



Research Article

Effects of Scale and Land Cover on Loggerhead Shrike Occupancy

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ABSTRACT The loggerhead shrike (*Lanius ludovicianus*) is a species of concern throughout its range because of severe population declines over the past 7 decades. Grassland habitat loss and fragmentation is widely viewed as contributing to the decline. Habitat associations have primarily been studied up to the territory scale, with few studies assessing shrike habitat selection at landscape scales. We conducted roadside passive-active point counts for loggerhead shrikes in the coastal plain of South Carolina to evaluate detection probability and how land use and habitat fragmentation at multiple spatial scales influenced the occupancy of loggerhead shrikes. Detection probability increased with fair weather to 63% with clear skies, with temperature to 83% at 28°C, and with declining noise to 63% at no noise. High probability of detection at mean covariate conditions ($P = 0.49 \pm 0.08$ [SE]) indicated support for the application of passive point counts in future loggerhead shrike monitoring. Occupancy was best predicted by percentage of pasture within 1 km, where predicted occupancy increased from $2 \pm 0.02\%$ when there was 0% pasture in a 1-km radius, to $98 \pm 0.03\%$ occupancy when there was 43% pasture in a 1-km radius, though our model-averaged occupancy estimate was low ($\psi = 0.17 \pm 0.05$). Extrapolation of our averaged model suggested that only 8% of the South Carolina coastal plain was occupied by loggerhead shrikes in the 2016 and 2017 breeding seasons. Our results highlight the importance of habitat beyond breeding territories and highlight the need for the establishment of higher concentrations of pasture and grassland at a 1-km scale to increase the proportion of area in South Carolina occupied by shrikes. © 2018 The Wildlife Society.

KEY WORDS detection probability, fragmentation, grassland birds, land cover, *Lanius ludovicianus*, loggerhead shrike, occupancy.

The loggerhead shrike (*Lanius ludovicianus*), a mid-sized passerine endemic to North America, has exhibited a range-wide decline of at least 74% since the 1940s (Cade and Woods 1997, Rosenberg et al. 2016). Loggerhead shrikes breed across most of Mexico, the United States except for part of the Midwest and New England, and into parts of central Canada (Cade and Woods 1997). Northern populations are migratory and spend the winter in the southern United States and Mexico. The Breeding Bird Survey estimated an average annual decline of 3.0% from 1966 to 2015 (Sauer et al. 2017), although annual population trends in key ecoregions range from -7.3% in oaks (*Quercus* spp.) and prairies in Texas, to -1.3% in the intermountain west (Rosenberg et al. 2016). The Atlantic Coast has

exhibited average annual short-term population declines (-2.9%) but has experienced one of the highest regional population loss percentages between 1970 and 2014 (-89.0% ; Rosenberg et al. 2016). The southeast experienced an average population decline of 6.0% per year prior to 1985 (Luukkonen 1987). Consequently, the loggerhead shrike has become a species of conservation concern and is now listed as a species of conservation priority in numerous state wildlife action plans, including all southeastern and coastal states from Texas to New York, USA.

Despite the extent and longevity of loggerhead shrike declines, the causes are still unknown. One plausible reason for the decline is the loss of available breeding habitat (Luukkonen 1987, Smith and Kruse 1992, Gawlik and Bildstein 1993, Prescott and Collister 1993). The loggerhead shrike has 4 essential habitat requirements: open grassy areas for foraging, trees or large shrubs for nesting, elevated perches for foraging, and spiky vegetation for prey

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impalement (Pruitt 2000). These requirements have historically been filled by numerous ecosystems such as sage-steppes, prairies, desert scrubs, pinyon (*Pinus* spp.)-juniper (*Juniperus* spp.) woodlands, and pine (*Pinus* spp.) savannas (Pruitt 2000), but these land cover types are being converted to human-modified land (e.g., crop fields, residential areas). As a result, native grasslands, savannas, and barrens represent 55% of all critically endangered ecosystems (identified by having $\geq 98\%$ decline in area) in the United States (Noss et al. 1995). Decline in these native ecosystems could be detrimental to loggerhead shrikes because non-native land cover types are likely to support novel plant and animal communities, though shrikes may be able to adapt to these changing conditions.

As native grasslands have become increasingly rare, researchers have observed shrikes using a variety of human-modified land cover types, from pasture to croplands (Luukkonen 1987, Chabot et al. 2001, Esely and Bollinger 2001, Walk et al. 2006). Indeed, in South Carolina, pasture comprised over half of the area within 100 m of nests (Gawlik and Bildstein 1990). However, it is unclear whether other human-modified land cover types, such as row crop agriculture, are useful for shrikes. Shrikes have been reported to use fencerows by roads and cropland, and edges of cropland (Stewart 1975, Smith and Kruse 1992) but have also been reported to avoid cropland (DeGeus 1990). Conflicting evidence for selection or avoidance of cropland has even been observed within studies. For example, Gawlik and Bildstein (1990) found no cropland within 100 m of nests, yet they observed shrikes using cropland and pasture equally during the breeding season, and shrike use of cropland increased in the non-breeding season. Additionally, some large pasture areas seemingly ideal for shrikes have remained unoccupied leading some researchers to suggest that habitat is not a limiting factor to shrike populations (Jobin et al. 2005). Collectively, this suggests that there is still more to discover about what specifically constitutes habitat for shrikes and how this may differ regionally.

