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Effects of patch-burn grazing and rotational grazing on grassland bird abundance, species richness, and diversity in native grassland pastures of the Midsouth USA

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ABSTRACT

Declines in native warm season grasslands have been linked to grassland bird population decline due to habitat loss including conversion to non-native grasses. Rotational grazing (ROT) and patch-burn grazing (PBG) are two possible tools to restore native warm-season grasses (NWSG) on working-lands in the Mid-South USA and thus aid in the recovery of grassland bird populations. This project compares ROT, PBG, and before treatment implementation to assess their effects on grassland-associated bird species. At three research sites between KY and TN, 14–10 ha NWSG pastures were established and randomly assigned 7 pastures each to ROT and PBG treatment and monitored avian relative abundance during the breeding season from 2014 to 2017. Avian call count data and vegetation characteristics were collected in 2014 and treated as a before treatment year. Following 2014, ROT and PBG treatments were implemented across each respective research site. We used the open N-mixture model framework to estimate avian relative abundance related to year, treatments, research site, and landscape and within-field variables. Avian species richness and diversity were calculated for each treatment, research site, and year. Landscape variables, within-field variables, and research sites exerted more influence on relative abundance than ROT or PBG. Grassland-associated bird species relative abundance and species richness/diversity were affected by habitat disturbances (both ROT and PBG) but varied by species and site. Field sparrows [*Spizella pusilla*] had the highest increase in relative abundance (9.68 ± 1.24 birds/point count location or 1.77 ha) while northern cardinals [*Cardinalis cardinalis*] exhibited a significant decrease in relative abundance (3.44 ± 1.54 birds/point count location) following treatment implementation on specific research sites. Species diversity and richness did not differ between ROT and PBG treatments. However, a site and year difference were observed for both estimates. Using ROT and/or PBG to create habitat disturbances can alter within-field variables (i.e., vegetation height) which, taken into context with landscape variables, could impact grassland bird populations and diversity depending on grassland bird species habitat requirements. Our research provided the baseline information for ROT and PBG impacts on grassland birds in the east/southeastern USA. However, we believe future research should focus on breeding and annual fecundity to better understand how populations will change over time and how working lands conservation might aid this conservation effort without a reduction in livestock productivity.

1. Introduction

The United States has experienced an estimated > 98% decline in dry savanna or steppe, grassy savanna, prairie, and/or shrub savanna ecosystems since European settlement (Bailey, 1980; Noss et al., 1995; White et al., 2000; Wilsey et al., 2019). This decline is attributed to

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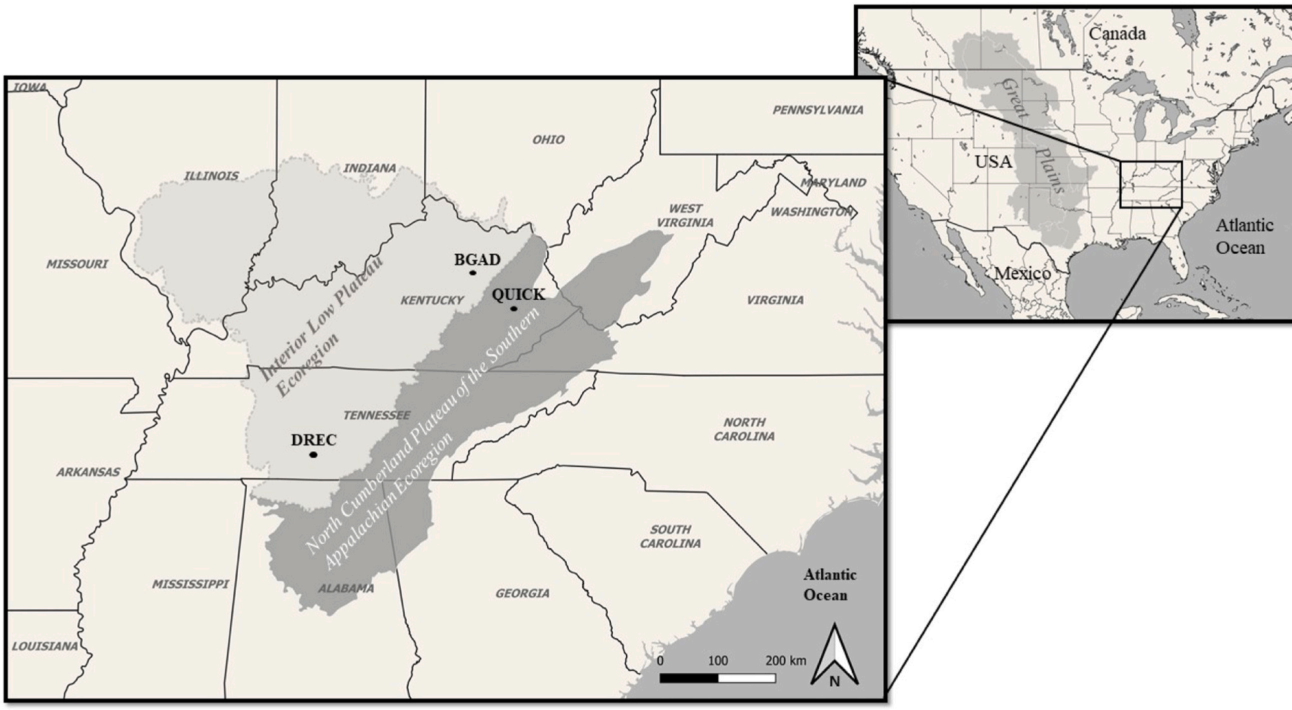

Fig. 1. Study site location for DREC, BGAD, and QUICK within 2 ecoregions to assess the impacts of rotational grazing (ROT) and patch-burn grazing (PBG) on grassland-associated bird relative abundance, species richness, and diversity on native warm-season grasses pastures in the Mid-South in Tennessee and Kentucky, USA.

habitat degradation, habitat fragmentation, conversion of native grasslands to non-native grasses and row crop agriculture, fire suppression, and mismanaged livestock grazing (Green et al., 2005; White et al., 2000). Conservation groups (e.g., Central Hardwoods Joint Venture, National Bobwhite Conservation Initiative) and government agencies (e.g., United State Fish and Wildlife Service, Natural Resources Conservation Service) have attempted to promote sustainable grazing practices on private lands to benefit both livestock production and wildlife populations, notably grassland birds, within a "working-lands conservation" framework (Keyser et al., 2019; Kremen and Merenlender, 2018; Monroe et al., 2016). Under the working-lands model, land management techniques, such as conversion of non-native pastures to native warm-season grasses (NWSG), are used to provide wildlife habitat while also maintaining livestock production on the same land base.

Two strategies for improved grazing management that could contribute to working lands conservation are rotational grazing (ROT) and patch-burn grazing (PBG). The more widely used, ROT, relies on systematic shifting of livestock (i.e., predominately cattle, *Bos tarus*), temporally and spatially to achieve uniform utilization of forage within a given paddock (Briske et al., 2011; Holling, 1978). The more recent innovation in grazing management, PBG, also referred to as pyric-herbivory (i.e., periodic fires with large ungulate grazing), incorporates natural disturbances associated with herbivory, fire, and their interaction in a manner that resembles historical patterns for North American grasslands (Fuhlendorf et al., 2009).

