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DENSITY AND NEST SUCCESS OF SHRUB-DEPENDENT BIRDS ON FORMERLY STRIP-MINED LANDS

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A Thesis Submitted to the Graduate School in Partial Fulfillment of the Requirements for the Degree of Master of Science

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DENSITY AND NEST SUCCESS OF SHRUB-DEPENDENT BIRDS ON FORMERLY STRIP-MINED LANDS

An Abstract of the Thesis by
Luke A. Headings

As bird populations continue to decline across North America, it is important to understand the benefits that disturbed habitats can have for breeding birds. One of the major land disturbances and causes of habitat loss in the United States is surface mining, which often results in altered vegetative communities. The primary goal of this study was to evaluate the relationships between bird populations, habitat, previous and current land use, and densities of invasive plant species on formerly strip-mined land. Due to the proliferation of invasive shrub species in post-mined landscapes, we sought to determine the effects of post-mined habitat features on three shrub-nesting bird species: Bell's Vireo (*Vireo bellii*), Northern Cardinal (*Cardinalis cardinalis*), and Indigo Bunting (*Passerina cyanea*). In addition to assessing their densities, we estimated each species' reproductive success to understand future population trends. We conducted point count surveys, and searched for and monitored nests of these shrubland birds at 84 sites varying in land use and mining history. Overall, we detected 7,999 individuals from 87 bird species. Forested mined lands had the most diverse bird communities. We found that habitat type (i.e., forest, grassland, or rangeland) best described patterns in each focal species' density, with densities differing by habitat type for all three shrub-dependent species. We located 178 nests, the majority of which belonged to Bell's Vireos and Northern Cardinals. Logistic exposure models predicted daily nest survival for Bell's Vireos as a function of habitat type between post-mined grasslands and rangelands, while Northern Cardinals

daily nest survival was a function of nest age. If demographic rates were consistent across the study region, Bell's Vireo reproductive rates were not high enough to maintain their populations. Particularly as woody invasion continues, invasive shrub populations grow, and land cover changes occur in the Midwest, both species' breeding success may be negatively impacted, resulting in their population declines. This information will be useful for creating a more informed management plan for non-game birds and exotic plant species on reclaimed mined lands.

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CHAPTER I

DIVERSITY AND DENSITIES OF THREE SHRUB-DEPENDENT BIRD SPECIES ON FORMERLY STRIP-MINED LANDS

ABSTRACT

Shrubland habitats occupy a crucial position in ecological succession and host a wide variety of species, including shrub-dependent birds. Unfortunately, these habitats are often overlooked and understudied. Understanding the causes of bird declines in response to landscape changes is imperative, especially in the era of biodiversity loss. In this study, we examined the relationships between habitat features and densities of three shrub-dependent bird species on previously strip-mined land. We used fixed radius point counts to survey the bird communities on 84 locations in southeast Kansas and fit generalized linear mixed models to estimate densities of Bell’s Vireos, Northern Cardinals, and Indigo Buntings. We detected 7,999 individual birds from 87 species, including 13 species of conservation concern in Crawford and Cherokee counties. Habitat type was the best-supported model for predicting densities of all focal species. Bell's Vireos, Northern Cardinals, and Indigo Buntings had the highest densities in rangelands, forests, and grasslands, respectively. We demonstrated that formerly mined areas can support a diverse range of species, with the most diverse areas being the forested

sections. Management that creates a habitat matrix of multiple habitat types may support the greatest diversity of bird species.

INTRODUCTION

Birds in North America have suffered persistent and widespread population declines over the past 50 years. In a highly publicized paper, Rosenberg et al. (2019) reported trends that indicated 2.9 billion birds have disappeared from the continent since 1970, representing 29% of all individuals. These declines represent birds across multiple habitat types and life history traits. Grassland birds were the most affected group, exhibiting a 53% total loss and 74% of species in decline. Eastern forest birds have not shown declines as steep as grassland birds, but the trend is still concerning, with a 17% population decrease. The decline in populations of generalist species may be even more telling of population trends. A group of 38 habitat generalist species showed a 23% decrease in the same time frame, suggesting that even the most adaptable species are having trouble conforming to human-altered landscapes. Many factors influence these population trends, the most consistent across all regions being habitat loss (Rosenberg et al., 2019). Other important factors include increased pesticide use, exotic species, building collisions, predation by cats, emerging diseases and global climate change (Faaborg et al., 2010).

In the Midwest, the two primary threats to breeding birds are habitat loss and fragmentation, which are closely associated (Robinson et al., 1995). Fragmented habitats disrupt the interconnectivity of populations and may serve as population sinks for some specialist species. Agriculture is a main driver of habitat loss in this region, but urbanization and industrial land uses, such as mining, also have high impacts. For example, more than 95% of all tallgrass prairies were converted to agricultural land during the 19^{th} and early 20^{th} centuries, and remaining patches are often too small and isolated to support grassland specialists (Johnson & Igl, 2001; Powell, 2008).

Surface mining is a major form of land disturbance in the United States. Surface mining has resulted in the destruction of over 2.4 million hectares of terrestrial habitat since the 1930s (Lemke et al., 2013). Mining is distinct from most other disturbance types because of its comprehensive impact on ecosystems. Surface mining, in particular, changes the entirety of the ecosystem structure starting at the soil level. Soil horizons and pH levels in mined soils can take decades or centuries to return to suitable conditions for the original plant community (Skousen et al., 1994). The long-term impacts of mining on vegetation and wildlife communities are determined by the initial reclamation efforts on the mined site, which are highly variable depending on when the mining occurred. Land mined before the passing of the Surface Mining and Control Act (SMCRA) in 1977 was more likely to be abandoned to natural succession (Holl et al., 2018; Skousen et al., 1994; SMRCA, 1977). Following the passage of the SMCRA, the key reclamation objectives are typically to restore soil horizons and vegetation structure to the original status after mining operations are completed. Mined lands are often reclaimed with herbaceous plants because soil conditions and compaction from large machinery prevent tree regeneration (Lautenbach et al., 2020). These grasslands are dominated by seeded plants, usually exotic cool season grasses and legumes, for at least 20 years after reclamation (Rummel & Brenner, 2003). Percent biomass of seeded species is positively related to topsoil depth during the reclamation process (Pinchak et al., 1985). This suggests that older pre-

SMCRA mined lands are more vulnerable to aggressive successional species with high environmental tolerances.

As land uses, such as agriculture and mining, and habitat fragmentation continue to change the landscape, it is important to understand the response of bird communities to habitat disturbance. Reclaimed mined lands can benefit a wide variety of wildlife including birds, small mammals, reptiles, and amphibians (Carrozzino et al., 2011; Rummel & Brenner, 2003). Restoration principles implemented for post-mined landscapes include establishing suitable soil for the target plant species, providing seed sources for recolonization, using non-aggressive ground cover and planting a variety of species (Holl et al., 2018). Mined lands are difficult to restore to original habitat conditions because of the scope of the disturbance and due to their poor soil conditions (Wali, 1999). Examples of indicators of successful grassland reclamation in the Midwest include tall ground vegetation, dense ground cover (40–85%), low canopy cover (< 40%), and patch size minimums for target species (Rummel & Brenner, 2003). In reclaimed forests, heterogeneity of the vegetation structure may be the most important factor affecting bird species diversity (Karr, 1968). Reclamation goals for each habitat type are necessary for creating adequate habitat to support associated bird populations and communities (Reiley & Benson, 2020).

Habitat restoration and management is essential to maintain native plant communities, especially in the forest-prairie ecotone of the Midwest. Invasive plants exhibit characteristics that make them highly competitive, such as growth under variable moisture conditions, clonal growth, extended flowering periods, and allelopathy (Cadotte et al., 2006). Mined lands are especially vulnerable to invasion because invasive plants respond positively to disturbance, early successional environments, low diversity of native species and high environmental stress (Lemke et al., 2013). Additionally, woody encroachment continues to threaten grassland ecosystems in the Midwest due to fire suppression, heavy grazing, climate change, and introduction of exotic species (Anadon et al., 2014). Woody cover provides perches for birds, which encourages a positive feedback loop of encroachment through seed defecation from perches (Lautenbach et al., 2020). Eastern Red Cedar (*Juniperus virginiana*) has had a particularly prolific expansion in the forest-prairie ecotone. Though a native species, the growing stock volume of red cedar increased in Kansas by 15,000% from 1965–2010. Eastern Red Cedar not only encroaches on grasslands, but also into forests, suppressing the oak-dominated forests that constitute just 5% of Kansas's land base (Galgamuwa et al., 2020).

Avian community composition often changes dramatically with succession following disturbance. In some cases, reclaimed mined lands can support similar diversity of birds to unmined areas and provide quality habitat for grassland, shrub- and forest-dependent birds (Carrozzino et al., 2018; Graves et al., 2010; Karr, 1968). However, when the percentage of woody cover increases and distance to woodlands decreases, grassland obligate birds are quickly replaced by shrubland species, such as Bell's Vireos (*Vireo bellii*), Northern Cardinals (*Cardinalis cardinalis*), and Indigo Buntings (*Passerina cyanea*) (Graves et al., 2010). Shrubland bird species may remain in these habitats between 10–12 years post-disturbance, though soil loss from surface mining may delay the transition from shrublands to forests (Hollie et al., 2020).

The primary goal of this study was to evaluate the relationships between bird communities, vegetation structure, land use, and mining history on strip-mined land. We described bird diversity and evenness, and modeled species responses to vegetation structure. We tested the impacts of land cover type, the presence of exotic plants, and overall plant structure on the densities of three common shrub-dependent bird species that occur at high densities on mined areas: Bell's Vireos, Northern Cardinals, and Indigo Buntings. We predicted that bird densities would be positively related to shrub vegetation structure, but negatively related to invasive plant cover. Information on bird use of abandoned strip-mined land should guide the prioritization of habitat features in formerly mined landscapes.

METHODS

Study Area

We studied shrub-dependent birds on abandoned mined lands in southeast Kansas, which is part of the Cherokee Lowlands ecoregion. This ecoregion spans Bourbon, Crawford, Cherokee, and Labette counties, totaling about 259,000 hectares (Buchanan & McCauley, 2010). The variable climate is characterized by cold winters and hot, dry summers. Monthly average temperatures ranged from 0.66ºC in January (coldest month; average daily min. -4.55ºC, max 5.94ºC) to 26.72ºC in July (hottest month; average daily min. 21.17ºC, max 32.33ºC) (NOAA, 2023). Average annual precipitation was 121.64 cm, with the most precipitation falling in spring (39.19 cm) and summer (36.09 cm; NOAA 2023).