How an individual selects habitat can be an intricate process that occurs on multiple spatial scales (Johnson 1980), yet most shrike studies have only examined habitat selection up to the territory scale (Luukkonen 1987, Gawlick and Bildstein 1990, Prescott and Collister 1993, Walk et al. 2006). For many avian species, it is increasingly understood that larger-scale habitat connectivity is important (Brennan

and Kuvlesky 2005), and fragmentation negatively affects the occurrence of shrikes in Ontario, Canada (A. A. Chabot, Queens University, unpublished report). In the southeastern United States, landscape composition has become increasingly fragmented, particularly over the past several decades (Griffith et al. 2003). Thus, including landscape-scale characteristics in habitat analyses is important to building an understanding of how land-use change and the patchiness or connectivity of habitats influences shrike populations. Our objective was to examine how land use and habitat fragmentation at multiple spatial scales influences the breeding season occupancy of loggerhead shrikes in the coastal plain of South Carolina to inform future conservation decision-making. We evaluated support for several competing hypotheses of how land use and habitat fragmentation at multiple spatial scales influenced the occupancy of loggerhead shrikes (Table 1). Specifically, we hypothesized that amount of pasture, degree of pasture aggregation, degree of aggregation of pasture and crop combined, amount of forest, amount of crop, and vegetative productivity would influence shrike occupancy.

STUDY AREA

The South Carolina Breeding Bird Atlas (Cely 2003) and eBird records (eBird 2012) suggest that shrikes are most often sighted in the coastal plain and portions of the Piedmont regions in South Carolina, with a majority in the coastal plain. Accordingly, we identified a study area with 2 focal regions in the coastal plain of South Carolina totaling 368,086 ha. We selected the study area to maximize our ability to collect shrike detections and to represent a range of land cover conditions for testing our habitat selection hypotheses relative to pasture and cropland cover. One portion of the study area was in Calhoun and Orangeburg counties and was composed of 42% forest, 22% crop, 14% shrub-scrub, 10% pasture, and 12% other land cover types as determined by the 2011 National Land Cover Database (NLCD; Homer et al. 2015). The other part of the study area was in Colleton County and was composed of 61% forest, 15% shrub-scrub, 6% each of pasture and crop, and 12% other land cover types as determined by 2011 NLCD. The coastal plain is characterized by flat topography (0–100 m) and fertile soil that was once under the ocean. Corn and cotton were the dominant crop types, and pasture included residential lawns, overgrown fields, hay, and livestock

Table 1. Covariates hypothesized to influence breeding season loggerhead shrike occupancy in South Carolina, USA, 2016–2017, and their predicted effect. We also present the scales at which we examined each covariate in *a priori* models.

Occupancy covariate	Predicted effect	Spatial scales examined			
		300 m	1 km	5 km	15 km
Percent pasture	(+)	x	x	x	
Percent crop	(-)		x	x	
Percent forest	(-)	x	x	x	x
Aggregation of pasture	(+)			x	x
Aggregation of pasture and crop	(+)			x	
Average max. breeding season NDVI ^a	(-)	x	x	x	

^a Normalized difference vegetation index.

pastures. During the data collection seasons (Mar–Jun 2016 and 2017), average monthly precipitation ranged from 5 to 10 cm, and average daily temperature extremes ranged from a low of 13°C to a high of 27.5°C.

METHODS

Data Collection Methods

We conducted point count surveys to determine loggerhead shrike occupancy of selected survey sites. To select survey sites, we used the 2011 NLCD layer (Homer et al. 2015) in ArcGIS (Environmental Systems Research Institute, Redlands, CA, USA) to create polygons of pasture and cultivated crop cover types in our study area. We chose to use 2011 NLCD data because we determined through ground truthing that it more accurately identified pasture patches than other data sets, including the yearly National Agriculture Statistics Service Cropscape layer (www.nass.usda.gov, accessed 15 Feb 2016). We chose to use a roadside survey methodology because our study area had a high road density allowing 90% of all the pasture and crop polygons to be considered for survey site selection, and because previous research suggests roadside surveys are not likely to affect grassland bird detection probabilities (Lituma and Buehler 2016). Accordingly, we retained only polygons adjacent to non-major roadways. We also eliminated polygons <0.5 ha because they would be too small for a loggerhead shrike territory (Pruitt 2000). To ensure that a range of polygon sizes were equally represented in our study, we separated the remaining polygons into 4 size classes, small (0.5–3.49 ha), medium (3.5–7.49 ha), large (7.5–11.49 ha), and extra-large (≥ 11.5 ha), with the small size reflecting the minimum territory size of a shrike (Pruitt 2000), the medium and large size classes being representative of different published average shrike territory sizes (Pruitt 2000), and the extra-large size capable of supporting larger than average territories or multiple territories. We then used the generalized random tessellation stratified (GRTS) sampling design with the `spsurvey` package (Kincaid and Olsen 2016) in Program R version 3.3.3 (R Core Team 2017) to select the final polygons for establishing survey sites. The GRTS sampling allowed for a spatially balanced, random sampling distribution so that each polygon land cover type and size was represented in our survey sites across the study area. When first visiting a GRTS selected polygon, we established a roadside point that allowed the maximum viewing area of the polygon, which we used as the center point for the survey site.

