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Impacts of biomass production at civil airports on grassland bird conservation and aviation strike risk

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Abstract. Growing concerns about climate change, foreign oil dependency, and environmental quality have fostered interest in perennial native grasses (e.g., switchgrass [*Panicum virgatum*]) for bioenergy production while also maintaining biodiversity and ecosystem function. However, biomass cultivation in marginal landscapes such as airport grasslands may have detrimental effects on aviation safety as well as conservation efforts for grassland birds. In 2011–2013, we investigated effects of vegetation composition and harvest frequency on seasonal species richness and habitat use of grassland birds and modeled relative abundance, aviation risk, and conservation value of birds associated with biomass crops. Avian relative abundance was greater in switchgrass monoculture plots during the winter months, whereas Native Warm-Season Grass (NWSG) mixed species plantings were favored by species during the breeding season. Conversely, treatment differences in aviation risk and conservation value were not biologically significant. Only 2.6% of observations included avian species of high hazard to aircraft, providing support for semi-natural grasslands as a feasible landcover option at civil airports. Additionally, varied harvest frequencies across a mosaic of switchgrass monocultures and NWSG plots allows for biomass production with multiple vegetation structure options for grassland birds to increase seasonal avian biodiversity and habitat use.

Key words: aviation risk; biomass; bird strike; conservation; grassland birds; *Panicum virgatum*.

INTRODUCTION

Growing concerns about climate change, dependency on foreign oil, and overall environmental quality have fostered interest in bioenergy (Farrell et al. 2006, Sagar and Kartha 2007, U.S. EPA 2011). Although corn (*Zea mays*), sugar cane (*Saccharum officinarum*), and wheat (*Triticum aestivum*) are the predominant global biofuel crops, there is an increased focus on using perennial native grasses (e.g., switchgrass [*Panicum virgatum*]) in the United States because they provide multiple ecosystem services including carbon sequestration and wildlife habitat (Fargione et al. 2009, Knight 2010, Hartman et al. 2011). However, bioenergy crop production has drawn concern as it might compete with food production (Campbell et al. 2008). Thus, marginal agricultural land or other non-productive human-managed sites could offer a solution to the land use dilemma. For example, areas such as airport grasslands typically do not produce crops but do provide a substantial land base of over 3,300 km² within the United States, equivalent to the area of Rhode Island (Blackwell et al. 2009, DeVault et al. 2012, 2013a). However, changing land use near airports could affect animal use, subsequently affecting aviation strike risk (Martin et al. 2011, 2013).

Current grasslands at certificated airports (those approved for regularly scheduled passenger flights) occupy on average 297 ha per airport (DeVault et al. 2012) and are composed primarily of turfgrass maintained through frequent mowing at a short height (i.e., 15–25 cm; Dolbeer et al. 2000). The primary focus of managing wildlife on airport lands is to minimize wildlife–aircraft collisions (Sodhi 2002, Blackwell et al. 2009, DeVault et al. 2013b). However, airports could mitigate aviation risk associated with hazardous avian species by converting land cover from turfgrass to native semi-natural grasslands. This also may reduce mowing costs and generate income through biomass production (Fletcher et al. 2010, Robertson et al. 2012, DeVault et al. 2013a, Martin et al. 2013).

Management practices such as biomass production on semi-natural grasslands (Allen et al. 2011) can directly influence subsequent avian habitat use, potentially altering the aviation strike risk or conservation value of a given grassland treatment (Robertson et al. 2011a, DeVault et al. 2013a, Blank et al. 2014, Hovick et al. 2014). Birds attempt to maximize survival and fitness by selecting habitat based on vegetation structure or composition that best matches their foraging and flight behaviors or provides suitable breeding locations (Grinnell 1917, Hildén 1965, Fretwell and Lucas 1969, Stephens and Krebs 1986, Whittingham et al. 2006). Differences in vegetation structure may interact with both food availability and accessibility to influence real or perceived predation risk, resulting in varying avian foraging strategies (predation risk hypothesis; Lima and Dill 1990, Butler and Gillings 2004, Butler et al. 2005b). For example, taller, dense vegetation can limit visibility of predators and

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hinder escape for larger-bodied birds and flocking species, including those that pose a high risk of damaging aircraft (Beck and Watts 1997, Whittingham and Devereux 2008, Blumstein 2010, DeVault et al. 2013a). Additionally, avian species might prefer grasslands with variation in vegetation species composition and foliage structure (heterogeneous habitat hypothesis; Simpson 1949, MacArthur and MacArthur 1961), especially during the breeding season due to greater availability of potential nest sites and arthropods as food for nestlings (McCoy et al. 2001, Hovick et al. 2012). However, limited research addresses grassland bird habitat use across multiple seasons, with most studies focusing primarily on a single (e.g., breeding or wintering) period (reviewed in Vickery and Herkert 2001). As such, there is a lack of information regarding the seasonal variation of year-round avian habitat use in semi-natural grasslands and resulting effects those use patterns may have on mitigating aviation risk or grassland bird conservation, two conflicting land management practices (Blackwell et al. 2013).

Our objectives were to (1) investigate the year-round effects of vegetation composition and structure (heterogeneous habitat hypothesis) and harvest frequency (predation risk hypothesis) on species richness and species-specific habitat use of grassland birds and (2) relate this to aviation strike risk and conservation value of birds associated with biofuel crops. For migration and winter periods, avian use of a given habitat influenced by predation risk should vary by vegetation structure relative to species-specific antipredator strategies. Species dependent on flocking behaviors or powerful escape flights should be more common in shorter, more frequently harvested vegetation, although the strength of this relationship can fluctuate depending on underlying food needs (Lima 1993, Butler et al. 2005a, Blackwell et al. 2016). Conversely, species more prone to solitary behaviors that are weak fliers or rely on crypsis should select habitat with greater visual obstruction (Lima and Dill 1990, Whittingham et al. 2006). During the breeding season, overall bird density and diversity should be greater in native warm-season grasslands (hereafter NWSG) than in switchgrass monocultures, as bird habitat use is positively correlated with increased vegetation structural heterogeneity and habitat complexity present in mixed species plantings, based on the heterogeneous habitat hypothesis (Simpson 1949, MacArthur and MacArthur 1961, Wiens 1974, Robertson et al. 2011b). We also expected aviation risk and conservation value to demonstrate similar seasonal patterns, with greater risk during migration periods and winter in more frequently harvested treatments with shorter vegetation favorable to hazardous large-bodied, flocking species that are most likely to cause damage when struck by aircraft (Lima 1993, DeVault et al. 2011). Additionally, we expected greater risk and conservation importance in NWSG during summer based on availability of potential nesting sites and arthropods for breeding birds, especially for species of higher conservation priorities (Panjabi et al. 2012).

METHODS

Study area

We conducted the study from January 2011–December 2013 on 16 adjacent plots (range 5.03–8.41 ha) previously

used for row crop agriculture arranged in a randomized complete block design at B. Bryan Farm in Clay County, Mississippi, USA; treatment blocks were assigned based on prevailing soil type (Fig. 1). B. Bryan Farm lies within the historical range of the Blackland Prairie (Barone 2005) and consists mostly of row crop agriculture, pastureland, and conservation easements. The region experiences mean daily temperatures from May to August of 25.3°C and average monthly precipitation of 7.8 cm. While B. Bryan Farm is not an active civil airport, it provided a suitable surrogate location for examining the effects of experimental land manipulation, as current FAA regulations discourage agriculture (including hay crops) at civil airports (DeVault et al. 2013a). Furthermore, there is an airport <3 km from our study area in a very similar landscape indicating that our study area is representative of a landscape commonly used for airports (DeVault et al. 2009).