The native ecosystems in this region included tallgrass prairie with smaller patches of oak-hickory forests. However, over 90% of historical prairie habitat has been converted to row crop agriculture, creating a diverse matrix of croplands, grasslands, and forest. Strip mining activity also played a prevalent role in land use change for this region. Strip mining for coal occurred from the 1860s to the early 1970s using a variety of methods, with the majority using large electric draglines (Kansas Geological Survey, 2021). All mining activity in this region ceased in the face of incoming federal legislation for reclamation and restoration of mined lands (SMCRA, 1977), so many of these areas were abandoned to natural succession. The enormous electric shovels used for strip mining created a landscape of alternating overburden piles and water-filled pits that is still prevalent on mined lands today. The pits and overburden piles range in size from 2 m to 20 m deep/tall. The variability in terrain, in conjunction with dense vegetation, makes many mined areas impractical or unsafe to traverse by foot for ecological surveys.

After the cessation of mining activity, the Pittsburg & Midway Coal Company donated a large portion of their land to the state of Kansas, which resulted in the creation of the Mined Land Wildlife Areas (MLWAs). The majority of our study sites were on the MLWAs and other public lands in southeast Kansas (Figure 1.1). Sites were primarily located in Cherokee (n=63) and Crawford (n=9) counties in Kansas. In addition, we selected sites in the adjacent Barton (n=9) and Jasper (n=3) counties in Missouri. The MLWAs consist of 47 individual units totaling 5,868 hectares, including 1,619 hectares of grassland, 3,642 hectares of forest, and 607 hectares of open water. All but 809 hectares of the property was mined (KDWP, n.d.). We determined the mining history of these areas with a combination of characteristics. The most obvious indicator of mining history being the presence of strip pits. Other indicators included lack of topsoil and location on geological maps (Kansas Geological Survey 2021). The MLWAs included a wide range of successional stages due to the 100-year range of mining activity and varied restoration practices. We classified habitat types across the study region as forests, grasslands, or rangelands. Rangelands were classified as any area that was observed to be pasture for livestock during any part of the study. Forests were characterized by having thin rows of overburden piles and pits running through the entire area (Figure 1.2). Grasslands and rangelands were typically graded flat and had much deeper and wider pits (Figure 1.3). Now under the management of the KDWP, a variety of management practices were used on the majority of study sites including prescribed burns, native grass restoration, water level management, mowing, food plots, and livestock grazing (KDWP, n.d.).

Site Selection

We identified 84 point count locations, twenty of which were located in forests, 37 in grasslands and 18 in rangelands. The distribution of sites between habitat types was determined by availability, with far fewer rangelands available and many forests MLWAs unsuitable for this study. Sites were selected to achieve representative spatial coverage of the region while allowing for accessibility. Prior to sampling, each site was visited to evaluate the location for accessibility, noise, habitat type, and any other factors affecting suitability for the project. To select point count sites, we overlaid a 100 x 100 m grid on Google Earth satellite view, assigned a number to each box on the grid, and used a random number generator to select the box where the point count location would be located. Grids were placed 100 m from any habitat borders to prevent bias from adjacent habitats. Points were placed 200 m apart in forests and 250 m apart in grasslands and rangelands to avoid double-counting individuals (Hutto et al., 1986). We placed sampling locations farther apart in grasslands and pasture because noise carries further in those habitats.

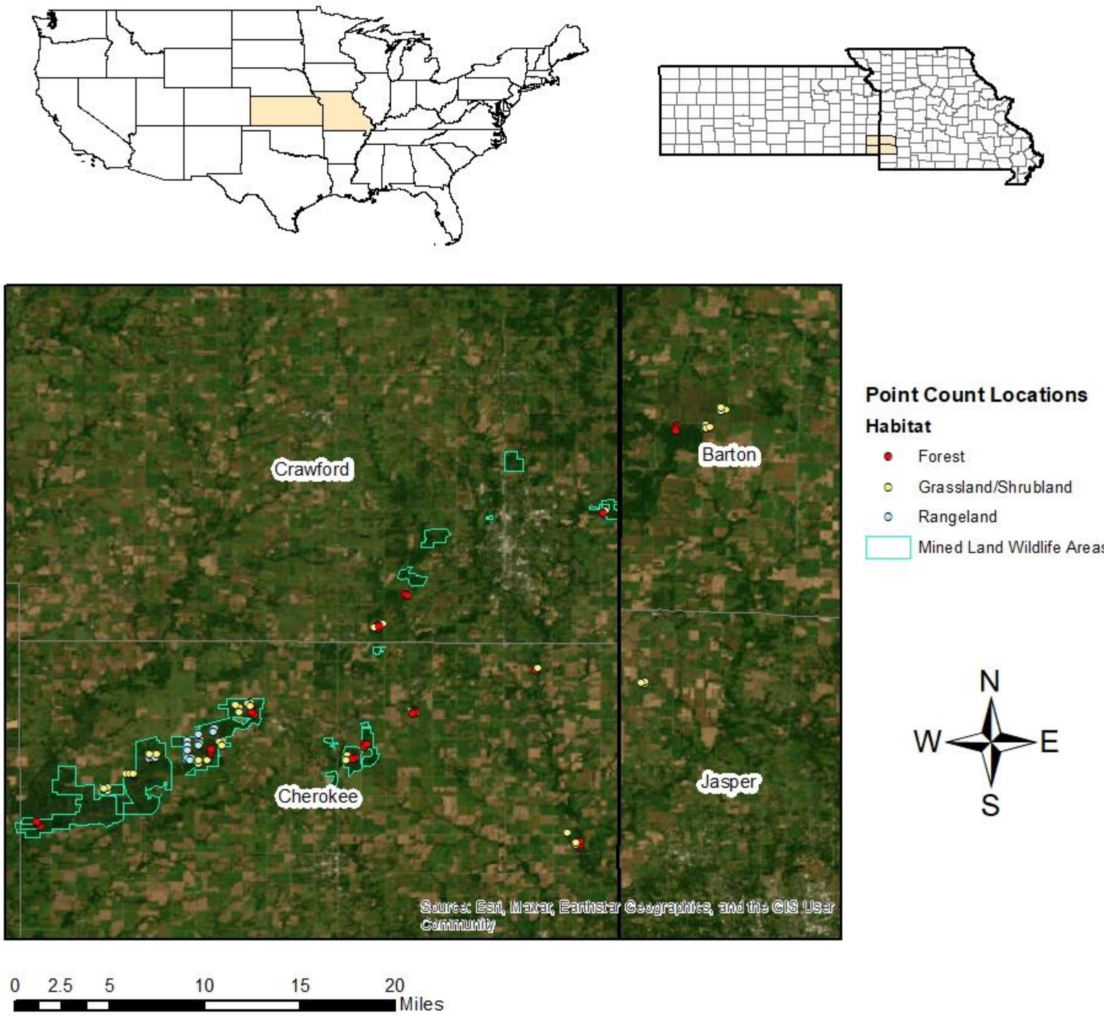

Figure 1.1. Map of surveyed mined land areas in southeast Kansas (Crawford and Cherokee counties) and southwest Missouri (Barton and Jasper counties). Point count locations are represented by their habitat categories.

Figure 1.2. Aerial view of the typical landscape of a forested unit on the Mined Land Wildlife Areas. Photograph from LJWorld.com by Mike Belt, 2007.

Figure 1.3. Typical MLWA habitats: MLWA 40 grassland (top photograph, foreground), MLWA 21 rangeland (top photograph, background), and MLWA 17 managed grassland, with MLWA 17 forest fragments in the distance (bottom photograph).

Field Methods

We performed five-minute fixed-radius point counts at each site three times during the breeding season (May 16–June 30) for three years (2020–2022) (Bibby et al., 2000). Counts were conducted from sunrise until four hours post sunrise. We did not conduct point counts if wind speeds surpassed 8 km/h, during sustained rain, in temperatures above 35ºC, or if other noisy conditions occurred (i.e., construction or road noise; Buckland, 2006). Counts were broken into four distance classes 0–24 m, 25–49 m, 50–99 m, and 100+ m (Ralph et al., 1995). We recorded the following detection variables with every point count: site ID, date, observer, start time, end time, cloud cover, wind speed, air temperature, visit number, and any additional notes specific to that visit. Before conducting a point count, we became familiar with the site by identifying landmarks for each of the distance classes with a rangefinder. Sites were approached as quietly as possible to avoid flushing birds. If any birds were flushed, a note was made of species and distance from point count location. The environmental readings, wind speed, air temperature, and cloud cover were collected prior to the count to give the area time to quiet down and accustom birds to the presence of the observer, typically two minutes. For each bird detected, we recorded time of detection, species (alpha code), distance class from observer, type of detection (i.e., fly through, seen, or heard), cardinal direction of the detection, and additional notes such as breeding behavior. If a flock was too large to count, we recorded an estimated range. During collection periods with multiple observers, we alternated point count sites between observers to minimize bias.

We used a variety of methods to collect vegetation data within a 11.3-m radius circle centered on each point count site (James & Shugart, 1970). We sampled five random locations using a Daubenmire frame (30 x 50 cm) to estimate ground cover of artificial surface, bare soil, forbs, grass, leaf litter, rock, shrubs, trees, woody litter, and water (Bonham et al., 2004). We identified trees (> 8 cm DBH) and shrubs (< 8 cm DBH, > 1 m height) to species, and classified the species as exotic or invasive according to the Kansas Forest Service's invasive species list (Kansas Forest Service, n.d.) and the Kansas Department of Agriculture's noxious weed list (Kansas Department of Agriculture, n.d.). We measured tree diameter-at-breast-height (DBH) using a D-tape (Metric Fabric Diameter Tape, Forestry Suppliers Inc., 205 W Rankin St, Jackson, MS 39201). Snag trees were included in these measurements because they are important habitat features for cavity nesting species. We visually estimated percent shrub cover of the plots and measured vertical vegetation density using a Nudd's board (Nudds, 1977). The board was placed in the center of each plot and the observer recorded how much of the board's five sections was covered by vegetation when viewed from 11.3 m away and from each cardinal direction. Tree canopy cover was estimated with a spherical densiometer (Spherical densiometer, Model-A, Forest Densiometers, 10175 Pioneer Ave Rapid City, SD 57702-4756). We also measured grass or other dominant ground cover height with a meter stick.

Statistical Analysis

We used generalized linear mixed models within an information theoretic framework to examine relationships between habitat features and densities of our focal species (Burnham and Anderson 2002, Bolker et al 2009). To account for differences in detection probability, we first estimated p (availability) and q (perceptibility) for each species and visit using time-removal models and distance models, respectively, based on the raw point count values binned by distance and time interval (Sólymos et al., 2013). The log product of both parameters was then included as an offset term in Poisson models with raw point counts as the response variable and different combinations of habitat characteristics as the predictor variables. We included a random intercept in all models to account for non-independence of point counts from the same sites. The candidate models represented *a priori* hypotheses regarding effects of grass height, invasive shrub cover, invasive forb cover, shrub cover, basal area, vertical vegetation density, habitat type (i.e., grassland, rangeland, forest), and past mining occurrence (i.e., mined or unmined). Correlated variables ($r > 0.7$) were excluded from the same models. We then ranked and sorted models using Akaike's Information Criteria (AICc) and model weights. Informative models with ΔAICc < 2 were considered supported (Burnham and Anderson, 2002, Arnold, 2010). We also calculated Shannon diversity of each habitat using package vegan (Oksanen et al., 2022). All analyses were conducted in Program R, version 2021.09.0 (R Core Team, 2021).