There remains a substantial knowledge gap regarding avian responses to management strategies for NWSG in eastern grasslands, including two strategies often recommended for managing NWSG pastures, ROT and PBG. Additionally, the adoption of working lands conservation based on NWSG in the eastern USA has been slow, in part because of questions regarding grazing management (i.e., expected yields for livestock) (Keyser et al., 2019). Improved understanding of both management strategies and their effects on grassland bird populations could increase adoption by producers, and ultimately contribute to stabilizing or reversing grassland bird population declines. With the potential increase in demand for livestock production as human population increases (Thornton, 2010), it is important to understand how these 2 grazing strategies impact wildlife, more specifically grassland birds. Grassland bird populations have declined by ~ 45% since the 1970s and have experienced a reduction in species diversity across North American grassland ecosystems (Benton et al., 2003; Martin and Possingham, 2005; Rosenberg et al., 2019; Stanton et al., 2018). There is an especially pressing need to understand how ROT and PBG management strategies affect grasslands and grassland birds in the humid temperate eastern United States. It is also important to understanding how obligate (avian species that require some grassland habitat for survival) and facultative (avian species that can utilize grassland habitat but do not require for survival) species respond to ROT and PBG. Management responses, plant species composition, and landscape context of eastern grasslands all may be different than those of semi-arid grasslands of the Great Plains where current research for ROT/PBG on wildlife has been most prevalent. Relative abundance of eastern meadowlarks [*Sturnella magna*] and grasshopper sparrows [*Ammodramus savannarum*] were positively related with PBG pastures in northern Missouri and southern Iowa (Pillsbury et al., 2011). A PBG study in Tennessee reported the practice created spatial heterogeneity (i.e., grass height, litter cover); however, the research only examined vegetation and not avian responses (McGranahan et al., 2013). Thus, data on ROT and PBG on grassland bird abundance and diversity in the grasslands of the eastern United States are limited.

Our primary objective was to evaluate the effects of ROT and PBG in NWSG pastures on grassland-associated avian species abundance, diversity, and richness in the humid, temperate eastern United States, which differ from the historical Great Plains regions where these methods originated. Our secondary objectives were to determine if 1) landscape and/or within-field vegetation conditions influence these same avian metrics and 2) how avian species abundance, diversity, and richness for ROT and PBG treatments compared to ungrazed-unburned conditions on these same pastures. We predict that grassland-associated bird abundance, diversity, and richness would be greater on PBG than ROT pastures. We also predicted that grassland obligate species' relative abundance (i.e., grasshopper sparrow, eastern meadowlark) would be negatively associated with the amount of forest edge cover in the landscape (Jacobs et al., 2012) and species associated with tall and dense vegetation structure (i.e., Henslow's sparrow) would be negatively affected by PBG management (Pillsbury et al., 2011; Powell, 2006).

2. Materials and methods

2.1. Study area and site preparation

We conducted research on three sites: 1) Madison County in east-central Kentucky (Blue Grass Army Depot [BGAD]; 37°41′31″ N, 84°10′56″ W; elevation, 283 m), 2) Breathitt County in eastern Kentucky (Quicksand, Robinson Center for Appalachian Resource Sustainability [QUICK]; 37°25′42″ N, 83°10′22″ W; elevation, 383 m) and 3) Marshall County in south-central Tennessee (Dairy Research and Education Center [DREC]; 35°24′58″ N, 86°48′50″ W; elevation, 251 m; Fig. 1). The BGAD and DREC are located within the Bluegrass and Highland Rim sections, respectively, of the Interior Low Plateau (Griffith, 2010; The Nature Conservancy, 2005) while QUICK is located in the North Cumberland Plateau of the Southern Appalachian ecoregion (Griffith, 2010; The Nature Conservancy, 2003). The Interior Low Plateau consists of irregular plains, open hills, and smooth plains with an elevation between 200 and 300 m and an average annual precipitation of 111 cm. The Interior Low Plateau is generally described as a predominately oak [*Quercus* spp.]-hickory [*Cary* spp.] forested region with sections of prairie (The Nature Conservancy, 2005). The North Cumberland Plateau is characterized by oak-hickory, oak-pine [*Pinus* spp.] mixed forest with pastures dominated by tall fescue and reclaimed coal surface mines; elevations range from 365 to 609 m with annual precipitation of 88–139 cm (Griffith, 2010; The Nature Conservancy, 2003). Pastures (9.73 ± 0.47 ha each) at each site were previously established to NWSG during the growing seasons of 2012–2013 (Keyser et al., 2015b). Pasture size was selected to coincide with the surrounding working pasture lands of the Mid-South. Stands were sown with a grass mixture that included 6.7 kg ha^{-1} (pure live seed basis) big bluestem [*Andropogon gerardii*], 3.3 kg ha^{-1} Indian grass [*Sorghastrum nutans*], and 1.1 kg ha^{-1} little bluestem [*Schizachyrium scoparium*]. Six pastures were established at BGAD, a property that also included significant amounts of tall fescue pastures and hayfields, and oak-dominated woodlots. Four pastures were connected 1890 m northeast of 2 isolated pastures. Four pastures were established at DREC (376 m average distance between all pastures), which had similar land use as BGAD. At QUICK, four pastures (642 m average distance between pastures) were established with the surrounding landscape being a reclaimed surface mine (reestablished 2004–2012) dominated by tall fescue, sericea lespedeza [*Lespedeza cuneata*], and stands of various planted hardwoods including autumn olive [*Elaeagnus umbellate*] and American sycamore [*Platanus occidentalis*].

2.2. Treatments and management protocol

We divided each experimental pasture into thirds (3.2 ha paddocks) using temporary polywire fencing (ROT pastures only) with permanent fencing to enclose each pasture. Prescribed burns were conducted based on a randomly assigned sequence such that a different paddock within each PBG pasture was burned each year during the three years of the study. Rotationally grazed pastures were not burned during the study. Pastures were not grazed or burned for either treatment during 2014 to allow them to complete establishment and to allow for pre-treatment data collection. We considered 2014 as a "before" year to estimates baseline pre-treatment data for avian species abundance, diversity and richness, for all 3 sites.

We used an initial stock density of cattle based on previous NWSG research in the Mid-South and was adjusted across the sites based on pasture conditions (Keyser et al., 2015a). On the previously restored mine site (QUICK), stock density was 260–350 kg ha^{-1} while at BGAD it was 500–600 kg ha^{-1}, and at DREC, 620–700 kg ha^{-1}. Stocking density within sites was similar between treatments for all years. In all cases, the objective was to maintain an optimal pasture condition for cattle production across all experimental pastures (Backus et al., 2017). A mixture of heifers and steers was used for grazing. On PBG pastures, cattle had access to the entire 10 ha. Cattle on rotational grazing pastures were rotated among the three paddocks based on residual vegetation height (target = 35–45 cm); in practice, this resulted in cattle being rotated approximately once every 4–7 days. Cattle occupied each pasture from mid-May until late August each year, 2015–2017. Grazing did not occur on DREC during 2017 and the site was not included in the study for that year.

2.3. Avian surveys

We conducted avian community and species abundances surveys at fixed radius (75 m) point count locations within each pasture (Elzinga et al., 1995). We placed detected birds into distance bands 0–25 m, 26–50 m, 51–75 m, and > 76 m (Farnsworth et al., 2002) and each bird was then "removed" for the duration of that survey to prevent double detection (Elzinga et al., 1995). We truncated observations > 75 m because they were outside our fixed detection radius and to avoid double-counting on subsequent counts. We conducted point count surveys in May, June, and July (n = 1 visit/month) from 2014 to 2017. We spaced fixed point count locations > 150 m apart within each pasture to avoid repeat detections of individuals, so there were ≤ 5 points/pasture. Within each pasture, we started point counts ~ 30 min before sunrise and ended no later than ~ 5 h after sunrise, and surveys were only conducted on days with no rain, fog, or high winds (> 16 km/h). We conducted surveys for 3 min at each point count location and observers recorded all individuals seen and heard. Point count data was collected by 8 observers per year (3 each at BGAD, DREC, and 2 at QUICK). Point count observers varied among years but were trained in survey protocols and use of rangefinders to estimate distances before each years' data collection began. Researchers also used rangefinders to estimate distances during surveys. We collected pre-treatment information during the breeding season of 2014 on all 14 pastures. We conducted surveys at the same point count locations each year (2014–2017).