We used previously reported detection and occupancy probability rates to determine the number of survey sites and replicate visits used in our study. Shrikes occurred at an average of 62% of sites in Big Bend National Park, Texas (Gutzwiller and Barrow 2001) with an estimated detection probability of 0.24, which we derived from reported occupancy and mean abundance of shrikes detected per count per site (Gutzwiller and Barrow 2001, 2002). Using these estimates and power analyses (MacKenzie et al. 2006), we determined that we needed to survey up to 96 sites, 7–10

times each to be 95% confident that we would detect shrikes during ≥ 1 survey if they were present at a site. We surveyed 96 points (1 point at each site) over 2 years, where we surveyed 12 points adjacent to each polygon type (crop or pasture) and size class (small to extra large). To ensure independence between sampling sites, we maintained a minimum of 1 km between survey sites based on the diameter of the largest territory size reported (47 ha; A. A. Chabot, unpublished report). To maintain closure within the sampling period, we conducted all surveys during the breeding season when shrikes are territorial and most likely to remain in one location (Pruitt 2000). In 2016, we surveyed each site 9 times between 24 March and 16 June. We obtained relatively higher detection estimates than expected during 2016 ($P=0.50$ in 2016 vs. $P=0.24$ expected based on literature surveys); therefore, we adjusted our survey effort in 2017 to survey sites only 4 times between 24 March and 22 May.

At each survey site we conducted unlimited-radius point counts using a combined methodology of a passive point count followed by a call playback period. Before we started a survey, we used a rangefinder to establish visual distance landmarks to ensure accurate distance estimates. Our surveys started with a 10-minute passive point count where we recorded all bird individuals of any species that we saw or heard, immediately followed by a 6-minute call playback sequence during which we looked and listened for adult loggerhead shrikes. The playback sequence was as follows: 20 seconds of song, 40 seconds of silence, 20 seconds of begging call, 40 seconds of silence, 20 seconds of alarm call, 40 seconds of silence. This sequence repeated for the second 3 minutes of the callback survey. For shrike and all other avian detections, we recorded distance to the bird and whether the detection was auditory or visual. We also recorded detection variables at the start of each survey either by a direct measure (temperature, time of day, ordinal date) or as an index (noise, wind, sky conditions). We indexed noise from 0 to 4 where 0 was no noise and 4 was loud noise preventing the detection of any birds beyond 50 m (Huff et al. 2000). We indexed wind using the Beaufort wind scale from 0 to 5 where 0 was calm, and 5 was 27–34 km/hour. We indexed sky conditions by the following: 0 = clear, 1 = partly cloudy, 2 = overcast, 3 = fog, 4 = drizzle or light rain, and 5 = rain. A single observer conducted all point count surveys within 4 hours after sunrise, and varied the time of day between visits to an individual point.

We used the 2011 NLCD to obtain spatial data for landscape characteristics (Homer et al. 2015) and calculated percent cover and aggregation index values (an area-weighted like-adjacency metric to address class-specific degree of fragmentation) using the `SDMTools` package (VanDerWal et al. 2014) in Program R, which uses calculations according to FRAGSTATS (McGarigal et al. 2012). We estimated vegetation productivity using normalized difference vegetation index (NDVI) data obtained through the National Air and Space Administration Moderate Resolution Imaging Radiospectrometer (MODIS) data (Didan 2015) and reformatted to the

same spatial scale as the NLCD data (30 m). We standardized all covariates using the scale function in Program R prior to analysis.