RESULTS

We recorded 87 bird species from 7,999 total detections during point counts (Appendix I), including 13 species of conservation concern for southeast Kansas counties (Appendix II). Our most frequently detected species was the Dickcissel *(Spiza americana*), with a total of 1286 detections. We detected 276 Bell's Vireos, 757 Northern Cardinals, and 496 Indigo Buntings. Across habitats, forests had the highest Shannon diversity score (H' = 2.81) (Figure 1.4).

Our best-supported density models for all three shrub-dependent species indicated that habitat type was the leading variable in predicting their densities (Table 1.1). Bell's

Vireos occurred at similar densities on grasslands and rangelands, with no individuals detected in forest habitats (Figure 1.5; Table 1.1). Northern Cardinals had the highest densities in forest habitat, intermediate densities on grassland habitat, and the lowest densities on rangeland habitat (Figure 1.6; Table 1.2). Indigo Buntings had highest densities on grasslands, intermediate densities in forests, and lowest densities on grasslands (Figure 1.7, Table 1.2).

Vegetation structure in forested locations differed from that in either rangeland or grassland, but was similar between grassland and rangeland. Forested areas had the greatest canopy cover, invasive shrub cover, basal area, vertical vegetation cover, and overall shrub cover (Table 1.3). Grassland and rangeland had similar vegetation makeup with a few notable differences. Grassland had the tallest grass height, double that of rangeland, as well as the greatest invasive forb cover, which was made up of over 90% Sericea Lespedeza (*Lespedeza cuneata*) (Table 1.3). The most common invasive shrub species overall were Bush Honeysuckle (*Lonicera maackii*), Japanese Honeysuckle (*Lonicera japonica*), and Autumn Olive (*Elaeagnua umbellate*). Grassland also had greater vertical vegetation density and shrub cover than rangeland (Table 1.3).

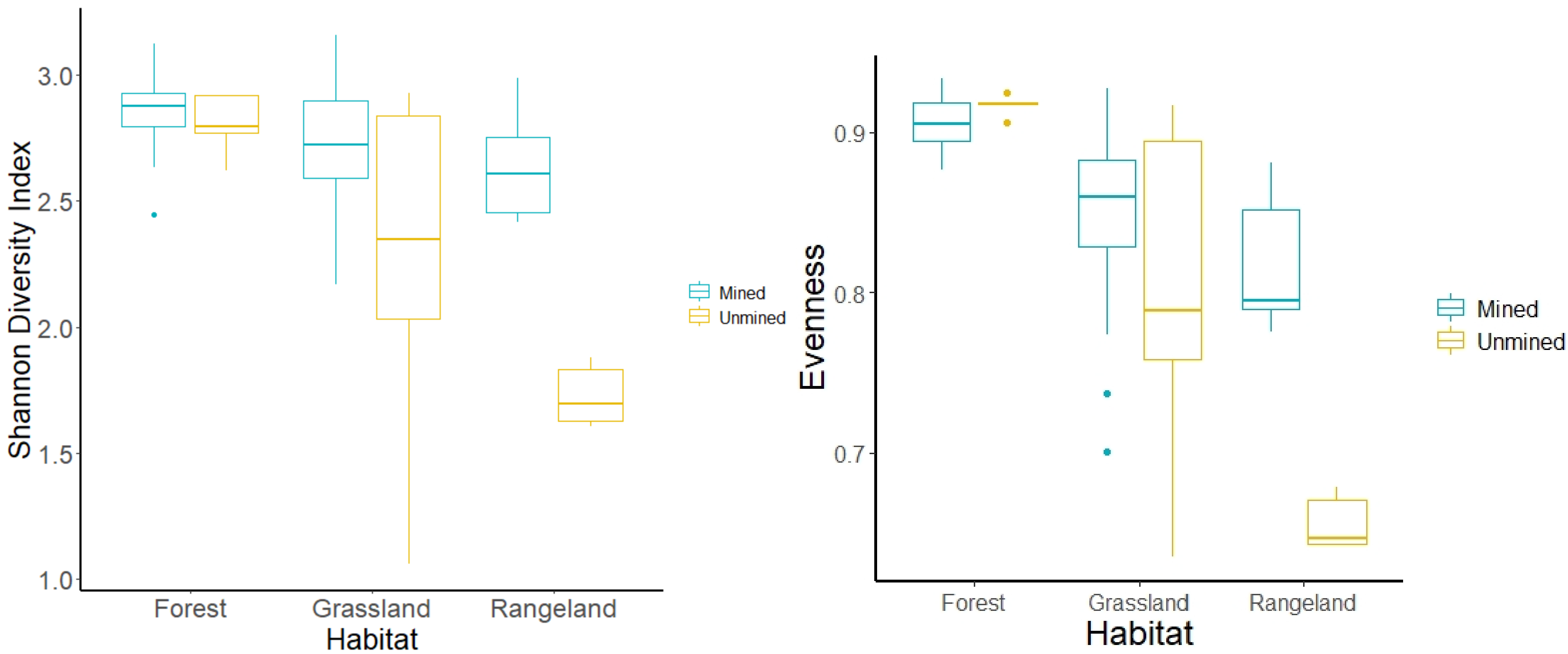

Figure 1.4. Shannon diversity index and species evenness of point counts across habitat type and mining history. Forests (mined n=26, unmined n=4), Grassland (mined n=25, unmined n=11), Rangeland (mined n=15, unmined n=3).

Table 1.1. A priori candidate models for Bell's Vireo, Northern Cardinal, and Indigo Bunting densities, estimated from point count data. Models included habitat types (i.e., forest, grassland, rangeland), basal area, canopy coverage (out of 100%), grass height ("Grass"), mining history (i.e., mined vs. unmined), shrub coverage ("Shrub"), invasive shrub coverage ("Invasive"), invasive forb coverage ("Forbs"), vertical vegetation density ("Nudds"), and forb ground coverage (i.e. forb, grass, shrub). Each model's ∆AIC, parameters (K) and weights (wi) are included.

Species	Model	∆AICc	K	w*i*
Bell's Vireo	Habitat	0	4	0.87
	Basal area	3.77	3	0.13
	Canopy	16.34	3	0.00
	Soil depth	32.26	3	0.00
	Invasive	40.31	3	0.00
	Mining	50.19	3	0.00
	Grass	50.49	3	0.00
	Shrub	52.42	3	0.00
	Nudds	53.01	3	0.00
	Forbs	53.06	3	0.00
	Null	53.77	2	0.00
Northern Cardinal	Habitat	0	4	1.00
	Nudds	24.63	3	0.00
	Basal area	36.53	3	0.00
	Invasive	51.20	3	0.00
	Mining	53.08	3	0.00
	Grass	57.34	3	0.00
	Forbs	58.95	3	0.00
	Soil depth	58.95	3	0.00
	Null	58.96	2	0.00
Indigo Bunting	Habitat	0	4	0.74
	Mining	3.23	3	0.15
	Grass	6.40	3	0.03
	Forbs	8.01	3	0.01
	Basal area	8.02	3	0.01
	Soil depth	8.03	3	0.01
	Nudds	8.08	3	0.01
	Shrub	8.08	3	0.01
	Null	8.08	2	0.01

Table 1.2. Coefficients of the best-supported models for Bell's Vireo, Northern Cardinal, and Indigo Bunting, based on point count data. The beta coefficients, standard errors (SE), and the lower and upper 95% confidence intervals (CI) for each parameter are included. Forest habitat is the reference level.

Species		Coefficient	SE	Lower 95% CI	Upper 95% CI
Bell's Vireo	Intercept	-6.96	1.05	-9.02	-4.90
	Grassland	4.84	1.07	2.74	6.94
	Rangeland	5.00	1.10	2.84	7.16
Northern Cardinal	Intercept	-0.67	0.08	-0.83	-0.51
	Grassland	-0.62	0.11	-0.84	-0.40
	Rangeland	-1.31	0.15	-1.60	-1.02
Indigo Bunting	Intercept	-1.23	0.11	-1.45	-1.01
	Grassland	0.21	0.14	-0.06	0.48
	Rangeland	-0.29	0.18	-0.64	0.06

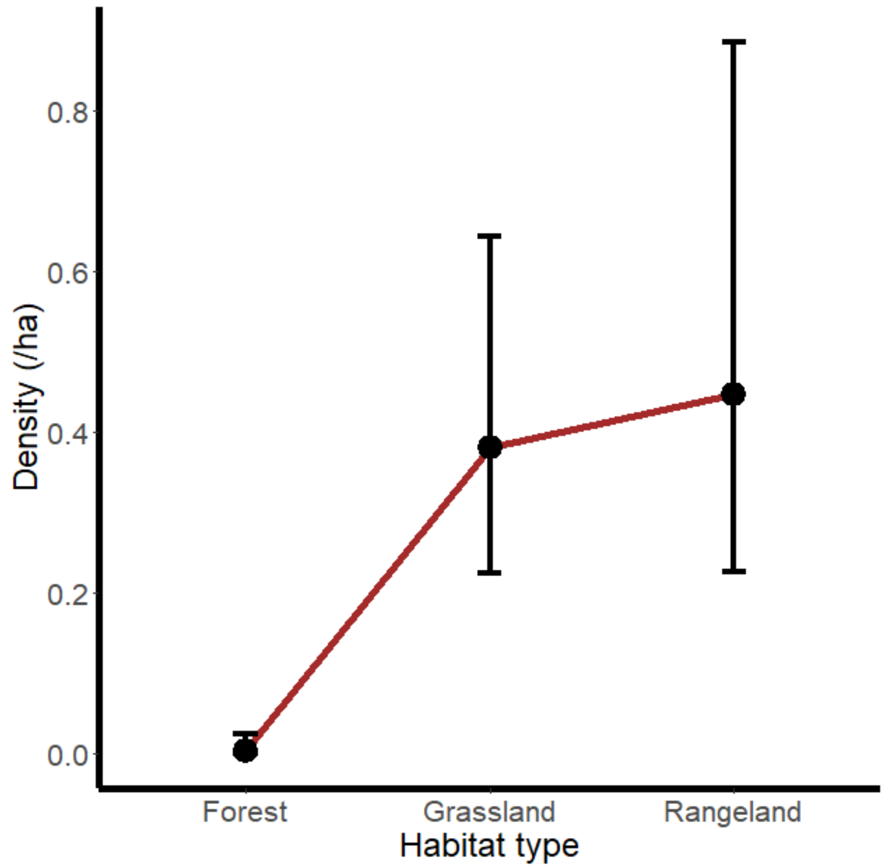

Figure 1.5. Model predictions from the best-supported models of the effects of habitat type on Bell's Vireo densities in southeast Kansas. Error bars indicate 95% confidence intervals.