2.4. Habitat sampling

We collected vegetation samples within pastures at each avian point count in May, June, and July. We sampled within-field habitat measurements for vegetation height (cm), litter depth (cm), and used a Daubenmire frame to estimate the percent cover of grass, forbs, bare ground, and litter. Starting at each point count center, we measured within-field habitat variables along a 25 m transect located in a single randomly selected cardinal direction (N, S, E, or W; Elzinga et al., 1999). We recorded vegetation metrics every 5 m and averaged among visits for analyses. We used these measurements to document within-field changes in vegetation structure resulting from treatment implementation. We created a 250 m buffer around each point count center using ArcGIS 10.7.1 (ESRI ArcGIS Desktop Develop Team, 2019) to estimate surrounding landscape variables [grass cover (%), forest canopy cover (%), bare ground/developed (%), and forest edge density (m/ha)]. We used the National Agricultural Statistics Service (NASS) CropScape data layer for land cover analysis (Han et al., 2012). We chose to use NASS rather than National Land Cover Database (NLCD) because NASS data are produced annually which allowed us to explicitly match our year of interest with the corresponding NASS data layers. Resolution of the NASS CropScape was 0.08 ha/pixel (USDA National Agricultural Statistics Service Cropland Data Layer, 2019). We calculated annual sample mean values and standard errors (*SE*) for all landscape and within-field variables.

2.5. Data analysis

Of all grassland bird species detected across all study sites, we selected species for data analysis based on conservation status (i.e., species listed on the Birds of Conservation Concern List; U.S. Fish and Wildlife Service, 2008), whether occupied vegetation types that would represent possible responses to cattle grazing treatments, had > 30 detections (Smith et al., 1997), and a number that permitted models to converge properly (Moineddin et al., 2007). Before fitting models, we assessed response variables multicollinearity by calculating variance inflation factors (VIF) with the VIF function in the R package CAR, version 3.5.0 (Fox and Weisberg, 2018). We created a linear regression model with all response variables and removed variables with VIF values > 5. We utilized an open N-mixture model to estimate relative abundance for selected grassland-associated bird species from point count survey data that was replicated at temporal and spatial scales (Dail and Madsen, 2011). Dail and Madsen's (2011) model is a generalization of the Royle (2004) N-mixture model approach that allows for an open population between surveys. We ran models under the pcountOpen function in the unmarked statistical package (Fiske et al., 2020; Fiske and Chandler, 2011) in Program R, version 3.6.2 (R Core Team, 2019) to estimate relative abundance (λ) and detection probabilities (ρ). Prior to running models, we used the scale function in the unmarked package to standardize all variables. We assess the appropriate distribution for our count data by comparing a Poisson, zero-inflated Poisson, and negative binomial distribution for each bird species (Kéry, 2018). We held recruitment rate (γ) and survival probability (ω) among years constant (Zipkin et al., 2014) for each selected species and site. We pooled avian species detections across sites to establish a detection function for each species to maintain consistency and to increase sample size. We assessed significant detection predictors for each species by creating models containing time-since-sunrise (TSS), day, year, and site as covariates. We added year as a detection covariate based on previous research which showed northern bobwhite (which inhabit NWSG areas) calling behavior significantly differed between 2 years (Lituma et al., 2017). We evaluated relative abundance models using 5 subsets for each selected grassland bird species. Each subset consisted of dummy coded covariates: 1) year (1–4 yrs), 2) site (BGAD, DREC, QUICK), 3) year × site interactions, 4) treatments (PBG, ROT, before treatment), and 5) landscape [grass cover (%), forest canopy cover (%), bare ground/developed (%), and forest edge density (m/ha)] and within-field variables [vegetation height (cm), litter depth (cm), percent cover of grass, forbs, bare ground, and litter]. For each model subset, for detection and relative abundance, we used Akaike's Information Criterion corrected for small sample sizes (AIC_c) to evaluate model performance and determine competing models (≤ 2.0 ΔAIC_c; Anderson, 2008; Burnham and Anderson, 2002). Lastly, we placed the top competing models (≤ 2.0 ΔAIC_c) from the previous 5 subsets into a combined model set to determine covariate effects on relative abundance estimates for each selected grassland bird species. For each top model, we deemed variables important if 95% confidence intervals of β estimates did not overlap zero (Arnold, 2010). We backtransformed each variable prior to estimating relative abundance predictions to retain their original scale. We calculated predicted relative abundance for each supported variable from top supported models. For those supported variables, we created a range based on the 1st and 3rd quartile from the mean for each variable (i.e., vegetation height 32–62 cm) one at a time and held all other model variables at their mean value. Finally, to explicitly test for pre-post-treatment effects, we ran post-hoc models for top models, but added a site × year 1 interaction. Thus, post-hoc models were a final step to examine pre-post-treatment effects on abundance, but after accounting for landscape and/or within field variables from the other model subsets.

We calculated species diversity with the Shannon–Wiener species diversity index (H', Hutcheson, 1970; Shannon, 1949) for each study site and across the 4 years of which was defined as:

$$H' = -\sum (n_i/N \times \ln n_i/N)$$

where the number of individuals of each of the species (n_i) divided by the total number of species (N) times the natural log of the individual species divided by the total number of species. We calculated equivalent estimates for sites and years to compare diversity for the Shannon–Wiener index (Jost, 2007) and were defined as:

$$H^e = [\exp(H')]$$

where the inverse of the natural logarithm (ln) raised to the power of the Shannon–Wiener index. We calculated equivalent estimates (H^e) mean and standard error for each research site, year, and treatment. We calculated Menhinick (1964) species richness (D) which was defined as:

$$D = n/\sqrt{N}$$

where the number of species (n) divided by the square root of the total number of individuals (N). We calculated Shannon–Wiener species diversity index and Menhinick's species richness estimates with the vegan package in Program R (Oksanen et al., 2019). We treated the 2014 breeding season data collection as a "before" year to estimate baseline data for relative abundance and species diversity/richness for all 3 sites.

Table 1
Open population N-mixture model results from the combined model set for top ranked models (Δ AICc < 2.0) and the closest competing model for all 7 selected grassland-associated species. Model selection was based on Akaike's information criteria for small sample sizes (AICc), the difference between ranked models (Δ AICc), and model weight or likelihood (w_i).