During spring 2010, eight plots were planted with a NWSG seed mixture including big bluestem (*Andropogon gerardii*), little bluestem (*Schizachyrium scoparium*), Indian grass (*Sorghastrum nutans*), tickseed sunflower (*Bidens aristata*), and Illinois bundleflower (*Desmanthus illinoensis*) and eight plots were planted with switchgrass. Additional species prevalent in the existing seedbank included annual ragweed (*Ambrosia artemisiifolia*), broadleaf signalgrass (*Urochloa platyphylla*), and *Sesbania* spp. No trees were present within treatment plots. All treatment plots were harvested once in late April 2012, while four plots of each vegetation type were also mowed to ~20 cm and baled in late June 2012 and 2013. No harvests occurred in 2011 to allow plots to continue establishing vegetation. This resulted in four distinct treatments: NWSG single harvest (NWSG single), NWSG multiple harvest (NWSG multiple), switchgrass single harvest (switchgrass single), and switchgrass multiple harvest (switchgrass multiple). Two of the initial 16 plots (one switchgrass single harvest and one switchgrass multiple harvest) failed to establish the treatment vegetation so we removed them from subsequent analyses.

We defined five sampling periods each year (late winter, 1 January–15 March; spring, 16 March–31 May; summer, 1 June–15 August; fall, 16 August–31 October; early winter, 1 November–31 December) and conducted three surveys per period at 10–14 d intervals. Due to logistical constraints, spring surveys were not conducted in 2011; all other survey periods were sampled for the 3-yr duration of the study (January 2011–December 2013). We used multiple sampling methods to detect birds including flush transects, point counts, and line transects (Ralph et al. 1993, Butler et al. 2009). Methods varied by sampling period, with point counts during summer breeding seasons when birds are vocal and often readily visible, whereas flush and line transects are considered to be more efficient methods for sampling migratory or wintering birds (Igl and Ballard 1999, Fletcher et al. 2000, Butler et al. 2009). We conducted all surveys from sunrise until 3 h after sunrise during favorable weather conditions (i.e., no precipitation and wind <16 km/h) to minimize the effects of weather on detections and recorded percent cloud cover, temperature, average wind speed, and humidity with a Kestrel 3000 pocket weather meter (Nielsen-Kellerman, Boothwyn, Pennsylvania, USA).

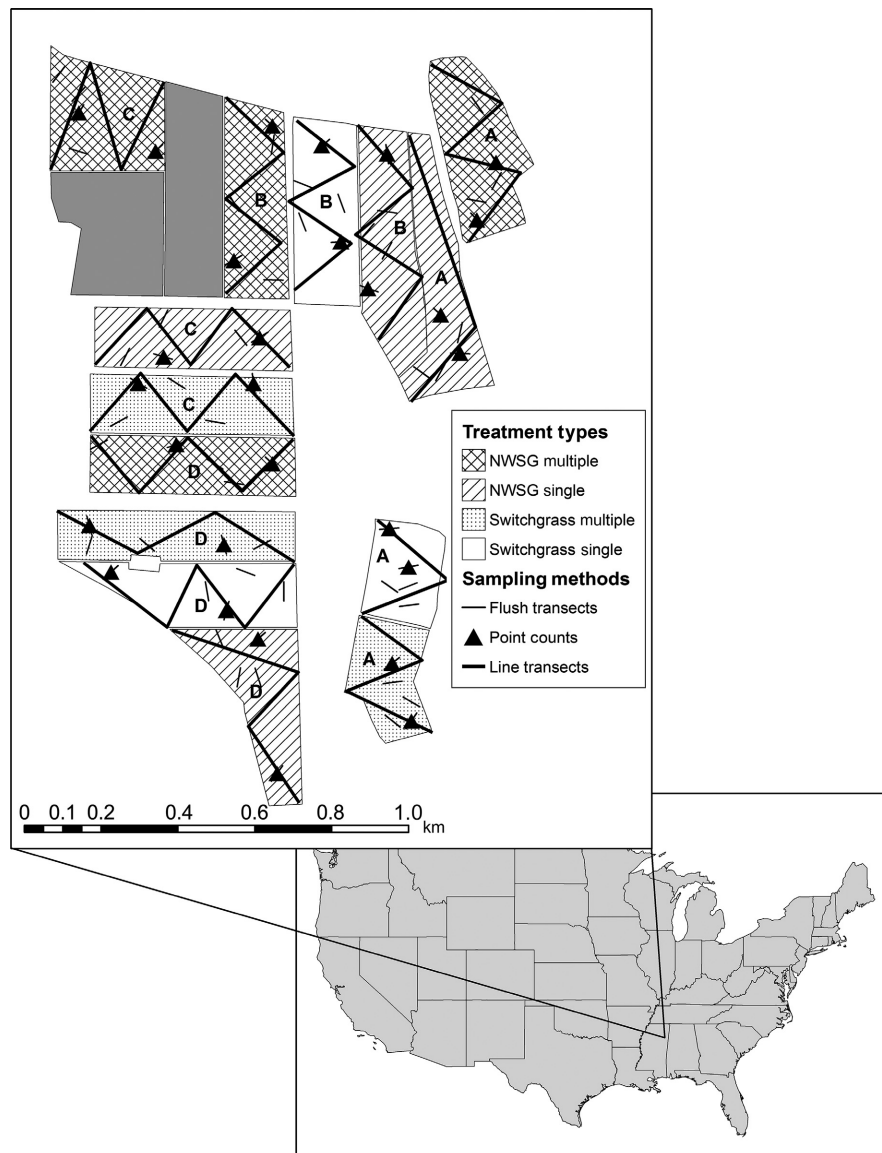


FIG. 1. Map of treatment plots and survey locations at B. Bryan Farm in Clay County, Mississippi, USA from January 2011 to December 2013. Letters A–D designate treatment blocks assigned based on prevailing soil type. One switchgrass single and one switchgrass multiple plot (in gray) failed to establish treatment vegetation and were removed from subsequent analyses.

Flush transects

During late winter, we used a geographic information system (ESRI 2011) to overlay a 50×50 m grid on each study plot and randomly selected five grid squares per plot for strip transect locations after analyzing preliminary transect survey data (T. Conkling, *unpublished data*) to determine the minimum sample size of transects (Buckland et al. 2001). Each 50-m strip transect was centered on the grid square midpoint. We randomly assigned transect orientation and started all flush transects from the eastern-most endpoint to prevent observers from facing the sun during surveys. Two observers pulled a 25-m rope with noisemakers (1-L bottles containing rocks attached to the rope every 6 m) across the vegetation, with a third observer near the center of the rope serving as the data recorder; all observers identified flushed

birds to species using visual, aural, and flight cues (Butler et al. 2009). When existing vegetation precluded efficient rope-dragging (vegetation > 1.5 m in height), each edge observer walked approximately 12 m from the center observer and made noise with hand-held noisemakers to flush any birds. If multiple birds flushed simultaneously, the observer recorded flock size according to Grzybowski (1983b). To avoid double-counting individuals (Grzybowski 1983a,b), the total number of birds for each transect only included individuals or flocks initially located in the transect that completely left the area during the survey.