Analytical Methods

We used a 2-step occupancy modeling approach (MacKenzie et al. 2006) to evaluate habitat selection at multiple spatial scales. In the first step, we examined how weather (temperature, sky conditions, wind speed), variability in timing (time of day or ordinal date), and ambient noise levels, affected the probability of detection. We predicted that increasing temperature, wind, and worsening sky conditions would hinder detection as strong winds, clouds, and extreme temperatures can all decrease activity level of birds because of energy budgeting (Richards 1981, Robbins 1981a). We predicted that unlike most passerine species, shrikes would be more active later in the morning, and therefore, detections would increase with time of day (Robbins 1981b). We also predicted that more ambient noise would decrease our detection ability because the observer would have difficulty hearing a calling bird, and because shrikes would have difficulty hearing and reacting to the call-playback. Finally, we predicted that detectability of shrikes would peak during the breeding season when most individuals are breeding (Skirvin 1981) and highly territorial (Pruitt 2000). Peak breeding occurs during April and May depending on location (Yosef 1996), so we used a quadratic form ($x + x^2$) of ordinal date to represent this hypothesized relationship in our survey period, March–June. We developed 8 *a priori* models to evaluate hypothesized effects of individual and combined covariates on detection while holding occupancy constant (Table S1, available online in Supporting Information). We tested for correlations between covariates and did not include correlated covariates ($r > 0.7$) in the same model. We ran all occupancy models using the unmarked package (Fiske and Chandler 2011) in Program R. We compared models using Akaike's Information Criterion corrected for small sample sizes (AIC_c) and by examining model weight (Burnham and Anderson 2002). We considered all models within the 90% cumulative model weight to influence detection and carried over all covariates within this confidence set to the detection portion of our models in step 2.

In step 2 we examined support for hypothesized habitat and landscape variables at 4 different scales to predict shrike site occupancy. Our smallest scale was a 300-m radius from the center of the survey site. This was the farthest we detected a shrike from a point count; therefore, this distance represented the survey site scale and the scale at which we were able to make occupancy predictions. We also used a 1-km-radius scale, which is representative of the maximum distance a shrike would travel in the breeding season (A. A. Chabot, unpublished report). Finally, we analyzed the 5-km and 15-km scales because they were significant in other shrike studies looking at landscape factors (Burton and Whitehead 1990, Chabot et al. 2001). We hypothesized that the influence of land cover on occupancy depends on spatial

scales (Table 1). For example, because shrikes use pastures (Luukkonen 1987, Gawlik and Bildstein 1990, Eshel and Bollinger 2001, Walk et al. 2006), we predicted that percent pasture at the 3 smaller scales would have a positive effect on occupancy probability. In addition, because shrike occupancy can decline in more productive grasslands (Shen et al. 2013), we predicted increasing average maximum breeding season NDVI (index for productivity) at the 3 smaller scales would have a negative effect on occupancy probability. We predicted that percent crop cover at the 1-km and 5-km scales would have a negative effect on occupancy probability because of previous reports of row crop avoidance (DeGeus 1990), and we predicted that percent forest at all scales would have a negative effect on occupancy probability because shrike occurrence has also been found to be negatively associated with forest cover (Johnson 2017). For the forest cover covariate, we combined all forest types in the NLCD, but we were unable to separate out longleaf pine (*Pinus palustris*) savannas, which are native habitat for shrikes (Askins et al. 2007), because this cover type data is not yet compiled. We also predicted that the aggregation of pasture at the 2 largest scales, as a measure of connectivity and fragmentation, would have a positive effect on occupancy probability, and that the aggregation of pasture and cropland combined into 1 open land cover type at the 5-km scale would have a positive effect on occupancy probability (A. A. Chabot, unpublished report).

We used our 15 occupancy covariates (Table 1) to formulate 23 *a priori* occupancy models, evaluating hypothesized effects of land cover, aggregation, and primary productivity at each selected scale separately, and between scales on occupancy probability (Table S2, available online in Supporting Information). We tested for correlations between covariates to determine if any model contained correlated covariates ($r > 0.7$). In instances where correlated covariates occurred in the same model, we ran separate occupancy models with each singular covariate, and retained the covariate that produced the lower AIC_c score. We compared our *a priori* models using AIC_c (Burnham and Anderson 2002). We determined that models carrying any portion of the upper 90% AIC_c cumulative weight were important to occupancy rates and fell within our confidence set of models. We calculated model-averaged occupancy, detection, and covariate estimates, and their 95% confidence intervals based on all models in our 90% confidence set. We assumed that covariates with 95% confidence intervals not overlapping zero were important predictors of detection or occupancy.

We used a Pearson chi-square test with 10,000 bootstrap replications to determine if our averaged model fit our data (MacKenzie and Bailey 2004). We also used the bootstrap replications to estimate the overdispersion parameter \hat{c} , where a \hat{c} value near 1 indicated that the model adequately described the data (MacKenzie and Bailey 2004). To further assess model performance, we conducted a 10-fold cross validation (Boyce et al. 2002) by randomly separating our data by a 75:25 ratio to establish our training and testing datasets, running our averaged model with the training