Models	K	AICc	Δ AICc	w_i
Field sparrow				
λ(QUICK*year 1 + DREC*year1) γ(.) ω(.) ρ(year3)	10	1623.32	0.00	1.00
λ(ROT*DREC + ROT*QUICK) γ(.) ω(.) ρ(year3)	10	1669.98	46.66	0.00
Red-winged blackbird				
λ(Veg hgt + %forb + forest edge + %grass) γ(.) ω(.) ρ(year2 + day)	10	934.22	0.00	0.98
λ(ROT*DREC) γ(.) ω(.) ρ(y2 + day)	9	938.68	4.45	0.02
Indigo bunting				
λ(QUICK*year1 + DREC*year1) γ(.) ω(.) ρ(year1 + year2 + year3)	12	1544.99	0.00	0.45
λ(QUICK*year2 + DREC*year2) γ(.) ω(.) ρ(year1 + year2 + year3)	12	1545.24	0.26	0.40
λ(Veg hgt) γ(.) ω(.)p(year1 + year2 + year3)	8	1548.96	3.97	0.06
Common yellowthroat				
λ(QUICK*year1 + DREC*y1) γ(.) ω (.) ρ(day + tss)	11	1026.33	0.00	0.69
λ(ROT*DREC + ROT*QUICK) γ(.) ω(.) ρ(day + tss)	11	1028.52	2.19	0.23
Northern cardinal				
λ(QUICK*year1 + DREC*year1) γ(.) ω(.) ρ(tss)	10	871.79	0.00	0.60
λ(PBG*year2) γ(.) ω(.) ρ(tss)	8	874.46	2.67	0.15
Eastern meadowlark				
λ(Veg hgt + litter depth) γ(.) ω(.) ρ(year1 + day)	8	742.22	0.00	0.75
λ(%Bare ground*year2 + DREC*year2) γ(.) ω(.) ρ(year1 + day)	11	745.69	3.46	0.13
Henslow's sparrow				
λ(%forb + forest edge + %grass) γ(.) ω(.) ρ(year1 + day)	9	628.31	0.00	1.00
λ(ROT*year2) γ(.) ω(.) ρ(year1 + day)	9	645.03	16.72	0.00

K is the number of parameters for each model; λ: relative abundance predictive variable; γ: recruitment rate (held constant); ω: survival probability (held constant); ρ: detection probability (year(#): year 1–3, day: day of the study; tss: time-since-sunrise); QUICK and DREC research site; ROT: rotational grazing; PBG: prescribed-burn grazing; forest edge: forest edge density (m/ha); % grass: cover at the landscape scale, Veg hgt: vegetation height (cm); % forb: cover at the within field scale; litter depth (cm); %Bare ground: at the within field scale.

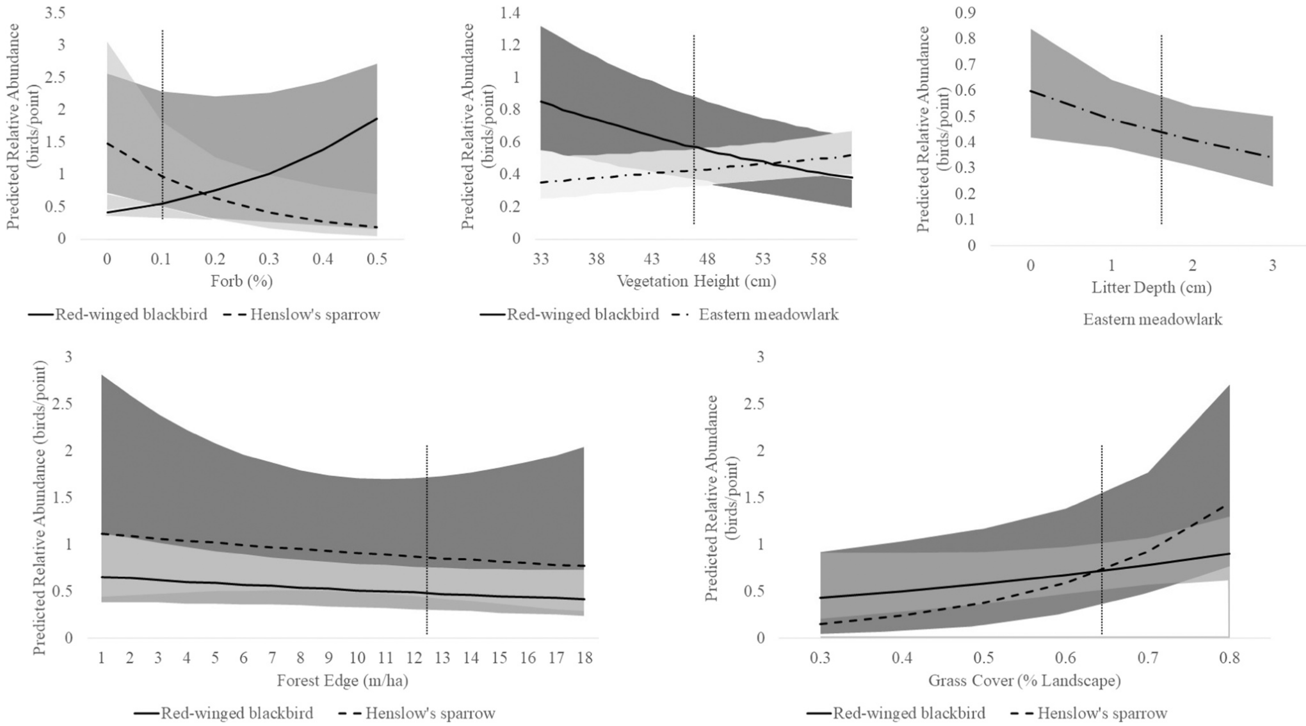

Fig. 2. Predicted relative abundance with standard error bands (dark and light gray) for grassland-associated bird species for significant landscape and within-field variables derived from combined model analysis to assess the impacts of rotational grazing (ROT) and patch-burn grazing (PBG) on 2 research sites located in Kentucky and 1 in Tennessee, USA between 2014 and 2017. Dash vertical line indicates mean variable estimates for the focal landscape or within-field variable.

3. Results

A total of 7460 individual birds were counted across all 3 sites and 4 years (BGAD = 4690; DREC = 1629; QUICK = 1141) representing 84 avian species collected from 744-point counts visits. Seven grassland-associated bird species met our selection criteria for data analysis: 2 grassland obligates (eastern meadowlark, Henslow's sparrow); 4 facultative grassland species (field sparrow; indigo bunting, [*Passerina cyanea*]; common yellowthroat, [*Geothlypis trichas*]; red-winged blackbird); and a habitat generalist (northern cardinal, [*Cardinalis cardinalis*]). Field sparrow, common yellowthroat, indigo bunting, red-winged blackbird, and northern cardinal were located at all 3 sites each breeding season, while Henslow's sparrow was only detected on BGAD. Eastern meadowlarks were identified on BGAD and DREC, but only one individual was detected at QUICK during the study. Of the 7 selected species, 2150 individuals were detected across all years and sites which field sparrow accounted for 28% ($n = 614$), indigo bunting 25% ($n = 569$), common yellowthroat 11% ($n = 249$), red-winged blackbird 9% ($n = 200$), northern cardinal 7% ($n = 174$), Henslow's sparrow 7% ($n = 173$), and eastern meadowlark 7% ($n = 171$).

3.1. Avian relative abundance

The results of each combined model set for Henslow's sparrow, eastern meadowlark, and red-winged blackbird provided considerable support that relative abundance was influenced by landscape and within-field variables (ΔAICc < 2; Table 1). Red-winged blackbird relative abundance was negatively related to forest edge in the landscape ($\beta = -0.053$, 95% CI = −0.091 to −0.016). Our results also indicate Henslow's sparrows exhibited a weak negative associated with forest edge however, the 95% confidence intervals overlapped zero ($\beta = -0.021$, 95% CI = −0.092 to 0.049). Red-winged blackbird ($\beta = 1.46$, 95% CI = 0.06–2.86) and Henslow's sparrow ($\beta = 4.47$, 95% CI = 1.42–7.53) relative abundance was positively related to percent grass cover in the landscape (Fig. 2). Within-field variables were related with relative abundance of red-winged blackbird (% forb: $\beta = 3.16$, 95% CI = 1.77–4.55, vegetation height: $\beta = -0.70$, 95% CI = −0.09 to −0.001), Henslow's sparrow (% forb: $\beta = -4.26$, 95% CI = −6.88 to −1.65), and eastern meadowlark (litter depth: $\beta = 0.070$, 95% CI = 0.00–0.13) (Fig. 2). Combined model sets for field sparrow, indigo bunting, and northern cardinal supported a pre- and post-treatment effect (2014 vs. 2015–2017) on relative abundance (Table 2). Field sparrow and indigo bunting relative abundance increased following treatment implementation by an estimated 9.68 ± 1.24 and 4.28 ± 0.53 birds/point count location (or 1.77 ha) at DREC, respectively, while at BGAD, only indigo bunting relative abundance increased by 2.73 ± 0.37 birds/point count location (Fig. 3). Northern cardinal relative abundance decreased by 3.44 ± 1.54 birds/point count location at QUICK following initiation of ROT and PBG. A site × year interaction was supported for field sparrow, indigo bunting, northern cardinal, and common yellowthroat (Table 1).