Point counts

During summer, we conducted 5-min single-observer point-counts at the midpoint of two previously defined flush

transects in each plot (Buckland et al. 2001). We randomly selected the first flush transect and then chose the second survey location to maximize sampling distance between the two points; all points were ≥ 25 m from plot edge (Fig. 1). We recorded distance to all observed individuals within the plot, identified to species and sex (if possible) based on visual and aural cues. If multiple birds of the same species were detected in a flock (as defined in *Flush transects*), we recorded flock size and distance to the center of the flock. We truncated 10% of observations to account for uncertainty of detecting individuals at large distances and to define a fixed survey radius of 100 m for subsequent analyses (Buckland et al. 2001).

Line transects

During spring, fall, and early winter, we conducted single-observer line-transects at each plot to maximize detection of inconspicuous, migratory species as well as facilitate observations of birds in tall vegetation (Buckland et al. 2001). We used preliminary data to establish one transect per plot in a zig-zag pattern to maximize transect length (650–850 m) while minimizing the potential to double-count individuals, randomly selecting the start corner for each transect (Fig. 1). We walked each transect at 1.6–3 km/h and visually estimated perpendicular distance to all observed individuals or song locations within the plot and identified species and sex based on visual and aural cues. When flocks were observed, we recorded flock size and distance to the center of the flock. We excluded any birds only observed as flyovers from subsequent analyses and truncated 5% of observations based on recommendations from Buckland et al. (2001) by removing observations >36 m from the transect.

Vegetation sampling

We collected vegetation data to determine structure and species composition among treatments. We sampled each transect in March 2011 and then monthly from June 2011–December 2013. We collected vegetation measurements using the point-intercept method (FIREMON 2007) to classify species composition at 5-m intervals along each transect (10 measurements/transect); transect locations corresponded to the existing flush transects. At each point, we recorded litter depth and species identification and height (cm) for the three most common vegetation species. At 10-m intervals, we quantified vegetation structure by recording maximum height of visual obstruction (VOR) from each of the four cardinal directions with a Robel pole (Robel et al. 1970). We collected VOR each month, but we excluded species and height data from November–February each year because we expected these measurements to be stable during winter dormancy. We used VOR as the main vegetation metric in subsequent analyses because we were primarily interested in the effects of vegetation structure on habitat use, aviation strike risk, and conservation value.

Statistical analyses

Many analytical methods to estimate densities of animal populations incorporate detection probabilities (MacKenzie

2006, Royle and Dorazio 2008), but we were unable to fit these models because of insufficient detections for most species (Blank et al. 2014, Hovick et al. 2014). Furthermore, the species with the fewest detections were often those with the greatest risk to aviation, thus, removing those species to incorporate detection probabilities would have affected our inference. As such, we did not include detection probability in our models in order to include the greatest number of species in our analyses, and instead we assumed detection differences among treatments were minimal because all plots consisted of structurally similar native grass species. We used unadjusted counts as an index of relative density for species with sufficient sample sizes for subsequent analyses (species detected at $>10\%$ of survey locations each biological season). These totals may be biased low, as preliminary hierarchical distance sampling analyses of line transect data for our most abundant sparrow species indicated that detection probability decreased with increasing distance from the observer (Buckland et al. 2001, Fiske and Chandler 2011). However, treatment type had no effect on detection probability (T. Conkling, *unpublished data*). Other factors that could possibly affect detection, such as weather and observers, were minimized through sampling design by sampling only during favorable weather conditions and using the same four primary observers for the duration of this study.

Using R 3.1.0 (R Core Team 2015) we calculated relative density (birds detected per 10 ha surveyed) by summing detections across all visits at the plot level divided by total area surveyed, which standardized survey efforts across years and species while accounting for differences in survey area. We were only able to compare density metrics within seasons, but not across survey periods (e.g., late winter vs. spring) due to differences in assumptions for each sampling method (Buckland et al. 2001). We determined values (mean \pm SE) for VOR, avian relative density, aviation risk, and conservation value for each plot by treatment and year. Aviation risk and conservation values for each plot were determined by summing species densities weighted with the relative hazard each species poses to aircraft (DeVault et al. 2011) or the relative conservation priority for a given species using Partners in Flight (PIF) scores (Nuttall et al. 2003), respectively.

For relative hazard score, DeVault et al. (2011) calculated scores based on the percentage of total bird strikes in the Federal Aviation Administration National Wildlife Strike Database from 1990–2009. First, they determined the percentage of strikes for species, genus, or family group (e.g., unknown sparrows) that caused damage to the aircraft, the severity of the damage (e.g., if major repair or replacement of components was required), and if there was an adverse effect on flight. The resulting 77 species groups were sequentially ranked for each of the three categories, and the composite score was determined based on the sum of the hazard score across all three categories, scaled from 1 (no hazard)–100 (maximum hazard). As such, high hazard species were those most likely to cause damage when struck by aircraft, including large-bodied, flocking birds (e.g., relative hazard score for Canada Goose [*Branta canadensis*] = 76 vs. Cliff Swallow [*Petrochelidon pyrrhonota*] = 2). We categorized detected species with a relative hazard score of 1 (lowest relative score possible) if they were not identified as one

responsible for strikes used by DeVault et al. (2011). We also included individuals identified only to genus or family group (e.g., unknown sparrows) if there was a relative hazard score available for that group.

For conservation values, we used PIF regional assessment scores for the Southeastern Coastal Plain Bird Conservation Region (BCR; Beissinger et al. 2000, Panjabi et al. 2012) following Nuttle et al. (2003). PIF calculates scores based on relative extinction risk by region and the resulting rank prioritizes conservation efforts for species in North America: a higher rank indicates species with greater conservation priority. For species not included in the Southeastern Coastal Plain BCR, we calculated conservation values based on PIF global assessment scores. We did not include unidentified birds (e.g., unknown warblers, unknown sparrows) in models to estimate conservation value, as PIF scores were species specific.

We used analysis of variance (ANOVA) and Tukey's test ($\alpha = 0.05$) to compare vegetation characteristics among treatments by year and survey period. We examined bird species composition across treatment types and years with nonmetric multidimensional scaling (NMDS) (Faith et al. 1987, Minchin 1987) using relative density of all identified species with sufficient detections using the package *vegan* (Oksanen et al. 2013) in R to create a summary of avian community habitat use. NMDS calculates a distance matrix based on the rank-order correlation of species abundances in Euclidian distance, allowing for two-dimensional matrix visualization of the avian communities. We used the Bray-Curtis distance metric for dissimilarity and Shepard diagrams and STRESS values to examine goodness of fit and determine the appropriate number of dimensions. We overlaid treatment type on the NMDS plots based on dispersion ellipses with 95% confidence intervals and plotted mean VOR as fitted contour lines using the function *ordisurf* to examine species distributions among treatments and vegetation structure. The arbitrary NMDS axes are not directly related to multidimensional space and cannot be interpreted independently; instead, axes only relate to distances between observations (Legendre and Legendre 2012). Independent of axis values, species in closer proximity to each other in the resulting plots are more similar in distribution than species spaced further apart, while a greater degree of overlap for treatment dispersion ellipses suggests an increasing similarity in the avian communities across treatments. We also tested whether species ordination was correlated with treatment type or vegetation structure using the function *envfit*, based on 10,000 permutations for all survey seasons and years. *Envfit* calculates both the strength of the correlation and the direction of maximum correlation between the NMDS structure and the corresponding environmental variables to determine if those variables are associated with the differences between sites as defined by the NMDS ordination (Jongman et al. 1995).