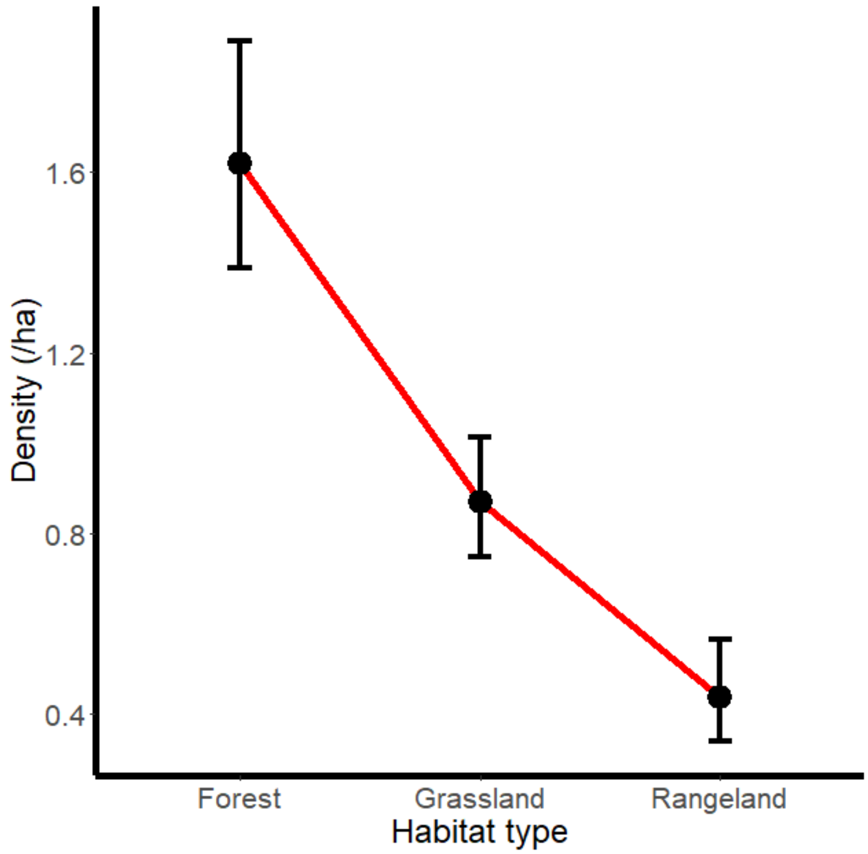

Figure 1.6. Model predictions from the best-supported models of the effects of habitat type on Northern Cardinal densities in southeast Kansas. Error bars indicate 95% confidence intervals.

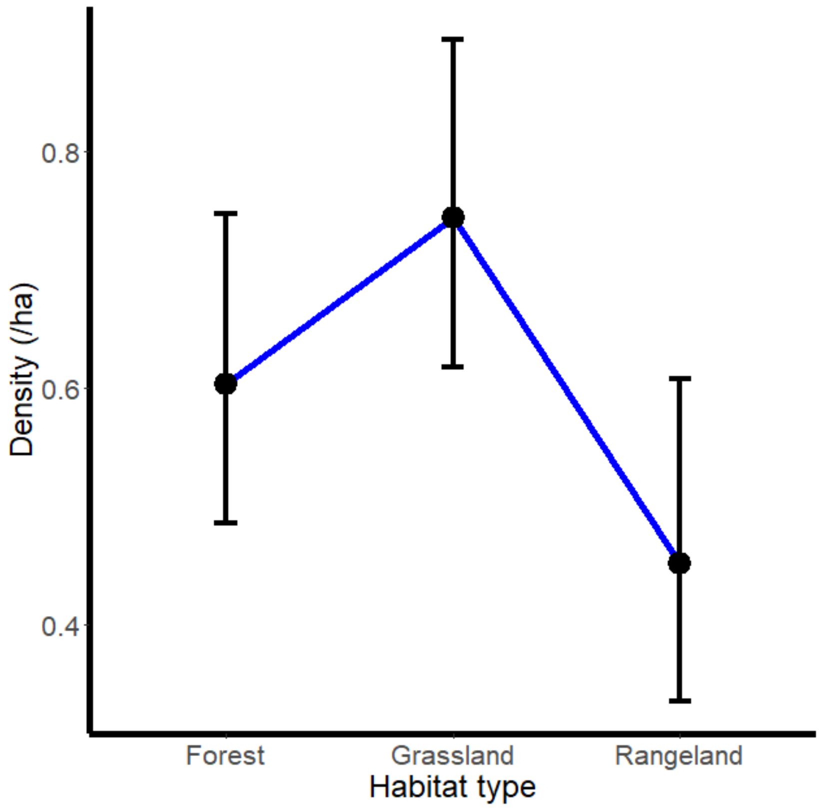

Figure 1.7. Model predictions from the best-supported models of the effects of habitat type on Indigo Buntings densities in southeast Kansas. Error bars indicate 95% confidence intervals.

Table 1.3. Mean vegetation values sampled each year at point count locations across three sampled habitat types. The number of point count locations surveyed for each habitat type are indicated in the first row.

Habitat Feature	Forest (n=30)	Grassland (n=39)	Rangeland (n=15)
Canopy Cover (%)	80.18 ± 26.84	4.55 ± 14.71	0.98 ± 4.43
Grass Height (cm)	32.84 ± 39.96	74.64 ± 48.33	38.82 ± 20.72
Basal Area	72.09 ± 39.11	3.63 ± 13.32	2.77 ± 10.41
Invasive Tree Cover (%)	0.77 ± 2.02	0.14 ± 0.64	0.06 ± 0.31
Shrub Cover (%)	42.64 ± 25.38	13.54 ± 13.87	9.53 ± 14.46
Invasive Shrub Cover (%)	17.92 ± 22.99	1.88 ± 4.20	0.57 ± 1.46
Invasive Forb Cover (%)	4.19 ± 9.01	5.44 ± 12.69	3.37 ± 6.33
Vertical Vegetation Density (%)	63.81 ± 24.05	38.97 ± 22.13	18.56 ± 16.70

DISCUSSION

We found that habitat type was the best predictor of population density for all three focal shrub-dependent bird species. Models with mining history were not supported, suggesting that whether a site was mined or not was less important than other characteristics. Habitat type, as determined by a variety of vegetation metrics, was the best explanation for bird diversity. Understanding how past land use history and current habitat conditions influence bird communities is imperative for habitat managers to effectively manage specific species or groups of birds.

There have been extensive studies globally of the relationships between bird community diversity and habitat (Goetz et al., 2014; Reif et al., 2022; Tu et al., 2020). In our study, forests had higher estimates of Northern Cardinal densities and overall diversity. One explanation for the lower species diversity on grasslands is the overall size and fragmentation of the grasslands in the study region (Herkert, 1994). Grassland units on the MLWAs were typically between 20 and 50 ha, while the minimum recommended size for a continuous grassland patch to support sensitive grassland species (e.g., Henslow's Sparrows, *Ammodramus henslowii*, and Grasshopper Sparrows, *Ammodramus savannarum*), range from 10–200 ha (Herkert, 1994; Vickery et al., 1994). The exception to this rule is Dickcissels, as they are less sensitive to grassland patch size (Herkert, 1994). This explains why Dickcissels were our most detected bird and also why evenness was lower on grasslands, as high densities of Dickcissels may have resulted in overall lower diversity. If area effects are contributing to the lower bird diversity that we observed on formerly mined grasslands, then management to increase the size and connectivity of grassland patches, may support more grassland obligate birds. Many mined areas had either hedgerows of trees running through grasslands or sparse numbers of trees scattered throughout. Decreasing the amount of woody vegetation would be beneficial to a number of species that we detected in low densities, such as Eastern Meadowlarks (*Sturnella magna*), Grasshopper Sparrows and Henslow's Sparrows. For forest habitats, there are limited management options for abandoned mined sites in this region. The difficult terrain and soil conditions make fire and mechanical techniques difficult and other approaches, such as regrading the surface to reseed vegetation may not be economically viable.

We observed major differences in vegetation characteristics between forested units and both grasslands and rangelands, with only minimal differences between grasslands and rangelands. Habitat designations for this project included a wide range of vegetation characteristics in each habitat. The designation of habitat types was difficult to distinguish between heavily shrubbed prairie, late stage shrubland and early successional forest. Using this designation system contributed to some of the high variation we observed in diversity differences between habitat types. Invasive shrubs made up a large portion of shrub cover in forests. However, we did not observe relationships between invasive plant densities and shrubland bird densities. Invasive plant species have complicated interactions with native bird species, with both positive, neutral and negative interactions (Maresh Nelson et al., 2017). Species such as bush honeysuckle, which was the most common invasive species in our forested habitats, often create monocultures. Even so, monocultures of invasive shrubs may not be harmful to bird species like Northern Cardinals, which often use bush honeysuckle as a food source (Ingold & Craycraft, 1983).

Northern Cardinals are typically associated with habitats characterized by shrubs and small trees, such as forest edges and openings within patch interiors (Halkin & Linville, 2021). Our findings support this habitat association, as we observed the greatest Northern Cardinal densities in forested units, and lower densities on grasslands and rangelands. Of the three focal species, Northern Cardinals have the least conservation and habitat concerns, as they adapt well to altered and anthropogenic landscapes and select for shrubby forest habitats like those found throughout previously mined lands.

Bell's Vireos were equally abundant on grassland and rangeland sites and absent from forested sites, which was expected, as they are a shrubland obligate species (Budnik et al., 2000). Bell's Vireos rely on grassland-shrub habitat that has largely been removed from the landscape with the removal of associated prairie habitat (Budnik et al., 2000). Early-successional wildlife habitat is largely overlooked and has become increasingly uncommon (King and Schlossberg 2014, DeGraaf & Yamasaki, 2003). Management of abandoned mined lands may provide opportunities to protect large areas of early-successional habitat. Formerly mined areas of the Midwest may be especially responsive to management because of the matrix of grassland and adjacent forested sites that harbor large populations of difficult to manage shrubs. Management that prioritizes the conservation of shrubland habitats will benefit shrub-dependent species such as Bell's Vireos. Approaches such as rotational prescribed burns and mechanical control could create a habitat matrix ranging from grassland to early successional forests.

Of our three focal species, densities of Indigo Buntings varied the least between habitat types, with the highest densities in grasslands. Indigo Buntings are associated with forest edges, and they prefer habitats with complex patch shapes. Thus, it is not surprising that Indigo Buntings showed the lowest variation among habitats; smaller, irregularly shaped patches were abundant throughout the formerly mined sites in our study area (Weldon & Haddad, 2005). Similar to Northern Cardinals, Indigo Buntings are a common songbird species throughout the Midwest and on mined lands.

CONCLUSION

Reclamation efforts following intense human disturbances can supply habitat for a wide variety of wildlife. Even with minimal restoration efforts, the strip-mined land in our study region hosts considerable habitat variation and associate species diversity. We observed that habitat type was the best model for predicting density of three shrub-dependent bird species. While managing for shrubs in restored mined lands may not be suitable for all species, focusing efforts to improve habitats for shrub-dependent species of conservation concern could benefit bird diversity overall. Formerly mined lands provide an excellent opportunity to manage a diverse habitat matrix that may benefit a wide range of species throughout the region.