For the selected species which had a site × year interaction, the post-hoc analysis indicated landscape and within-field variables influenced relative abundance (Fig. 4). Forest edge ($\beta = 0.03$, 95% CI = 0.008–0.06) and grass cover ($\beta = 1.49$, 95% CI = 0.42–2.56) were positively related with field sparrow relative abundance across all sites (Fig. 4). Northern cardinal relative abundance was negatively related with % grass cover at the landscape scale ($\beta = -2.27$, 95% CI = −3.65 to −0.90) but following treatment relative abundance increased on QUICK. Indigo bunting relative abundance was negatively related to vegetation height ($\beta = -0.01$, 95% CI = −0.02 to −0.007) (Fig. 4) however, relative abundance following treatment implementation increased for all sites. Grass cover at the within-field scale was negatively related with common yellowthroat relative abundance ($\beta = -1.27$, 95% CI = −2.16 to −0.38), yet after ROT and PBG treatments, relative abundance increased each year across all sites. Based on model selection results, at least 1 or more covariates influenced detection probability and varied among the seven selected bird species (Tables 1 and 2).

Table 2
Open population N-mixture top racked models (Δ AICc < 2.0) results for post-hoc analysis of focal species that had model support for grassland bird relative abundance in the before treatment year differed from all other years. Model selection was based on Akaike's information criteria for small sample sizes (AICc), the difference between ranked models (Δ AICc), and model weight or likelihood (w_i).

Models	K	AIC_c	Δ AIC_c	w_i
Field sparrow				
λ(QUICK*year1 + DREC*year1 + %grass + forest edge) γ(.) ω(.) ρ(year3)	12	1622.17	0.00	0.442
λ(QUICK*year1 + DREC*year1) γ(.) ω(.) ρ(year3)	10	1623.32	1.16	0.248
λ(QUICK*year1 + DREC*year1 + Veg hgt) γ(.) ω(.) ρ(year3)	11	1624.20	2.03	0.160
Indigo bunting				
λ(QUICK*year1 + DREC*year1 + Veg hgt) γ(.) ω(.) ρ(year1 + year2 + year3)	13	1543.62	0.00	0.477
λ(QUICK*year1 + DREC*year1 + Veg hgt + litter depth) γ(.) ω(.) ρ(year1 + year2 + year3)	14	1545.38	1.76	0.197
λ(QUICK*year1 + DREC*year1 + Veg hgt) γ(.) ω(.) ρ(year1 + year2 + year3)	13	1545.63	2.01	0.174
Common yellowthroat				
λ(QUICK*year1 + DREC*year1 + Veg hgt) γ(.) ω(.) ρ(day + tss)	12	1023.95	0.00	0.50
λ(QUICK*year1 + DREC*year1 + Veg hgt + litter depth) γ(.) ω(.) ρ(day + tss)	13	1024.54	1.81	0.20
λ(QUICK*year1 + DREC*year1 + %bare ground)γ(.) ω(.) ρ(day + tss)	12	1026.33	2.19	0.17
Northern cardinal				
λ(QUICK*year1 + DREC*year1 + %grass) γ(.) ω(.) ρ(tss)	11	866.22	0.00	0.381
λ(QUICK*year1 + DREC*year1 + %forb + % grass) γ(.) ω(.) ρ(tss)	12	867.41	1.19	0.215
λ(QUICK*year1 + DREC*year1 +forest edge + %grass) γ(.) ω(.) ρ(tss)	12	868.13	1.91	0.153
λ(QUICK*year1 + DREC*year1 +forest edge + %forb + %grass) γ(.) ω(.) ρ(tss)	13	869.35	3.12	0.084

K is the number of parameters for each model; λ: relative abundance predictive variable; γ: recruitment rate (held constant); ω: survival probability (held constant); ρ: detection probability (year(#): year 1–3, day: day of the study; tss: time-since-sunrise); QUICK and DREC research site; ROT: rotational grazing; PBG: prescribed-burn grazing; forest edge: forest edge density (m/ha); % grass: cover at the landscape scale, Veg hgt: vegetation height (cm); % forb: cover at the within field scale; litter depth (cm); %Bare ground: at the within field scale.

3.2. Avian species diversity and richness

A total of 84 avian species (Supplementary Table S1) were detected during data collection and differed based on sites and years ranging from 21 species at QUICK to 67 species at BGAD. Overall, species diversity and richness did not differ between ROT and PBG treatments (Supplementary Table S2). However, at the site-level, there were statistically significant variation in community metrics for both diversity and richness (Fig. 5). Both community measures increased post-treatment on BGAD then decreased consecutively between 2016 and 2017 yet, each metric remained higher than the pre-treatment estimates. Species diversity and species richness decreased for QUICK post-treatment but partially rebounded in 2016. Species diversity and richness declined at DREC all 3 years sampled.

3.3. Within-field habitat and landscape structure

Mean vegetation height across all sites and pastures declined following initial treatment implementations in 2015 (Table 3). All other within-field habitat variables varied among sites and treatment pastures (Table 3). Forest landscape cover was highest at QUICK (0.48%, ± 0.02) and moderate to low on BGAD (0.20%, ± 0.02) and DREC (0.06%, ± 0.02), respectively. Landscape grass cover for BGAD and DREC was 66% (± 0.02) while QUICK was 51% (± 0.02). Forest edge was highest on QUICK (22.17 m/ha, ± 0.70), moderate on BGAD (9.51 m/ha, ± 0.98), and lowest on DREC (3.32 m/ha, ± 0.88).

4. Discussion

Our data did not support the predictions that grassland-associated bird relative abundance, species diversity, or species richness would be greater on PBG than ROT pastures. To the best of our knowledge, this research represents the first comparison of ROT and PBG management in the humid southeastern USA. Much of the previous research on ROT and PBG has been conducted in the Great Plains and on larger pastures (100–900 ha) than we utilized for our research sites. By comparison to the more humid environment (i.e., greater annual precipitation and greater humidity) where our research was conducted, grasslands where past research has been conducted are much more extensive, inclusions of forest or woody cover are much less common, grass species composition differs, and structure is much shorter (Augustine and Derner, 2015; Ball et al., 2015; Holechek et al., 2010; Holechek and Herbel, 1982). The more rapid growth of vegetation in our environment may have minimized treatment differences relative to these more xeric environments in the Great Plains. Furthermore, this lack of response could be attributed to the structural conditions on our research sites having been within the range of adaptation for these specific bird species (i.e., species habitat plasticity). Regardless, none of the obligate or facultative grassland bird species relative abundance, species diversity, or richness we examined displayed a direct response (positive or negative) related directly to either grazing strategy during our study.