We modeled relative density, aviation risk, and conservation value using generalized linear mixed models (GLMM) with R package *lme4* (Bates et al. 2015) and linear and mixed models with R package *nlme* (Pinheiro et al. 2013) following model construction and selection procedures outlined in Zuur et al. (2009). We first constructed the most complex model, incorporating fixed effects of treatment,

year, and a treatment-year interaction (treatment \times year). For density models, we then used likelihood ratio tests to determine the appropriate model distribution (e.g., Poisson, negative binomial). If data were overdispersed, we examined data to determine if a negative binomial model better fit the data, then continued the model selection procedure. Next, we used likelihood ratio tests to determine support for the random effects (block), and then fit sequentially more parsimonious models to determine best model fit. When applicable, we validated models by examining residual plots (Zuur et al. 2009) and calculated the proportion of variance explained by the fixed effects in the best-fit models with pseudo R^2 (Nakagawa et al. 2013) for GLMM and McFadden's pseudo R^2 (McFadden 1973) for linear models fitted using generalized least squares. We modeled relative density (number of individuals per 10 ha) of avian species by survey visit with total birds as our response variable, treatment as a fixed effect, year as a repeated measure, and total area (ha) surveyed as an offset. We included unidentified species groups (e.g., unknown warblers, unknown sparrows) within the total relative density estimates for each plot if there were sufficient detections (observed at $>10\%$ of survey locations). We modeled overall estimated aviation risk and conservation value using the same model selection procedure while allowing for alternative variance structures for linear models as outlined above.

RESULTS

We recorded 7,973 individuals of 48 species and four unknown categories (unknown sparrows, swallows, warblers, or unknown small birds) across all survey periods and years (Appendix S1: Table S1), including 17 species in late winter, 14 in spring, 26 during summer, 24 in fall, and 18 in early winter. Thirty-six of the 52 species did not have sufficient detections (observed at $\leq 10\%$ of survey locations across years) and were eliminated from subsequent analyses. We used the remaining data on two unknown categories (unknown sparrows and unknown warblers) and 14 overall species including 6 species in late winter, 7 in spring, 5 in summer, 11 in fall, and 8 in early winter for NMDS, aviation risk, and conservation value modelling. For vegetation measurements, VOR differed among treatments and years, with switchgrass plots generally having greater VOR, especially in 2012 and 2013 (Appendix S1: Fig. S1) as the vegetation reached maturity in later years. Additionally, VOR in mature single-harvested treatments was greater than multiple-harvested treatments, although this difference was more pronounced in switchgrass (2012 and 2013) than NWSG (2013 only), suggesting that the resulting avian community differences should also be greater between switchgrass harvest treatments when compared to NWSG single and NWSG multiple plots.

Avian community

We observed a greater number of species in NWSG, but observed more individuals in switchgrass plots, especially multiple-harvest treatments (Appendix S1: Table S1). Savannah Sparrows (*Passerculus sandwichensis*) accounted for 51.0% ($n = 7,973$) of all observations, including most

observations in multiple-harvest plots. More species were detected during summer and fall migration than during spring or winter seasons, and the number of species declined annually across all treatments as plots became fully established (Appendix S1: Table S1, Fig. S1). Treatment and VOR were correlated with avian species NMDS scores during late winter and spring as plots reached maturity in 2012 and 2013, but not during fall and early winter, except for treatment in 2011 (Appendix S1: Table S2). Correlation between these metrics provides support that treatment type

and vegetation structure were associated with differences in species ordination, with individual species selecting habitat in winter months based on both environmental variables. Savannah Sparrows, which form loose flocks of solitary individuals, and flock-prone species such as Eastern Meadowlarks (*Sturnella magna*) preferred shorter vegetation of either species composition (Figs. 2 and 3), whereas Le Conte's Sparrow (*Ammodramus leconteii*) were associated with greater VOR (Fig. 3). Other species observed during the winter including Song (*Melospiza melodia*) and Swamp