CHAPTER II

NEST SUCCESS OF SHRUB-NESTING BIRDS IN A POST-MINED LANDSCAPE

ABSTRACT

As bird populations continue to decline in North America, it is important to understand the benefits that disturbed habitats can have for breeding birds. In this study, we tested how daily nest survival (DSR) responded to land use, habitat type, and vegetation characteristics across forest, grassland, and rangeland sites in a formerly strip-mined landscape. We searched for and monitored nests of shrubland birds at 84 sites in southeast Kansas and southwest Missouri. We located 178 nests, the majority of which belonged to our focal species: Bell's Vireo (*Vireo bellii*, 69% of nests), Northern Cardinal (*Cardinalis cardinalis*, 16%), and Indigo Bunting (*Passerina cyanea*, 4%). Logistic exposure models estimated daily nest survival for Bell's Vireo as a function of habitat type, with grasslands having a DSR of 94% and rangelands a DSR of 89%. This relationship could be the result of 10% more invasive plant cover and 17% higher chance of Brown-headed Cowbirds (*Molothrus ater*) brood parasitism in rangeland habitats. Northern Cardinals had an average DSR of 91%, with nest age negatively associated with their DSR. If demographic rates are consistent across the study region, Bell's Vireo reproductive rates are not high enough to maintain their populations.

INTRODUCTION

Surface mining is one of the most destructive forms of land use, and has resulted in over 2.4 million hectares of terrestrial habitat disturbance in the United States since the 1930s (Lemke et al., 2013). Mining is distinct from most other disturbance types because of the comprehensive impacts on ecosystems. Surface mining disturbance is thorough and persisting because it destroys both the soil and vegetation. Soil horizons and pH levels can be altered to the point that it takes decades or centuries for conditions to be suitable for the original plant community to reestablish, which has direct consequences on wildlife habitat (Skousen et al., 1994).

The long-term impacts of mining on vegetation and wildlife communities are determined by the initial site conditions following the conclusion of active mining. A wide range of reclamation efforts may occur on previously mined lands, some of which are a result of the passing of the Surface Mining and Control Act in 1977 (Holl et al., 2018; Skousen et al., 1994; SMCRA, 1997). Current reclamation efforts implemented for mined lands include establishing suitable soil for vegetation regrowth, providing seed sources for recolonization, using non-aggressive ground cover, and planting a variety of species (Holl et al., 2018). Reclaimed grasslands are typically dominated by seeded plants, usually exotic cool season grasses and legumes, for at least 20 years post-reclamation (Rummel & Brenner, 2003). Land mined before the passing of the SMRCA was more likely to be abandoned to natural succession due to the lack of guidelines for its remediation and reclamation. Thus, older pre-SMCRA mined lands are more vulnerable to aggressive successional species with high environmental tolerances.

Most mined lands are reclaimed by seeding the area with quick-growing ground cover species mixed with a wide range of planted tree species, such as hardwoods and pines (Holl et al., 2018). If reclaimed mined lands are not managed as grasslands post-restoration, natural succession will proceed and the area will transition to shrublands and forests due to fire suppression, heavy grazing, and the introduction of exotic species (Anadon et al., 2014). Introduced and naturalized species have increasing impact in encroachment due to their highly competitive traits, tolerance of a variety of moisture conditions, clonal growth, extended flowering periods, and allelopathy (Cadotte et al., 2006). Mined lands are especially vulnerable to invasion because they exhibit habitat attributes that coincide with invasive species establishment such as heavily disturbed soils, early successional environments, low diversity of native species, and high environmental stress (Lemke et al., 2013). To compound this problem, many reclamation efforts included seeding with exotic species that were eventually listed as invasive, including Autumn Olive (*Elaeagnus umbellate*; Oliphant et. al., 2016) and Sericea lespedeza (*Lespedeza cuneata*; Zipper et al., 2011). Other common invasive species on mined lands included, Bush Honeysuckle (*Lonicera maackii*), Japanese Honeysuckle (*Lonicera japonica*), Tree of Heaven (*Ailanthus altissima*), Multiflora Rose (*Rosa multiflora*) and Silktree (*Albizia julibrissin*; Adams et al., 2019; Holl et al., 2018). Eastern Red Cedar (*Juniperus virginiana*), which is a native species, has had particularly prolific expansion in the central and eastern Great Plains. Considered a pioneer species for mined lands, it grows quickly, is well adapted to drought conditions, and has a long growing season (Burns & Service, 1990). For example, the growing stock volume of Easter Red Cedar increased by 15,000% in Kansas between 1965 and 2010 (Galgamuwa et al., 2020). Eastern Red Cedar not only encroaches on grasslands, but also into forests, suppressing native hardwood species (Galgamuwa et al., 2020).

As land use legacies and invasive plant species establishment continue to change the landscape, it is important to identify positive and negative biodiversity impacts of previously disturbed areas. Invasive plants in post-mined lands have a number of concerning ecological effects, as they alter habitat structure, change ecosystem processes and decrease native biodiversity (McNeish & McEwan, 2016). The primary concern is that these aggressive and prolific species will outcompete slower growing native species that have higher ecological value, lowering the overall diversity of the area. Invasive plants can also provide less adequate habitat and forage for invertebrates, resulting in decreased food availability for organisms at higher trophic levels (Love & Anderson, 2020; George et al., 2013). For example, Bush Honeysuckle was consumed by larval insects ten times less than native shrubs in the same environment (Love & Anderson, 2020). Even so, reclaimed mined lands can be beneficial to a wide variety of wildlife, including birds, small mammals, reptiles, and amphibians (Carrozzino et al., 2011; Rummel & Brenner, 2003). While numerous wildlife species do use reclaimed mined lands, habitat use does not necessarily indicate habitat quality (Stauffer et al., 2011). Assessing a species' reproductive effort is imperative to understand its future population trends.

Relationships between exotic plant species and native birds are complex and cannot be easily summarized, as they can have negative, neutral, or positive outcomes, depending on the situation. The ecological trap hypothesis is one of the most discussed ideas as to why invasive plant species can potentially decrease songbird productivity.

This hypothesis describes decreased nest success in exotic plants due to differences in leaf phenology and insect biomass, compared to their native counterparts (Donovan & Thompson, 2001; McChesney & Anderson, 2015; Rodewald et al., 2010). However, other studies indicate that birds may nest in invasive plant species without any negative effects to nesting success (Gleditsch and Carlo, 2014). A recent meta-analysis showed that bird species richness was negatively related to invasive plant densities, while some birds preferred to nest in invasive shrubs, and nest success typically remained neutral (Nelson et al., 2017). Additional ecosystem-specific information is needed to continue developing our understanding of how invasive plant species affect native bird reproduction.

Due to the proliferation of invasive shrub species in post-mined landscapes, we sought to estimate reproductive rates of three shrub-nesting bird species: Bell's Vireos (*Vireo bellii*), Indigo Buntings (*Passerina cyanea*), and Northern Cardinals (*Cardinalis cardinalis*). Several studies have examined relationships between vegetation or habitat relationships and reproductive rates of Northern Cardinals and Indigo Bunting in the Midwest, but few have examined the reproductive success of Bell's Vireos outside of the southwestern United States (Budnik et al., 2002; Chapa-Vargas & Robinson, 2013). Our goal was to identify the habitat and vegetation variables that were most likely to affect reproductive success for our target species in a post-mined landscape. We predicted that areas with higher densities of invasive plants would have lower reproductive rates and areas with high vertical vegetation density would have higher reproductive.

METHODS

Study Area

We studied bird nesting success on post-mined lands in southeast Kansas, which is part of the Cherokee Lowlands ecoregion. This region covers about 259,0002 hectares in Bourbon, Crawford, Cherokee, and Labette counties (Buchanan & McCauley, 2010). The variable climate is characterized by cold winters and hot, dry summers. Monthly average temperatures ranged from 0.66ºC in January (coldest month; average daily min. -4.55ºC, max 5.94ºC) to 26.72ºC in July (hottest month; average daily min. 21.17ºC, max 32.33 ºC) (NOAA, 2023). Average annual precipitation was 121.64 cm with the most precipitation falling in spring (39.19 cm) and summer (36.09 cm) (NOAA, 2023).

Southeast Kansas contains a number of different ecotones, particularly with the transition from oak hickory forests in the east to the tallgrass prairie regions to the west. This ecotone also coincides with increases in intensive row crop agriculture, which has replaced over 90% of the native prairie habitat. In addition to habitat conversion due to agriculture, southeast Kansas was strip mined for coal from 1860–1974 (Kansas Geological Survey, 2021). The mining operations were performed with enormous electric draglines, which created a landscape of alternating overburden piles and water-filled pits that have now revegetated to grasslands, shrublands, or forest. All mining activity in the region ended with new federal legislation for reclamation or restoration of mined lands in the process of being enacted, so many of these areas were abandoned to natural succession (SMRCA 1977). Even so, some restoration actions have occurred to grade the overburden piles and pits to a relatively flat surface, leaving a smaller number of large deep pits instead of numerous narrow pits. After mining activity was completed, the mining companies donated 5,867 ha of land to Kansas Department of Wildlife and Parks, which resulted in the creation of the Mined Land Wildlife Areas (MLWAs) across Cherokee, Crawford and Labette counties.

The majority of the MLWAs were characterized by dense rows of forest, split by shallow strip pits. Forests (3,642 ha) were dominated by Pin Oak (*Quercus palustris*), Black Walnut (*Juglans nigra*) and Eastern Red Cedar. The remaining area was split into 607 ha of open water and 1,618 ha of prairie, shrubland, and rangeland. We classified habitat types on the MLWAs as either forest (35%), grassland (44%), or rangeland (21% sites). The MLWAs are managed using a variety of standard approaches including prescribed fire, livestock grazing, mowing, native plant restorations, wetland restoration and mechanical removal of vegetation. However, the majority of the forested sites remain inaccessible to management due to the complex strip pit topography.

We surveyed 84 sites for bird nesting activity. Most of our study sites were located in the MLWAs and other public lands in southeast Kansas. In Kansas, we surveyed sites in Cherokee (n = 63) and Crawford (n = 9) counties, including focused efforts on MLWAs 4, 9, 12, 14, 17, 21, 38, 40 41, 42, and 45, as well as the Buche Wildlife Area, Monahan Outdoor Education Center, and Spring River Wildlife Area. We also surveyed sites in Missouri in Barton (Prairie State Park, n = 9) and Jasper (Wah-Sha-She Prairie State Natural Area, n = 3) counties. To identify study sites, we first scouted the location to evaluate each area for safety and accessibility. Then, we identified areas with high densities of our three focal species early in the point count season and from previous years' experience (Chapter 1). Sites were selected on a year-to-year basis due to the high occurrence of prescribed fire and brush clearing activity, which caused shifts in local densities of shrub-nesting birds, especially Bell's Vireo.

Field Methods

We searched for and monitored nests according to the BBird protocol (Martin et al., 1997). We searched for nests from mid-May through late July in 2020–2022. We used two primary methods to find nests: systematic searching of nesting habitat and observing bird behavior. The behaviors we looked for during nest searching included carrying nesting material or food items, carrying fecal sacs and defending a territory. We also found nests opportunistically during point count surveys or by flushing birds.