dataset, and then using the resulting averaged beta coefficients to predict occupancy with the testing data set. We assessed differences between predicted and known occupancy of the testing datasets using a receiving operator characteristic (ROC) and calculating the area under the curve (AUC; Boyce et al. 2002). We ultimately determined model performance using the AUC value, which could range from 0.0 (predicts opposite of the truth 100% of the time) to 0.5 (no better than a null model) to 1.0 (predicts truth 100% of the time). If our model performed well (AUC > 0.7; Morelli et al. 2017) in the cross validation analysis, we applied it across a grid of the coastal plain of South Carolina with a resolution equal to the size of our sampling unit (530 × 530-m resolution) to predict occupancy probabilities. Because occupancy probability is also a measure of proportion area occupied (MacKenzie et al. 2002), we binned the extrapolated occupancy probability values by tenths, multiplied the bin area by the bin mean occupancy probability, and scaled it to the whole South Carolina coastal plain area to determine the area and the proportion of area occupied by loggerhead shrikes in each probability bin. Finally, we summed all bins together to get total predicted area and proportion of area occupied by loggerhead shrikes in the whole coastal plain of South Carolina during the 2016 and 2017 breeding seasons.

Additionally, to inform future loggerhead shrike monitoring efforts, we used our model confidence set to determine if our detection and occupancy probabilities would change if we did not include a playback period. We removed detection data collected during the playback section of our surveys and ran our confidence set with this modified dataset. We then compared 95% confidence intervals of detection and occupancy probability between the original dataset and the modified dataset to determine if estimates changed when we removed the playback survey period. We determined if occupancy and detection probability estimates were similar when confidence intervals overlapped between datasets.

RESULTS

We detected loggerhead shrikes at 20 of 94 survey sites. Our detection model confidence set from step 1 included 5 models (Table 2) and included all of our detection covariates. Therefore, we carried over all detection covariates and used them in our second model-fitting step. Five models were in the 90% confidence set in step 2 (Table 2). When we held covariates at their mean value, our model-averaged detection rate was 0.486 ± 0.082 (SE), and occupancy was 0.166 ± 0.0465 . Within the confidence set, sky was the only model-averaged detection covariate whose 95% confidence interval of the beta estimate did not overlap zero, though confidence intervals for temperature and noise indicated moderate support because they barely overlapped zero (Table 3). Detection estimates increased from 26% when skies were cloudy to 63% when skies were clear, from 4% at our minimum survey temperature (2°C) to 83% at our maximum survey temperature (28°C), and from 41% when moderate noise was present to 63% when there was no background noise. Pasture within 1 km was the only model-averaged occupancy covariate with a confidence interval not overlapping zero (Table 3). Predicted occupancy increased from 2% when amount of pasture at a 1-km radius was 0% to 26% at 15% pasture, and reached 98% occupancy at our maximum pasture percentage of 45% (Fig. 1).

Our final model from our confidence set (Table 3) fit our data ($\chi^2 = 220.42$, $P = 0.35$, $\hat{\epsilon} = 1.06$). The occupancy portion of this final model set performed well in our cross validation with an AUC of 0.78. Extrapolation of our model across the coastal plains of South Carolina indicated few areas with high occupancy probability (Fig. 2) and predicted that the proportion of area occupied by loggerhead shrikes in the South Carolina coastal plain was 8% (Table 4).

The detection portion of our averaged model did not perform very well in our cross validation with an AUC of 0.57,

Table 2. Rankings of models of breeding season loggerhead shrike detection (ρ) and occupancy (ψ) in South Carolina, USA, 2016–2017. We present only models retained in the 90% confidence sets for step 1 and step 2. All occupancy covariates include the scale of measurement (e.g., 1 km = 1-km radius).

Model ^a	K^b	AIC _c ^c	ΔAIC_c	w_i^d	Log L ^e
Detection (step 1) confidence set					
$\psi(.)\rho(\text{wind+temp+sky})$	5	232.27	0.00	0.31	−110.80
$\psi(.)\rho(\text{wind+temp+sky+noise})$	6	322.01	0.73	0.22	−110.02
$\psi(.)\rho(\text{time})$	3	234.37	2.10	0.11	−114.05
$\psi(.)\rho(\text{wind+temp+sky+date2})$	7	234.52	2.14	0.11	−109.56
$\psi(.)\rho(\text{wind+temp+sky+time})$	6	235.37	2.25	0.10	−110.78
$\psi(.)\rho(.)$	2	235.57	3.29	0.06	−115.72
Occupancy (step 2) confidence set					
$\psi(\text{pasture1 km})\rho(\text{global})$	10	216.54	0.00	0.47	−96.94
$\psi(\text{pasture1 km+NDVI1 km})\rho(\text{global})$	11	218.67	2.13	0.16	−96.73
$\psi(\text{pasture1 km+forest15 km})\rho(\text{global})$	11	219.10	2.56	0.13	−96.94
$\psi(\text{pasture1 km+forest5 km})\rho(\text{global})$	11	219.10	2.56	0.13	−96.94
$\psi(\text{pasture1 km+forest15 km+NDVI300 m})\rho(\text{global})$	12	220.31	3.77	0.07	−96.23

^a Wind = wind strength during the survey, temp = ambient temperature at the time of survey, sky = sky conditions at time of survey (e.g., clear, cloudy, rainy), noise = ambient noise level during the survey, date² = ordinal date of survey in a quadratic form, pasture = percent pasture, global = all covariates from step 1, NDVI = normalized difference vegetation index, forest = percent forest.