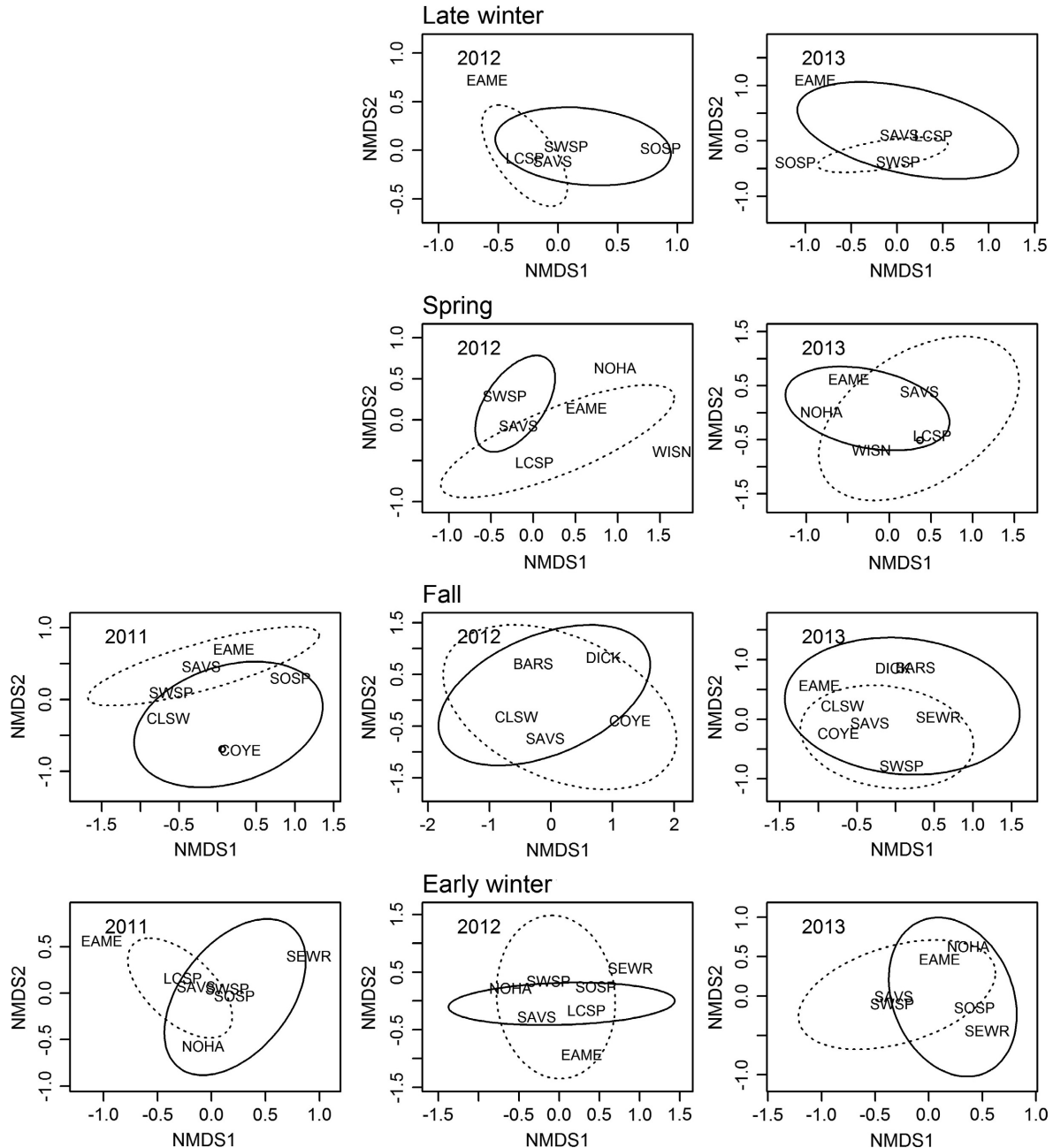


FIG. 2. Nonmetric multidimensional scaling (NMDS) plot of the first two axes of avian species with dispersion ellipses (95% CI) by native warm-season grass (NWSG; solid line) and switchgrass (dashed line) treatments by survey type in Clay County, Mississippi, 2011–2013. There were not sufficient species detections across all treatment types for 2011 data in late winter and spring survey periods. Bird species are listed by the four-letter alpha codes created by the Institute for Bird Populations (see Appendix S1: Table S1).

(*Melospiza georgiana*) Sparrows were primarily found in greater VOR and NWSG plots (Figs. 2 and 3).

There was considerable overlap in treatment dispersion ellipses for all seasons and years, except for spring 2012, indicating that the avian communities were similar among treatments. This overlap increased annually as the plots matured, although the number of individuals present varied. During spring migration, both treatment and VOR influenced habitat use of the avian community, whereas neither factor was important during fall except treatment during 2011 (Appendix S1: Table S2; Fig. 2). Ellipse spread was

similar or reduced in switchgrass treatments relative to NWSG for most years during winter months and fall migration (except 2012; Fig. 2), indicating species plotted within the smaller switchgrass ellipses selected that habitat more frequently than NWSG. We lacked sufficient detections in switchgrass treatments during summer to generate NMDS scores, but 84.0% ($n = 823$) of all individuals detected and 86.3% ($n = 684$) of observations for species occurring at $\geq 10\%$ of survey locations during summer were in NWSG plots, in congruence with the heterogeneous habitat hypothesis.

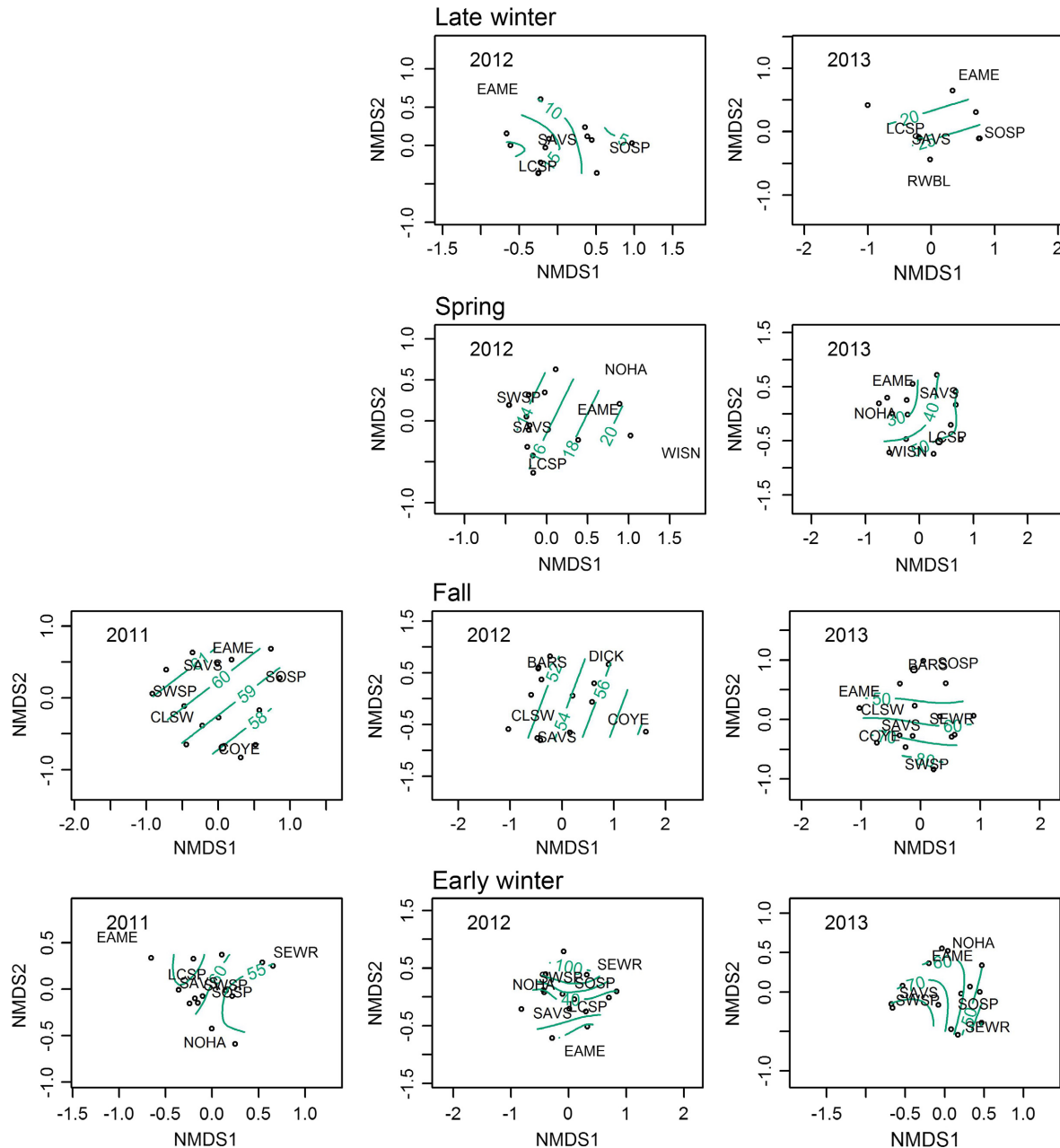


FIG. 3. Nonmetric multidimensional scaling (NMDS) plot of the first two axes of avian species with contour lines for vegetation visual obstruction (cm) by survey type in Clay County, Mississippi, 2011–2013. There were not sufficient species detections across all treatment types for 2011 data in late winter and spring survey periods. Bird species are listed by the four-letter alpha codes created by the Institute for Bird Populations (see Appendix S1: Table S1).