When a nest was found, we recorded the species, a location description to aid in finding the nest again, GPS coordinates, search method, date, time, observer, nesting stage, contents of nest and any comments. Once found, the nest was checked every 2–3 days. On each subsequent check, we recorded the date, time, observer, time at nest, nest stage, nest contents, adult location and activity, and the minimum and maximum age of the nestlings, if any. When possible, we checked the nests from a distance to minimize potential stress and likelihood of abandonment. We took different paths during each nest check to avoid creating a physical or scent trail leading to the nest. Upon nest fate completion (i.e., fledging of young, depredation, or abandonment), we attempted to determine nest fate and collected vegetation data. The fate clues we primarily looked for were presence of fledglings in the area, aggression from parents, and fecal sacs on the rim of the nest or below it.

We used a Daubenmire frame (30 x 50 cm) to estimate ground cover in vegetation plots at each nest (Bonham et al., 2004). The cover classes included artificial surface, bare soil, forbs, grass, leaf litter, rock, shrubs, trees, woody litter, and water. Trees (> 8 cm DBH) and shrubs (< 8 cm DBH, > 1 m height) were identified to species. Exotic and invasive plants were classified according to the Kansas Forest Service's invasive species list (Kansas Forest Service, n.d.) and the Kansas Department of Agriculture's noxious weed list (Kansas Department of Agriculture, n.d.). Each tree's diameter-at-breast-height (DBH) was measured with a D-tape (Metric Fabric Diameter Tape, Forestry Suppliers Inc., 205 W Rankin St, Jackson, MS 39201). Shrubs were not counted individually; instead, the percent total cover of the plot was visually estimated. We measured vertical vegetation density using a Nudd's board (Nudds, 1977). The board was placed in the center of each plot and the observer recorded how much of each section of the board was covered by vegetation standing 11.3 m away in each cardinal direction. Tree canopy cover was estimated with a spherical densiometer (Spherical densiometer, Model-A, Forest Densiometers, 10175 Pioneer Ave Rapid City, SD 57702-4756). Grass or other dominant ground cover height was recorded as well. We also recorded the nest substrate species and any other climbing or intertwined plant associated with the substrate plant, substrate height, nest height, and nest orientation. Nest cup visibility was measured using a 6.5 cm plastic disc divided into eight alternating black and white sections. We took measurements by placing the disc in the nest cup and recording how many of the sections were covered from directly above and from 1 m away for each of the four cardinal directions (Stauffer et al., 2011).

Statistical Analysis

We estimated daily survival rates (DSR) of nests following methods from Rotella et al. (2004). Estimates of nest success are an important metric for evaluating habitat management strategies and an essential component of demographic modeling (Jehle et al., 2004; Rotella et al., 2004). There are many ways to estimate nest success. Daily survival rate (DSR) calculates the probability that an individual nest will survive any given day. To be included in nest survival analysis, nests had to contain at least one host species egg for two or more nest checks. We used logistic exposure and program MARK to examine the relationships between DSR and habitat characteristics. Models were then ranked and sorted using Akaike's Information Criteria (AICc). Models within 2 ΔAICc were considered informative (Arnold, 2010). We included both null and global models in candidate model sets. We tested a variety of predictor variables including habitat type, nest age, proportion of shrub coverage on the nest vegetation plot, proportion of invasive shrub cover on the plot, invasive or native nest substrate, nest height, and year (Table 2.1). Correlated predictor variables ($r > 0.7$) were excluded from the same models. All models were fit and ranked in Program R, version 2021.09.0 (R Core Team, 2021) using package MARK (White and Burnham, 1999).

Table 2.1. Description of daily nest survival model variables.

Variable	Habitat Characteristic
NestAge	Age of nest in days
Time	Time series since day 1 of monitoring
PpnInv	Proportion of invasive vegetation within nest plot
NuddsAVG	Average vertical vegetation density as measured with a Nudds board
PpnShrub	Proportion of shrub coverage within nest plot
Year	Year
Habitat	Habitat Type (i.e., grassland, rangeland, forest)

RESULTS

From 2020–2022, we monitored 178 nests that included 159 nests of our three focal species (Table 2.2). We only found eight nests from Indigo Buntings, none of which were successful. Thus, we excluded Indigo Buntings from the analysis due to a lack of statistical power.

The leading cause of failure for Bell's Vireo nests was depredation, followed closely by nest parasitism (Figure 2.1; Table 2.2). Of all observed species, Bell's Vireo had the most nest failures due to livestock disturbance (Figure 2.1; Table 2.2). Bell's Vireo nests were found in only two of the three habitat designations for our study: grassland (n = 90) and rangeland (n = 32) (Table 2.3). Bell's Vireo nests had an average DSR of 93% (Figure 2.2). Habitat plus nest age was the best-supported model for DSR (Table 2.4). Nests in grasslands had an average DSR of 94%, while nests in rangeland had a DSR of 89% (Figure 2.3). On average, Bell's Vireo nests had 23.1% shrub cover on the nest vegetation plot, 13.7% of which was invasive shrub cover. Only ten (8%) of nests were placed in an invasive substrate (i.e., Bush Honeysuckle, Black Locust [*Robinia pseudoacacia*], and Autumn Olive). Brown-headed Cowbirds parasitized 50% of Bell's Vireo nests with at least one egg (Table 2.2). Nest parasitism was more likely to occur on rangeland habitat, with 66.7% of nests being parasitized compared to 46.7% on grassland. Damage from livestock was the cause of 4% of Bell's Vireo nest failures, all of which occurred on rangeland habitat. Additionally, rangelands had 9.7% greater invasive plant cover around the nests than those in grassland habitats.

Table 2.2. Number of nests detected with our nest searching efforts 2020–2022. Values include the number of nests that succeeded in fledging young and those that failed, listed by cause of failure.

		Failure Cause					
Species	Successful	Depredation	Parasitized	Livestock	Weather	Unknown	Total
Bell's Vireo	21	58	33	5	3	2	122
Northern Cardinal	6	19	3	0	0	1	29
Dickcissel	3	8	1	0	0	0	12
Indigo Bunting	0	6	1	1	0	0	8
Common Nighthawk	1	2	0	0	0	0	3
Scissor-tailed Flycatcher	1	1	0	0	0	0	2
Lark Sparrow	0	1	0	0	0	0	1
Kentucky Warbler	0	1	0	0	0	0	1
Total	32	96	38	6	3	3	178

Figure 2.1. Examples of a Bell's Vireo nest failure due to livestock disturbance on MLWA 21 in 2022 (left) and Brown-headed Cowbird nest parasitism and depredation on MLWA 17 in 2021 (right).

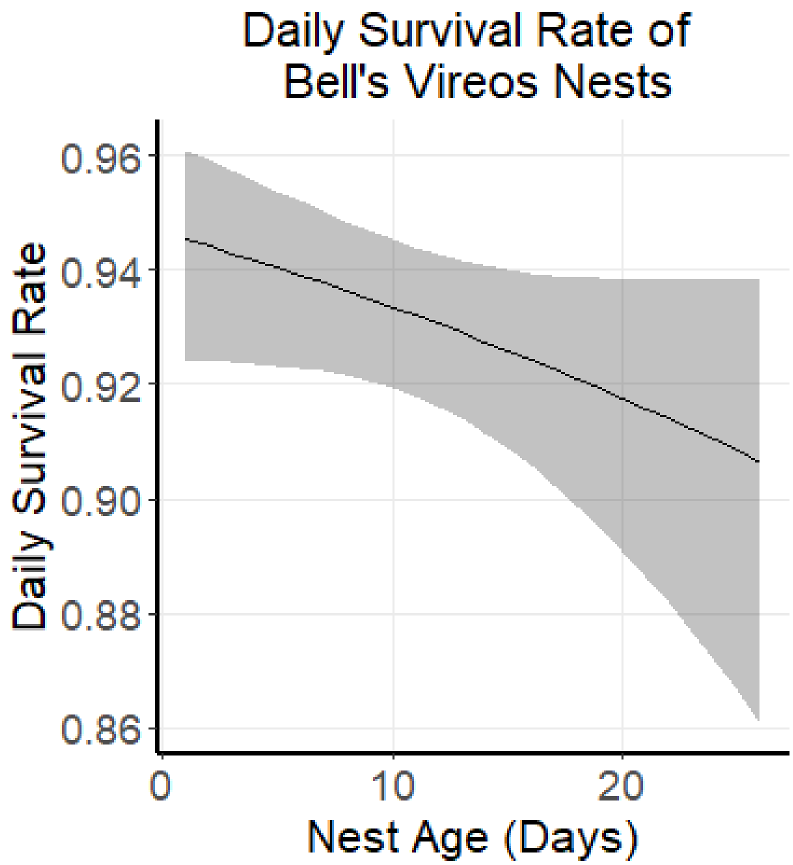

Figure 2.2. Daily survival rate of Bell's Vireo nests (n=122) over the 26-day nesting period in southeast Kansas. Shaded regions represent 95% confidence interval for daily survival rate.

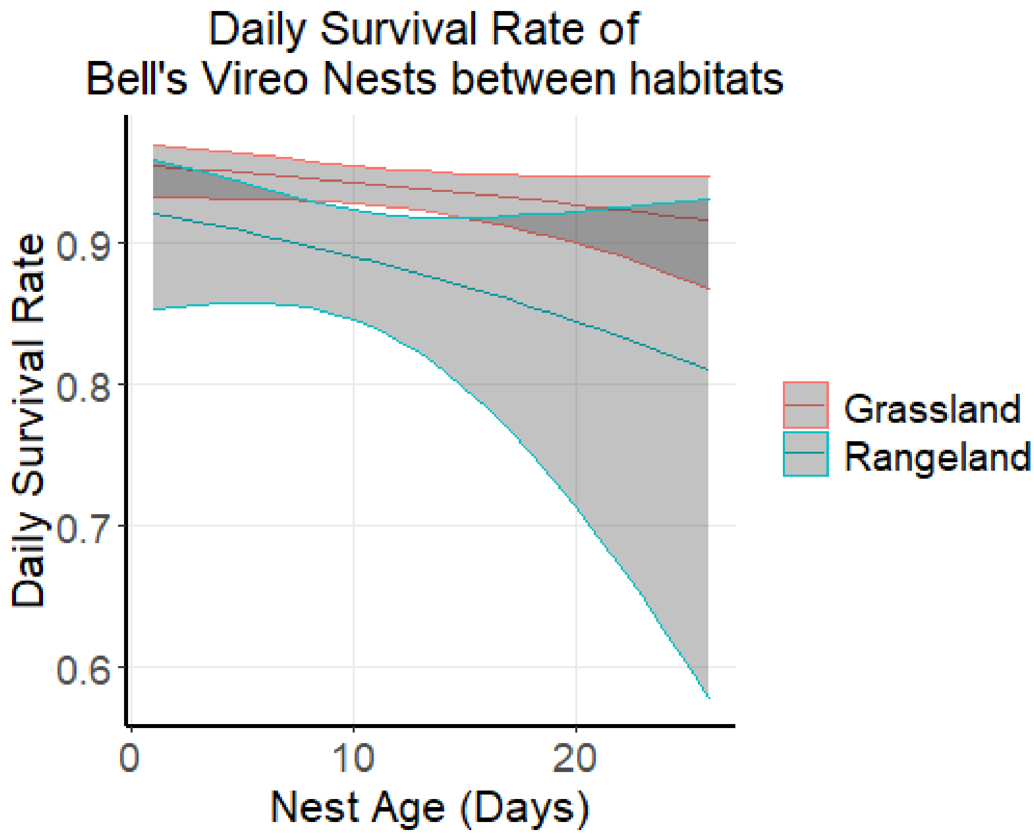

Figure 2.3. Daily survival rate by habitat type for Bell's Vireo nests across a 26-day nesting cycle. More nests were found in grasslands (n=90) versus rangelands (n=32). Error bars represent 95% confidence intervals for daily survival rate.