We documented landscape and within-field variables as influential factors for obligate and facultative grassland bird species that were not linked to grazing treatment. For two of the grassland bird species we examined, Henslow's sparrow (obligate) and red-wing blackbird (facultative), relative abundance was directly linked to landscape and within-field variables and a third, eastern meadowlark (obligate), was linked to within-field variables only. The landscape influence on Henslow's sparrows and red-winged blackbirds were similar in that both species relative abundance was negatively related with forest edge density and positively related with % grass cover. Yet the % forb, a within-field variable, exhibited a positive influence on red-winged blackbird relative abundance and a negative impact for Henslow's sparrow, likely due to each species' specific breeding habitat preference. Our data confirm previous research that Henslow's sparrow density increased with the total area of grassland in the surrounding landscape (Winter, 1998). Within fields and pastures, Henslow's sparrows avoid recently disturbed (i.e., grazed or burned) areas where standing vegetation is significantly reduced (Herkert, 2002, 1994; Zimmerman, 1988). Additionally, Johnson and Igl (2001) suggested red-winged blackbird's relative abundance was tied more to proximate habitat features (i.e., tall grass and dense forb cover) rather than the size of grassland patch or landscape variable. Habitat disturbance (i.e., livestock grazing, burn grazing, or both) altered structure to make it more favorable to eastern meadowlark. Previous research indicated eastern meadowlarks were more abundant in pastures that were exposed to light or moderately grazed in grasslands with tall vegetation (Powell, 2008). Vickery (1996) noted that cattle grazing would create bare patches of ground which some grassland bird species (i.e., Henslow's sparrow and eastern meadowlark) need for foraging. Without such habitat disturbance, there is a potential that grassland bird species could experience a reduction in relative abundance or be extirpated from an area as grassland ecosystems would move into the next successional stages (i.e., increased shrub-density) (Brennan and Kuvlesky, 2005). Based on the loss of NWSG and the long- and short-term population trends for our study area there has been a significant decrease in all but one of our focal bird species (BBS) (Supplementary Table S4).

Even though our research did not directly link ROT or PBG to relative abundance, we did document pre- and post-treatment differences in

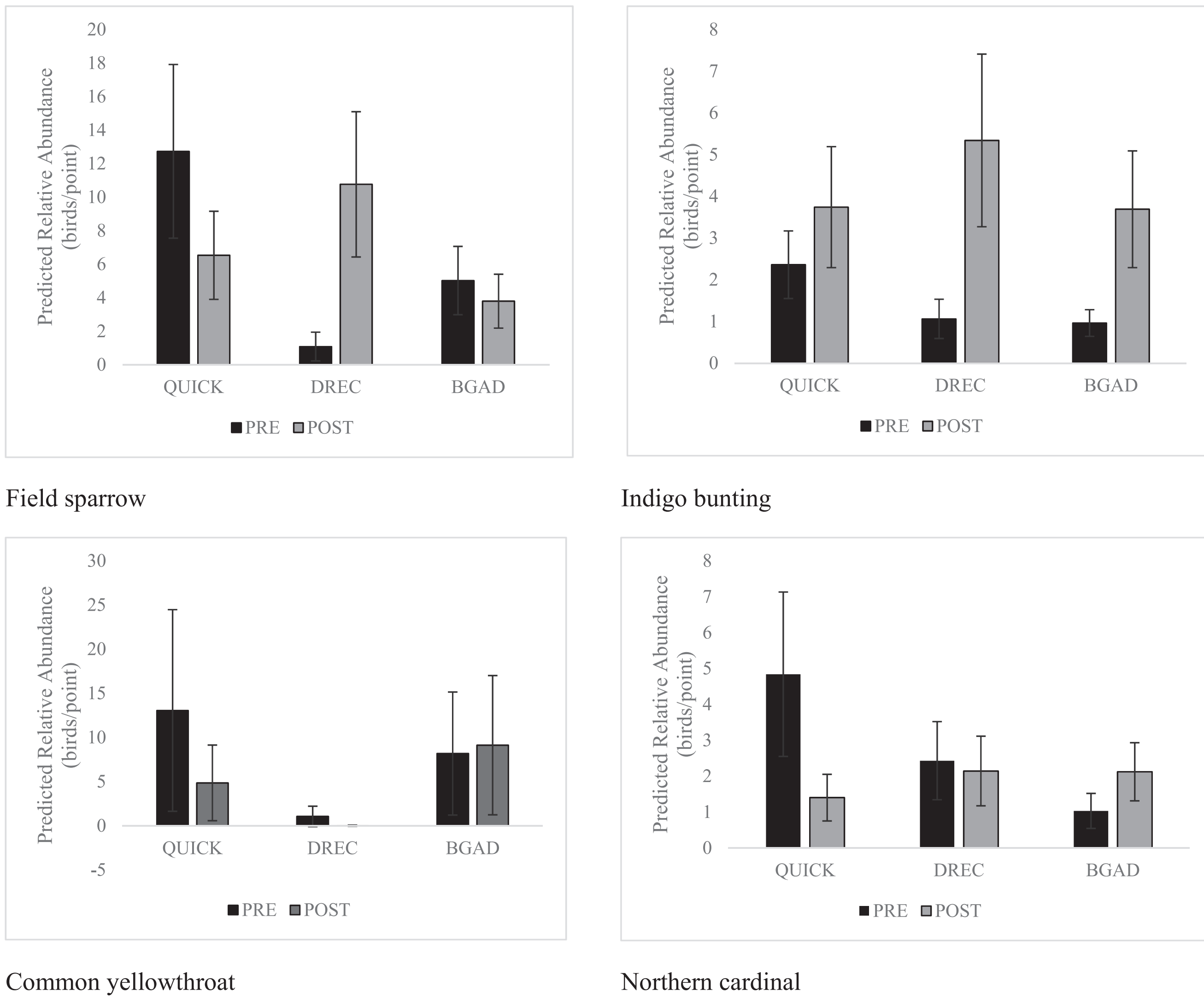

Fig. 3. Facultative grassland bird species predicted relative abundance estimates for pre-treatment (2014; PRE) and post-treatment (2015–2017; POST) to assess the impacts of rotational grazing (ROT) and patch-burn grazing (PBG) management on 3 research sites (QUICK, DREC, and BGAD) in the Mid-South in Tennessee and Kentucky, USA.

relative abundance for three facultative grassland species: field sparrow, indigo bunting, and northern cardinal. For these species, the habitat disturbances that we imposed (i.e., ROT and PBG) had a site- and species-specific influence on relative abundance. Our research indicated landscape effects likely mediated the species-specific differences among sites. Thus, at DREC, the combination of landscape features (forest edge density 3.32 m/ha and 66% grass cover) and grazing management (PBG and ROT) created the most favorable conditions for field sparrow, as reflected in the greater increase in relative abundance (0.95 birds/point count location in 2014–9.38 birds/point count location in 2015–2016) at that site. Field sparrows have been positively associated with lower forest edge density and an increase in grass cover at the landscape scale during the breeding season (Best, 1979). Grazing at DREC may have created openings in grass cover thereby reducing vegetation height (Table 3), which in combination with edge density at the site created favorable conditions for field sparrows.

As vegetation height increased indigo buntings predicted relative abundance decreased at DREC and BGAD but not at QUICK indicate within-field and treatment implementation was important, but site was also influential. These patterns could be due to indigo bunting habitat requirements at each of these scales. This assertion is supported by two studies, Fletcher and Koford (2002a) and Renfrew and Ribic (2008), which found multiple scales (landscape and within-field variables) can influence species-specific relative abundance for various grassland-associated birds.