Relative density

All GLMM models for relative density incorporated a negative binomial distribution to account for overdispersion and block as random effect (Appendix S1: Table S3). Overall, relative density declined annually as plots matured across all survey periods (except for summer; Appendix S1: Tables S3 and S4; Fig. 5). Additionally, by the third year, mean densities were greater in multiple-harvest plots than single-harvest plots of the same vegetation type (Appendix S1: Table S1), aligning with predictions under the predation risk hypothesis; these density estimates were driven largely by flock-prone species such as Savannah Sparrows. However, 95% CI for estimated treatment means in GLMM overlapped (Fig. 4). Likelihood ratio tests supported inclusion of a treatment \times year interaction for both early ($L = 15.13$, $df = 6$, $P = 0.02$) and late winter ($L = 19.08$, $df = 6$, $P < 0.01$), with more birds detected in switchgrass than NWSG across all years, except for NWSG multiple (2013 only; Appendix S1: Tables S3 and S4). For summer, the best-fit density models contained both treatment and year as the inclusion of year improved fit ($L = 18.14$, $df = 2$, $P < 0.01$). It also provided support for the heterogeneous habitat hypothesis, with significantly greater avian relative density in NWSG treatments than either single-harvest switchgrass ($\beta = -1.38$, 95% CI: -1.82 , -0.94) and multiple-harvest switchgrass plots ($\beta = -1.54$, 95% CI: -1.99 , -1.08 ; Appendix S1: Tables S3 and S4; Fig. 4).

Aviation risk

Overall, 99.7% ($n = 7,973$) of observations included species with a relative hazard score ≤ 10 (maximum hazard score = 100). Most birds observed (92.4%; e.g., sparrows) were considered a “very low” aviation safety hazard, based primarily on small body size (Dolbeer and Wright 2009). All aviation risk models included year (Appendix S1: Tables S3 and S5), with aviation risk declining annually across all survey periods as plot vegetation became well-established 2–3 yr post-planting, resulting in increased VOR (Appendix S1: Fig. S1) that can limit predator visibility, thereby restricting the effectiveness of flocking behaviors as an escape strategy. For both spring and fall, likelihood ratio tests indicated the best-supported model included year effects, but not both year and treatment (spring, $L = 0.53$, $df = 3$, $P = 0.91$; fall, $L = 7.08$, $df = 3$, $P = 0.07$), with lower aviation risk in 2013 (spring, $\beta = -152.24$, 95% CI: -208.90 , -95.58 ; fall, $\beta = -106.20$, 95% CI: -152.55 , -59.86 ; Fig. 5). The best-fit models for the other survey periods supported the additional inclusion of treatment as an additive effect (late winter, $L = 24.96$, $df = 3$, $P < 0.001$; summer, $L = 21.29$, $df = 3$, $P < 0.01$) or treatment \times year interaction (early winter, $L = 10.51$, $df = 2$, $P < 0.01$; Appendix S1: Tables S3 and S5). Overall, the treatment type with greater aviation risk varied by survey period. Switchgrass treatments had greater risk during late winter (switchgrass multiple, $\beta = 864.69$, 95% CI: 365.75 , $1,363.62$; switchgrass single, $\beta = 573.90$, 95% CI: 74.97 , $1,072.84$; Appendix S1: Table S6; Fig. 5). During early winter,

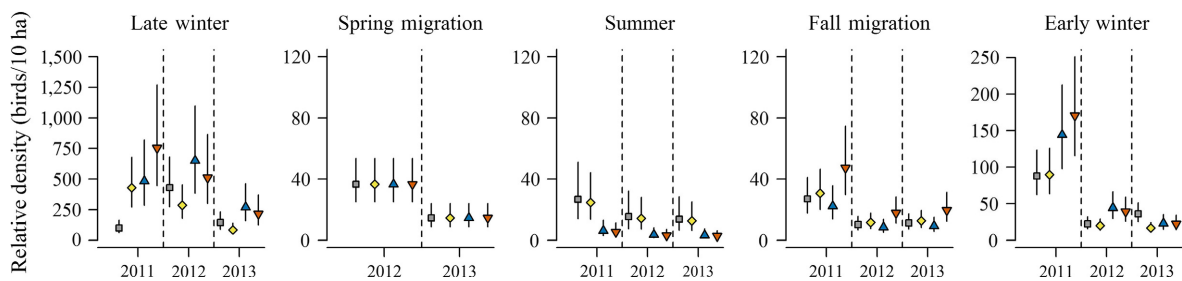


FIG. 4. Predicted values and 95% CI for avian relative density by survey period for native warm-season grass (NWSG) multiple harvest (squares), NWSG single harvest (diamonds), switchgrass multiple harvest (up-pointing triangles), and switchgrass single harvest (down-pointing triangles) treatment types in Clay County, Mississippi, 2011–2013.

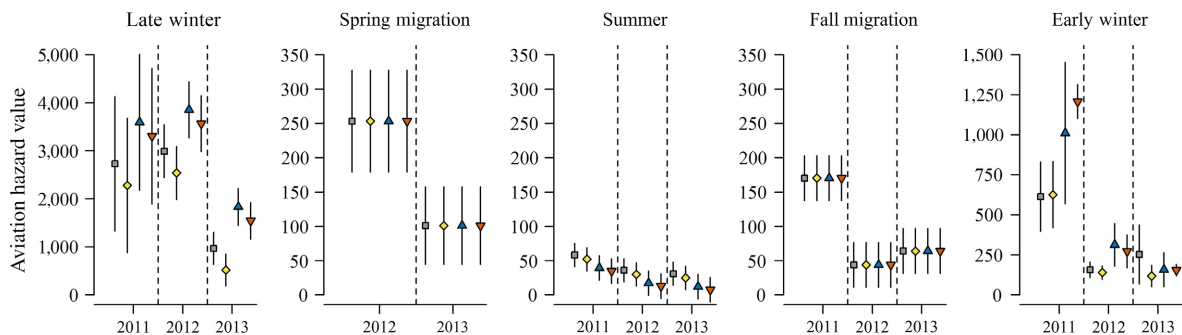


FIG. 5. Predicted values and 95% CI for avian aviation hazard score by survey period for native warm-season grass (NWSG) multiple harvest (squares), NWSG single harvest (diamonds), switchgrass multiple harvest (up-pointing triangles), and switchgrass single harvest (down-pointing triangles) treatment types in Clay County, Mississippi, 2011–2013.

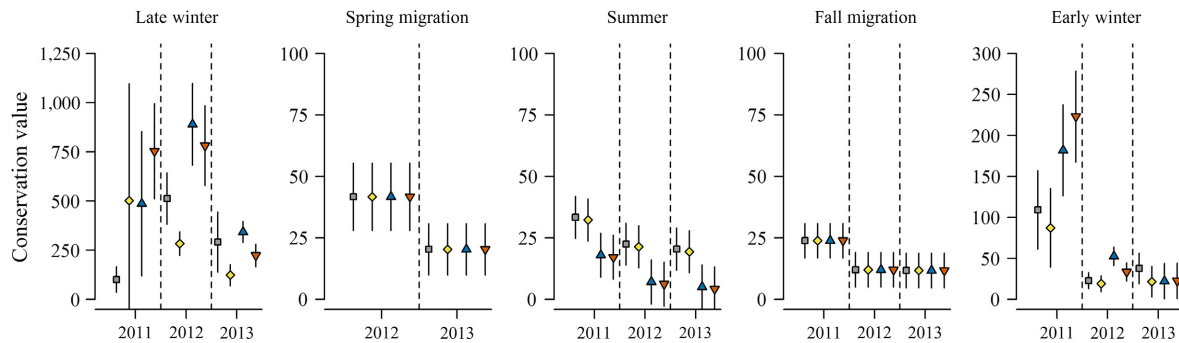


FIG. 6. Predicted values and 95% CI for conservation value by survey period for native warm-season grass (NWSG) multiple harvest (squares), NWSG single harvest (diamonds), switchgrass multiple harvest (up-pointing triangles), and switchgrass single harvest (down-pointing triangles) treatment types in Clay County, Mississippi, 2011–2013.

switchgrass had a significantly greater avian hazard score than NWSG plots in 2011 and 2012 but, by 2013, the hazard scores were similar across all treatment types. Switchgrass plots had lower hazard scores than both NWSG treatments during summer (switchgrass multiple, $\beta = -18.77$, 95% CI: $-32.91, -4.63$; switchgrass single, $\beta = -23.25$, 95% CI: $-37.40, -9.11$), as most birds favored the heterogeneous NWSG plots during breeding.

