiring land distant from paved roads where Brown-headed Cowbird (*Molothrus ater*) brood parasites may be particularly abundant (Sliwinski and Koper 2012) may especially benefit nesting Dickcissel. Prairie Kingsnake (*Lampropeltis calligaster*) was a primary nest predator, although it has not been previously documented as such. Contemporary management suggestions of reducing woody edge cover to lower nest depredation rates by snakes (Thompson and Ribic 2012) may be inadequate for Prairie Kingsnake, since it does not associate with woody edges (Richardson et al. 2006). Dickcissel typically selected nest sites with lower but more variable visual obstruction and less bare ground/litter cover compared to availability. Sites characterized by these vegetation variables may increase nesting density (Fondell and Ball 2004).

My transect surveys in the non-breeding season revealed that grassland-obligate diversity is comparable between sites, although one small isolated restored prairie had consistently lower grassland bird diversity than other field sites. Two species, Dickcissel and Sedge Wren (*Cistothorus platensis*) appeared sensitive to woody edge proximity, and therefore may require large habitat patches in the non-breeding season. Meanwhile, Le Conte's Sparrow (*Ammodramus lecontei*) was detected most frequently at sites featuring more woody vegetation, which may provide protective cover. Detection rates increased for Eastern Meadowlark (*Sturnella magna*) and Savannah Sparrow (*Passerculus sandwichensis*) in burned areas. By retaining some shrubs while rotationally burning small sections of a prairie, managers may establish diverse grassland habitats that satisfy diverse habitat requirements of non-breeding grassland-obligate birds.

My thesis work demonstrates that remnant and restored prairies in Northwest Arkansas are used similarly by grassland birds during migration and winter, and that Dickcissel nesting success does not vary on a restoration basis. Therefore, restored sites could be important additions to relic tallgrass prairies for nesting Dickcissel and non-breeding grassland birds. Careful consideration should be given to parcel size, patch-burn management, and distance from paved roads when managing restored tallgrass prairies.

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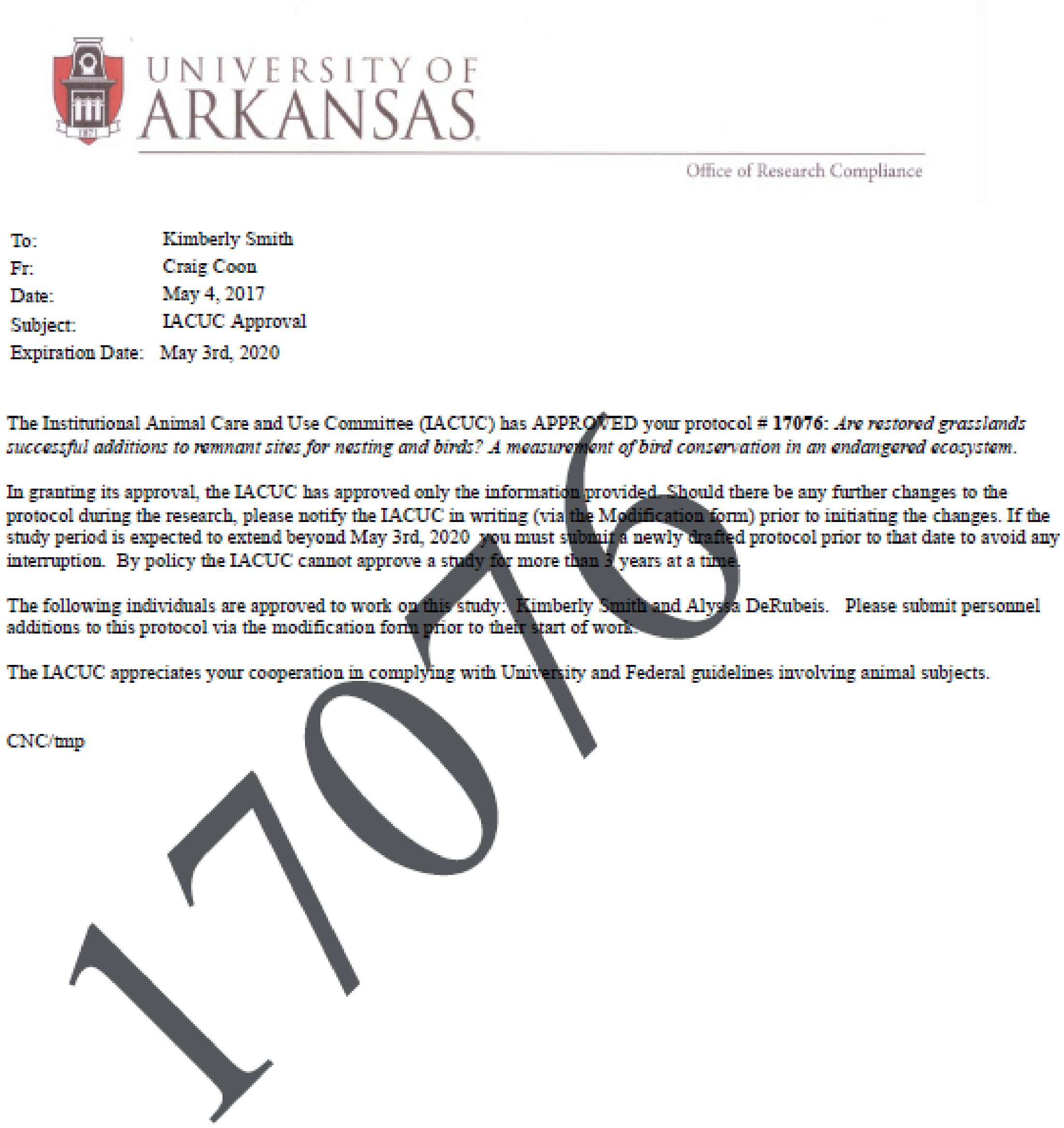
Appendix A: Research Compliance Protocol Letter



UNIVERSITY OF ARKANSAS
Office of Research Compliance

To:	Kimberly Smith
Fr:	Craig Coon
Date:	May 4, 2017
Subject:	IACUC Approval
Expiration Date:	May 3rd, 2020

The Institutional Animal Care and Use Committee (IACUC) has APPROVED your protocol # **17076**: *Are restored grasslands successful additions to remnant sites for nesting and birds? A measurement of bird conservation in an endangered ecosystem.*

In granting its approval, the IACUC has approved only the information provided. Should there be any further changes to the protocol during the research, please notify the IACUC in writing (via the Modification form) prior to initiating the changes. If the study period is expected to extend beyond May 3rd, 2020 you must submit a newly drafted protocol prior to that date to avoid any interruption. By policy the IACUC cannot approve a study for more than 3 years at a time.

The following individuals are approved to work on this study: Kimberly Smith and Alyssa DeRubeis. Please submit personnel additions to this protocol via the modification form prior to their start of work.

The IACUC appreciates your cooperation in complying with University and Federal guidelines involving animal subjects.

CNC/tmp