^b Number of parameters in the model.

^c Akaike's Information Criterion corrected for small sample sizes.

^d Akaike weight.

^e Log likelihood of the model.

Table 3. Model-averaged covariate estimates, standard errors (SE), and 95% confidence intervals (CI) for covariates in the final 90% confidence model set for loggerhead shrike detection and occupancy in the South Carolina Coastal Plain, USA, 2016 and 2017 breeding seasons. Occupancy covariates reflect the normalized difference vegetation index (NDVI) or percent of land cover at the scale shown in the covariate name.

Model-averaged covariate	Estimate	SE	CI
Detection covariates			
Sky conditions	-0.62	0.27	-1.16, -0.09
Wind	0.47	0.30	-0.12, 1.06
Temp	0.69	0.36	-0.01, 1.39
Noise	-0.39	0.22	-0.69, 0.04
Time	-0.27	0.36	-0.98, 0.45
Date ²	0.22	0.22	-0.20, 0.65
Date	-0.42	0.31	-1.04, 0.19
Occupancy covariates			
Pasture 1 km	1.70	0.50	0.71, 2.69
NDVI 1 km	-0.21	0.31	-0.82, 0.31
Forest 15 km	0.03	0.34	-0.64, 0.69
NDVI 300 m	-0.40	0.33	-1.05, 0.25
Forest 5 km	-0.02	0.31	-0.64, 0.59

suggesting it does not have a strong predictive ability. The passive survey period was sufficient to determine occupancy at all but 3 sites where shrikes were observed only during 1 survey, and only during the active survey period. Indeed, removing data gathered in the active survey period from the model did not greatly affect detection rates (0.44 ± 0.09 without playback, compared to 0.49 ± 0.08 with playback included), or occupancy rates (0.10 ± 0.04 without playback, compared to 0.17 ± 0.05 with playback) although both detection and occupancy did increase slightly with playback.

DISCUSSION

Our finding that occupancy of loggerhead shrikes in the South Carolina Coastal Plain during the breeding season was best predicted by the availability of pasture at a 1-km scale suggests that shrikes are sensitive to habitat at a broader spatial scale than previously thought. Previous shrike research has focused on habitat associations within 100–300 m of nest sites (Luukkonen 1987, Gawlik and Bildstein 1990, Esely and Bollinger 2001, Walk et al. 2006), or on breeding territories (Yosef and Grubb 1994) that are smaller (4–13 ha; Pruitt 2000) than our 1-km scale. Our landscape-scale findings are consistent with recent findings elsewhere in the southeast that predicted shrike occupancy to decline as percent forest cover increased at the 2.5-km scale (Johnson 2017). We investigated only breeding season occupancy, but pasture availability within 1 km could be important to shrikes for securing specific resources throughout the rest of the year. Shrike territories expanded during experimental prey scarcity (Yosef and Deyrup 1998), suggesting that fluctuations in prey availability could explain changes in territory size and selection of more habitat than what is used on an average day-to-day basis. Further, territories expand during the post-fledging period (A. A. Chabot, unpublished report), suggesting extra buffer habitat around the smaller breeding season territory could provide essential resources during the vulnerable post-fledging period. Thus, to better understand the influence of these moderate scale effects of pasture on site occupancy,

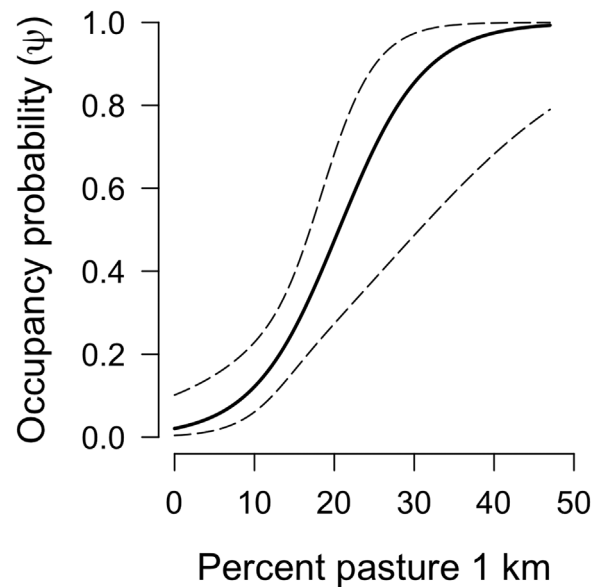


Figure 1. Predicted model-averaged effect of percent pasture within 1 km on occupancy probability for loggerhead shrikes in the South Carolina Coastal Plain, USA, during the 2016 and 2017 breeding seasons. Dashed lines represent the 95% confidence interval.

further research of shrike space use is likely needed during the fledgling and little-studied non-breeding seasons.