Conversely, northern cardinal relative abundance decreased at QUICK (4.84 birds/point count location in 2014–1.4 birds/point count location in 2015–2016) following treatment implementation. This suggests that PBG and ROT, in conjunction with lower % grass cover at the landscape scale, created unfavorable or reduced preferred nesting habitat conditions during the breeding season for this species. At DREC and BGAD, northern cardinal relative abundance did not differ from pre- to post-treatment, an outcome likely mediated by the more open and less forested landscape context combined with the ~ 15% less grass cover at the landscape scale for QUICK when compared to DREC and BGAD. Decreased relative abundance for northern cardinals following treatment implementation indicates that this species in the North Cumberland Plateau ecoregion could be more susceptible to grazing pressure when landscape cover is ~ 50% grassland and ~ 50% forest than compared to northern cardinals in the Interior Lower Plateau ecoregion (~ 60% grassland and ~ 20% forest).

Our research also emphasizes the importance of site-specific responses of obligate and facultative grassland bird species when land managers and those interested in grassland bird conservation develop

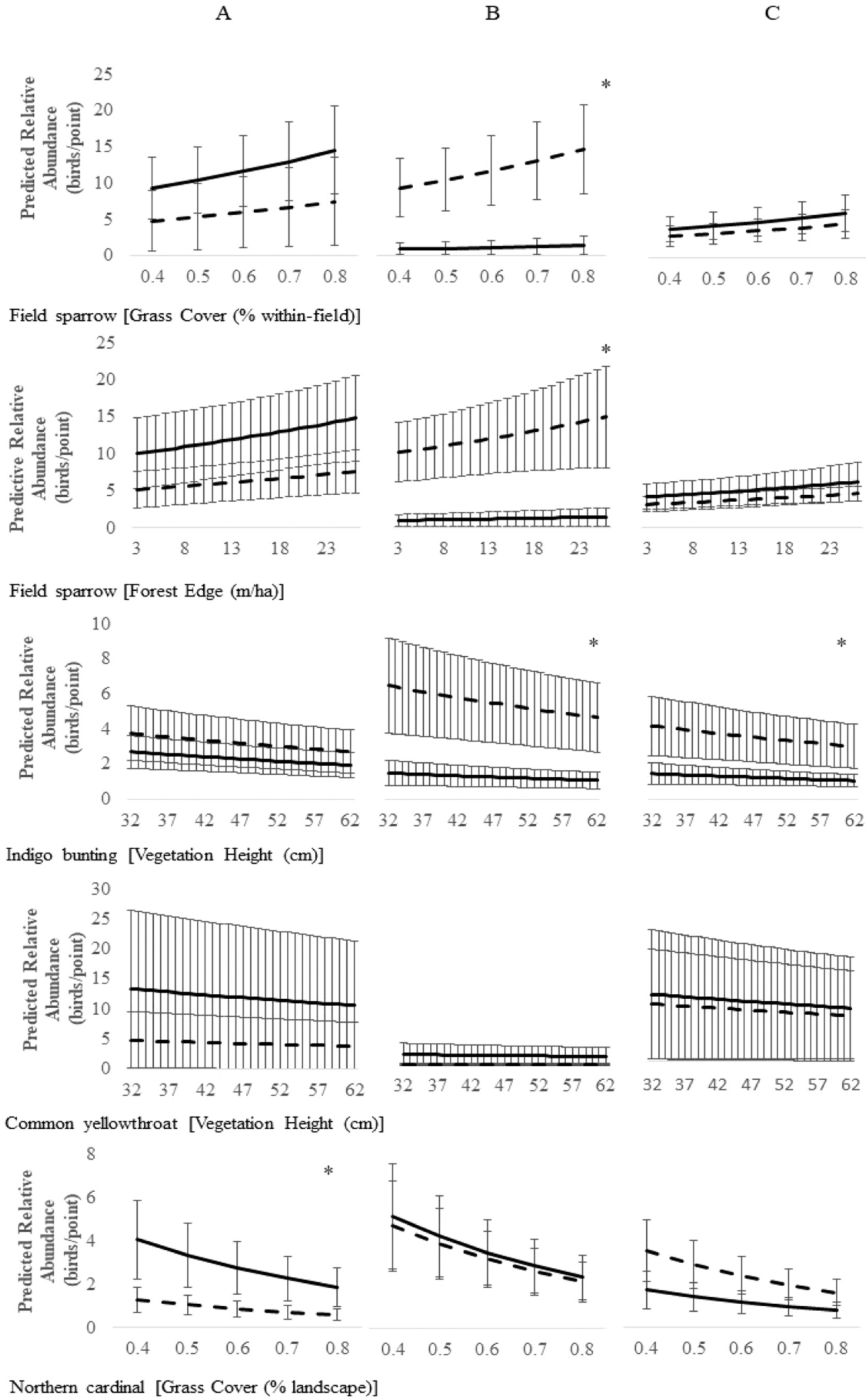

Fig. 4. Landscape and within-field variable effects on predicted relative abundance for selected grassland-associated bird species from post-hoc analysis on pre-(solid black line) and post-(dash black line) treatment pastures during a rotational grazing (ROT) and patch-burn grazing (PBG) project on QUICK (A), DREC (B), and BGAD (C) research sites in Tennessee and Kentucky, USA between 2014 and 2017. The * indicate a significant difference between pre and post-treatment for the respective bird species and landscape or within field variable.