Conservation value

Conservation value was best explained during both winter survey periods by a treatment \times year interaction (late winter, $L = 21.60$, $df = 6$, $P < 0.001$; early winter, $L = 20.46$, $df = 6$, $P < 0.01$), whereas the best-fit model for summer only included additive effects of treatment and year ($L = 6.08$, $df = 6$, $P = 0.41$; Appendix S1: Table S3). During migration periods, likelihood ratio tests indicated the best-fit model for conservation value including year was better supported than the null model for both spring ($L = 10.95$, $df = 2$, $P < 0.01$) and fall ($L = 12.20$, $df = 2$, $P < 0.001$; Appendix S1: Fig. S5). Conservation value declined annually in all other models that included year.

Additionally, conservation value differed among treatments (Fig. 6; Appendix S1: Table S6), although this treatment effect differed between summer and both winter periods. Switchgrass plots had lower conservation value than NWSG treatments (switchgrass multiple, $\beta = -15.39$, 95% CI: $-21.96, -8.83$; switchgrass single, $\beta = -16.25$, 95% CI: $-22.82, -9.69$; Fig. 6; Appendix S1: Table S6) during Summer, whereas there was greater conservation value for switchgrass plots in late winter and early winter, respectively, relative to NWSG (Appendix S1: Table S6). In 2013, conservation value was greater for multiple-harvested plots during late winter (both switchgrass and NWSG) and early winter (NWSG only); however, 95% CI for estimated early winter NWSG treatment means overlapped. Only 6.3% ($n = 48$) of known species detected (Le Conte's Sparrow, Northern Harrier, Eastern Meadowlark) were considered species of PIF conservation priority based on rank values (Appendix S1: Table S1). Two of these species, Le Conte's Sparrow and Eastern Meadowlark, were detected on 18.2% and 11.8% of all survey visits ($n = 1,215$), accounting for 6.7% and 3.6%, respectively, of total observations ($n = 7,973$), whereas Northern Harriers comprised only

0.9% of observations on $<5\%$ ($n = 59$) of survey visits. Of these three species, Eastern Meadowlarks were detected across all survey periods, whereas Le Conte's Sparrows and Northern Harriers were not present during summer.

DISCUSSION

During late winter and spring, birds selected habitat with vegetation structure in accordance with their species-specific escape behaviors, flight patterns, and flocking strategies, which may also minimize their perceived predation risk (Lima and Dill 1990, Lima 1993, Wirsing et al. 2010). Savannah Sparrows and Eastern Meadowlarks, two commonly detected habitat generalists that forage in loose and compact flocks, respectively, during the winter (Grzybowski 1983a, Jaster et al. 2012), routinely selected habitats with lower VOR (Figs. 2 and 3). Not surprisingly, relative densities for these species by season declined annually as VOR increased with vegetation establishment and maturity 2–3 yr post-planting (Keyser et al. 2012; Appendix S1: Table S1, Fig. S1), with the increased vegetation structure limiting both flocking ability and visibility of predators (Whittingham and Devereux 2008, Blumstein 2010). This annual decline also decreased overall avian hazard scores and conservation value, as Savannah Sparrows alone accounted for 51.0% ($n = 7,973$) of all observations. Alternatively, Le Conte's Sparrow, a PIF species of conservation concern known for solitary behaviors and relying on crypsis rather than flocking or strong escape flights (Lowther 2005), demonstrated a strong preference for greater VOR, along with Swamp and Song Sparrows, two solitary-foraging species typically associated with woody vegetation and old fields (Mowbray 1997, Arcese et al. 2002). Watts (1990) observed similar patterns in habitat use based on species-specific escape tactics; Song Sparrows demonstrated a reliance on the screening cover of herbaceous vegetation by avoiding mowed old fields, whereas Savannah Sparrows used cover-independent escape tactics (often flying up to exposed perches) to avoid predators and showed no aversion to open grassland habitat.

These responses to vegetation structure have been noted in other studies on wintering bird use of altered grassland habitats (Beck and Watts 1997, Hovick et al. 2014, Monroe and O'Connell 2014) because of food availability, reducing predation risk, and providing thermoregulatory cover for

energy conservation (Houston et al. 1993, Beck and Watts 1997, Tsurim et al. 2009). Although perceived predation risk and food availability may concurrently influence habitat selection by wintering birds (Moorcroft et al. 2002, Whittingham et al. 2006), we did not capture birds to measure fat reserves or quantify winter seed availability in the study plots. As a result, it is unclear to what extent food resources and adaptive energy management (Lima 1986, Tsurim et al. 2009, Hammers et al. 2015) affected habitat selection choices in addition to perceived risk (see also Blackwell et al. 2016). However, the observed annual decline in the relative density of birds in established vegetation plots where greater seed production from mature plants likely increased food availability provides support that birds actively incorporated some measure of habitat structure and perceived risk into habitat selection decisions.

Contrary to non-breeding periods when vegetation structure, rather than composition, appeared to be the driving factor towards habitat selection, habitat heterogeneity was more important to grassland birds during summer. In addition to greater avian densities on NWSG relative to switchgrass plots (Fig. 4; Appendix S1: Table S4), nest densities for Dickcissels (*Spiza americana*) were 54.0–64.6 times greater in NWSG than switchgrass plots with the same harvest frequency (Conkling et al. 2017), suggesting that breeding birds responded positively to the vegetation structural heterogeneity in mixed species plantings. Of the three most commonly observed breeding bird species, Dickcissels, often associated with habitat containing a high proportion of forbs (Temple 2002), and Red-Winged Blackbirds (*Agelaius phoeniceus*), a marsh and prairie-breeding species (Yasukawa and Searcy 1995), primarily used NWSG plots. In contrast, Eastern Meadowlarks were observed in all treatment types, although they nested primarily in immature switchgrass plots (T. Conkling, *unpublished data*; Keyser et al. 2012). Forb-dominated structurally complex landscapes (especially NWSG) provide more potential nest sites and abundant food resources, especially during the breeding season, when arthropods (e.g., Orthoptera) are important for breeding birds and nestling nutrition (Winter et al. 2005, Mitchell et al. 2012).