Table 2.3. Mean vegetation characteristics of Bell's Vireo nests (± standard deviation). No Bell's Vireo nests were found in forested habitats. Shrub coverage and invasive vegetation represent plants within the 11.3 m plot surrounding the nest. Brown-headed Cowbird parasitism represents the percentage of all nests with at least one Brown-headed Cowbird egg.

Habitat	n	Shrub Coverage (%)	Vertical Density (%)	Invasive Shrub Coverage (%)	Invasive Vegetation (% of plot)	Nest Height (m)	Brown-headed Cowbird Parasitism (%)
All Nests	122	23.16 ± 16.68	62.99 ± 21.22	8.19 ± 5.52	13.70 ± 21.30	1.02 ± .45	50.00
Grassland	90	22.56 ± 23.91	63.47 ± 8.43	10.00 ± 2.06	11.17 ± 16.94	1.05 ± .52	45.56
Rangeland	32	24.88 ± 2.08	61.63 ± 8.37	3.13 ± 10.55	20.84 ± 6.56	0.93 ± .69	62.50

Table 2.4. Model variables for Bell's vireo nest daily survival rate, including number of model parameters (npar) and model weights (w*i*).

Model	npar	AICc	ΔAICc	w*i*
(NestAge +Habitat)	3	495.57	0	0.57
(Habitat)	2	497.14	1.56	0.26
(NestAge)	2	501.56	5.99	0.03
(NestAge + PpnShrub)	3	501.65	6.07	0.03
(Null)	1	501.87	6.29	0.02
(PpnShrub)	2	502.30	6.72	0.02
(NestAge + PpnInv)	3	503.30	7.73	0.01
(NestAge + NuddsAVG)	3	503.33	7.76	0.01
(Year)	2	503.61	8.03	0.01
(PpnInv)	2	503.65	8.08	0.01
(NuddsAVG)	2	503.71	8.13	0.01
(Time)	2	503.87	8.29	0.01

Of the total 29 Northern Cardinal nests, 19 were depredated, three failed due to parasitism, and one failed due to an unknown reason. Only six were successful in fledging at least one young (Table 2.2). Northern Cardinals had an overall DSR of 91% (Figure 2.4). Northern Cardinals nested in all three habitat types, with the most nests observed in grasslands (Table 2.5). Of these nests, 28% were placed in invasive substrates, with 20% in Bush Honeysuckle, 4% in Autumn Olive, and 4% in Black Locust. Nest age was the best predictor of DSR. Northern Cardinals had higher DSR during their incubation period (96%) (i.e., the first 12 days of the nesting period) than Bell's Vireos (94%).

On average, Northern Cardinal nests had 37% shrub coverage on their nest plots, 75% vertical vegetation density, and 28% were placed in an invasive substrate, 13% of the plots were covered by invasive plant species, and 44% of nests were parasitized by at least one Brown-headed Cowbird egg (Table 2.2). Nests on grasslands had a greater likelihood of being placed in an invasive substrate (53%), compared to rangelands (33%) and forests (18%); however, the proportion of invasive substrates in which Northern Cardinals nested did not vary between habitats (Table 2.6).

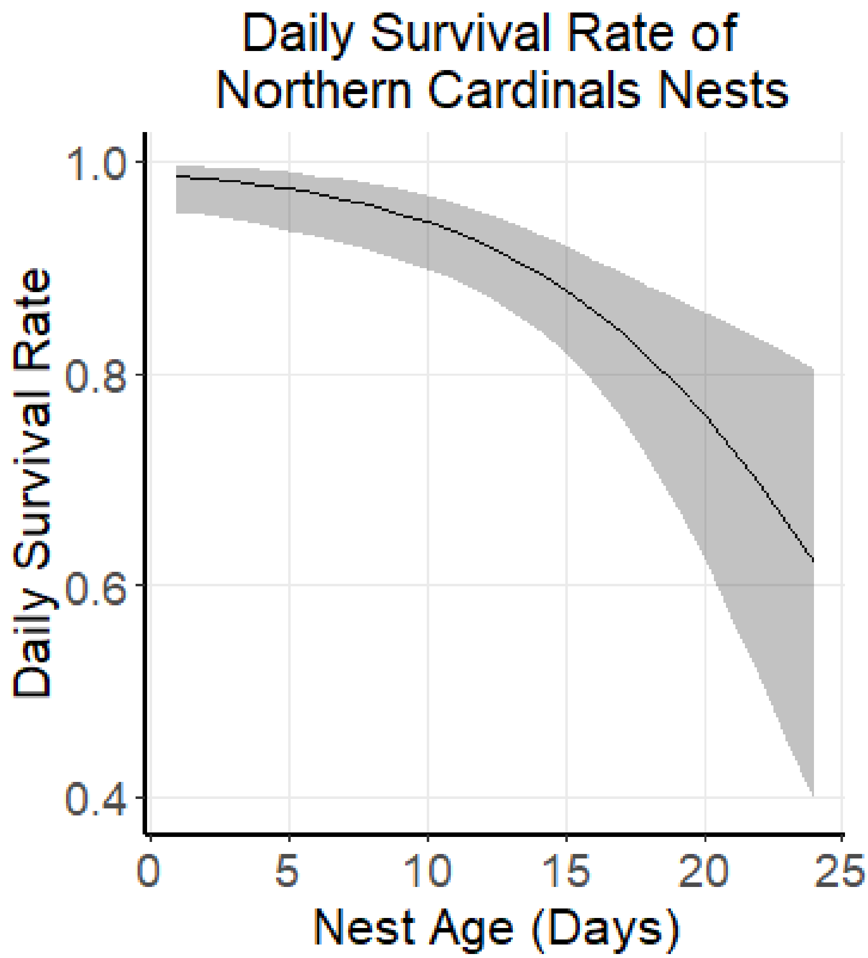

Figure 2.4. Daily survival rate of Northern Cardinal nests in southeast Kansas across a 24-day nesting cycle. Shaded regions indicate 95% confidence intervals of daily survival rate.

Table 2.5. Model variables for the daily survival rates of Northern Cardinal nests, including number of model parameters (npar) and model weights (w*i*).

Model	npar	AICc	ΔAICc	w*i*
(NestAge)	2	94.79	0.00	0.41
(NestAge + PpnInv)	3	95.98	1.18	0.22
(NestAge + PpnShrub)	3	96.76	1.96	0.15
(NestAge + NuddsAVG)	3	96.84	2.04	0.14
(NestAge + Habitat)	4	98.86	4.06	0.05
(Null)	1	108.97	14.17	0.01
(Time)	2	109.88	15.09	0.01
(PpnInv)	2	110.83	16.03	0.01
(NuddsAVG)	2	110.89	16.09	0.01
(PpnShrub)	2	110.94	16.15	0.01
(Year)	2	111.00	16.21	0.01
(Habitat)	3	113.05	18.25	0.01

Table 2.6. Mean vegetation characteristics of Northern Cardinal nests (± standard deviation). Shrub coverage and invasive vegetation represented plants within the 11.3 m plot surrounding the nest. Brown-headed Cowbird parasitism represents the percentage of all nests with at least one Brown-headed Cowbird egg.

Habitat	n	Shrub Coverage (%)	Vertical Density (%)	Invasive Shrub Coverage (%)	Invasive Vegetation (% of plot)	Nest Height (m)	Brown-headed Cowbird Parasitism (%)
All Nests	29	36.65 ± 21.59	74.81 ± 13.81	37.93 ± 9.55	12.82 ± 13.97	1.32 ± 0.49	44.82
Grassland	15	35.86 ± 23.92	79.76 ± 8.43	53.33 ± 10.06	16.06 ± 16.94	1.23 ± 0.52	46.66
Rangeland	3	39.33 ± 2.08	69.08 ± 8.37	33.33 ± 1.15	6.00 ± 6.56	1.20 ± 0.69	66.66
Forest	11	37.00 ± 22.37	69.60 ± 18.56	18.18 ± 9.35	10.27 ± 9.95	1.48 ± 0.40	36.36

DISCUSSION

We found that habitat type plus nest age and nest age were the leading variables for predicting the DSR of Bell's Vireos and Northern Cardinals, respectively, in a disturbed, post-mined landscape. As woody encroachment continues throughout the world, invasive shrub species and vegetative succession will alter wildlife habitat, especially for shrub-nesting birds. Understanding the continuing larger role of these issues is imperative for wildlife managers presently and in the future.

The presence of livestock and grazed pastures may be an important feature affecting Bell's Vireo nest success. Previous literature has mixed findings regarding livestock presence and stocking rates on daily survival rates of bird nests. The majority of literature indicates that livestock are responsible for less than 2% of all nest failures and do not significantly influence daily nests survival rates of bird nests (Bleho et al., 2014, Johnson et al., 2012) However, some studies have shown drastic decreases in daily survival rates in some species of grassland birds in pastures with high stocking rates (Fromberger et al., 2020). All highly publicized research has been completed on obligate grassland bird species, so improved knowledge base is necessary to understand the impacts of livestock stocking rates on shrubland nesting birds. We observed that Bell's Vireo nests placed in rangelands had on average 5% less daily survival than nests placed in grasslands. Nests in grasslands were less likely to be parasitized by Brown-headed Cowbirds and had fewer incidental failures from livestock. Cattle could either snap off the branches on which the nests were located or knock down the entire nest substrate. Increased invasive plant cover is often associated with livestock presence (Hobbs, 2001). Livestock can act as transmission vectors for plant seeds as they are moved from pasture to pasture, by grazing and path creation (Chuong et al., 2016). Assemblages of livestock can also influence the abundance and distribution of Brown-headed Cowbirds, potentially influencing the likelihood of nest parasitism (Goguen & Mathews, 1999).