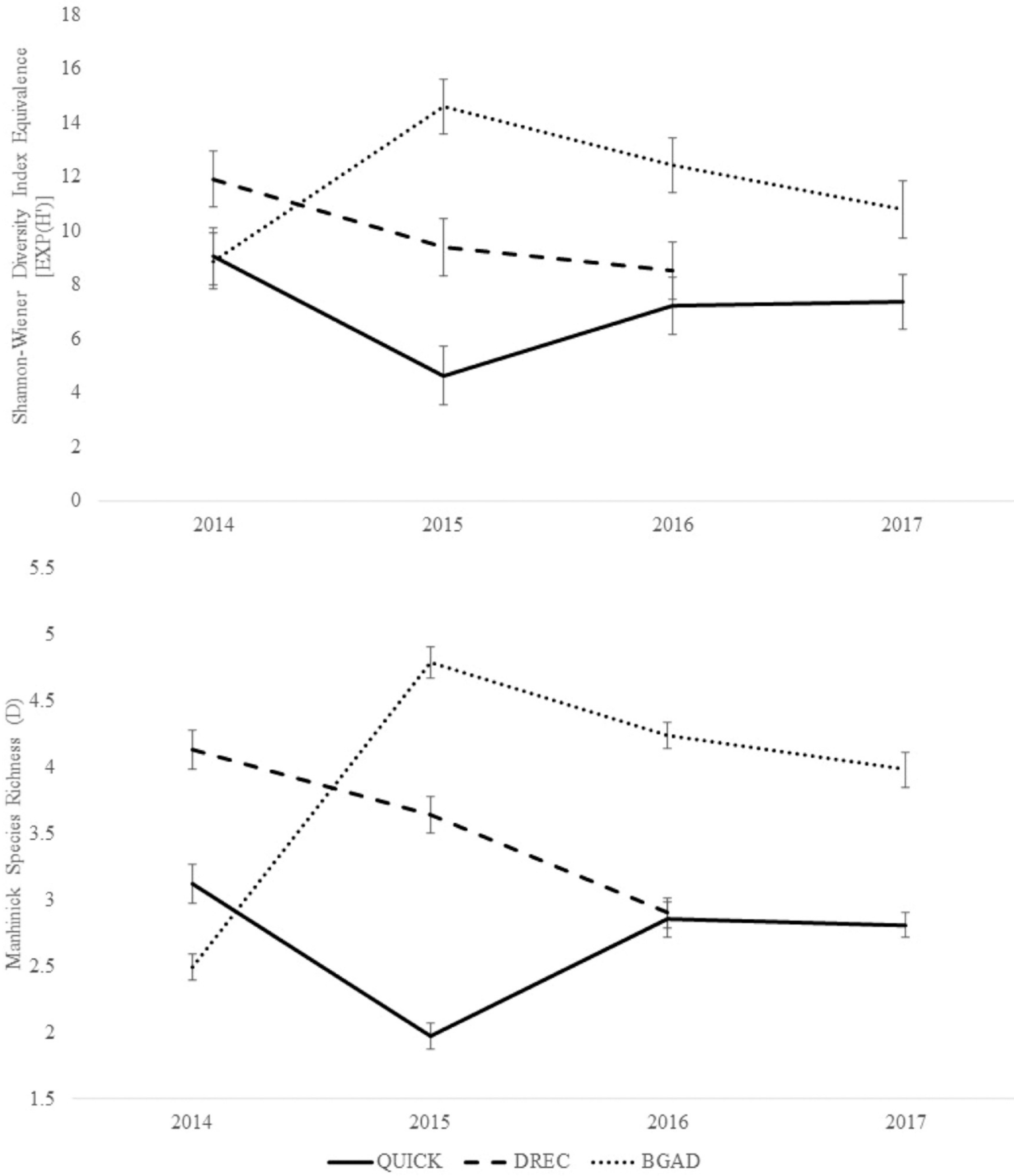

Fig. 5. Grassland-associated avian species diversity and richness estimates on 3 research locations (1: Tennessee; 2: Kentucky) to assess rotational grazing (ROT) and patch-burn grazing (PBG) effect on native warm-season grassland pastures between 2014 and 2017.

management plans. We did not find a consistent response pattern for our species among our research sites. Developing appropriate working-lands approaches for recovering grassland bird populations will necessitate landscape- and site-specific considerations (Concepción et al., 2020; Concepción and Díaz, 2019; Díaz and Concepción, 2016; Fletcher and Koford, 2002b; Vander Yacht et al., 2016), an assertion supported by our results. In our case, differences in species abundance among sites consistently had the greatest explanatory power, and responses to PBG and ROT were minimal, although there was evidence of species-specific responses to pre- and post-treatment conditions.

Even though habitat disturbance can potentially impact grassland bird relative abundance, ROT and PBG treatments were not a critical factor influencing species diversity and richness. Previous researchers suggested differences in grassland bird communities may have been due less to grazing management than landscape features (i.e., forest edge density or canopy cover; Cerezo et al., 2011; Sliwinski et al., 2019). A potential explanation for differences we observed in species diversity and richness across our research sites and years could be grassland-associated bird's responses to habitat disturbance more generally rather than specific grazing strategy. Brawn et al. (2001) stated that avian responses to prescribed burning and grazing in the grassland ecosystem were: 1) species that recolonized immediately after the prescribed burn or habitat disturbance (i.e., lark sparrow), 2) recolonized ~ 2 yrs. post-burn but before woody encroachment (i.e., grasshopper sparrow), or 3) species that required woody encroachment and greater time post-disturbance (i.e., common yellowthroat). We believe these are plausible explanations for the differences in avian diversity and richness across our 3 grassland research sites.

We conducted our research at multiple spatial and temporal scales to assess the impacts of ROT and PBG management on grassland bird

Table 3
Totals samples collected (*N*), sample means, and standard error (*SE*) results for within-field vegetation variables for 3 research sites across Tennessee and Kentucky during a rotational grazing (ROT) and patch-burn grazing (PBG) study to ascertain effects on grassland-associated bird relative abundance, species diversity and richness from 2014 to 2017.

Site	Year	*N*	Veg height (cm)	(*SE*)	Litter depth (cm)	(*SE*)	Grass (%)	(*SE*)	Forb (%)	(*SE*)	Litter (%)	(*SE*)	Bare ground (%)	(*SE*)
BGAD	2014	576	76.30	(1.19)	0.91	(0.07)	83.36	(0.94)	14.27	(0.87)	0.61	(0.18)	0.55	(0.19)
BGAD	2015	576	29.12	(0.81)	2.20	(0.20)	46.43	(1.15)	14.32	(0.84)	31.97	(1.16)	6.85	(0.73)
BGAD	2016	576	45.28	(0.74)	0.19	(0.02)	63.81	(1.04)	8.18	(0.65)	19.46	(0.83)	8.19	(0.68)
BGAD	2017	582	63.86	(0.98)	0.89	(0.04)	63.15	(0.98)	17.50	(0.87)	13.85	(0.64)	5.24	(0.54)
DREC	2014	378	70.63	(1.99)	3.33	(0.21)	58.21	(1.76)	3.90	(0.56)	28.20	(1.48)	9.46	(1.01)
DREC	2015	288	41.94	(1.28)	3.46	(0.18)	47.22	(1.64)	1.58	(0.36)	41.23	(1.89)	9.98	(1.22)
DREC	2016	306	53.38	(0.87)	2.49	(0.09)	77.04	(0.01)	1.07	(0.00)	19.72	(0.01)	1.58	(0.00)
QUICK	2014	324	46.01	(1.07)	2.85	(0.22)	56.51	(1.61)	17.55	(1.04)	6.94	(0.58)	18.92	(1.46)
QUICK	2015	324	25.94	(1.05)	5.28	(0.31)	53.07	(1.56)	5.83	(0.68)	26.34	(1.36)	13.92	(1.34)
QUICK	2016	342	21.15	(0.77)	0.89	(0.07)	20.07	(0.87)	6.49	(0.58)	50.18	(1.61)	23.41	(1.49)
QUICK	2017	342	42.32	(0.85)	0.95	(0.06)	54.59	(1.33)	15.78	(1.09)	16.45	(0.86)	13.00	(1.27)

relative abundance and diversity. We acknowledge our study lacks a Before-After-Control-Impact (BACI) design which would be desirable but with the small pastures in the southeastern USA and much of the land in private ownership we deemed the Before After Impact (BAI) appropriate because the climate conditions were similar during the summer months for at each site across all years of the study (National Oceanic and Atmospheric Administration local weather station data; Supplementary Table S3). An additional concern may be raised due to the nearness of 4-pastures at BGAD which could indicate a lack of independence for our call count sampling. We addressed this potential issue by ensuring our call point placement was at least 150 m from the nearest next call point and truncated observation at 75 m to avoid making inferences to individuals beyond paddock boundaries. We believe the 3 min call point duration was also an ensure bird did not move between paddocks during call counts or be double-counted across the paddocks.

5. Conclusions

Our results provide information that either ROT or PBG could be used within a working-lands conservation model for native grass pastures within the eastern United States. The similarity between these two grazing strategies in terms of outcomes for avian species suggests there may be a great deal of flexibility in managing NWSG pastures, with respects of avifauna, in the southeastern USA. Indeed, given the higher precipitation in the eastern USA, we believe that without the use of livestock grazing and/or prescribed burning that grassland bird populations will continue to decline due to a loss of early successional grasslands. Utilizing ROT and/or PBG to create habitat disturbance while accounting for landscape context appears to provide appropriate within-field composition and structure to maintain or, depending on surrounding habitat, increase relative abundance and diversity/richness for the grassland-associated avian species we evaluated. Managers and biologists interested in grassland bird management must account for this site-specific variability when implementing ROT and PBG management. Species responses in our study were mediated by landscape-level factors. Keyser et al. (2019) addressed the working-lands model for grasslands of the eastern United States and the importance of having clearly defined management goals. Our research provided the baseline information for ROT and PBG impacts on grassland birds in the humid eastern USA. Because relative abundance, diversity, and richness may not fully address the full impacts of ROT and PBG, future research should focus on breeding and annual fecundity to better understand how grazing management of NWSG might influence working lands conservation efforts.

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Declaration of Competing Interest

None.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2021.107710.

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