Limited research exists on species habitat selection during migration due to difficulties in sampling migrating species (Robertson et al. 2011a, 2013). Our species identification was lower during migration when many species are non-vocal or have cryptic non-breeding plumages; over 16% ($n = 1,892$) of observations during migration periods included unknown individuals (e.g., unknown sparrows). Additionally, with only three surveys over a 20–30 d period during spring and fall, we may have inadequately detected some migrating species. As such, these counts may be conservative regarding the total number of individuals and species using the plots during migration. Increasing identification efforts and survey frequency to better detect birds based on species-specific migration timings would help mitigate existing data gaps on avian habitat use during migrations.

Many breeding grassland bird species benefit from diverse native-warm-season grasslands as opposed to large patches of switchgrass. However, the reverse may be true for migrating and wintering birds that select habitat based more on

structure than species composition (Fletcher and Koford 2002, Fletcher et al. 2010, Blank et al. 2014). This distinction is important to consider for biomass cultivation. Switchgrass produces more potential ethanol than NWSG (McLaughlin and Kszos 2005), but there is mixed support regarding the benefits of monoculture grass cultivation vs. NWSG on avian diversity and conservation (Meehan et al. 2010, Blank et al. 2014, Uden et al. 2014). Growing switchgrass concurrently with NWSG plots instead of as a monoculture allows for biomass production while still providing multiple cover and habitat options for grassland birds. Additionally, because switchgrass was rarely used as breeding habitat (Conkling et al. 2017), mid-season harvest of switchgrass plots to optimize biomass production may have a limited effect on breeding grassland birds.

Conversion of existing airport grasslands to native grasses for both bird habitat and biomass production does not appear to pose risk to aircraft (Blackwell et al. 2009, DeVault et al. 2013a, Schmidt et al. 2013), supporting the concept of native grasslands as a viable landcover for some portions of airport properties outside the air operations areas (DeVault et al. 2012). Most birds (e.g., sparrows) detected (92.4%) pose a “very low” hazard to aircraft (Dolbeer et al. 2000, Dolbeer and Wright 2009), and larger-bodied species posing a moderate to extremely high hazard to aircraft (e.g., Turkey Vultures [*Cathartes aura*]) accounted for <1% of all observations. Only 2.6% ($n = 1,215$) of all surveys yielded observations of high-hazard (e.g., classified as moderate or greater hazard; Dolbeer and Wright 2009) species. However, most of these individuals (76%, $n = 55$) consisted of moderate risk species including Mourning Dove (*Zenaidura macroura*) and Wilson’s Snipe (*Gallinago delicata*). Dolbeer et al. (2016) noted that only 2.4% ($n = 7,648$) of reported strikes involving these two species resulted in aircraft damage, providing further evidence that native grass plantings may not negatively affect aircraft safety.

Although logistical constraints precluded us from directly comparing aviation risk for turfgrass relative to NWSG or switchgrass across multiple years, we conducted bird surveys (2012 only) on managed Bermuda grass (*Cynodon dactylon*) fields near the study area as part of a concurrent study. We observed high-hazard species during 3.5% ($n = 57$) of all Bermuda grass surveys, with these species of moderate or greater hazard comprising 40% ($n = 10$) of all avian species detected (T. Conkling, *unpublished data*). We acknowledge that we conducted our study solely within Mississippi and as such, our results regarding avian habitat use and aviation strike risk in semi-natural grasslands are most applicable within the southeastern United States. However, even with our limited scope of inference, our values are similar to those reported by Schmidt et al. (2013), who found that birds categorized as moderate or greater hazard accounted for 6% of all birds surveyed in extant airfield grasslands in western Ohio, USA, compared to 2% in adjacent NWSG areas (Schmidt et al. 2013). Additionally, although our relative hazard scores used for analyses were calculated using data derived solely from strikes involving civil aircraft (DeVault et al. 2011), our results appear broadly applicable across airport types. For example, although aircraft flight operations differ somewhat between civil and military airports, the same types of birds (e.g., large-bodied, flocking

species) that are most dangerous to civil aircraft also pose the greatest risk to military aircraft (Zakrajsek and Bissonette 2005).

Model results indicated both aviation hazard and conservation value differed between grassland treatments and years (Figs. 4 and 5), but these differences are likely not biologically relevant. Treatment was a significant factor in aviation hazard models for every sampling period except Spring, but this was primarily due to the large number of sparrows that collectively posed a very low aviation risk (DeVault et al. 2011). As such, use of either NWSG or switchgrass plantings do not appear detrimental to aviation safety. Aviation hazard scores were greater in NWSG than switchgrass plots during Summer, but maximum relative density for NWSG was <25 birds/10 ha surveyed across all three survey visits and all species detected during summer (Appendix S1: Table S1) posed minimal threats to aircraft. Hazard scores also declined annually across all treatments as the vegetation reached maturity by the third year (Keyser et al. 2012), with the increased vegetation structure limiting numbers of flock-prone species (e.g., Savannah Sparrows). Similar to models for density, conservation value was largely influenced by the number of flocking species using the plots, with conservation value declining as plots matured. Conservation value may be a reliable metric for comparing conservation importance between distinct habitat classes such as pine forest and switchgrass (Loman et al. 2014), but these scores were not useful for comparing similar grassland treatments. Partners in Flight conservation priority species accounted for only 3 of the 57 species detected during surveys (Le Conte's Sparrow, Northern Harrier, Eastern Meadowlark). However, the presence of all three species had minimum effect on the weighted conservation value scores that were heavily influenced by densities of low-priority species such as Savannah Sparrows.

Although wildlife management strategies at airports focus on reducing hazards to human safety, our results demonstrate that it may be feasible to use mature native-grass plantings to concurrently minimize aviation risk and provide habitat for some grassland bird species on some portions of airport properties, especially at very large airports. Both NWSG and switchgrass provided sufficient vegetative structure to minimize the presence of high-hazard, flocking avian species while also providing quality overwintering and breeding habitat for grassland birds. As such, a mosaic of both treatment types could provide airport managers simultaneous opportunities for landcover management, biomass production, and avian conservation (Conkling et al. 2017).

It is also important to consider additional factors that may influence avian habitat use. Although we did not quantify food abundance, reductions in habitat structure can improve prey detectability and accessibility, causing birds to modify foraging behaviors and habitat use accordingly to balance predation risk and energy acquisition (Getty and Pulliam 1993, Atkinson et al. 2004, Butler et al. 2005b, Whittingham et al. 2006, Blackwell et al. 2016). For example, although managed grasslands support populations of small mammals, habitat use by foraging raptors is influenced more by the prey's vulnerability to predation, rather than overall abundance (Baker and Brooks 1981, DeVault et al. 2011, DeVault and Washburn 2013). As such, these behavior

modifications may enhance aviation risk if food availability increases avian densities or reduces forager vigilance, thereby slowing response times required to successfully escape approaching aircraft (Blackwell et al. 2013).

Overall, by addressing avian responses specifically within a managed system, this study provides airport professionals with evidence to help make informed decisions regarding the risk produced by native grasses vs. traditional landcover strategies at airports. As technological advancements further increase interest in cellulosic biofuels, the benefits from both risk and cost management perspectives are promising for enhancing grassland bird conservation and biomass production in airport landscapes.

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SUPPORTING INFORMATION

Additional supporting information may be found online at: <http://onlinelibrary.wiley.com/doi/10.1002/eap.1716/full>

DATA AVAILABILITY

Data available from the Dryad Digital Repository: <https://doi.org/10.5061/dryad.k44303c>