Our lack of support for land cover aggregation effects on shrike occupancy could be explained by the strength of our percent pasture predictor and by temporal limitations. Simulations testing the importance of habitat variables to extinction thresholds suggest that the abundance of habitat on a landscape has a much larger effect on extinction than does fragmentation (Fahrig 2001), and grassland percentage has been reported to be a more consistent predictor of grassland bird abundance than fragmentation (Renfrew and Ribic 2008). Additionally, because fragmentation occurs over time, effects may appear only when analyzing long-term data from when fragmentation occurred. For example, Ethier et al. (2017) reported negative effects of large-scale fragmentation on bobolinks (*Dolichonyx oryzivorus*) when analyzing data over 25 years, but when they analyzed data in 5-year increments, they did not detect effects of fragmentation. Our snapshot analysis suggests that fragmentation was not an important predictor of occupancy during our survey period, though low occupancy probabilities and high fragmentation of pastures in our study area could indicate that loggerhead shrikes in our study could have already suffered from fragmentation. Additionally, our use of 2011 NLCD data could be inaccurate in capturing the amount of fragmentation during our data collection period, though, as stated in the methods, the NLCD more accurately identified pasture patches than other more current data sets.

Although our study identified only pasture as a strong land cover predictor of shrike occupancy (Table 3), NLCD lumped rural residential areas, hay, old fields, grassy areas, and pasture into a single pasture category and we did not distinguish between crop type. Delineating land cover data

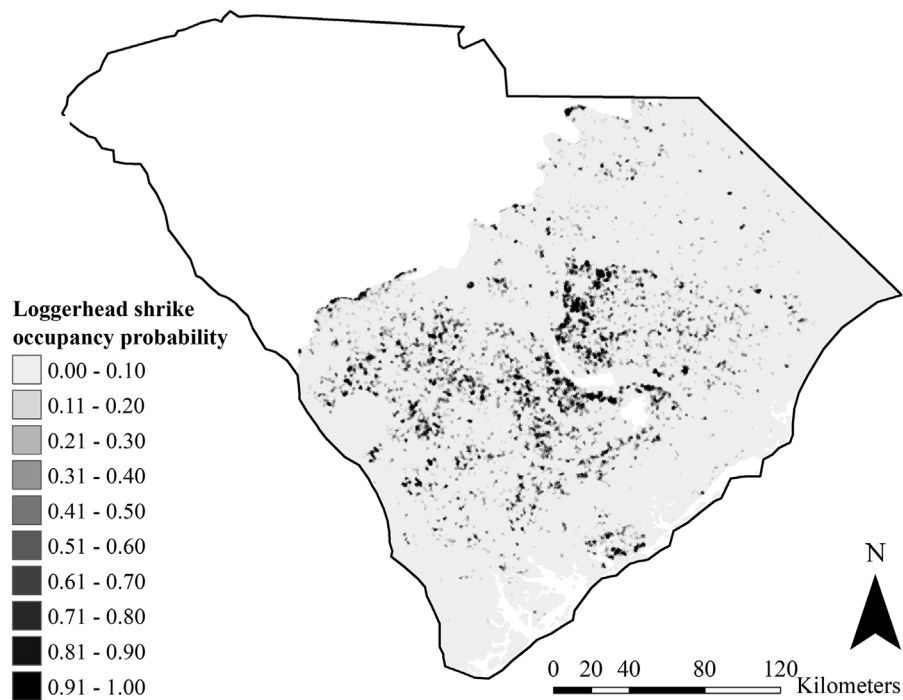


Figure 2. Predicted loggerhead shrike occupancy probability across the South Carolina Coastal Plain, USA, during the 2016 and 2017 breeding seasons.

into more specific cover types and monitoring shrike occupancy through time could allow researchers to determine more specific habitat preferences, and how shrikes are responding to land cover change. Further, because occupancy measures only presence, not abundance or vital rates, it is unknown whether abundance or fitness varies between different open land cover types. For example, by examining population growth rates in different land cover types, Arlt et al. (2008) were able to determine that northern wheatears (*Oenanthe oenanthe*) in pastures and farmyards acted as population sources, but those in crop fields and tall grasslands were population sinks, and that population decline was influenced by the decline of source habitats. Further investigation of shrike demographic parameters, including adult survival, juvenile survival, nesting success, and annual fecundity, is needed to determine if different open land cover types act as population sources or sinks. This is especially important in areas like South Carolina that are primarily composed of small private landholdings. Identification of source and sink habitats will help identify private land that is most able to provide conservation benefits and thus also identify where agencies should focus on landowner engagement.