The leading causes of nest failure reported in the literature for Bell's Vireo are nest depredation and parasitism by Brown-headed Cowbirds (Budnik et al., 2000; Parker, 1999; Rivers et al., 2010). Bell's Vireos are one of the preferred host species for Brown-headed Cowbirds and rates of parasitism vary from 29% to 70.5% (Budnik et al., 2000; Rivers et al., 2010). Nest parasitism can influence host productivity in numerous ways. Parasite eggs or nestling presence can reduce incubation of host eggs and divert food resources away from host young. Adult cowbirds also regularly remove host eggs or destroy established nests to induce renesting (Kosciuch & Sandercock, 2008). We have little evidence to suggest why parasitism rates were higher on rangeland than on grasslands in our study. Brown-headed Cowbirds are associated with livestock, but they are known to feed, breed, and roost in distinct habitats that are separated by up to 10 km (Chace et al., 2005). Thus, local abundances of cowbirds on rangeland may not always indicate higher rates of local parasitism. Research on Plumbeous Vireos (*Vireo plumbeus*) indicated that brood parasitism rates were associated with distance to pasture, with parasitism rates greater than 80% in actively grazed pastures to 33% in areas 8–12 km away from a pasture (Goguen & Mathews, 2000). In the landscape matrix of southeast Kansas, all of our sites were within close proximity (<10 km) to pasture, forest and grassland habitat.

Bell's Vireos were by far the most sampled nesting species we monitored. With an overall DSR of 93%, Bell's Vireo nests were slightly more productive on grassland units compared to rangeland units. The habitat variables that differed the most between rangelands and grassland were invasive species coverage and rate of nest parasitism. Rangelands had more invasive cover with more parasitized nests. In central Missouri, Bell’s Vireo had a DSR of 95.6%, which was too low to compensate for annual mortality, resulting in a shrinking local population (Budnik et al., 2000). Thus, while nationwide populations of Bell’s Vireo are rebounding from a record low in the 1980s (Ziolkowski et al., 2022), local populations in our sampled region and Missouri may decline in the near future due to low nest daily survival rates.

Nest age was the most informative variable for predicting Northern Cardinal nest daily survival rates. Northern Cardinal nests were placed in invasive substrates at a much higher rate than Bell’s Vireo. Nest placement might be attributed to the habitat preferences, or lack thereof, for Northern Cardinals. Northern Cardinals were more likely to place nests in habitats with more tree and shrub coverage. Many nests were found on formerly mined lands that have transitioned to mature forests. Northern Cardinal nests were surrounded by high vertical vegetation densities, particularly on grassland habitats. Rates of brood parasitism were less than those of Bell’s Vireos, but still high at 45%. Northern Cardinal DSR dropped consistently over time with major decreases occurring after the completion of incubation. This pattern may be due to increased activity around the nest during the nestling stage, which may alert predators to the nest location.

Our results indicated a lack of a relationship between the proportion of invasive shrub species and daily survival rates of nests. Invasive shrubs could influence the nest success of in a variety of ways, both positively and negatively. The ecological trap hypothesis, which is one of the most common explanations why invasive shrubs negatively affect bird species, may not apply to some of our study sites. For instance, most studies that support this hypothesis describe increased depredation rates in patches of invasive shrubs (Gleditsch & Carlo, 2014). The invasive vegetation found throughout mined lands differ, as bush honeysuckle created such large continuous monocultures within our study sites. Thus, the ecological trap hypothesis may not apply. It is difficult to determine underlying reasons for decreasing DSR rates without further study in this system.

CONCLUSION

While Bell's Vireo populations are increasing nationwide, they remain a species of conservation concern in Kansas (Rohweder, 2022). Our management recommendations to increase Bell's Vireo nest success are to remove livestock or reduce stocking densities from sites that have high densities of birds during peak breeding season. This change could reduce direct livestock destruction of nest substrates and potentially reduce Brown-headed Cowbird abundance. We also recommend continuing efforts to mitigate woody encroachment, as Bell's Vireo prefer early successional shrublands to early forest succession. While Northern Cardinals are a common songbird species, maintaining nesting habitat may support their persistence in this region. Removing invasive shrub species on all habitat types may reduce ecological traps for shrub dependent birds and improve overall breeding bird diversity in a disturbed landscape.

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APPENDIX

Appendix I. Bird species and individuals observed at point count sampling locations across three sampling years. Values indicate species counts. Focal species are bolded.

Species		2020	2021	2022
Acadian Flycatcher	*Empidonax virescens*	0	1	10
Alder Flycatcher	*Empidonax alnorum*	0	0	7
American Crow	*Corvus brachyrhynchos*	63	105	114
American Goldfinch	*Spinus tristis*	48	54	35
American Kestrel	*Falco sparverius*	1	0	0
Baltimore Oriole	*Icterus galbula*	5	7	10
Barred Owl	*Strix varia*	5	0	3
Barn Swallow	*Hirundo rustica*	6	8	13
Black-and-white Warbler	*Mniotilta varia*	0	2	0
Bell's Vireo	***Vireo belli***	**57**	**116**	**103**
Belted Kingfisher	*Megaceryle alcyon*	0	0	1
Blue-gray Gnatcatcher	*Polioptila caerulea*	29	54	49
Brown-headed Cowbird	*Molothrus ater*	82	102	143
Blue Grosbeak	*Passerina caerulea*	4	0	2
Blue Jay	*Cyanocitta cristata*	27	62	41
Brown Thrasher	*Toxostoma rufum*	7	4	16
Carolina Chickadee	*Poecile carolinesis*	41	54	74
Carolina Wren	*Thryothorus ludovicianus*	95	42	45
Canada Goose	*Branta canadensis*	6	50	8
Cedar Waxwing	*Bombycilla cedrorum*	0	0	6
Chipping Sparrow	*Spizella passerina*	0	2	3
Common Gallinule	*Gallinula galeata*	2	0	0
Common Grackle	*Quiscalus quiscula*	5	2	1
Common Nighthawk	*Chordeiles minor*	2	3	2
Common Yellowthroat	*Geothlypis trichas*	24	48	56
Chuck-will's-widow	*Antrostomus carolinensis*	0	2	0
Dickcissel	*Spiza americana*	284	496	523
Downy Woodpecker	*Picoides pubescens*	12	15	25
Eastern Bluebird	*Sialia sialis*	1	1	5
Eastern Kingbird	*Tyrannus tyrannus*	6	4	6
Eastern Meadowlark	*Sturnella magna*	21	31	32
Eastern Phoebe	*Sayornis phoebe*	1	1	2
Eastern Towhee	*Pipilo erythrophthalmus*	35	31	35
Eastern Wood Peewee	*Contopus virens*	52	64	51
Eurasian Collared Dove	*Streptopelia decaoto*	0	0	1
European Starling	*Sturnus vulgaris*	1	0	1
Fish Crow	*Corvus ossifragus*	8	23	26
Field Sparrow	*Spizella pusilla*	146	137	106
Great Blue Heron	*Andrea herodias*	12	3	3
Great-crested Flycatcher	*Myiarchus crinitus*	30	57	46

Gray Catbird	*Dumetella carolinensis*	5	21	10
Great Egret	*Andrea alba*	2	1	4
Hairy Woodpecker	*Leuconotopicus villosus*	0	1	4
Henslow's Sparrow	*Ammodramus henslowii*	0	7	15
House Sparrow	*Passer domesticus*	1	0	0
Indigo Bunting	***Passerina cyanea***	**159**	**156**	**181**
Kentucky Warbler	*Geothlypis formosa*	2	1	7
Killdeer	*Charadrius vociferus*	15	12	6
Lark Sparrow	*Chondestes grammacus*	1	0	1
Least Flycatcher	*Empidonax minimus*	0	0	4
Louisiana Waterthrush	*Parkesia motacilla*	0	0	1
Mourning Dove	*Zenaida macroura*	80	74	65
Mourning Warbler	*Geothlypis philadelphia*	0	0	1
Northern Bobwhite	*Colinus virginianus*	52	80	41
Northern Cardinal	***Cardinalis cardinalis***	**215**	**281**	**261**
Norther Flicker	*Colaptes auratus*	5	8	1
Northern Mockingbird	*Mimus polyglottos*	21	7	13
Northern Parula	*Setophaga americana*	20	33	28
Orchard Oriole	*Icterus spurius*	12	11	12
Painted Bunting	*Passerina ciris*	0	0	2
Pileated Woodpecker	*Dryocopus pileatus*	9	25	33
Prothonotary Warbler	*Protonotaria citrea*	11	12	11
Purple Martin	*Progne subis*	0	1093	5
Red-bellied Woodpecker	*Melanerpes carolinus*	68	100	95
Red-eyed Vireo	*Vireo olivaceus*	30	32	34
Red-headed Woodpecker	*Melanerpes erythrocephalus*	14	5	6
Red-shouldered Hawk	*Buteo lineatus*	5	5	8
Red-tailed Hawk	*Buteo jamaicensis*	11	5	5
Ruby-throated Hummingbird	*Archilochus colubris*	4	7	6
Red-winged Blackbird	*Agelaius phoeniceus*	47	87	70
Song Sparrow	*Melospiza melodia*	2	0	0
Scissor-tailed Flycatcher	*Tyrannus forficaatus*	3	11	3
Summer Tanager	*Piranga rubra*	13	16	17
Tree Swallow	*Tachycineta bicolor*	3	4	1
Tufted Titmouse	*Baeolophus bicolor*	73	121	100
Turkey Vulture	*Cathartes aura*	13	7	4
Warbling Vireo	*Vireo gilvus*	12	10	13
White-breasted Nuthatch	*Sitta carolinesis*	3	1	11
White-eyed Vireo	*Vireo griseus*	5	20	23
Willow Flycatcher	*Empidonax traillii*	7	3	1
Wild Turkey	*Meleagris gallopavo*	2	0	1
Wood Duck	*Aix sponsa*	0	1	1
Wood Thrush	*Hylocichla mustelina*	4	1	3
Yellow-breasted Chat	*Icteria virens*	66	115	120
Yellow-billed Cuckoo	*Coccyzus americanus*	107	116	77
Yellow Warbler	*Setophaga petechia*	1	0	3

Yellow-throated Vireo	*Vireo flavifrons*	1	3	0

Appendix II. List of detected Kansas bird species of conservation concern (Rohweder 2015). Each species is listed by the number of individuals observed during point count sampling across all sampling years.

Species		2020	2021	2022	Total
Baltimore Oriole	*Icterus galbula*	5	7	10	22
Bell's Vireo	*Vireo belli*	57	116	103	276
Chuck-will's-widow	*Antrostomus carolinensis*	0	2	0	2
Dickcissel	*Spiza americana*	284	478	524	1286
Eastern Kingbird	*Tyrannus tyrannus*	6	4	6	16
Eastern Meadowlark	*Sturnella magna*	21	31	32	84
Eastern Wood-Pewee	*Contopus virens*	52	64	51	167
Kentucky Warbler	*Geothlypis formosa*	2	1	7	10
Lark Sparrow	*Chondestes grammacus*	1	0	1	2
Northern Bobwhite	*Colinus virginianus*	52	80	41	173
Prothonotary Warbler	*Protonotaria citrea*	11	12	11	34
Red-headed Woodpecker	*Melanerpes erythrocephalus*	14	5	6	25
Scissor-tailed Flycatcher	*Tyrannus forficaatus*	3	10	3	16