Our study illustrates the utility of passive point counts in monitoring shrikes during the breeding season, although poor detection model validation results suggest that more research is needed. Our model suggests that surveying in fair weather, higher temperatures, and low noise levels result in higher detection probability. Our results also suggest that a passive point count is sufficient for detecting shrikes in our study system, though occupancy and detectability did slightly increase with the addition of the active survey period, indicating that active surveys may be helpful for finding shrike territories in areas of low occupancy.

The effective use of passive-only point counts to monitor shrike population trends could be beneficial to researchers because other species of interest can also be observed at the same time without biasing shrike-specific detection. Currently, shrikes are surveyed differently between researchers, and standardization of survey methodology is needed so that comparisons can be made between or within studies (A. A. Chabot, personal communication). Our results suggest that passive point counts could be used as part of a standardized survey protocol in the breeding season. Further, in areas like South Carolina that are host to both residents and overwintering migrants, we suggest that surveying in the non-breeding season could be a vital addition to shrike monitoring. Detection could vary by season (Ralph 1981) and it is unknown if a passive survey would be as effective in the non-breeding season because shrikes may be less active. Habitat use could similarly vary by season and by life-history strategy, where optimal habitat may be more limited in the non-breeding season because of the influx of migrant individuals (Johnson 2017), and migrants may be settling for more marginal habitat because residents get first pick of the best habitat (Lynn and Temple 1991). Thus, pairing breeding and non-breeding surveys could provide an understanding of temporal factors influencing shrike detection and habitat selection, which we did not investigate in this study.

Overall, our exploration into habitat-scale relationships for loggerhead shrikes confirms that landscape-scale habitat conditions influence shrike breeding habitat selection. The scarcity of shrike habitat in our predictive map of the South Carolina Coastal Plain illustrates the importance of maintaining open pastures in higher concentrations on the South Carolina landscape for the recovery of the shrike population. Maintaining high concentrations of pasture is also likely to benefit other grassland bird species of

Table 4. Extrapolated area in hectares (ha) occupied by loggerhead shrikes in the South Carolina Coastal Plain, USA during the 2016 and 2017 breeding seasons.

Estimated occupancy (bins)	Total ha in bin	Mean ha occupied	Mean proportion area occupied
0.00–0.10	4,169,342.5	125,080.3	0.025
0.11–0.20	367,445.3	52,177.2	0.010
0.21–0.30	154,691.6	37,899.4	0.008
0.31–0.40	89,270.0	30,976.7	0.006
0.41–0.50	62,107.0	27,886.0	0.006
0.51–0.60	47,135.0	25,877.1	0.005
0.61–0.70	38,370.9	24,864.4	0.005
0.71–0.80	31,629.3	23,722.0	0.005
0.81–0.90	26,882.1	22,984.2	0.005
0.91–1.00	33,033.8	31,580.4	0.006
Total	5,019,907.7	403,047.8	0.080

conservation concern, which as a guild, have experienced more severe declines than any other avian guild in North America (Rosenberg et al. 2016). Savanna sparrow (*Passerculus sandwichensis*), grasshopper sparrow (*Ammodramus saviannarum*), and bobolink abundance was reported to be best explained by variables including percent forest and percent grassland at the 1.2-km scale (Renfrew and Ribic 2008), and occupancy of 11 out of 19 grassland species was best explained by percent tree cover at the 1.2–1.6-km scales (Cunningham and Johnson 2006). These correlative results suggest that preserving grassland and pasture at a 1-km scale for shrikes could help to protect habitat for numerous grassland bird species. Thus, conservation and restoration of grassland in high densities within 1 km will contribute to the conservation of loggerhead shrikes and many other declining grassland species.

MANAGEMENT IMPLICATIONS

Ultimately, conserving grassland in South Carolina and other southeastern states is likely to be challenging but necessary for continued and enhanced occupancy of loggerhead shrikes. Land-use predictions through 2051 for the southeastern United States highlight the susceptibility of open areas to land use change, predicting a 49.3% loss of pasture and a 24.2% loss of natural rangeland under increased crop demand (Martinuzzi et al. 2015). Our finding of the importance of pasture at moderate spatial scales, and predicted loss of grasslands, highlight the urgent need for maintaining existing grasslands and creating new grassland area. In South Carolina where 90% of land is privately owned, managers could use conservation incentive programs (e.g., farm bill cost share programs; Hellerstein 2017) to maintain and create grasslands on large private properties. Additionally, because many occupied sites occurred in residential and fragmented areas, managers should explore opportunities to inform residents about grassland conservation strategies, and to engage residents in meaningful conservation experiences that will help instill a personal reason to enact conservation on their land. However, our research highlights the importance of providing grassland for shrikes on a larger scale than a typical residential plot, suggesting coordination of conservation efforts at the neighborhood or community level will be paramount for effective shrike conservation in these fragmented areas.

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